SUSTAINABLE MANAGEMENT OF BIOGEOCHEMICAL CYCLES IN SOILS AMENDED WITH BIO-RESOURCES FROM LIVESTOCK, BIOENERGY, AND URBAN SYSTEMS

A Dissertation

by

RONNIE WAYNE SCHNELL

Submitted to the Office of Graduate Studies of
Texas A&M University
in partial fulfillment of the requirements for the degree of

DOCTOR OF PHILOSOPHY

August 2010

Major Subject: Agronomy

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Approved by:

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ABSTRACT

Sustainable Management of Biogeochemical Cycles in Soils Amended with Bio-Resources from Livestock, Bioenergy, and Urban Systems.

(August 2010)

Ronnie Wayne Schnell, B.S., Sam Houston State University; M.S., Texas A&M University

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Bioresources are generated in a variety of environments and each presents unique risks and benefits associated with land application. Bioresources from livestock, urban and bioenergy systems were selected and evaluated through field, greenhouse and laboratory studies of potential risk and benefits of recycling to agricultural and urban landscapes.

The waste stream, including feedstock sources and treatment processes, affects composition and properties of bioresources and effects on biogeochemical cycles of amended soils. Variation of decomposition and nutrient mineralization rates among bioresources used to amend soil for turfgrass and forage reflected variation among contrasting feedstock sources and treatments prior to application. During turfgrass establishment, plant available nitrogen and nitrogen mineralized from a bioresource from livestock waste streams, (Geotube® residual solids, supplied N in excess of crop uptake potential and contributed to leaching loss of N. In contrast, N mineralization rates from bioresources generated during methane production from dairy manure (manure solids) were not sufficient to maximize crop production, necessitating N fertilizer application.

In addition to variation of composition, bioresource effects on crop productivity and environmental quality vary among management practices and between forage and turfgrass cropping systems. Large application rates of bioresources increase soil nutrient concentration and potential crop productivity, but contribute to increased nutrient loss in drainage and surface runoff. Yet, incorporation or Alum treatment of bioresources will reduce runoff loss of dissolved P and protect water quality without sacrificing crop productivity. Alum treatment of bioresources prior to land application effectively reduced runoff loss of dissolved P to levels observed for control soil.

For situations in which large, volume-based bioresource rates are top-dressed or incorporated, export of applied nutrients environmental impacts were compared between forage and turfgrass systems. Starting during the initial year of production, annual export of applied N and P in Tifway bermudagrass sod was greater than export through forage harvests of Tifton 85. Low forage yield limited N and P export from Tifton 85 during the year of establishment, but increased forage yield during the second year increased export of manure N and P to levels more comparable to sod. As variation between compost sources, turfgrass and forage production systems, and application methods indicated, effective management of bioresources is necessary to balance benefits and risk in cropping systems. Integrated assessment of bioresource composition and crop-specific management of application method and rate will enable sustainable bioresource cycling and crop productivity.

ACKNOWLEDGEMENTS

I would like to thank my committee chair, Dr. Vietor, and my committee members, Dr. Provin, Dr. White, and Dr. Munster, for their guidence and support throughout the course of this research. Thanks also to the department of Soil and Crop Sciences faculty and staff for making my experience at Texas A&M University enjoyable.

This research was supported, in part, by United States Geological Service-Texas Water Resource Institute Graduate Student Grant Program, Texas State Soil and Water Conservation board and the Southern SARE Graduate Student Grant Program. I also want to extend my gradtitude to Bill Allen, the staff of the Soil, Water & Forage Testing Laboratory, and all of the student workers for their support and cooperation.

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CHAPTER I

INTRODUCTION

Growth of urban populations is increasing demand for food and limited energy supplies in Texas and the Nation. As a growing population expands markets for meat and milk and for biofuels from bioenergy production systems, byproducts of production and consumption must be managed. These byproducts comprise carbon and nutrients that could serve as bioresources for improving soil physical and chemical properties and crop productivity. Diminishing supplies of mineral nutrients and the carbon costs associated with extraction, processing and transport of plant nutrients are among incentives for recycling bioresources. Yet, applying bioresources at rates that exceed crop nutrient requirements can diminish soil and water quality. Sustainable cropping systems and bioresource management practices need to be identified and evaluated to protect soil and water quality.

The expansion of milk production in Texas has contributed 5 million dry tons of animal manure each year. Advances in on farm manure management practices and development of off-farm anaerobic digesters have accompanied the increased manure supply. The Microgy facility in Huckabay Ridge is expected produce over 1 billion cubic feet of methane per year from the digested manure of 10,000 cows. In addition to methane generation from manure, crop residues and dedicated bioenergy crops are being evaluated

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for the production of biofuels across Texas. The emergence of bioenergy facilities will produce significant amounts of by-products that could be applied to agricultural soils.

In addition to byproducts of livestock and energy production, consumption of agricultural and industrial products results in urban waste streams comprising carbon and mineral nutrients. Urban centers across Texas produce about 8 million tons of fresh biosolids each year, 650,000 dry tons of which are disposed of in landfills (www.texasep.org). Limited landfill space and increasing cost of mineral nutrients provide incentives for cycling biosolid sources of carbon and nutrients back to agricultural fields. The source and composition of biosolids and other bioresources produced in Texas vary, yet all are composed of carbon and nutrients that could benefit soil properties and crop production.

Recycling bioresources is not only encouraged for conservation of nutrients but may be necessary for sustainable crop production. Annual harvest of entire crop biomass for bioenergy production removes large quantities of nutrients and would necessitate application of fertilizer nutrients or recycling of byproducts to maintain productivity (Heggenstaller et al., 2008). Hons et al. (1986) reported that an additional 45, 10 and 140 kg ha^{-1} of total N, P and K are removed when energy sorghum biomass is harvested in addition to grain. If bioenergy facilities are located near agricultural land used for energy crop production, systems for recycling byproducts back to production fields may be feasible and contribute to sustainable bioenergy production.

In contrast to bioenergy systems, distances between areas of consumption and generation of bioresources and areas of production have uncoupled nutrient cycles for urban and animal agriculture systems. In lieu of costly disposal, bioresources are often applied to near by agricultural land. Repeated application of bioresources to land can result in excessive nutrient accumulation and deterioration of water quality (Berka et al., 2001). A variety of forage cropping systems have been evaluated to estimate nutrient export potential from soils amended with bioresources (Brink et al., 2004; Ketterings et al., 2007; Woodard et al., 2007). Yet, application rates often exceed crop uptake potential. Export of excess nutrients in value-added products may be a feasible option to forage production (Bergstrom et al., 2006). Previous studies have evaluated exporting excess manure nutrients in value-added products such as turfgrass sod (Vietor et al., 2002). The uptake and export of bioresource nutrients from amended soils will be dependent on concentration and form of nutrients.

The specific nutrient and carbon content of bioresources varies among sources and suppliers. Analysis of 24 samples of fresh dairy manure indicated mean total N (TN), total P (TP) and organic C content was 18.9, 3.49 and 351 g kg^{-1} respectively (Sharpley and Moyer, 2000). In addition, TP concentration in composted dairy manure (16.25 g kg^{-1}) was greater than fresh manure. Composted municipal biosolids were found to have TN (27.3 g kg^{-1}) and TP (14.7 g kg^{-1}) concentration comparable to composted dairy manure (Loschinkohl and Boehm, 2001). Although TP concentration was greater for composted manure and biosolids compared to fresh manure, comparative concentrations of water extractable P (WEP) do not necessarily reflect variation of TP between bioresources. The ratio of WEP to TP for fresh dairy manure can be 3 to 19 times greater than that of composted dairy manure or biosolids (Leytem et al., 2004). The concentration of WEP in manures and compost is reported to directly influence runoff loss of P from amended soils (Kleinman et al., 2002b).

Mineralization of carbon and nutrients following application of composted manure and biosolids provides mineral nutrients for crop growth and increased productivity. Tognetti et al (2008) showed that up to 33% of carbon and 101 mg kg^{-1} of N was mineralized over a 16 week period in soil amended with composted biosolids. Similarly, 24 to 33% of applied carbon and up to 8 % of applied N was mineralized for soil receiving composted dairy manure (Gale et al., 2006). As the previous studies suggest, carbon and nutrient mineralization rates of various bioresources need be evaluated in amended soil to estimate contributions to crop production. Flavel and Murphey (2006) demonstrated that mineralization of N in soils amended with composted biosolid warranted only partial reductions in fertilizer N requirements of a given crop.

In contrast to reported mineralization rates for nutrients and carbon supplied by composted manures and biosolids, properties of bioenergy byproducts may limit mineralization of carbon and nutrients. Anaerobic digestion of animal manure produces a slurry byproduct comprising a solid fraction that can be separated and composted. The residual solid fraction contains non-digestible substances, including lignin and bacterial biomass (Pesta, 2006). Biological recalcitrance of the residual solids in soil could limit mineralization rates of carbon and nutrients compared to composted manures and biosolids (Van Kessel et al., 2000).

In addition to the residual solids remaining after anaerobic digestion, byproducts of pyrolysis of manure and biomass feedstocks are a potential bioresource for carbon and nutrients. The bio-char byproduct of bio-oil and syngas production through pyrolysis is a carbon rich, residual solid that is highly resistant to decay (Lehmann, 2007). The stability of biochar against decay provides the opportunity to sequester carbon in soil while increasing the capacity of soil to retain nutrients and reduce environmental pollution (Lehmann, 2007). Furthermore, cycling biochar to bioenergy crop fields may be necessary for sustainable bioenergy production (Anex et al., 2007; Lehmann et al., 2006). Significant quantities of N, P, K and C can be exported in crop biomass and residues used for bioenergy production. Recycling byproducts of bioenergy production may be necessary to maintain levels of C and nutrients in soil (Anex et al., 2007; Johnson et al., 2004).

In addition to benefiting crop growth, environmental risks are associated with application of bioresources to soil (Vietor et al., 2004). Yet, bioresources can be managed to manipulate biogeochemical processes in soil and preserve surface and subsurface water quality. Incorporating manure can reduce runoff concentration of dissolved reactive P (DRP) compared to surface application (Kleinman et al., 2002a). Furthermore, Miller et al., (2006) found that composting dairy manure before application to soil reduced dissolved reactive P (DRP) in runoff compared to fresh manure. Composting manures and biosolids can reduce WEP of compost and reduce leaching and runoff loss of P of amended soils (Miller et al., 2006; Sharpley and Moyer, 2000). In addition, chemical agents can be mixed with bioresources to reduce extractable P and reduce runoff loss of P from amended soils (DeLaune et al., 2006). For bioenergy byproducts, limited data exist concerning impacts on water quality following application to soil. Low mineralization rates and concentrations of extractable nutrients in byproducts of bioenergy production could reduce runoff loss of nutrients compared to manures and compost. Evaluations of options for management of varied bio-resources are needed to support development of sustainable methods of cycling organic C and nutrients and protecting environmental quality.

Sustainable bio-resource cycling systems need to be developed and evaluated for the diverse array of byproducts available from livestock and bioenergy production and municipalities. System development will depend on bio-resource properties and supply; responses of crop productivity and quality; effects on soil physical, chemical, and biological properties; and impacts on environmental quality. Four diverse sources of bioresources were chosen for study. These bio-resources will be evaluated for grass crops capable of rapid establishment and coverage and conservation of added mineral nutrients and available soil and water resources.

CHAPTER II

CYCLING OF SOLIDS FROM GEOTUBE® TREATMENT OF DAIRY LAGOON WASTEWATER

Geotube® Dewatering Systems (Miratech Division of Ten Cate Nicolon Corporation; Commerce, Ga.) collect solids (GRS) from waste liquids of municipal wastewater and septic systems, confined animal feeding operations, and contaminated marine environments. Solids are retained during pumping of waste liquids into large porous tubes or socks (~ 300 m^3) constructed of synthetic fibers. In applications designed to separate solids from runoff and process wastewater of confined dairy feeding operations, Alum (Aluminum sulfate) and polyacrylamide are injected during pumping of waste liquid into the Geotube® (Mukhtar et al., 2007). Compared to untreated waste liquid, the metal salt and polymers enhanced flocculation of particulates and associated nutrients and increase water flow rate through the Geotube® wall (Worley et al., 2008). Water draining from the Geotube® was returned to source lagoons or separate retention ponds. For liquid pumped from an agitated dairy wastewater lagoon, the Geotube® system reduced total solids 93.5%, soluble P 85%, and total P 96% (Mukhtar et al., 2007).

Although GRS solids separated from liquid wastes of industrial and municipal sources are often landfilled, GRS from wastewater of livestock operations is a potential bioresource for agricultural lands. Similar to manure, GRS could increase soil organic C (SOC) and provide essential mineral nutrients on fields proximate to livestock and composting operations (Gregorich et al., 1998; Hay et al., 2007; Tisdale et al., 1993). In addition, the GRS sources of nutrients could produce and be exported through harvests of marketable crop products. Yet, repeated land application of GRS at rates exceeding crop nutrient requirements could exceed regulatory limits on soil nutrient concentrations (O'Connor et al., 2005). The high soil concentrations contribute to nutrient transport and loss in water draining through soil (Hay et al., 2007; Maguire and Sims, 2002). Sustainable systems for managing GRS will require maintenance of field, landscape, and regional nutrient balances (Bergstrom et al., 2006).

Turfgrass sod is among the marketable crops through which GRS sources of nutrients and organic C offer environmental and economic benefits (Munster et al., 2004). Harvest of a shallow layer of GRS-amended soil with turfgrass sod, a non-food crop, exports more nutrients and organic C from fields on which GRS are applied than forage, grain, or fiber crops (Vietor et al., 2002). In addition, the GRS and associated polymer within the layer of harvested sod could reduce removal of native soil and enhance soil structure and water infiltration and storage compared to sod grown in mineral soil (Green and Stott, 2001; Schnell et al., 2009). Moreover, the GRS sources of nutrients within the harvested sod layer will reduce fertilizer requirements of the transplanted sod (Munster et al., 2004).

Although Alum injection enhances flocculation and separation of solids and P from waste liquid in the Geotube®, Alum hydrolysis products could reduce pH and plant growth in GRS-amended soil (Tisdale et al., 1993). Yet, Alum mixed with composted biosolids reduced neither pH or grass establishment compared to soil amended with biosolids only (Vietor et al., 2010a). Before volume-based GRS rates are used to amend soil for turfgrass establishment under field conditions, effects on plant growth and soil physical, chemical and biological properties need to be evaluated (Aggelides and Londra, 2000; Malecki-Brown et al., 2007; Wang et al., 1998).

OBJECTIVES

1. Evaluate effects of increasing rates of GRS on physical, chemical, and biological properties of contrasting soil types and turfgrass establishment.
2. Evaluate leaching losses of nutrients from contrasting soil types with and without incorporation of GRS during turfgrass establishment.

MATERIALS AND METHODS

Mineralization Study

A randomized block design composed of four replications of the factorial combination of two soil types and three rates of GRS was implemented in incubation vessels (0.5 L) to evaluate C and N mineralization rates. A control and two volume-based GRS rates (0.125, and 0.25 m^3 m^{-3}) were mixed in 80 g dry soil, wetted to 60% of soil water holding capacity, and incubated for 56 days at 25° C (Haney et al., 2004). Alkali traps (0.25 N NaOH) absorbed CO_2 and were titrated with acid (0.25 N HCl) at 1, 2, 3, 7, 14, 21, 28, 35, 42, 49, 56 days to determine carbon mineralization rates (Haney et al., 2004). Extractable inorganic N (NO_3-N + NH_4-N) concentrations in soil mixtures were determined colorimetrically before and after incubation to estimate N mineralization rates (Sims et al., 1995).

The percent of GRS applied C remaining was calculated for each sampling date as described previously (Kaboneka et al., 1997). The C respired from soil without GRS was subtracted from the amount respired from soil with GRS for each soil type before subtracting from the amount of C applied with GRS. The percent C remaining for each soil type and level of GRS rate was plotted versus time. The datum was fit to a four parameter double exponent decay model described by Berndt (2008). The model included parameters for fast and slow decomposing fractions of GRS. Fitting the model for percent GRS C remaining for Tifway and Tifton 85 plots was done using SPSS 18.0, which used the Levenberg-Marquardt algorithm to find the best fit.

Column Lysimeter Study

A randomized block design comprised four replications of a factorial arrangement of two soil types and three rates of GRS in column lysimeters (10-cm diameter x 30-cm depth) under greenhouse conditions. Soil for each a Windthorst fine sandy loam (fine, mixed, thermic Udic Paleustalfs) and Weswood silt loam (fine, mixed, thermic Fluventic Ustochrept) was air dried and sieved (< 10 mm). The GRS was collected after 1 yr of storage in a Geotube® that was used to remove sediment and P from wastewater of a confined dairy feeding operation (Mukhtar et al., 2007). The air-dried GRS was sieved through a 10-mm mesh screen before sampling, analysis, and incorporation in soil. Soil with or without GRS was packed within 5-cm increments to achieve a consistent bulk density throughout a 30-cm depth over a layer of glass fiber cloth, which separated soil from a 5-cm depth of washed pea gravel. Two volume-based GRS rates (0.125, and 0.25 m^3 m^{-3}) were incorporated in soil packed within the top 10 cm of lysimeters as two 5-cm depth increments. The volume base rates of GRS were equivalent to 75.8- and 152.8- Mg of dry GRS per hectare. Tifway bermudagrass (Cynodon dactylon L. Pers. X C. transvaalensis Burtt-Davey) was sprigged in all treatments after soil was firmed into columns. After sprigging, the hydrostatic pressure of a water column was used initially to wet soil from the bottom to soil surface within lysimeters. Excess water in each column, which drained through a fitting at the bottom, was collected for analysis. Well water was deionized through an in-line cartridge prior to use for irrigation of lysimeters.

One pore volume of leachate was displaced and collected from each column through surface irrigation with distilled water at 45 and 90 days after sprigging. Pore volumes were estimated using weights and particles densities of soil mixtures loaded into lysimeters. Leachate volumes were measured, sub-sampled, and filtered (< 0.45 μm). Dissolved reactive P (DRP), NH_4-N, and NO_3-N in filtrate were analyzed colorimetrically within 24 hr after leachate was collected (Pierzynski, 2000; Sims et al., 1995). Inductively coupled plasma optical emission spectroscopy (ICP) was used to measure total dissolved P (TDP) in filtrate (Eaton and Franson, 2005). Dissolved unreactive P (DUP) was calculated as the difference between TDP and DRP. An Elementar Rapid Liquid TOC analyzer (Hanau, Germany) was used to analyze dissolved organic carbon (DOC) in filtrate.

Before packing in lysimeters, soil and GRS were sampled, dried, ground (< 2 mm) for analysis. Soil was digested using nitric acid and total P determined using ICP (Havlin and Soltanpour, 1980). In addition, P was extracted using water (WEP) and the Mehlich-3 solution (M3P) (Mehlich, 1984; Pierzynski, 2000). Soil concentration of total carbon and N was determined using combustion procedures (McGeehan and Naylor, 1988). In addition, lysimeters were cut into segments at 90 d after sprigging for sampling and analyses of soil from 0- to 10-cm and 10- to 30-cm depths. Soil water content was determined gravimetrically for oven-dried soil (60° C, 48 hr). Turfgrass was clipped to a 5-cm height when plant height exceeded 15 cm. Clippings were dried, composited over cutting dates, weighed, ground, and digested for analysis of total N and P (Havlin and Soltanpour, 1980). Total dry matter production over 90 days was used to compute average daily biomass production (g m^{-2} d^{-1}).

Analysis of variance was performed using SPSS 18.0 (Chicago, Illinios) to compare leaching losses and soil concentrations of nutrients and organic C among establishment treatments. Leaching events were analyzed separately. Soil and leaching data were analyzed as a complete randomized block design. If interaction was observed between soil type and GRS rate ($P < 0.05$), soil types were analyzed separately. Fisher's least significant difference test ($P = 0.05$) was used to compare treatment means. Regression analysis was used to evaluate relationships between variation of soil and leachate concentrations of P and organic C.

RESULTS AND DISCUSSION

Soil and GRS Properties

Total and extractable P and total N concentrations were similar between the Windthorst and Weswood soils before incorporation of GRS (A-1). In contrast, pH and total organic carbon (TOC) concentration were greater for the Weswood than the Windthorst soil. In addition, extractable NO_3-N concentration of the Windthorst soil was 24 times greater than the Weswood soil. After air drying and storage in the Geotube® under field conditions, total P concentration in GRS was similar to values reported for semisolid dairy manure (Leytem et al., 2004). In contrast, WEP concentration was 98% lower in GRS than in the semisolid dairy manure (A-1). The Alum and polymer injected during pumping of waste liquid into the Geotube® could have adsorbed dissolved P forms during dewatering of waste liquid and prevented P dissolution in water extracts of GRS (Mukhtar et al., 2007). Previous laboratory-scale studies indicated incorporation of Alum at an Al to total P molar ratio of 1.7 minimized water-soluble P release from anaerobically-digested biosolid (Huang and Shenker, 2004).

Similar to Alum acidulation of soil, pH in GRS collected after Alum injection in dairy waste liquid was relatively low compared to values ranging from 8.1 to 8.5 for composted or fresh dairy manures (A-1) (Sharpley and Moyer, 2000). In addition, nitrification of mineralized N in GRS during storage under field conditions could have lowered pH. The NO_3-N concentration in GRS was far greater than values reported for dairy manures and municipal organic waste (Johnson et al., 2006; Tognetti et al., 2008).

Large application rates of GRS could reduce soil pH and affect growth of acid-sensitive crops.

Analysis of soil with and without GRS following 90 d of turfgrass establishment revealed soil pH was lowest ($P < 0.001$) for the highest GRS rate, but pH of the Windthorst soil increased with or without GRS (A-1, A-2). Increases in both soil pH and Na concentration within the 10-cm depth at the 90-d sampling indicated Na bypassed the filtration cartridge and was applied in irrigation water. Before filtration of well water, Na concentration was 246 mg L^{-1} and pH was 8.06. Mehlich-3 extractable Na concentration of soil without GRS was an average of 4.2-fold greater than concentration in soil before treatments were imposed (A-1, A-2). Although irrigation water was a potential source of Na and dissolved salts, hydrolysis products of Alum in GRS and nitrification of mineralized N could have prevented increases in soil mixed with GRS compared to controls for each soil (A-2). Previous reports indicated increasing rates of compost or organic amendments increased or decreased soil pH, depending on composition (Aggelides and Londra, 2000). Similar to the present study, increasing rates of sewage sludge treated with Alum reduced soil pH (Wang et al., 1998). In contrast, incorporation of 0.25 m^3 m^{-3} of composted biosolids minimized Alum effects on pH of a sandy loam soil during establishment of Tifway bermudagrass (Vietor et al., 2010a).

Similar to soil pH, irrigation water likely moderated effects of GRS sources of Alum hydrolysis products on extractable soil Al. Mehlich-3 Al concentration was greater ($P < 0.001$) for Windthorst (50 mg kg^{-1}) compared to Weswood (16.5 mg kg) soil, but similar with or without GRS for each soil. The GRS effects on soil pH and Mehlich-3 Al will depend on interactions among Alum injection rates into waste liquids and buffering effects of irrigation water, amended soil, and organic matter in GRS.

Incorporation of the respective GRS rates within the 10-cm soil depth increased concentrations of soil organic carbon (SOC) to 3 and 5 times that of soil alone (A-2). In addition, incorporation of 0.25 m^3 GRS m^{-3} soil decreased ($P < 0.05$) soil bulk density 14.4% compared to soil without GRS. Previous reports corroborate this inverse relationship between soil bulk density and application rate of biosolids (Evanylo et al., 2008). Similar to the present study, a volume-based rate (300 m^3 ha^{-1}) of composted municipal biosolids reduced bulk density 19.7% after incorporation in a loamy soil (Aggelides and Londra, 2000).

In contrast to reports of increased soil gravimetric water content for soil amended with compost (Aggelides and Londra, 2000), water content was lowest ($P < 0.05$) for soil mixed with 0.25 m^3 m^{-3} of GRS (Table A-2). In the present study, variation of turfgrass uptake and transpiration of water prior to soil sampling at 90 d could have accounted for differences in soil water content among treatments. Turfgrass biomass production was greater ($P = 0.05$) for turfgrass grown in Windthorst soil than in Weswood soil (A-3). In addition, biomass production increased ($P = 0.05$) for both soil types as GRS application rate increased (A-3). An inverse relationship ($r^2 = 0.55$, data not shown) was observed between soil water content and turfgrass biomass production. Increasing GRS rates could have increased soil N and mineral nutrient concentrations and contributed to greater turfgrass biomass production and water use (Evanylo et al., 2008).

Analyses of soil sampled 90 d after incorporation of GRS indicated increasing GRS rates increased ($P < 0.001$) plant-available nutrients in both soil types, including concentrations of Mehlich-3 and water extractable P (WEP) and extractable NH_4-N (A-4). In addition, both total N and P increased ($P < 0.01$) as GRS rate increased (A-2). Variation of extractable soil NO_3-N concentration did not differ ($P < 0.05$) between soil types or among GRS rates, but soil NO_3-N concentration was linearly related to GRS rate in both soil types (A-4). The coefficients of determination for regression relationships between GRS application rate and soil extractable NO_3-N concentration were $r^2 = 0.95$ for Windthorst and $r^2 = 0.91$ for the Weswood soil (data not shown). The NO_3-N extracted from soil sampled at 90 d comprised NO_3-N available at incorporation of GRS and NO_3-N mineralized from TN during the ensuing 90 d. As variation of grass biomass among GRS rates indicates (A-3), the NO_3-N added in GRS and derived from mineralization of TN in GRS could have contributed to enhanced turfgrass establishment and growth (A-4).

Carbon and Nitrogen Mineralization

Cumulative carbon mineralization was similar ($P > 0.05$) between Windthorst and Weswood soil types and was pooled to evaluate GRS rate effects. For all sampling dates over a 56-d period, increasing rates of incorporated GRS increased ($p < 0.001$) cumulative soil carbon mineralization in the order $0 < 0.125 < 0.25$ m m^{-3} of GRS (A-5). Although respective increases in GRS rate increased cumulative carbon mineralization 62% and 289% compared to soil without GRS, only 1.8 to 2.9% of GRS carbon was mineralized over a 56-day period (A-5). The relatively low mineralization rates for GRS were comparable to soil amended with 75 to 88% less carbon from straw-based compost (Flavel and Murphy, 2006). The GRS sampled from Geotube® socks under field conditions could have comprised relatively recalcitrant forms of carbon that remained after storage for 1 yr under field conditions.

Variation of cumulative carbon mineralization among GRS rates reflected the amount of organic matter added rather than differences in degradability. The percent carbon remaining plotted versus time was fitted to a four-parameter double exponent kinetic decay model. The decay constant of fast and slow decomposing C pools in GRS was greater for Weswood than for Windthorst soil (A-6). In addition, the slow decomposing fraction comprised greater than 99% of carbon added with GRS (A-6). Similarly, a previous study observed that greater than 98% of C from bermudagrass thatch was associated with the slow pool (Berndt, 2008). Compared to various other bioresources, the decay rate of GRS was relatively low (Gale et al., 2006), which was attributed to possible reductions in fresh organic matter before collection from Geotube® socks.

Similar to organic C, TN mineralization rate over 56 days was similar between Windthorst and Weswood soils. Averaged between soil types, the respective increases of GRS rate increased ($P < 0.001$) TN mineralization of NH_4-N and NO_3-N by 3- and 4-fold during the 56-day incubation period (A-7). The percentage of the initial amount of TN applied with GRS, excluding NO_3-N, that was mineralized to NH_4-N and NO_3-N over 56 d was 8.0 to 9.4% for the low GRS rate (0.125 m^3 m^{-3}) and 5.5 to 6.6% for the high GRS rate (0.25 m^3 m^{-3}). The N mineralization rates were similar to values reported for compost (Flavel and Murphy, 2006), but the percentage of TN mineralized was slightly greater for

GRS than for municipal organic waste (Tognetti et al., 2008). High initial NO_3-N concentrations in GRS collected from Geotubes® could have contributed to greater mineralization of TN than for municipal wastes after both were mixed with soil. Concentrations and mineralization rates of initial N forms in GRS need to be quantified to optimize rates and timing of applications for turfgrass establishment. In addition, increases in soil concentrations of SOC, WEP, and NO_3-N associated with large, volume-based GRS rates need to be monitored and managed to prevent leaching losses during turfgrass establishment (Maguire and Sims, 2002).

Leaching Loss of Nutrients and Carbon

Consistent with observed increases in soil WEP, both GRS rates (0.125 and 0.25 $m^3\ m^{-3}$) contributed to greater ($P < 0.01$) concentrations of dissolved P forms in leachate compared to controls without GRS (A-8). Dissolved un-reactive P (DUP) concentration in leachate, calculated as concentrations of TDP – DRP, made up a large proportion of TDP at 45 and 90 d after GRS incorporation. The DUP is operationally determined through TDP and DRP analyses, but could comprise organic P and small amounts of inorganic P, including polyphosphates and pyrophosphates (Broberg and Persson, 1988). At 90 d, the percentage of TDP that was DUP ranged from 60 to 90% for soils without GRS and from 87 to 100% for soils with GRS. Similarly, Chardon et al. (1997; 2007) reported that 90% of TP in leachate from soil columns amended with manure solids or slurries was DUP. Previous studies evaluating P species in animal manure indicated 30% of the total P in manure is organic P (Hansen et al., 2004).

In contrast to large proportions of DUP in TDP for the current study, Kleinman et al. (2005) observed that surface applied poultry manure increased the fraction of DRP in leachate TDP from 7 to 72%. In the present study, injection of Alum and polymers during dewatering of waste liquid could have adsorbed or precipitated DRP in GRS and minimized concentrations and transport of DRP in leachate from soils amended with GRS. In addition, soil sorption and turfgrass uptake could have reduced soil concentration of DRP compared to DUP concentration in leachate. Low concentrations of DRP could limit the portion of TDP that is bioavailable, yet hydrolysis of DUP could serve as a long-term source of P in aquatic ecosystems (Toor et al., 2003).

Similar to GRS effects on soil and leachate concentrations of P forms, increasing GRS rates increased SOC concentration and dissolved organic carbon (DOC) concentration in leachate at 45 days after planting. Incorporating 0.125 m^3 m^{-3} of GRS increased leachate concentration of DOC 59% and incorporating 0.25 m^3 m^{-3} increased leachate concentration of DOC 76% compared to soil without GRS (data not shown). A previous evaluation of leaching losses from transplanted sod indicated DOC loss was two times greater for sod grown with composted biosolids than for sod grown with inorganic fertilizer (Dai et al., 2009). After turfgrass was established in lysimeters for the present study (90-d sampling), DOC concentration in leachate was similar with or without incorporation of GRS. Leaching loss of DOC from turfgrass biomass at 90 days could have masked contributions of GRS to DOC leaching loss and variation of DOC leaching loss among treatments (Dai et al., 2009; Wright et al., 2008). For sods transplanted from turfgrass grown with composted biosolids, the proportion of SOC attributed to turfgrass sources increased during a 10-month period of establishment (Dai et al., 2009).

The concentration of DOC in leachate at 45 days after planting was linearly related to the concentration of DUP (A-9). The relationship between leaching loss of DUP and DOC is not unexpected considering DUP comprised organic P and both DOC and DUP concentrations reflected the application rates of GRS. Yet, it does not eliminate the possibility of DOC contributing to transport of inorganic P forms in soil (Donald et al., 1993). Injection of Alum during dewatering of waste liquid could have formed phosphate-metal-humic complexes less than 0.45 µm in diameter, which were recovered in leachate at 45 d, before being masked by turfgrass sources of DOC at 90 d (Guardado et al., 2007).

The effects of soil NO_3-N on leachate NO_3-N concentrations, whether applied with GRS or released through mineralization of TN, differed ($P < 0.05$) between leachate collection dates (A-4, A-10). Similar to soil NO_3-N concentrations, NO_3-N concentrations in leachate collected at 90 d were similar with and without GRS for the Weswood soil and greater for 0.25 $m^3\ m^{-3}$ of GRS in the Windthorst soil. Similarly, leachate concentrations of NO_3-N were greatest ($P < 0.05$) at 45 d for 0.25 $m^3\ m^{-3}$ of GRS mixed with the Windthorst soil (A-10). For the Windthorst soil, NO_3-N concentrations in leachate were 87% and 170% greater for respective increases in GRS rate than without GRS. Leachate concentrations of NO_3-N were greater with than without GRS for the Weswood soil, but were more than 50% less than the Windthorst soil at respective GRS rates at 45 d. Reductions in leachate NO_3-N concentrations from 45 to 90 d after planting indicated slow mineralization rates of TN in GRS were not sufficient to replace root uptake and leaching losses of NO_3-N during turfgrass establishment (A-5, A-10). For both soils, the high NO_3-N concentration in leachate collected 45 d after planting exceeded USEPA regulatory limits for drinking water. In previous studies, incorporation of composted manures increased leachate NO_3-N concentrations less than observed for GRS in the present study, but concentrations similarly exceeded regulatory limits (Evanylo et al., 2008).

CONCLUSION

Incorporation of GRS enhanced turfgrass growth and physical and chemical properties of contrasting soil types, but leaching loss of dissolved nutrients could be problematic during turfgrass establishment. Although volume-based GRS rates provided plant-available N and P and mineralizable C and N forms in soil, NO_3-N and dissolved P forms exceeded turfgrass uptake and increased leachate concentrations during turfgrass establishment. Low decay rates after mixing with soil and high NO_3-N concentration in GRS indicated substantial mineralization of organic N occurred during 1 yr of storage in Geotube® socks before collection and incorporation in soil. At high soil pH, residual Alum and associated hydrolysis products in GRS did not increase extractable soil Al, but did minimize leachate concentrations of dissolved reactive P forms. Composition, timing and application rate of GRS need to be managed to optimize turfgrass establishment while preventing leaching loss of dissolved nutrients.

CHAPTER III

CHEMICALLY-TREATED COMPOSTED BIOSOLIDS ENHANCE WATER CONSERVATION AND QUALITY ON URBAN LANDSCAPES

Water conservation is a priority in many cities that routinely face water shortages and drought in Texas and similar regions of the world. Soil management is fundamental to water conservation efforts. Incorporation of composted biosolids (CB) in low quality soils can enhance water conservation and provide organic carbon and nutrients that improve growth of vegetation and limit sediment loss (McCoy, 1998; Schnell et al., 2010). Schnell et al. (Schnell et al., 2009) demonstrated up to 53% greater turfgrass coverage of the soil surface during establishment and 49% greater soil water content at harvest for sod grown in soil mixed with CB (25% v/v) than for sod grown in soil alone. In addition, the volume-based rate of CB reduced soil bulk density 34% and sod weight at harvest 19% compared to sod without CB.

The contribution of CB to improvements in water conservation and soil properties can be offset by potential impacts on water quality. Volume-based CB rates can increase total and extractable P concentration in soil, which contributes to increased concentration and mass loss of dissolved P in runoff and drainage from CB-amended soils (Kleinman et al., 2002b; Schnell et al., 2010). Increased P concentrations in surface waters can contribute to algal blooms and result in depletion of oxygen and potential fish kills (Sharpley et al., 1992). The challenge is to develop and evaluate practices that will immobilize P in CB and preclude detrimental effects of volume based CB rates on water quality.

Both CB and soil chemical properties need to be considered during CB management. For example, increasing clay concentration increases potential sorption of dissolved P incorporated in soil with CB. In an effort to further limit dissolved P loss from CB amended soil, Alum [$Al_2(SO_4)_3 \cdot 18 H_2O$], calcium hydroxide [$Ca(OH)_2$], and other chemical agents have been incorporated with CB and byproducts of anaerobically-digested sludge to immobilize P and NH_4-N (Huang and Shenker, 2004). Yet, the amount of Alum or $Ca(OH)_2$ in CB must be managed carefully to limit P solubility without large changes in pH. The stability of P forms in chemically treated CB and potential transport of soluble P forms in runoff and drainage from soils amended with CB, Alum, and $Ca(OH)_2$ need to be evaluated for contrasting soil types.

OBJECTIVES

1. Evaluate Alum and $Ca(OH)_2$ effects on CB and soil properties.
2. Evaluate Alum and $Ca(OH)_2$ effects on runoff and drainage losses of dissolved P forms during turfgrass establishment on contrasting soil types with and without CB amendments.

MATERIALS AND METHODS

An incubation experiment was conducted to identify Alum or $Ca(OH)_2$ rates needed to minimize water extractable phosphorus (WEP) concentrations in CB. The CB was a composted municipal biosolid collected at the Hornsby Bend Biosolid Management plant in Austin, Texas. Similar to previous studies, increasing rates of Alum (0, 1.17, 2.34, 3.51, 4.68 and 7.02 g Alum per 100 g dry CB) and $Ca(OH)_2$ (0, 0.46, 0.91, 1.37, 1.82 and 2.73 g $Ca(OH)_2$ per 100 g dry CB) were dissolved in distilled water and mixed and incubated with CB at 30°C for 1 week (Huang and Shenker, 2004). A complete randomized design comprised three replications of each rate of Alum or $Ca(OH)_2$ in 25 g (dry weight, 60°C 48 hr) of fresh CB. After incubation, 4.0 g of CB was extracted during shaking in 40 ml deionized water for 1 hr. The solution was filtered (< 0.45 µm) after shaking and soluble reactive P was analyzed colorimetrically (Kleinman et al., 2007; Pierzynski, 2000).

Alum and $Ca(OH)_2$ effects on runoff and drainage losses of P were evaluated during turfgrass establishment in box lysimeters (45.5 x 33 x 15.0 cm). Three replications of eight establishment treatments were arranged in a randomized complete block design under greenhouse conditions. Two control treatments comprised Tifway Bermudagrass (*Cynodon dactylon* L. Pers. X *C. transvaalensis,* Burtt-Davey) sprigged in each a calcareous (Branyon clay) and acidic (Burleson clay) soil types (Fine, smectitic, thermic Udic Haplusterts). For the remaining six treatments, Tifway was sprigged after incorporation of untreated CB and of CB treated with 0.075 kg Alum kg^{-1} CB and 0.029 kg

$Ca(OH)_2$ kg^{-1} CB. A 15-cm depth of control or treated soil was packed into lysimeters over glass fiber cloth on a 5-cm depth of washed pea gravel. Alum or $Ca(OH)_2$ was dissolved in distilled water, sprayed on CB, mixed in a portable cement mixer, and incubated in 100-L plastic drums for one week before mixing with soil. After incubation, the CB was mixed with each soil (0.25 m^3 m^{-3}, ~167 Mg ha^{-1}) in the portable cement mixer before soil was packed into lysimeters.

Tifway sprigs were washed from sod and planted at a 1-cm depth after soil was packed in lysimeters. Measurements of dielectric Aquameter sensors(McMichael and Lascano, 2003) in soil and solar radiation and air temperature sensors were recorded on a data logger and used to manage daily water applications to lysimeters. Turfgrass was mowed to a 5-cm height and clippings were removed, dried, and pooled during turfgrass establishment and maintenance. An oscillating, indoor multiple-intensity (10 cm hr^{-1}) rainfall simulator was used to apply 30 min. of rain at 15, 30, and 60 d after planting of Tifway sprigs (Birt and Persyn, 2007). Lysimeters were positioned on a metal frame beneath the rainfall simulator to impose a soil-surface slope of 7%. The runoff volume from each lysimeter was collected at 10-min. intervals from a flume attached to the down-slope lysimeter wall, measured, and sampled for analysis. In addition, runoff volumes were composited over the three 10-min. intervals of each rain event for sampling and analysis. Subsurface drainage (leachate) volumes were collected over each 30-min. rain event, measured, and sampled for analysis. Runoff and drainage samples were filtered (< 0.45 μm) before dissolved P forms in filtrate were analyzed colorimetrically using the molybdate-blue method (Pierzynski, 2000). In addition, an ICP was used to analyze total dissolved P in filtrate (Eaton and Franson, 2005).

Soil and CB were sampled before sprigging and soil was sampled again after the third rain event to quantify concentrations of total and extractable N and P forms and organic C with and without CB and Alum or $Ca(OH)_2$. Soil sampled after the third rain event was separated into 0- to 5-cm and 5- to 15-cm depths for analysis of chemical and physical properties. Soil bulk density and water content were measured gravimetrically. Mehlich-3 solution or deionized water were used to extract P from CB and soil samples (Mehlich, 1984; Pierzynski, 2000). The ICP was used to analyze total P in nitric acid digests of soil and CB and in Mehlich-3 extracts (Havlin and Soltanpour, 1980). The NO_3-N in 1.0 M KCl extracts of CB and soil were analyzed through cadmium reduction. Dissolved reactive phosphorus in water extracts of CB and soil was measured colorimetrically within 24 hours of extraction and filtering (Pierzynski, 2000).

The GLM procedure in SPSS 18.0 (Chicago, Illinios) was used for analysis of variance among treatments, soil types, rain events, and sampling intervals. Rain events at 15, 30 and 60 days after planting were significantly different and were analyzed separately. Soil types were analyzed separately if statistically significant interaction between soil type and treatment were observed for soil variables. Fischer's test for least significant difference (LSD, $P = 0.05$) was used to separate means of soil treatments.

RESULTS AND DISCUSSION

Soil and CB Analysis

Separate analyses of soil and CB chemical properties indicated Mehlich-3 and water extractable P (WEP) concentrations were similar between the Burleson and Branyon soils (Table 1). In contrast, total P concentration and pH were higher for the Branyon soil and total N and organic C concentrations were higher for the Burleson soil. The application volume (2500 m^3 ha^{-1}) and total P concentration of CB with or without Alum or $Ca(OH)_2$ contributed an average of 1576 kg ha^{-1} of total P to both soil types. The CB and total P application rates were much greater than reported previously for top-dressed composted manure sources (Vietor et al., 2004) or for volume-based rates of composted biosolids incorporated within a shallower depth of soil (Schnell et al., 2009; Vietor et al., 2004). Large increases in soil concentration of total and extractable P can contribute to increased runoff loss of P and diminish water quality (Kleinman and Sharpley, 2003; Schnell et al., 2010).

Chemical Treatment of CB

Incubation of CB with increasing Alum or $Ca(OH)_2$ concentrations reduced ($P < 0.001$) WEP compared to untreated CB (B-1). Concentrations of WEP were similar among Alum concentrations ranging from 2.34 to 7.02 g 100 g CB and lower ($P < 0.001$) than untreated CB. The Alum treatment of 7.02 g per 100 g CB reduced WEP concentration of CB 84.5%. The reduction ($P < 0.001$) of WEP observed for 7.02 g Alum 100 g^{-1} CB compared to the control was comparable to an 85-93% reduction in WEP for fresh poultry litter treated with 10% Alum by weight (DeLaune et al., 2006). Similarly, concentrations greater than 0.46 g $Ca(OH)_2$ 100 g CB reduced WEP concentration compared to CB without $Ca(OH)_2$ (B-1). Similar WEP concentrations among $Ca(OH)_2$ concentrations greater than 1.37 g per 100 g CB were lower ($P < 0.05$) than un-treated CB. The highest $Ca(OH)_2$ concentration incubated with CB reduced WEP 65.6%, which exceeded reductions (46-49%) reported for composted cattle manure treated with calcium amendments (Whalen (2002). Similarly, Huang and Shenker (2004) reported reductions in water soluble P for a wide range of calcium oxide treatments of sewage sludge. The concentrations of 7.02 g Alum and 2.73 g $Ca(OH)_2$ per 100 g CB were selected for runoff experiments.

Soil Chemical and Physical Properties

Incorporation of large, volume-based rates (0.25 m^3 m^{-3}, ~167 Mg dry CB ha^{-1}) of CB, with or without Alum or $Ca(OH)_2$, increased ($P < 0.001$) soil concentrations of organic C, total P, and Mehlich-3 P nearly two-fold compared to the control soil without CB (B-3). In addition, the volume-based CB rate reduced ($P < 0.001$) soil bulk density 17.9% compared to soil without CB. Similar reductions in bulk density were reported for clay soils amended with large volumes of compost (300 m^3 ha^{-1}) (Aggelides and Londra, 2000). Aggelides et al. (2000) demonstrated greater than 2-fold increases in organic matter content for clay soils amended with large volumes of compost.

Treating CB with Alum before incorporation in both soil types resulted in greater reductions ($P < 0.001$) in soil concentration of WEP than the $Ca(OH)_2$ treatment of CB. Although CB with or without Alum or $Ca(OH)_2$ increased ($P < 0.001$) soil total and Mehlich-3 P compared to soil alone, WEP concentration was similar between control soil and soil amended with Alum–treated CB (B-3). Alum treatment reduced ($P=0.001$) soil WEP 70% compared to soil mixed with untreated CB. Similarly, mixing $Ca(OH)_2$ with CB lowered ($P < 0.005$) soil WEP concentration 30.4% below that of soil amended with untreated CB (B-3). Previous studies evaluating surface sprays of Alum and Alum incorporated with composted biosolids observed similar reductions in soil WEP concentration (Vietor et al., 2010b).

The reductions in soil WEP concentrations indicated Alum or $Ca(OH)_2$ treatments of CB could limit runoff or drainage losses of P compared to untreated CB. Previous studies demonstrated chemical treatments reduced soluble soil P and potential P transport in runoff or drainage from soil amended with poultry litter or composted biosolids (DeLaune et al., 2006; Vietor et al., 2010a). In the present study, the incubations and soil analyses indicated Alum more effectively reduced soluble P in CB and CB-amended soil than $Ca(OH)_2$ for the range of chemical additions evaluated.

A significant interaction ($P = 0.001$) between soil type and CB treatments indicated variation of soil pH with and without CB and with and without Alum or $Ca(OH)_2$ was dependent on soil type (B-3). For Burleson soil, incorporating CB with or without Alum or $Ca(OH)_2$ increased ($P < 0.001$) soil pH compared to soil without CB. In addition, $Ca(OH)_2$ increased soil pH compared to soil amended with CB with or without Alum (B-3). In contrast to Burleson soil, incorporating CB with or without Alum in Branyon soil decreased soil pH compared to soil without CB and soil with $Ca(OH)_2$-treated CB. Changes in soil pH due to additions of CB with and without chemical agents could affect the solubility of P forms in soil and CB and non-point source loss of P from amended soils (Penn and Bryant, 2008).

Phosphorus in Runoff

Runoff volumes collected over three 10-min. sampling intervals were similar among treatments for the three rain events. Mean runoff concentration of the dissolved reactive P form (DRP) was compared among treatments to evaluate Alum and $Ca(OH)_2$ effects on P loss from CB incorporated in soil (B-4). Variation of mean runoff concentrations of DRP among treatments was similar for each of three rain events. Compared to soil without CB, incorporation of untreated CB in soil increased runoff concentration of DRP an average of 150% over three rain events. Treating CB with Alum prior to incorporation prevented increases in mean runoff concentrations of DRP observed for both soil types compared to CB without Alum treatment during Tifway establishment. In contrast, runoff concentrations of DRP were greater ($P < 0.001$) for incorporated CB with or without $Ca(OH)_2$ than soil with or without Alum treated CB at 15 and 30 days after planting (B-4).

Compared to runoff at 15 and 30 d after sprigging of Tifway, lower mean DRP concentrations in runoff from all treatments at 60 d were attributed to previous runoff losses of DRP and Tifway uptake of P (Kleinman and Sharpley, 2003). Yet, DRP concentrations in runoff from CB-amended soil with Alum remained lower than soil amended with $Ca(OH)_2$-treated CB at 60 d (B-4). For all three simulated rain events, runoff concentrations of TDP were comparable to and comprised of primarily (89.3%) of DRP for all treatments and both soil types. The fraction of TDP that was DRP was greater than previously reported for soil amended with manure or fertilizer (Kleinman et al., 2002a).

Alum treatment of CB reduced mean DRP concentration in runoff 34.9% compared to un-treated CB over three rain events during Tifway establishment. Moore et al. (2000) reported a greater reduction in DRP (73%) in runoff from field applications of poultry litter with compared to without Alum over a 3 yr period. In contrast, the range of DRP concentration in runoff (1.60 – 6.29 mg L^{-1}) after poultry litter applications on tall fescue (*Festuca arundinacea*) were greater than observed for establishing Tifway (0.17 – 0.37 mg L^{-1} DRP) in the present study. Sorption of P in clay soils used in the current experiment could have limited mean concentrations of soil extractable P and P forms in runoff. Schnell et al. (2009) incorporated a similar volume-based CB rate for Tifway establishment, but reported a 2-times greater mean Mehlich-3 P concentration of soil (621 mg kg^{-1}) than observed in the present study. Greater soil-test P in the previous study was attributed to the lower adsorptive capacity of the sandy loam soil compared to the clay soils used for the present study.

Treating CB with $Ca(OH)_2$ reduced the concentration of WEP in CB (B-2) and soil, but was not effective in reducing runoff concentration of DRP from CB incorporated with the acidic or alkaline clay soils in lysimeters. In addition, treating CB with $Ca(OH)_2$ was not as effective in reducing soil WEP as Alum treatment of CB. Torbert et al. (2005) reported lime was less effective than ferrous sulfate as a surface amendment for reducing runoff loss of DRP from bermudagrass sod topdressed with manure. Insoluble P forms in $Ca(OH)_2$-treated CB could be less stable than Alum-treated CB in clay soils used in this study (Penn and Bryant, 2008). The collective effects of several mechanisms could explain higher soil concentration of WEP and runoff loss of DRP. For example, the increase in pH of Branyon soils with $Ca(OH)_2$ treated CB to 7.9 due to hydrolysis of calcium hydroxide may have contributed to calcite formation and concomitant decrease in soil Ca concentration and increased solubility of calcium-phosphate minerals (Penn and Bryant, 2008). In addition, coating of Ca-phosphates by organic matter can inhibit crystal growth resulting in Ca-P precipitation as non-crystalline nanoparticles (Huang and Shenker, 2004). Phosphorus associated with nanoparticles or colloids in runoff water filtrate (< 0.45 μm), which may include calcium phosphates, are likely hydrolyzed during colorimetric phosphorus determination (McDowell and Sharpley, 2001; Turner et al., 2004).

Phosphorus in Leachate

No leachate was collected during the first rain event 15 d after planting. At 30 d after planting, leachate was collected from all treatments but random variation in DRP concentrations precluded statistical differences between soil treatments. At 60 d after planting, chemically treating CB prior to incorporation reduced concentration of DRP in leachate for Burleson and Branyon soil types. Alum treatment of CB incorporated into both soil types reduced ($P < 0.05$) leachate DRP concentration 77.2% compared to soils amended with untreated CB. Leaching concentration of DRP was comparable between Alum-treated CB and soil without CB. In contrast to runoff from soil with $Ca(OH)_2$ treated CB , treating CB with $Ca(OH)_2$ was effective in reducing leaching loss of DRP compared to untreated CB. During the rain event at 60 d, $Ca(OH)_2$ treated CB reduced ($p < 0.05$) leachate concentration of DRP by 33.1% compared to untreated CB amended soils yet was greater ($P < 0.05$) than soil amended with Alum treated CB. Similarly, mixing Alum and Ca containing residuals from waste water treatment with biosolids reduced leaching loss of P (Elliott et al., 2002).

CONCLUSION

In summary, treating CB with 7.02% Alum or 2.73% $Ca(OH)_2$ on a dry weight basis effectively reduced WEP concentration in CB and in acidic and calcareous soils mixed with 25% by volume of the treated CB. Yet, Alum treatment of the CB prior to incorporation in soil reduced runoff and leachate concentrations of DRP compared to the CB treated with $Ca(OH)_2$. For three simulated rain events during establishment of turfgrass, runoff and leachate concentrations in water from soil mixed with Alum treated CB were similar to those observed for soils without CB. Results indicate Alum treated CB can be applied to soils with contrasting pH and limit P concentration in runoff and protect water quality. In contrast, insoluble complexes of DRP, Ca, and soil particles in soil amended with $Ca(OH)_2$-treated CB may be unstable in soils, runoff, and colorimetric assays of DRP. Reduced transport and loss of soluble P forms due to Alum treatment will enable recycling of large, volume-based rates of CB for improved water capture and storage in soils during turfgrass establishment and maintenance on urban landscapes.

CHAPTER IV

CROPPING SYSTEMS FOR SUSTAINABLE NUTRIENT MANAGEMENT AND DAIRY PRODUCTION

Similar to other regions in the US, the intensification of dairy production within Texas and other southeastern states has contributed to localized increases in cow numbers. Localized increases in cow numbers can contribute to nutrient loading on associated land and concerns about off-farm environmental impact. Manure and wastewater supply nitrogen (N) for cropping systems on dairies and surrounding watersheds, but 20% or less of associated phosphorus (P) is typically removed in forage harvests (Sanderson and Jones, 1997). Excess P remaining in soil is a potential nonpoint source of pollution and Texas regulations now mandate export of manure nutrients from dairies and specified watersheds to protect rivers and lakes. Similar situations exist across the USA (Beegle and Lanyon, 1994).

In an effort to prevent pollution from land-applied manure and wastewater, uptake of N and P in forage harvests have been evaluated for year-round cropping systems. Adapted forage crops are grown and harvested to remove P and prevent nonpoint-source losses of N and P. Woodward et al. (2007) reported annual harvests from a bermudagrass/rye cropping system removed 67 kg P ha^{-1} $cycle^{-1}$. Ketterings et al. (2007) reported a two-cut system applied to brown midrib sorghum removed up to 510 kg N ha^{-1} yr^{-1} and 101 kg P ha^{-1} yr^{-1}. Brink et al. (2004) observed average uptake and export of 300 kg N ha^{-1} yr^{-1} and 46.5 kg P ha^{-1} yr^{-1} for Tifton 85 bermudagrass.

Vietor et al. (2002) identified an option for export of two to three times more total P than the year-round forage harvests. A single harvest of Tifway bermudagrass sod exported up to 561 kg ha^{-1} of total N and 219 kg ha^{-1} of total P applied as raw or composted dairy manure. In addition, an economic analysis indicated turfgrass sod sales were sufficient to purchase forage to replace that grown on the land area allocated to sod production (Vietor et al. 2003).

The introduction of turfgrass sod will diversify forage systems designed to achieve the dual purposes of cow nutrition and nutrient management. Yet, crop and soil responses and export of applied nutrients had not been compared between forage and sod crops within the same space and time. The fate of applied manure nutrients in soil; which includes sorption reactions, mineralization of N, P and C, and uptake of nutrients by plants; will impact export and nonpoint-source losses (Flavel and Murphy, 2006; Jiao et al., 2007; Oehl et al., 2004). A side-by-side comparison between sod and forage production is needed to evaluate soil and crop responses and nutrient imports and exports for waste application fields and dairies. In addition, suitable waste management practices are needed for each crop to ensure the sustainability of dairy production in impaired watersheds.

The comparison between forage and turfgrass sod production systems will reveal advantages and disadvantages of diversifying crop enterprises used to manage nutrients in dairy production systems. In addition, measurements of soil, plant, and water quality responses will reveal the comparative effects of forage and turfgrass production practices on components of sustainability for dairy production systems. For example, amending manure and manure byproducts with Alum can significantly reduce concentrations of water extractable P and decrease runoff concentrations of soluble reactive P, which could enhance the feasibility of topdressing manure after sod or forage harvests (DeLaune et al., 2006). Conversely, incorporation of manure may be necessary to protect environmental quality. Incorporation of manure will increase P adsorption to soil particles and limit nonpoint source losses in runoff. In addition, incorporation of large manure rates could improve soil water retention and biomass production compared to surface application (Kleinman et al., 2002a). Knowledge of the integrated effects of forage or turfgrass production systems and associated practices on soil processes and plants will enable dairy producers to optimize nutrient management and land allocation to forage and turf crops.

OBJECTIVES

1. Compare forage and turfgrass sod production systems with respect to field-scale nutrient balance and effects of manure management practices on plants and soil properties and processes.
2. Compare runoff losses of P and organic C among manure management practices for forage and turfgrass sod production systems.
3. Compare mass balance of N and P on field scale between manure management practices and forage and turfgrass sod production systems.

MATERIALS AND METHODS

Field Study

Soil and crop responses to topdressed and incorporated manure solids (MS) were compared between Tifton 85 bermudagrass forage and Tifway bermudagrass turf. The MS comprised residues of anaerobic digestion and methane production from dairy manure collected April 2008 at the Microgy facility at Huckabay Ridge, Texas. The MS were applied with and without incorporation with commercially available Alum (1.0 g Al kg^{-1} Alum) to manage water-soluble P concentrations. Four replications of eight combinations of crop, application method, and Alum treatment, including controls for each crop, comprised a randomized block design. The MS with and without Alum were incorporated to a 5-cm depth of a Boonville fine sandy loam soil to provide 500 kg total P ha^{-1} before planting of forage and turfgrass in May of 2008. For topdressed treatments, 250 kg of total P ha^{-1} was applied as MS after planting and after the final harvest of each crop during 2008. Alum was mixed with MS to achieve a ratio of 0.1 kg Alum per kg of dry MS prior to application on designated treatments (Vietor et al. 2009).

Digital photography and ImageJ software were used to measure and analyze vegetation coverage of soil monthly during establishment of forage and turf crops (Schnell et al., 2009). Tifway turf was mowed after reaching a height of 7.5-cm to a 2.5-cm height and clippings were weighed, sub-sampled for analysis, and returned. Tifton 85 forage was cut to a 10-cm height, weighed, sub-sampled for analysis, and removed from plots every 28 days or as growth permitted following the initial harvest in 2008.

Soil and MS were sampled and analyzed before MS application in April of 2008 and soil sampled again in October of 2008 and 2009. Soil was sampled prior to second application of topdressed MS in October 2008. Soil was sampled from a 0- to 5-cm depth and 5- to 20-cm depth in May of 2008. Sod was cut (45-cm width, 2-cm depth) from Tifway bermudagrass using a mechanical sod harvester during October in each 2008 and 2009. Sod was sampled for analysis of soil and turfgrass by removing four 10-cm diameter cores from the harvested layer. Turfgrass was separated from soil in the sod layer at harvest. In addition, soil was sampled from 2- to 5-cm and 5- to 20-cm depths below the sod layer in October of 2008 and 2009. For Tifton 85 plots, soil was sampled from 0- to 5-cm and 5- to 20-cm depths following the final forage harvest in 2008 and 2009.

A mineralization study was conducted over a 28-day period for soil sampled from the 10 treatments three weeks after topdressing or incorporation of MS. Intact soil cores (6-cm diameter, 5-cm depth) were incubated at 25° C for 28 days in 1-L glass jars with an alkali CO_2 trap containing 10 ml 1.0 M NaOH. Traps were changed at 1, 3, 7, 14, 21 and 28 days and titrated with 1.0 M HCl to measure carbon mineralization. The portion of organic C in MS remaining was calculated for each sampling date as described previously (Kaboneka et al., 1997). Carbon (CO_2-C) evolved from control soil was subtracted from soils with MS to estimate MS carbon remaining. The mean g 100 g^{-1} of organic C remaining, as derived from the four MS treatments for Tifway or Tifton85, were used to identify and fit a model to the incubation data. A four-parameter, double exponent decay model was selected and calibrated to fast- and slow-decomposing components of MS (Berndt, (2008). The Levenberg-Marquardt algorithm was applied through SPSS 18.0

software to fit the decay model to the percent of organic C from MS remaining in amended soils of Tifway and Tifton 85 plots. Nitrogen mineralization was measured colorimetrically as the difference in soil inorganic N concentration (NO_3-N and NH_4-N) between samplings before and after the incubation over 28 days (Sims et al., 1995). The mineralization rates were used to estimate monthly application rates of fertilizer N for the field plots of turfgrass and forage bermudagrass amended with MS.

Sample Analysis

The Texas A&M AgriLife Soil, Water and Forage testing laboratory (College Station, TX) analyzed total and extractable N, P, K and C in soil and MS. Combustion procedures were used to measure concentrations of total N (TN) or organic C (SOC) in soil and grass biomass (McGeehan and Naylor, 1988). Inductively coupled plasma optical emission spectroscopy (ICP) was used to analyze concentrations of total P (TP) and minerals in nitric acid digests of grass and soil and of soil Mehlich-3 P (M3P) and Mehlich-3 extractable K (Havlin and Soltanpour, 1980). Soil concentrations of KCl-extractable NO_3-N and NH_4-N were analyzed colorimetrically (Mulvaney, 1996).

Total N, TP, TK and organic C (OC) in MS were analyzed as described for soil and grass. Water extractable P in MS was determined colorimetrically following a 1 hr extraction with deionized water (1:200 solid to solution) (Kleinman et al., 2002b; Pierzynski, 2000). In addition, a modified Hedley method was used to sequentially extracted and quantify concentrations of P forms in MS with and without Alum (Turner and Leytem, 2004). A filter bag system was used to characterizer fibrous components of

MS (Vogel et al., 1999). Fibrous components included neutral detergent fiber (NDF), acid detergent fiber (ADF), acid detergent lignin (ADL) and acid detergent lignin on an ash free basis (ADL_{om}).

Runoff Study

Three replications of four treatments comprising topdressed or incorporated MS with or without Alum, plus a control, were installed on a Boonville fine sandy loam soil in box lysimeters (44 x 34 x 15 cm). The treatments were compared in lysimeters for each Tifway turf and Tifton 85 forage. Runoff concentrations and losses of total and dissolved N, P and organic C were measured under simulated rain (10 cm hr^{-1}) during early establishment and after complete forage or turfgrass coverage of the soil surface (Birt and Persyn, 2007). The box lysimeters were mounted within support frames below the rain simulator to impose a 7% surface slope. The lysimeters were maintained under greenhouse conditions before and between the simulated rain applications.

Runoff volumes were measured and sampled at 10-min intervals over a 30-min period after runoff started. Samples were refrigerated and filtered for analysis of dissolved reactive P (DRP) within 24 hr (Pierzynski, 2000). An ICP was used to measure total dissolved P (TDP) in filtrate within 3 d after sampling (Eaton and Franson, 2005). An Elementar Rapid LiquidTOC analyzer (Hanau, Germany) was used to quantify dissolved organic carbon in filtrate of water samples.

Soil was sampled from 0- to 5-cm and 5- to 15-cm depths of lysimeters for analysis of TN, TP, and SOC after the second runoff sampling. The turfgrass or forage was clipped at the soil surface before soil samples were removed, dried, weighed, and sampled for analysis. An Elementar VarioMax (Hanau, Germany) analyzer was used for automated dry combustion and measurement of TN and organic C in soil, biomass and MS. In addition, the ICP was used to measure total P and minerals in nitric acid digests of soil, MS and biomass and in Mehlich-3 extracts of soil (Havlin and Soltanpour, 1980; Mehlich, 1984). Dissolved reactive P in water extracts (1:10 solid to solution) of soil and MS were determined colorimetrically (Pierzynski, 2000).

Nutrient Balance

Plot-scale measurements of N and P imports, exports, and soil storage during establishment, production, and harvest were used to estimate nutrient balance ha^{-1} for forage or sod produced with and without MS. Total export of TN and TP were calculated as the product of Tifton 85 dry matter and forage nutrient concentration or of dry weights of soil or turfgrass fractions and respective nutrient concentrations. The mass balances of TN and TP were calculated by subtracting TN outputs (Tifton 85 forage or Tifway sod export) from TN inputs (Fertilizer + MS). In addition, the proportion of MS sources of TN and P exported in annual sod or forage harvests were calculated for 2008 and 2009. Previous evaluations of the percentage of composted biosolid nutrients exported in sod subtracted the amount of nutrients in control soil sod from amended sod to account for antecedent soil nutrients (Schnell et al., 2009). In the current study, variation of sod soil weights between control sod and MS-amended sod would contribute to over estimates of antecedent soil nutrients. To account for antecedent soil nutrients, the sod soil weight from

MS-amended soil was divided by the sod soil weight of control soils. Antecedent soil nutrients were calculated by multiplying the ratio of MS-amended to control soil weight by the mass of nutrients found in control sod. The calculated mass of antecedent soil nutrients was subtracted from sod grown on MS-amended soil to calculate the percentage of MS-applied nutrients removed by one sod harvest.

Statistical Analysis

The General Linear Model procedure of SPSS 18.0 (Chicago, Illinois) was used for Analysis of Variance among crops, soil treatments and sampling dates. Fisher's least significant difference (LSD, $P = 0.05$) was used to separate means. Regression analysis was used to relate runoff losses and export of nutrients and organic C to concentrations and mass of nutrients and organic C in soil and applied MS.

RESULTS AND DISCUSSION

Field Study – Soil and CM Properties

Prior to imposing soil treatments and planting of sprigs, soil and MS were sampled and analyzed (C-1). The TP concentration of MS with and without Alum was used to calculate MS application rates. An average of 41.5 Mg ha^{-1} of dry MS with or without Alum was topdressed on Tifway and Tifton 85 plots in May of 2008 to achieve the target TP rate of 250 kg ha^{-1}. After sod removal or the final forage harvest in October of 2008, an additional topdressing of dry MS supplied 250 kg TP ha^{-1} (40.7 Mg ha^{-1} dry CM), with and without Alum. For treatments in which MS was incorporated to a 5-cm depth in soil, an average of 81.4 Mg ha^{-1} of dry MS was applied to achieve a TP rate of 500 kg ha^{-1} in May of 2008. For MS with Alum, Alum was dissolved in water (1:2 Alum to water ratio), applied to MS (0.1 kg of Alum per kg of dry MS), and mixed in a portable cement mixer for 5 min before application to soil.

Analysis of MS indicated that Alum addition reduced ($P = 0.05$) WEP concentration 95% (1:200 CM to water extraction ratio) compared to MS without Alum (C-1). Similar reductions in water extractable P for Alum-treated poultry litter compost were reported previously (DeLaune et al., 2006). In addition, sequential extracts of MS revealed shifts in extractable forms of P with compared to without Alum (C-2). Mixing Alum with MS reduced ($P < 0.001$) both WEP and $NaHCO_3$-extractable P. The WEP and $NaHCO_3$-extractable P fractions are considered readily soluble and could contribute to runoff loss of P (Turner and Leytem, 2004). The P fraction extracted in HCl is associated with Ca-phosphates and was not affected by Alum additions. Yet, Alum did significantly increase the concentration of P extracted from MS in NaOH, which comprises P associated with amorphous aluminum oxides and organic matter (Turner and Leytem, 2004). The shift of P concentrations from soluble fractions or forms to more stable forms indicated Alum additions could effectively reduce dissolved P in runoff for MS amended soils.

The SOC was analyzed in samples collected in May of 2008 after planting of Tifway and Tifton 85 and again after sod or final forage harvests in 2008 and 2009. Shortly after planting, concentrations of SOC were similar ($P > 0.05$) between Tifway and Tifton 85 plots and between treatments without or with Alum at the same respective MS rate and method of application (C-3). Topdressing 41.5 Mg ha^{-1} of MS increased SOC concentration 3.0-fold and incorporating 81.4 Mg ha^{-1} of MS increased SOC concentration 5.4-fold compared to control soil. Similar increases in SOC have been reported following incorporation of similar rates of composted biosolids (Wright et al., 2008).

On annual soil sampling dates after sod or final forage harvests in 2008 and 2009, a significant ($P < 0.05$) interaction between crop species and treatment occurred. In October 2008, mean SOC concentration was 49% greater in soil of Tifton 85 than soil of Tifway plots in which soil was removed with sod (C-3). Concentrations of SOC in soil remaining after harvest of sod from Tifway that was topdressed with MS in 2008 were similar to that of control soil. Similarly, sod harvest in 2009 removed MS that was incorporated in 2008 or top-dressed without Alum after sod harvest in 2008, which exposed soil similar in SOC concentration to control soil (C-3).

In contrast to soil used for sod production, topdressing MS with and without Alum on Tifton 85 (forage plots) in May and October of 2008 maintained SOC concentrations at 3- and 4-times greater than control plots (C-3). These results are consistent with a previous report demonstrating annual manure applications increased SOC concentration (Bhogal et al., 2009). In contrast, SOC declined over two seasons of Tifton 85 hay harvest after the single MS application was incorporated in soil with or without Alum (C-3). Concentration of SOC after incorporation of MS and Tifton 85 establishment was 2.8 times greater than soil topdressed with MS and 7.1 times greater than control soil. After the final forage harvest from Tifton 85 in 2009, SOC declined over two seasons of Tifton 85 production to concentrations similar to the control without MS and to top-dressed MS with Alum. A previous study indicated incorporation of dairy manure compost increased SOC initially, but SOC declined over three subsequent years of corn production (Butler et al., 2008). The SOC reductions were attributed to tillage effects on organic matter decomposition during corn production. In contrast, SOC concentration increased from the initial to final soil sampling dates for Tifton 85 top-dressed with MS and control soil in the present study. Similar to reports for turfgrass, Tifton 85 root biomass could have contributed to increases in SOC for control soil and added to organic C in top-dressed MS (Dai et al., 2009; Wright et al., 2008). The reduction in SOC over time for soil with incorporated MS indicated decomposition rates of MS were greater than SOC contributions from Tifton 85 roots.

Variation in soil TN concentration was similar to trends observed for SOC. Soil TN concentration shortly after planting was similar between Tifway and Tifton 85 bermudagrass plots and with or without Alum treatment for respective topdressed or incorporated MS treatments (C-3). On the initial sampling date, topdressing MS with or without Alum increased soil TN concentration 71%. Incorporating the larger MS rate increased soil TN 2.7 times compared to soil without MS. In October of 2008 and 2009, variation among treatments differed between Tifway harvested as sod and Tifton 85 harvested as forage. After each sod harvest, soil TN concentration below the harvest layer for Tifway was similar between topdressed MS without Alum and soil without MS. A previous report indicated that up to 47% of topdressed manure sources total Kjeldahl nitrogen (TKN) could be exported with a single sod harvest (Vietor et al., 2002). In contrast, TN concentration in soil below Tifway sod remained greater with than without incorporated MS compared on successive sampling dates, yet was reduced (P = 0.05) on the final compared to initial sampling date. Harvest and export of 2-cm of soil mixed with MS removed a portion of TN, but a portion of MS remained within the soil sampling depth on dates in October of 2008 and 2009 (Schnell et al., 2009).

Unlike soil sampled after Tifway sod harvests, MS top-dressed on Tifton 85 with or without Alum increased soil TN concentration compared to control soil after the final forage harvest during 2008 and 2009 (C-3). Soil TN concentration after the final forage harvest of Tifton 85 was 2.9 times greater with than without top-dressed MS in October of 2009. Similarly, soil concentration of TN was 90% greater with than without incorporated MS after the final forage harvest in 2009. Applications of fertilizer N (400 kg ha^{-1} yr^{-1}) were sufficient to balance TN exported in Tifton 85 forage harvests and maintain soil TN concentration over the 2-yr study.

After planting and on successive sampling dates, high antecedent soil TP concentrations precluded a relationship between variation in soil TP concentration among treatments and variation of TP amounts applied with MS. Antecedent soil M3P concentration of control soil was greater than 200 mg kg^{-1}, which exceeded regulatory limits established for Texas and typical for soils receiving repeated manure applications (Sharpley et al., 2000, Sharpley et al., 2004). However, differences in soil M3P concentration were observed between sod and forage production and among treatments with or without topdressed or incorporated MS. Similar to SOC and TN, soil M3P concentration was similar between sod and forage production shortly after planting. In contrast, topdressed or incorporated MS with or without Alum increased soil M3P concentration 68% compared to soil without MS shortly after planting (C-3). Application of MS to soil with high concentration of extractable P in excess of crop requirements could further increase risk for P loss in runoff (Kleinman et al., 2002a).

Export of P in two consecutive harvests of Tifway sod decreased mean M3P concentration 51% for soil exposed below the sod layer in October of 2009 compared to sampling shortly after planting (C-3). Although MS was topdressed at planting and after the first sod harvest, annual sod harvests removed a major portion of MS sources of P and reduced M3P in remaining soil to concentrations below 200 mg kg^{-1} and similar to soil without MS. In addition, repeated harvests of Tifton 85 biomass over two years reduced soil M3P concentration 52% for soil without MS and 36% for soil with incorporated MS. Yet, soil M3P concentration remained greater with than without incorporated MS after the final harvest of Tifton 85. In contrast, a second annual topdressings of MS with or without Alum maintained soil M3P after the final Tifton 85 harvest at concentrations comparable to shortly after planting. Whether incorporated or topdressed, P amounts applied as MS exceeded Tifton 85 uptake and removal in forage and excesses in soil could be susceptible to transport in runoff and drainage.

Field Study – C and N Mineralization

Similar to soil chemical properties, CO_2 evolution rates of soil cores did not differ between Tifway grown for sod and Tifton 85 harvested as forage. The CO_2 evolution rates were combined between crops to evaluate effects of soil treatments. After one week of incubation, CO_2 evolution reached a constant rate and significant differences were observed among soil treatments. An initial flush of CO_2 evolution during the first 7d of incubation was minimized through use of intact soil cores at water contents that existed at sampling (Franzluebbers, 1999). Topdressing MS with or without Alum increased CO_2 evolution rates to three times the rate of control soils (C-4). In addition, CO_2 evolution rate was five times greater for incorporated MS with or without Alum than the control soil (C-4). Variation of CO_2 evolution rates among treatments was attributed largely to variation of application rate of MS. An average of 12.5 Mg ha^{-1} of C was applied with topdressed MS and 24.4 Mg ha^{-1} of C was applied with incorporated MS, with or without Alum. Although CO_2 evolution rate increased in response to MS application, CO_2 evolution rates from MS-amended soils were much lower than reported previously for soils amended with a variety of compost sources (Flavel and Murphy, 2006).

As observed for CO_2 evolution rates, the proportion of applied MS organic C remaining in soil on each sampling date was similar between topdressed and incorporated MS and with or without Alum. A double exponential model was used to analyze and represent mean proportions of carbon remaining from applied MS that were calculated for each sampling date (Berndt, 2008). After 28 days of incubation, greater than 97 g remained per 100 g of C applied as MS (C-5). The model analysis indicated greater than 99 g per 100 g of C from MS declined at a slow rate. In contrast, a rapid rate of decline occurred for 0.32 g C per 100 g of C in MS (C-6). Given the decay rates, the half-life of the fast fraction ranged from 3.8 to 4.85 days while the half-life of the slow fraction ranged from 3.64 to 5.52 years (C-6). The decay rate of the fast fraction was similar to decay rates for the fast fraction reported for bermudagrass thatch (Berndt, 2008). Yet, the decay rate of the MS slow fraction was much lower than was reported previously for the slow fraction of bermudagrass thatch. Estimates from the model indicated 75.8 to 79.2 g C per 100 g of the original 12.5 Mg C ha^{-1} in topdressed MS with and without Alum will remain after one year. For incorporated MS with or without Alum, 82.4 to 83.2 g C per 100 g of the 24.4 Mg C applied ha^{-1} in MS will remain after one year.

Mineralized N was calculated as the increase in inorganic N during the 28-d incubation. Increases in mineralized N in soils amended with MS, with or without Alum, were similar (P=0.05) to control soils of Tifway grown for sod and Tifton 85 harvested as forage. Higher CO_2 evolution rates of MS-amended soil compared to control soil without corresponding differences in mineralized N suggested that immobilization of mineralized N occurred. Previous studies evaluating decomposition of pine tree substrate indicated that immobilization of inorganic N can occur along with increased microbial activity (Jackson et al., 2009).

The composition of MS could have affected N transformations in amended soil (Van Kessel et al., 2000). The C/N ration of MS was low, yet high lignin (> 190 g kg^{-1}) (ADL_{om}) indicated MS was more recalcitrant to microbial degradation than fresh organic matter sources from crop leaves and stems (C-1). Previous studies demonstrated that the proportion of lignin in residual solids from methane digestion were greater than original feedstock (Ivanova et al., 2008). Similarly, anaerobic digestion of cattle manure could have increased the fraction of lignin in MS, which decreased microbial degradation of MS and increased N immobilization in soil.

Estimated N mineralization rates did not warrant reductions in inorganic fertilizer rates to achieve forage and turfgrass sod production goals on field plots receiving topdressed or incorporated MS. In contrast, previous studies evaluating manure- and compost-amended soils indicated N mineralization rates were sufficient to warrant partial reductions in fertilizer N rates (Flavel and Murphey, 2006). For Tifway Plots, 200 kg N ha^{-1} $year^{-1}$ of ammonium sulfate (21-0-0) was applied. For Tifton 85 plots, 400 kg N ha^{-1} $year^{-1}$ of ammonium sulfate was applied.

Field Study – Biomass Production

In addition to measuring soil nutrients and mineralization of MS sources of nutrients, dry matter production of Tifton 85 was measured with or without Alum treatment for each topdressed and incorporated MS. Aerial biomass was harvested for Tifton 85 in July, August and September of 2008 and in May, June, August and November of 2009. Total dry matter production of Tifton 85 was greater ($P = 0.05$) in 2009 than in 2008 (C-7). Brink et al. (2004) observed a similar increase in Tifton 85 yield over four successive years of growth. Brink et al. (2004) hypothesized that increased root development and tillering at nodes contributed to increased biomass production of Tifton 85 over time.

Dry matter production of Tifton 85 varied among soil treatments in 2008, but was similar among treatments in 2009. Although 400 kg fertilizer N ha^{-1} was applied to all treatments, dry matter production of Tifton 85 in 2008 was greater ($P = 0.05$) for control soil than for topdressed or incorporated MS with or without Alum. Alum treatment of MS did not reduce Tifton 85 yield compared untreated MS. DeLaune et al. (2006) similarly reported Alum additions did not affect tall fescue biomass yield. As observed during incubations of cores sampled from soil with or without MS, N immobilization in soil amended with MS could have limited dry matter production of Tifton 85 compared to control soil. Kirchmann et al. (1991) similarly reported partial immobilization of N for soil amended with anaerobically-digested manure. In contrast, increasing rates of composted dairy and poultry manures with and without supplemental fertilizer N reportedly increased forage yields of bermudagrass (Helton et al., 2008; Sistani et al., 2008). The lack of increases in mineralized N and of dry matter yields with compared to without MS amendments to soil indicated MS provided less available N than fresh or composted manure.

Runoff Study – Soil Properties

Similar to the field study, Tifway and Tifton 85 bermudagrasses were established in Boonville soil with and without topdressed or incorporated MS that was applied with or without Alum in box lysimeters. Topdressing or incorporating MS with Alum reduced ($P = 0.05$) soil pH compared to control soil and MS-amended soil without Alum (C-8). Previous studies reported similar reductions of soil pH with compared to without incorporation of Alum (Vietor et al., 2010a). In addition, topdressed MS with Alum and incorporated MS with or without Alum increased concentrations of TN, TP, SOC, and Mehlich-3 extractable P (M3P) compared to control soil (C-8). Yet, Alum treatment before topdressing or incorporating MS reduced soil WEP to concentrations similar to control soil (C-8). Previous studies indicated Alum treatment before topdressing or incorporating poultry litter reduced WEP concentration of soil 88.8 to 90.1% compared to poultry litter applied without Alum (Moore Jr and Edwards, 2007). Reductions in WEP but not M3P for MS-amended soil with compared to without Alum indicated runoff loss of P could be reduced without reducing soil P availability and uptake in plants (Moore Jr and Edwards, 2007).

Runoff Study – Phosphorus Loss

Variation of mean runoff concentrations of dissolved P forms among treatments was similar between Tifway turf and Tifton 85 forage during establishment in lysimeters and data were combined for analyses. In contrast, variation among treatments differed between rain events and each event was analyzed separately. In contrast to previous reports, runoff concentration of TDP increased from the first to second rain event for incorporated MS with or without Alum and topdressed MS with Alum (Kleinman and Sharpley, 2003). Mean runoff concentration of TDP concentration was lower for incorporated than topdressed MS and lower with than without Alum treatment of MS for each rain event. Alum reductions in runoff concentration of TDP reflected Alum reductions in WEP of MS (C-1, C-8).

Mean TDP concentration comprised largely DRP in MS-amended soil without Alum during the initial rain event and variation of runoff concentration of DRP among treatments was similar to variation of TDP for both rain events. Mean DRP concentration was 78.1% of TDP concentration for the first rain event and 50.1% of TDP concentration for the second rain event (C-9). The fraction of DRP in runoff was similar to the fraction of DRP reported for leachate from soil amended with poultry litter (Kleinman et al., 2005). In addition, TDP concentration comprised a greater proportion of dissolved unreactive P

(DUP = TDP-DRP) for the second than the initial rain event (C-9). Mineralization of MS sources of P could have increased runoff concentrations of DUP compared to DRP during the second rain event.

Runoff concentrations of TDP were lower (P = 0.05) for top-dressed or incorporated MS with than without Alum for both rain events (C-9). Similar to reports for Alum treatment of biosolids, Alum treatment of MS reduced runoff TDP to concentrations comparable to control soil for both rain events (Vietor et al., 2010b). In addition, runoff concentrations of TDP were reduced 50.4 to 80.4% for incorporated compared to topdressed MS without Alum for both rain events. Yet, the concentration of TDP in runoff water from soil amended by incorporating MS without Alum was greater than soil with Alum treated MS. Kleinman et al. (2002a) observed similar reductions in runoff loss of P for incorporated compared to surface-applied manure. Incorporating MS during establishment reduces runoff concentration of TDP, but applications are confined to years in which Tifway sod or Tifton 85 forage stands are renovated. Treating MS with Alum provides an option for repeat application of MS during sod or forage regrowth for successive years without compromising water quality.

Variation in concentration of WEP of MS and application rate of topdressed and incorporated MS contributed to variation in soil WEP concentration (C-7). The rate of incorporated MS was two times greater than the rate of topdressed MS. Similarly, soil WEP concentration was 2.1 times greater for incorporated than for top-dressed MS without Alum. Regression analysis revealed a linear relationship (r^2 = 0.921) between variation of concentrations of soil WEP and runoff TDP during the first rain event for control soil and incorporated MS with or without Alum (C-10)(Pote et al., 1996). Similarly, a linear relationship (r^2 = 0.575) was observed between concentrations of soil WEP and runoff TDP during the first rain event for control soil and topdressed MS, with or without Alum (C-10). In addition, slopes of regression lines were significantly (P = 0.05) different between top-dressed and incorporated MS treatments with or without Alum (C-10).

Contrasting slopes between topdressed and incorporated MS demonstrated that relationships between soil WEP and runoff TDP concentrations and predictions of runoff losses depend on MS application method (Vietor et al., 2004). As relationships between soil WEP and runoff TDP concentrations indicated, incorporation reduces interaction of MS sources of P with rainfall and could reduce runoff loss of TDP. Previous reports indicated manure rather than soil was the principal source of dissolved P in runoff water when manure was surface applied (Kleinman et al., 2002a). In the current study, a linear relationship (r^2 = 0.916) existed between WEP in top-dressed MS without Alum and runoff concentration of TDP. The higher coefficient of determination for regression analyses comprising variation of WEP in MS rather than soil WEP in comparisons to variation of runoff concentration TDP indicated MS was the principal source of TDP. Previous studies have recommended that manure concentrations of WEP be used as indicators of potential P loss in runoff for surface applied manures (Kleinman et al., 2002b; Kleinman et al., 2007; Sharpley and Moyer, 2000).

Runoff Study – Carbon Loss

Similar to runoff loss of P, variation of mean runoff concentrations of dissolved organic C (DOC) among treatments was similar between Tifway turf and Tifton 85 forage during establishment and data were combined for analyses. In contrast, variation of runoff concentration of DOC among treatments differed between rain events and each event was analyzed separately. For the first rain event, topdressing MS without Alum increased runoff concentration of DOC 2.6 fold compared to soil without MS (C-11). In contrast, runoff concentration of DOC for the initial rain event was similar among control soil, incorporated MS with or without Alum, and top-dressed MS with Alum (C-11). Mean runoff concentration of DOC for top-dressed MS without Alum was lower for the second than initial rain event and similar ($P > 0.05$) to control soil and other MS-amended treatments.

High DOC concentration in runoff from top-dressed MS during the initial rain event and similar DOC concentrations among treatments during the second event indicated both MS and plant biomass were DOC sources (Dai et al., 2009; Wright et al., 2008). As observed for MS without Alum in the present study, Royer et al. (2007) suggested that DOC in liquid hog manure was more prone to runoff loss from top-dressed than from incorporated applications. Incorporation of MS increases potential interaction of DOC with soil, including adsorption to mineral particles and microbial degradation (Royer et al., 2007). Similar to interaction with soil minerals, interaction of DOC with Alum hydrolysis products could have reduced DOC concentration in runoff from top-dressed MS treated with Alum in the present study (Vietor et al., 2010a). Treating MS with Alum prior to land application can reduce both biologically available P and C in runoff and improve water quality.

Field Study – Nutrient Balance

Variation of soil nutrient concentrations among treatments with and without MS and Alum contributed to variation in nutrient amounts exported through Tifton 85 forage and Tifway sod harvests. Harvest of Tifway sod from MS-amended treatments in 2008 and 2009 removed ($P < 0.05$) more TN and TP than multiple forage harvests from Tifton 85 (C-12, C-13). Annual exports of TN and TP in Tifway sod were similar to those reported for sod grown with 200- to 400- kg P ha^{-1} from manure (Vietor et al., 2002). In the present study, export of TN and TP in Tifway sod was similar between the year of establishment (2008) and second harvest in 2009. In contrast, export of TN and TP in forage harvests was 2.8- and 2.3-times greater during 2009 compared to 2008 for Tifton 85. The increase in Tifton 85 export in 2009 was attributed to greater biomass production during the second growing season. Similarly, a previous report demonstrated an increase in TN and TP uptake by Tifton 85 due to greater biomass production during the fourth compared to the earlier growing seasons (Brink et al., 2004). The amount of TN (212 kg ha^{-1}) and TP (51 kg ha^{-1}) exported in Tifton 85 biomass during 2008 were similar to amounts reported by Brink et al. (2004) for the year of establishment. In addition, exports of TN and TP in Tifton 85 forage during 2008 were similar to annual N and P uptake from poultry litter applications to a bermudagrass-ryegrass forage system (Evers, 2002). The export of TN (593 kg ha^{-1}) and TP (118 kg ha^{-1}) during the second year of the current study was slightly greater than that reported by Brink et al. (2004) during a fourth growing season for Tifton 85 grown on soil amended with broiler litter.

Export of TN and TP in Tifway sod was greater for topdressed or incorporated MS with or without Alum than for control sod in both 2008 and 2009 (C-12, C-13). In contrast, TN and TP in Tifton 85 forage did not differ ($P = 0.01$) among treatments with and without top-dressed or incorporated MS for either year. Additionally, export of TN or TP in Tifway sod and Tifton 85 dry matter was similar with or without Alum treatment of MS. Harvest of Tifway sod removed 2.7 times more TN mass and 2.8 times more TP mass than three harvests of Tifton 85 forage in 2008. Greater biomass production and nutrient uptake for Tifton 85 in 2009 compared to 2008 reduced, in part, the difference in annual export of TN and TP between Tifway sod and Tifton 85 forage harvests. Export in Tifway sod was 48% greater for TN and 60% greater for TP than respective exports in Tifton 85 forage during 2009.

In addition to exports of TN and TP mass through sod, proportions exported from MS sources of TN and TP over 2 yr were calculated for Tifway sod (C-14). Over two growing seasons, two applications of top-dressed MS with and without Alum supplied TN and TP at rates comparable to TN and TP in MS incorporated at planting. The proportion of the MS source of TN exported in Tifway sod was similar among treatments during 2008. In 2009, the proportion of TN exported from MS topdressed with or without Alum was three times greater than from incorporated MS with or without Alum (C-14). Over both years, the proportion exported from TN of top-dressed MS was 1.9 times greater than from TN of incorporated MS. The percent export of MS applied TN was near 100% for

Tifway sod topdressed with Alum treated and untreated MS. Similarly, Schnell et al. (2009) reported that greater than 92% of TN applied with composted biosolids was exported with two harvest of Tifway sod. The proportion of TN in incorporated MS that was exported during the first year was consistent with the percent export reported for a single harvest of sod grown with manure (Vietor et al., 2002). Yet, the proportion of TN exported from incorporated MS decreased during the second year. Leaching and other losses of soil N forms from the sod layer could have reduced the proportion of MS sources of TN recovered in sod harvests during the second year (Schnell et al., 2009).

In contrast to TN, the proportion of MS sources of TP exported through Tifway sod did not differ ($P > 0.01$) among treatments for either year or over both years combined (C-14). The proportion of the MS source of TP exported in sod ranged from 470 to 790 g kg^{-1} was similar to a previous report for sod grown with manure sources of P (Vietor et al., 2002). The results indicate the proportion of MS sources of TP exported through sod will be comparable between topdressed and incorporated MS if sod harvests remove the soil depth in which MS are incorporated. Yet, runoff loss of soluble P forms will be less for incorporated than topdressed MS, which could improve water quality (Kleinman et al., 2002a). Conversely, topdressed MS could be treated with Alum to reduce potential transport of P in runoff while achieving export of a large proportion of MS sources of TP in sod.

The balance of TN (total applied – total export) differed between Tifway sod and Tifton 85 forage and between topdressed and incorporated MS over 2 yr. Except for incorporated MS without Alum, TN export through annual Tifway sod harvests was greater than TN in MS and fertilizer applications over 2 yr (C-15). Similarly, harvest and export of sod grown with similar rates of surface applied sludge exported more TN than was applied over 2 yr of sod harvest (Tesfamariam et al., 2009). In contrast, TN applications as inorganic fertilizer and top-dressed or incorporated MS, with or without Alum, supplied an excess of 1000 kg TN ha^{-1} above TN exports in forage harvests of Tifton 85 after 2 yr (C-15).

Net increases of soil TN for Tifton 85 compared to net reductions of soil TN for Tifway sod were reflected in greater soil TN concentrations after 2 yr of production for MS-amended treatments of Tifton 85 forage compared to Tifway sod (C-3). Greater net reductions of soil TN for Tifway sod than for Tifton 85 forage was attributed to TN removed with the layer of soil exported in annual sod harvests, which comprised N forms unavailable for root uptake. In contrast, only extractable, inorganic nutrient forms available from fertilizer N, MS, or soil were taken up and exported in Tifton 85 forage harvests. Greater net reductions of soil TN through Tifway sod exports for topdressed than incorporated MS, with or without Alum, reflected greater exports of TN and of proportions of TN in applied MS for topdressed treatments (C-12, C-13, C-14). Net increases of soil TN were similar for topdressed and incorporated MS, with or without Alum, over 2 yr of Tifton 85 forage production.

In contrast to net reductions of soil TN through Tifway sod harvests, net increases of soil TP were observed for incorporated MS with or without Alum and topdressed MS without Alum for Tifway Sod (C-16). Similar to TN balance for Tifton 85, similar net increases in soil TP occurred for all treatments amended with MS over 2 yr of Tifton 85 forage harvests. Net reductions of TP for control soils without MS were 161 kg ha^{-1} for Tifton 85 forage and 185 kg ha^{-1} for Tifway sod. Although TP rates in MS contributed to net increases in soil TP for Tifway sod and Tifton 85 forage, additional sod or forage harvests could remove more of the TP in topdressed or incorporated MS with or without Alum. For the annual rates and application methods used, gradual increases in soil TP could occur for both Tifway and Tifton 85 cropping systems. As observed for soil with high antecedent concentrations of total and extractable P, which occur on waste application fields, MS application rates will need to be managed to prevent gradual increases in soil P concentration.

Similar to previous reports, the soil layer comprised the greatest proportion of TN and TP exported in Tifway sod (Schnell et al., 2009; Tesfamariam et al., 2009; Vietor et al., 2002). From 57 to 83% of TN and 64 to 87% of TP mass were exported with the soil layer for each year. Removing a thin layer of soil (2-cm depth) enables export of nutrient forms unavailable for root uptake and recalcitrant forms of organic C applied in MS. The proportion of nutrients exported from a MS application through Tifton 85 forage harvests could increase as mineralization of MS nutrients continues over successive years and further MS applications are limited.

Although Tifway clippings were returned during mowing, sod harvests exported TN and TP from the clippings. Uptake of nutrients through Tifway roots and return of those nutrients to the soil surface through clippings could enable export of available nutrient forms from depths below the sod layer. Tifway clippings recycled 83 to 150 kg TN ha^{-1} and 16 to 28 kg TP ha^{-1} to the soil surface prior to the first sod harvest in 2008.

Clippings made up from 9 to 28% of the mass of TN and 7 to 28% of the mass of TP exported in sod in 2008.

CONCLUSION

Diversifying crop production systems to include Tifton 85 forage and Tifway sod and use of alternative MS management practices can affect soil chemical, physical and biological properties and improve environmental quality. Treating MS with Alum reduced the concentration of WEP in MS and MS-amended soils. Moreover, treating MS with Alum prior to land application reduced the concentration and loss of TDP in runoff water from MS-amended soils. In addition, treating MS with Alum did not affect crop biomass production or nutrient export in Tifway sod or Tifton 85 forage compared to MS without alum. Yet, mineralization of MS sources of TN may not be sufficient to reduce application rates of fertilizer N for Tifway and Tifton 85 production. Annual harvest of Tifway sod over two years exported greater amounts of TN and TP compared to multiple harvest of Tifton 85 forage over two years. Yet, increased biomass production of Tifton 85 for successive years of growth reduced differences between Tifway and Tifton 85 export of TN and TP. Diverse cropping systems and proper management of MS and nutrients will contribute to environmentally responsible and sustainable dairy production in the southeastern US.

CHAPTER V

CYCLING BIOCHAR AFFECTS CROP PRODUCTION AND WATER QUALITY

Increasing demand and cost of fossil fuel has resulted in the emergence of technologies for converting agricultural waste and biomass to energy. Pyrolysis, a thermo-chemical conversion process, generates a by-product known as biochar that offers an opportunity to cycle C and nutrients back to soils that produce dedicated bioenergy crops. The properties of biochar and affect on soil properties and crop production have been reported for the Terra Preta soils of the Amazon where historic burning of vegetation deposited large amounts of black carbon. The observed benefits associated with soils containing biochar in the Amazon increased interest for systems of cycling biochar from pyrolysis back to agricultural soils. Yet, additional research is needed to evaluate effects of commercially produced biochar on soil properties, biomass production and water quality.

The chemical nature of carbon compounds found in biochar depends on feedstock and pyrolysis conditions and reportedly limits microbial degradation in soil (Lehmann, 2007). Biochar sources of carbon that are recycled to soil could serve as a long-term sink for anthropogenic sources of CO_2 (Lehmann et al., 2006). In addition, systems for cycling biochar could enhance sustainability of bioenergy production through improved nutrient retention in soil and reduced environmental pollution (Anex et al., 2007; Lehmann, 2007).

Production and export of large amounts of biomass for bioenergy production can remove substantial amounts of nutrients from soil (Heggenstaller et al., 2008). In addition, repeated annual harvest of crop residues and biomass could reduce soil organic carbon and increase erosion of soil (Laird et al., 2009). Soil erosion and negative annual carbon and nutrient balances, in turn, decrease short- and long-term crop productivity, Cycling of biochar derived from pyrolysis of annual harvests of crop residues and dedicated bioenergy crops could maintain soil quality and fertility and sustain crop productivity (Laird et al., 2009).

Systems for cycling biochar back to crop production fields could offer multiple benefits in addition to nutrient cycling. Benefits reported for biochar-amended soils include increased soil CEC, nutrient retention, and biomass production. High surface area and charge density of biochar contributes to increased CEC and nutrient retention of amended soils. High surface charge densities were attributed to surface oxidation and/or adsorption of organic matter to biochar (Liang et al., 2006). In addition to increased CEC, amending soil with bio-char reportedly enhanced adsorption of inorganic P and limited transport of P in runoff and subsurface drainage (Lehmann, 2007). Novak et al. (2009) found that increasing biochar amounts increased soil nutrient content and decreased leaching losses of P. Consistent with biochar effects on CEC and nutrient retention, bio-char addition (20 % w/w) to tropical Anthrosol soils increased rice biomass production and reduced leaching loss of N and cations Lehman et al. (2003). In contrast, Zwieten et al. (2010) found that incorporation of 10 tons ha^{-1} of biochar with supplemental N, P, and K in soil increased biomass production of soybeans only. Similarly, corn (*Zea mays* L.) yield responses to biochar were smaller than expected even though peanut hull biochar rates up to 22 Mg ha^{-1} increased soil-test K, Ca, and Mg in amended soil (Gaskin et al., 2010).

OBJECTIVES

1. Evaluate sorghum biomass production and nutrient uptake in response to bio-char application rate and method and supplemental fertilizer nutrients over a 45-day period under greenhouse conditions.
2. Evaluate effects of biochar application rate and method and supplemental fertilizer nutrients on runoff loss of N, P and K under simulated rainfall during sorghum establishment.
3. Calculate mass balance of nutrients with and without biochar and supplemental fertilizer nutrients during sorghum establishment.

MATERIALS AND METHODS

Experimental Design

A complete randomized design comprised three replications of eight sorghum establishment treatments, which were randomly assigned to box lysimeters (45.5 x 33 x 15 cm) under greenhouse conditions. Soil (Booneville, fine sandy loam) was packed into box lysimeters in 5-cm depth increments to achieve a bulk density of 1.4 Mg m^{-3}. Five treatments were composed of top-dressed or incorporated (15 cm depth) biochar rates of 1.5 and 3.0 Mg ha^{-1} plus a control without biochar. Fertilizer N (30 kg N ha^{-1}) was incorporated in soil of all five treatments before planting of sorghum or topdressing of biochar. In three additional treatments, inorganic N (30 kg ha^{-1}), P (50 kg ha^{-1}), and K (100 kg ha^{-1}) fertilizers were incorporated in soil with or without 3.0 Mg ha^{-1} of biochar and before top-dressing of 3 Mg ha^{-1} of biochar. Ammonium sulfate (21-0-0), triple superphosphate (0-46-0) and potassium chloride (0-0-60) were used as fertilizer sources. The two biochar rates were selected to simulate 50 and 100% return of biochar derived from pyrolysis of expected sorghum biomass yields (15 Mg ha^{-1} DM). Biochar was collected from an auger-fed, fixed bed pyrolyzer set at 530° C. After biochar and fertilizer applications, sorghum (TAMU 08001) was seeded at a density of 80 plants m^{-2}. Biomass production (Mg ha^{-1}) was measured as aboveground dry matter (60 °C, 48 hr) at 45 days after planting. Total N was measured through automated dry combustion and inductively-coupled plasma optical emission spectroscopy (ICP) was used to quantify minerals in nitric acid digests of sorghum dry matter (Havlin and Soltanpour, 1980; McGeehan and Naylor, 1988).

Runoff Sampling and Analysis

Simulated rainfall was applied (10 cm hr^{-1}) at 21 and 34 days after planting and runoff was collected over a 30-minute period (Birt and Persyn, 2007). Runoff volumes were measured and sampled at 10-minute intervals and samples were composited for analysis. Sub-samples of the composite samples of runoff were filtered (< 0.45 μm) and dissolved reactive P (DRP) was analyzed colorimetrically using a microwell plate reader within 24 hrs of sampling (Pierzynski, 2000). The ICP was used to measure total dissolved P (TDP) and K (DK) in runoff filtrate. Inorganic N (NO_3 and NH_4) in filtrate of runoff was analyzed colorimetrically (Sims et al., 1995). In addition, concentrations of total organic C, total Kjeldahl N (TKN), total P (TP) and total K (TK) in unfiltered composite samples of runoff and drainage from each rain event were determined. An Elementar LiquiTox Analyzer was used to quantify total organic C in filtrate and in unfiltered composite samples (McGeehan and Naylor, 1988). In addition, Kjeldahl digestion procedures were used to quantify TKN in runoff. The ICP was used to analyze TP and TK in nitric acid digests of unfiltered runoff or filtrate of runoff and leachate (Eaton and Franson, 2005).

Soil and bio-char were sampled before planting to quantify total and extractable forms of N, P, K and organic C before treatments were imposed. Following harvest of biomass accumulated over 45 d, two soil cores (10-cm diameter) were separated into 0- to 5-cm and 5- to 15-cm depths for analysis of chemical and physical properties. Soil bulk density and water content were determined gravimetrically (60 °C, 48 hr). Mehlich-3 solution and deionized water were used to extract P from soil samples (Mehlich, 1984; Pierzynski, 2000). The ICP was used to analyze soil TP and TK in nitric acid digests and Mehlich-3 extractable P (Eaton and Franson, 2005). The NO_3-N in KCl extracts of soil was analyzed through cadmium reduction (Mulvaney, 1996). Concentration of DRP in water extracts of soil was measured colorimetrically within 24 hours of extraction and filtering (Pierzynski, 2000). Total N and organic C in soil and biochar were determined through automated dry combustion in an Elementar VarioMAX analyzer (Hanau, Germany). The ICP was used to measure total minerals in biochar after ashing at 900 ^{o}C and digestion in HCl.

One-way Analysis of Variance (SPSS 18.0) was used to compare soil, plant, and runoff variables among treatments. Rain events were analyzed separately. Orthogonal contrasts were used to compare treatments with and without biochar or inorganic fertilizer, biochar rates, and biochar application methods with respect to biomass production, crop nutrient uptake and runoff nutrient concentration. Regression analysis was used to relate runoff losses of P to concentrations and mass of extractable P in biochar. Mass balance of N, P and K nutrients for contrasting soil treatments were performed by subtracting total nutrient outputs (mass loss runoff + plant uptake) from total nutrient inputs (inorganic fertilizer + biochar). No leaching occurred during either rain event and was not included in the mass balance.

RESULTS AND DISCUSSION

Physical and Chemical Properties of Biochar and Soil

Total and extractable nutrient concentrations in soil and biochar were analyzed before treatments were imposed on lysimeters (D-1). Although respective application rates (1.5 and 3 Mg ha^{-1}) returned 50 and 100% of biochar derived from pyrolysis of 15 Mg ha^{-1} of sorghum biomass, percentages of N, P, and K recycled in applied biochar were low. For soil receiving 3.0 Mg ha^{-1} of biochar, 22 kg TN ha^{-1}, 7.0 kg TP ha^{-1}, and 12.4 kg TK ha^{-1} were applied. The nutrient loading rate from 3 Mg ha^{-1} of biochar in the present study were comparable to nutrient loading rates reported for application of 11 Mg ha^{-1} of pine chip biochar to Tifton soils in Georgia (Gaskin et al., 2010). In contrast, application of 11 Mg ha^{-1} of peanut hull biochar in the previous study supplied 2.6-fold greater total N and 4.7-fold greater total K compared to application of 3.0 Mg ha^{-1} of sorghum biochar in the current study. The low amounts of nutrients returned through 100% of biochar from the 15-Mg biomass yield indicated nutrient recovery in biochar was low for auger-fed, slow pyrolysis in this study. Loss of volatile forms of N and aerosols and particulate forms of P and other minerals in bio-oil and syngas could have limited mineral nutrient recovery in bio-char during pyrolysis (Agblevor et al., 1995). In contrast to nutrient recoveries in this study (D-1), Gaskin et al. (2008) reported that 27 to 90% of N, 60 to 100% of P, and 60 to 110% of K was conserved in char produced from pine chip, poultry litter and peanut hulls. Variations in pyrolyzer design, technologies and pyrolysis conditions may have contributed to variation in nutrient recovery between previous and current studies. Poor recovery of nutrients during pyrolysis increases the amount of inorganic fertilizer required to maintain productivity of sorghum biomass.

Nutrient cycling through 50 and 100% return of biochar ha^{-1} did not increase ($P > 0.05$) soil concentrations of TN, TP and TK in the 0- to 5-cm depth of soil compared to soil without biochar (D-2). In contrast, incorporation of 50 kg fertilizer P ha^{-1} increased ($P < 0.001$) soil TP concentration compared to soil without fertilizer P. In contrast, the 1.5- or 3.0-Mg rates of biochar increased ($P = 0.004$) soil organic C (SOC) concentration 24% (0- to 5-cm depth) compared to soil without biochar. Van Zwieten et al. (2010) similarly reported an 18 to 26% increase in soil organic C following incorporation of 10 Mg ha^{-1} of paper mill waste biochar within a 5-cm soil depth.

As observed for soil TP, 50 kg ha^{-1} of fertilizer P increased ($P < 0.001$) soil concentrations (0- to 5-cm depth) of Mehlich-3 extractable P 1.9-fold and WEP 3.9-fold compared to soil without fertilizer P (D-2). In contrast to soil TP, biochar increased soil concentrations of Mehlich-3 P ($P < 0.001$) and WEP ($P = 0.021$) compared to control soil without inorganic P fertilizer. Increases in soil extractable P forms due to biochar or inorganic fertilizer could increase potential P loss in runoff from production fields (Pote et al., 1996).

Compared to control soil, fertilizer K or biochar application increased ($P < 0.001$) concentrations of Mehlich-3 extractable K in 0- to 5-cm depth of soil. In addition, topdressed biochar increased ($P = 0.002$) Mehlich-3 K concentration in the 0- to 5-cm depth of soil compared to incorporated biochar (D-2). Incorporating biochar to a 15-cm depth diluted soil concentrations of Mehich-3 K compared to topdressed biochar. Similar to P, increased concentration of extractable K near the soil surface could increase potential runoff loss of K (Bertol et al., 2007). Excessive runoff loss of K following biochar application reduces the amount of K conserved in bioenergy cropping systems and could contribute to reductions in crop yields over time.

Biomass Production

Similar to variation of soil total and extractable N and P among treatments, sorghum biomass differences between control soil and soil top-dressed or mixed with biochar were small (D-3). In contrast, application of fertilizer N, P, and K with or without the 3-Mg rate of biochar increased sorghum leaf area index 58% and biomass production 86% compared to soil with or without biochar and fertilizer N. In contrast, a previous report indicated 11 Mg ha^{-1} of biochar and inorganic fertilizer increased corn stover production compared to soil receiving fertilizer only (Steiner et al., 2007). Similarly, biomass production for wheat and soybeans was greater for a combination of 10 Mg ha^{-1} biochar and inorganic fertilizer than for inorganic fertilizer only (Van Zwieten et al., 2010). Negligible biochar effects on biomass production without P and K fertilizers were consistent with small differences in total and extractable soil N, P, and K between treatments with or without biochar (D-2, D-3). Previous studies reported negligible corn yield responses to pine chip biochar application at nutrient loading rates similar to the current study (Gaskin et al., 2010).

Crop Nutrient Uptake

Variation in dry matter production was directly related to nutrient content of aerial vegetation during sorghum establishment with and without biochar (D-4). Both dry matter yield and total nutrient content of aerial vegetation were greater for sorghum fertilized with inorganic N, P, and K, with and without biochar, than treatments with biochar plus inorganic N. Soil supplemented with inorganic N, P, and K fertilizer increased N uptake 33%, P uptake 98% and K uptake 98%. Similarly, Gaskin et al. (2010) reported corn tissue uptake of N, P and K was greater for soil amended with inorganic fertilizer than with pine chip biochar. In the present study, increases in biomass and nutrient content in response to inorganic P and K indicated nutrient amounts in 3 Mg of limited sorghum growth during establishment over 45 d. Nutrient amounts returned in 3 Mg of sorghum biochar in the present study were similar to amounts in 11 Mg ha^{-1} of pine chip biochar as described previously. Poor conservation of feedstock nutrients during pyrolysis of sorghum biomass limited nutrient cycling through biochar and potential sustainability of sorghum biomass production.

Runoff

Analysis of runoff water revealed that biochar and fertilizer sources of P contributed to variation of DRP concentration in runoff. For the rain event 21 days after planting, runoff concentration of DRP was 2.6-fold greater ($P < 0.001$) for soils amended with topdressed or incorporated biochar than for soil without biochar (D-5). In contrast to the current study, previous reports indicated leaching loss of P was lower for soil amended with up to 2% biochar than for control soil (Novak et al., 2009). The reduction in leaching loss of P for the previous study was attributed, in part, to interactions of biochar with ortho-phosphate in soil. In contrast to the leaching study, biochar was a source of WEP and contributed to loss of DRP in surface runoff during sorghum establishment in box lysimeters. In addition, the DRP lost from biochar sources of P in runoff could serve as an immediate source of P for aquatic microorganisms (Sharpley et al., 1992).

Incorporation of biochar did reduce ($P < 0.001$) mean DRP concentration in runoff 78% compared to top-dressed biochar (D-5) during the first simulated rain event. Similarly, Kleinman et al. (2002a) reported a 67% decrease in runoff concentration of DRP for incorporated than for top-dressed fertilizer or manure P sources . Mean runoff concentration of DRP was lower during the second (34 d) compared to the first (21 d) simulated rain event (D-5) Reductions in DRP concentration in runoff were reported previously for successive rain events after manure application (Kleinman and Sharpley, 2003). Similar to a previous study, mean DRP concentration in runoff was 46% lower at 34 than at 21 d after seeding of all treatments (Shigaki et al., 2006). Yet, mean runoff concentration of DRP remained 62% lower for incorporated compared to topdressed biochar at 34 days after planting (D-5).

As observed for biochar, incorporated P fertilizer was a source of DRP in runoff during sorghum establishment. Applying fertilizer P increased ($P < 0.005$) runoff concentration of DRP 3-fold compared to the same treatments without fertilizer P at 21 days after planting. Runoff concentration of DRP was reduced at 34 d compared 21 d after planting, but, remained 1.7-fold greater with than without incorporation of inorganic fertilizer P. Similar to a previous report, mass loss of DRP during the initial rain event and sorghum uptake of P likely reduced runoff concentration of DRP between these successive rain events (Kleinman and Sharpley, 2003). Increased soil concentrations of Mehlich-3 P and WEP after incorporation of inorganic fertilizer P likely contributed to increased runoff concentrations of DRP (Pote et al., 1996).

Mean runoff volumes at 21 d (36.8 L m^{-2}) and 34 d (46.5 L m^{-2}) after planting were similar among treatments and variation of mass loss of P was attributed largely to variation of DRP and TP concentrations in runoff. Mass loss (kg ha^{-1}) of DRP and TP in runoff were greater ($P < 0.005$) with than without biochar, particularly if topdressed (D-6). In addition, mass loss of DRP from soil top-dressed with biochar was linearly related ($r^2 = 0.942$) to WEP mass in biochar (D-7). Similar to the current study, a linear relationship was observed previously between WEP concentration in surface-applied manure and DRP loss in runoff (Kleinman et al., 2002b; Kleinman et al., 2007). Application rates of biochar in excess of 50 tons ha^{-1} have been reported previously (Chan et al., 2008; Tryon, 1948). Although not evaluated in the current study, high application rates of biochar, especially if topdressed, could result in excessive runoff loss of P and compromise water quality.

Similar to mass loss of DRP, mean mass loss of TP in runoff was 2.9 times greater for topdressed than incorporated biochar over both rain events. For soils topdressed with biochar, mass loss of TP ranged from 1.6- to 2.6- kg ha^{-1}. Mass losses of TP ranging from 1.3- to 8.5- kg ha^{-1} were reported previously in runoff from soil amended with poultry litter at rates providing 8- to 82- kg P ha^{-1} (Tarkalson and Mikkelsen, 2004). The mass loss of TP from topdressed biochar (7 kg TP applied ha^{-1}) in the current study would be comparable to the mass loss of total P from a similar rate (8 kg TP applied ha^{-1}) of broiler litter in the previous study. Similar observations for poultry litter, incorporation of biochar in the current study reduced mass loss of total P compared to topdressed biochar (D-6) (Tarkalson and Mikkelsen, 2004).

Mean mass loss of DRP contributed nearly 100% of mass loss of total dissolved P, but only 8% of mass loss of TP in runoff over both rain events (D-6). Biochar or soil particles contained 91.6% of mean TP loss in runoff from all treatments. Similar mean mass losses of TP between treatments with and without incorporated biochar indicated P was adsorbed and transported largely with soil particles (> 0.45 μm). An increase ($P=0.001$) in mass loss of TP for top-dressed compared to incorporated biochar was attributed to P transport with biochar particles (D-6). To estimate the proportion of TP in top-dressed biochar that was lost in runoff, the difference in mass loss of TP between soils with and without biochar was divided by the TP applied as biochar. For top-dressed biochar, 14 to 20% of TP in applied biochar was lost in two rain events compared to less than 3% loss of TP from incorporated biochar (D-8). The contrasting losses of biochar sources of TP demonstrate the importance of incorporating biochar to protect water quality and to conserve P for cycling through bioenergy crop production.

In addition to affects on water quality, mass loss of P in runoff can diminish nutrient cycling and crop productivity (Tilman et al., 2002). Similarly, mass losses of N and K in runoff reduce the efficiency of nutrient cycling and are potential constraints to yield stability of biomass. Incorporating biochar reduced mass loss of TKN 47% and mass loss of TK 20% compared to topdressed biochar (D-9, D-10). For topdressed biochar, mean mass loss of TKN in runoff ranged from 5.5- to 8.3- kg ha^{-1}. Mean mass loss of TK ranged from 21.5- to 26.0- kg ha^{-1} for soil with topdressed biochar. In addition, application of biochar, especially when topdressed, increased ($P < 0.001$) mass loss of dissolved K (DK), which constituted 10% of TK in runoff (D-10). Excessive runoff loss of N, P and K nutrients can adversely affect soil nutrient balances and jeopardize sustainable crop production.

Mass Balance

Variation of TN mass balance reflected imports to soil through two rates of top-dressed or incorporated biochar with and without supplemental inorganic fertilizer. Losses comprised sorghum uptake during establishment over 45 d and transport in runoff during two simulated rain events. Initial soil N was excluded from the balance, but was a source for runoff and plant uptake and sink for excess N inputs. Application of inorganic P and K in addition to fertilizer N, with and without biochar, increased biomass production and N content of sorghum biomass. Mean TN in sorghum biomass made up 90% of mean TN removed from soil for all treatments. Greater mass loss of TN in runoff for soil with topdressed biochar contributed to reductions in net surplus of TN or deficits of TN compared to incorporated biochar (D-11). For soils receiving 3 Mg ha^{-1} of incorporated biochar with fertilizer N, a surplus of 17.6 kg ha^{-1} of TN was observed, compared to a deficit of -2.1 kg ha^{-1} of TN for topdressed biochar with fertilizer N, P and K. Similarly, biochar incorporation increased net TN to soil 113% compared to top-dressed biochar without fertilizer P and K. Incorporation of biochar may be necessary to conserve biochar sources of TN in soil.

In contrast to net TN increases observed for soil mixed with biochar, net reductions in soil TN (-9.6 to -24.1 kg ha^{-1}) occurred without biochar even though inorganic fertilizer N was applied (D-11). Net reductions of TN were greatest in soil fertilized with inorganic P and K fertilizer without biochar and were attributed to enhanced N uptake and biomass production with supplemental P and K fertilizer (Gaskin et al., 2010). Sorghum uptake of the extractable NO_3-N available in soil before fertilizer N applications was evident as net reductions in soil TN in computations of N balance without biochar. In addition, seedling uptake of the antecedent soil NO_3-N contributed to net reduction of soil TN for treatments topdressed with biochar. Net reductions of TN in soil indicated insufficient mineralization rates for biochar sources of N in soil could have limited sorghum seedling growth and accumulation of TN during establishment (Flavel and Murphy, 2006). Gaskin et al. (2010) reported that neither peanut or pine chip biochar enhanced N uptake in corn tissue.

Similar to the mass balance of TN, incorporated biochar reduced ($P < 0.05$) runoff losses and conserved TP in soil compared to topdressed biochar. Unlike TN, 3 Mg ha^{-1} of biochar was needed to achieve a net increase of TP in soil over 45-days of sorghum establishment without fertilizer P (D-12). For soils that received 50 kg ha^{-1} of fertilizer P, a net increase of soil TP from 42.7 to 50.4 kg ha^{-1} was observed after 45 days. The small net increase of soil TP (3.4 kg ha^{-1}) for the 3-Mg ha^{-1} rate of biochar without fertilizer P would likely be insufficient for full season production of sorghum. In contrast, supplementing the 3-Mg ha^{-1} rate of incorporated biochar with 50 kg ha^{-1} of fertilizer P could supply annual sorghum production requirements. In addition, runoff losses and P conservation in soil were comparable between biochar incorporated with fertilizer P and biochar topdressed without inorganic fertilizer P. Efforts to conserve P during pyrolysis and practices that conserve P when recycled back to soil will increase the sustainability of bioenergy crop production (Smit et al., 2009).

The mass balance of TK in soil revealed the importance of nutrient conservation during both pyrolysis of biomass and biochar recycling in bioenergy production systems (D-13). The 15-Mg yield of sorghum biomass will remove ~182 kg ha^{-1} of K from production fields. In the present study, the auger-fed, slow pyrolysis system conserved only 6.8% of K from biomass in biochar. Recycling up to 100% (3 Mg ha^{-1}) of biochar derived from 15 Mg biomass will supply only a fraction of the K removed ha^{-1} in biomass. Runoff loss of TK was reduced for incorporated compared to top-dressed biochar (D-10), but TK in sorghum biomass and runoff loss exceeded amounts applied in biochar and depleted TK available in soil (D-13). For topdressed or incorporated biochar without supplemental fertilizer K, TK in soil was reduced 35.9 to 53.9 kg ha^{-1} during sorghum establishment over 45 d. A large proportion of K depletion was attributed to runoff losses (20.7 kg TK ha^{-1}). The supplemental fertilizer K (100 kg ha^{-1}) offset both mean runoff loss (21.4 kg TK ha^{-1}) and TK in aerial biomass, which contributed to net TK increases in soil (6.4- to 20.7- kg TK ha^{-1}) during establishment over 45 d. Similar amounts off runoff loss of TK have been reported for grassland soil receiving cattle slurry (Misselbrook et al., 1995). Removal of K in crop biomass and excessive runoff loss of TK could result in declining crop yields over time (Benbi and Biswas, 1999).

CONCLUSION

Low recovery of biomass nutrients in pyrolysis biochar and runoff loss after biochar recycling to land necessitate biochar incorporation and supplemental fertilizer nutrients to sustain crop productivity. Cycling 100% (3 Mg ha^{-1}) of biochar derived from pyrolysis of a 15-Mg ha^{-1} yield of sorghum biomass per hectare did not increase soil nutrient concentrations or return sufficient nutrient for rapid sorghum growth and establishment. In contrast, fertilizer sources of nutrients did increase soil concentration of nutrients and contribute to increased biomass production. Yet, fertilizer and biochar sources of P contributed to increased concentration of P in runoff. Cycling biochar increased soil organic C concentration and biochar recalcitrance to microbial degradation could favor C sequestration. Improving nutrient conservation during pyrolysis, incorporation of biochar at rates derived from biomass yields ha^{-1}, and supplemental N, P, and K will contribute to environmentally responsible and sustainable production of bioenergy crops.

CHAPTER VI

SUMMARY

Recycling bioresources to agricultural land can pose both risks and benefits to sustainable agriculture. Variation in composition and quality of bioresources from urban, agricultural and bioenergy systems can affect biogeochemical cycles of amended soils and impact crop production and water quality. Through multiple field, greenhouse and laboratory studies, bioresources from urban, livestock and bioenergy systems have been characterized and management practices evaluated to identify potential risk and benefits for a variety of cropping systems. Appropriate management of recycled bioresources should enable environmentally responsible and sustainable crop production.

The first bioresource evaluated was a residual solid that remained after chemical flocculation and dewatering of sludge pumped from a secondary dairy manure lagoon through a synthetic fiber tube or Geotube®. The lagoon effluent was mixed with Alum and polymers to aid in separation of solids from liquid and to complex soluble P and reduce discharge of P in lagoon effluent (Worley et al., 2008). While the Geotube system was effective in reducing P discharge in dairy lagoon effluent, residual solids from Geotube's required disposal. Previous studies have identified turfgrass sod as a potential high value crop capable of exporting large percentages of manure applied nutrients (Vietor et al., 2002). Yet, properties of Geotube residual solids (GRS) and impact on soil physical and chemical properties and turfgrass growth were unknown.

A lysimeters study under greenhouse conditions revealed that GRS had a significant impact on soil properties, turfgrass growth and leaching loss of nutrients. Increasing rates of GRS applied to contrasting soil types decreased soil bulk density and increased soil nutrient content, especially soil NO_3-N concentration. The decay rate of GRS in amended soil was very low. Slow decay rate coupled with high concentration of NO_3-N in GRS suggests that significant decomposition, mineralization and nitrification had occurred before GRS solids were collected. Consequently, the potential for NO_3-N leaching from soil amended with GRS was high. Yet, the increase in soil concentration of NO_3-N for increasing rate of GRS applied to soils contributed to increased turfgrass biomass production. In addition, Alum and polymers injected during the dewatering process did not affect turfgrass growth but did contributed to reductions in leaching loss of dissolved reactive P (DRP). Careful selection of GRS application site, timing and rate will enable cycling of GRS that benefits both crop production and environmental quality.

In addition to evaluating a bioresource from livestock production, composted biosolids (CB) generated at the Hornsby Bend Biosolid Management plant near Austin, Texas was evaluated. Previous work has identified potential benefits to soil physical and chemical properties and plant growth for soil amended with large, volume based rates of CB (Schnell et al., 2009). Yet, excessive runoff loss of P from CB amended soils can negate benefits of CB (Schnell et al., 2010). To reduce environmental consequence associated with CB amended soils, Alum and calcium hydroxide were incorporated to reduce soluble P in CB (DeLaune et al., 2006). A study using box lysimeters with turfgrass grown under greenhouse conditions and a rainfall simulator evaluated the effects of chemical treatment of CB on water quality for contrasting soil types.

The results indicated that treating CB with Alum or calcium hydroxide was effective in reducing the water extractable P (WEP) concentration in CB and treated CB amended soil. Treating CB with Alum resulted in formation of insoluble P forms that were stable in soil with contrasting pH. Moreover, amending soil with Alum treated CB reduced the concentration of P in runoff to levels similar to both acidic and alkaline soil types without CB. In contrast, insoluble P compounds formed during treatment of CB with calcium hydroxide were not stable in soil with contrasting pH. Runoff concentration of P from soils amended with calcium hydroxide treated CB was similar to soil amended with un-amended CB. To improve soil condition and protect environmental quality, CB can be treated with Alum prior to incorporation of large volume based rates in soil.

In contrast to situations where bioresources are applied at large rates to improve soil physical and chemical properties, situations exist where large or repeated application of bioresources are necessary to dispose of bioresources in a feasible manner. In such situations, large export potential of affected cropping systems are necessary to reduce environmental impacts. A variety of cropping systems have been evaluated to estimate nutrient export potential which includes both forage and turfgrass sod production systems (Brink et al., 2004; Vietor et al., 2002). Yet, turfgrass sod and forage systems had not been compared in the same space and time along with alternative methods of bioresource management. A variety of field, greenhouse and laboratory studies was employed to evaluate various bioresource management practices and contrasting cropping system effects on soil biogeochemical processes and environmental quality. The bioresource evaluated in the study was the manure solids (MS) separated after methane production from dairy manure. Similar to manure on farms, efficient methods of bioresource nutrient export are required for sustainability.

Employing alternative MS management practices for turfgrass sod and forage systems affected soil chemical, physical and biological properties and environmental quality. Similar to effects observed for CB amended soil, treating MS with Alum reduced the WEP concentration of MS and MS-amended soil. Moreover, treating MS with Alum prior to land application did not affect crop biomass production or export of MS nutrients by turfgrass sod or forage biomass. Yet, similar to CB amended soil, treating MS with Alum did reduce runoff loss of P compared to un-treated MS and P loss was not different from control soil. Similar to GRS, methane production from dairy manure generated a bioresource with poor C quality and slow decay rates. Consequently, immobilization of N occurred in MS-amended soils and reduced forage biomass production compared to control soil during the establishment season. Greater export of total N and P was observed for turfgrass sod production over two years compared to forage production, largely due to low forage biomass production during the first year. Yet, neither cropping system removed greater total P than was applied over two years. Proper management of MS and MS application rate in conjunction with diverse cropping systems to include turfgrass sod has the potential to sustain nutrient export while protecting water quality.

In contrast to bioresources that producers apply to soil to improve soil condition or are as means of byproduct disposal, some bioresources should be recycled back to agricultural soil. Harvest and removal of entire crop biomass for bioenergy production can deplete soil C and nutrient content and reduce crop productivity (Anex et al., 2007). Harvest and pyrolysis of crop biomass offers an opportunity to recycle byproducts of bioenergy production, such as biochar, and contribute to sustainable bioenergy crop production. Yet, studies evaluating the affect of systems of crop and energy production on soil and water quality are limiting.

A study was conducted to evaluate the effect of recycling biochar with and without fertilizer nutrients at rates determined by the amount of biochar generated from pyrolysis of sorghum biomass per unit land area. The combination of poor nutrient recovery during pyrolysis and excessive runoff loss of nutrients from topdressed biochar necessitates application of fertilizer nutrients for sustainable bioenergy crop production. The need to incorporate biochar will limit tillage systems used in energy production systems. In addition, variations in pyrolyzer design can affect nutrient conservation and nutrient balances for energy crop production fields. Moreover, recycling biochar at rates designed to return produced biochar had negligible affects of soil nutrient content. Yet, modest rates of biochar return did increase soil C content and the recalcitrant nature of biochar could result in significant C sequestration over time (Lehmann, 2007). Efforts should be made to improve nutrient conservation during energy conversion processes and appropriate soil and crop management practice should be employed to achieve sustainable crop and energy production.

Bioresources are generated in a variety of environments and each have unique risks and benefits associated with land application. The source and process used generate bioresources can affect properties and composition of bioresources. In addition, the land area available for distribution and the conditions encountered for various land areas can affect cropping systems and bioresource management practices used. One-time applications of large volume based rates of bioresources require different management than bioresources that are frequently applied to land for disposal. Regardless of circumstance, management options have been identified that enable environmentally responsible and sustainable bioresource cycling.

REFERENCES

Agblevor, F.A., S. Besler, and A.E. Wiselogel. 1995. Fast pyrolysis of stored biomass feedstocks. Energy & Fuels 9:635-640.

Aggelides, S., and P. Londra. 2000. Effects of compost produced from town wastes and sewage sludge on the physical properties of a loamy and a clay soil. Bioresour. Technol. 71:253-259.

Anex, R., L. Lynd, M. Laser, A. Heggenstaller, and M. Liebman. 2007. Potential for enhanced nutrient cycling through coupling of agricultural and bioenergy systems. Crop Sci. 47:1327-1335.

Beegle, D., and L. Lanyon. 1994. Understanding the nutrient management process. Journal of Soil and Water Conservation. t: no. 2.

Benbi, D., and C. Biswas. 1999. Nutrient budgeting for phosphorus and potassium in a long-term fertilizer trial. Nutr. Cycl. Agroecosyst. 54:125-132.

Bergstrom, L., B. Bowman, and J. Sims. 2006. Definition of sustainable and unsustainable issues in nutrient management of modern agriculture. Soil Use Manag. 21:76-81.

Berka, C., H. Schreier, and K. Hall. 2001. Linking water quality with agricultural intensification in a rural watershed. Water, Air, & Soil Pollution 127:389-401.

Berndt, W. 2008. Double exponential model describes decay of hybrid bermudagrass thatch. Crop Sci. 48:2437-2446.

Bertol, I., F. Engel, A. Mafra, O. Bertol, and S. Ritter. 2007. Phosphorus, potassium and organic carbon concentrations in runoff water and sediments under different soil tillage systems during soybean growth. Soil and Tillage Research 94:142-150.

Bhogal, A., F. Nicholson, and B. Chambers. 2009. Organic carbon additions: Effects on soil bio-physical and physico-chemical properties. Eur. J. Soil Sci. 60:276-286.

Birt, L., and R. Persyn. 2007. Evaluation of an indoor nozzle-type rainfall simulator. Applied Engineering in Agriculture 23:283-287.

Brink, G., K. Sistani, and D. Rowe. 2004. Nutrient uptake of hybrid and common bermudagrass fertilized with broiler litter. Agron. J. 96:1509-1515.

Broberg, O., and G. Persson. 1988. Particulate and dissolved phosphorus forms in freshwater: Composition and analysis. Hydrobiologia 170:61-90.

Butler, T., K. Han, J. Muir, D. Weindorf, and L. Lastly. 2008. Dairy manure compost effects on corn silage production and soil properties. Agron. J. 100:1541-1545.

Chan, K., L. Van Zwieten, I. Meszaros, A. Downie, and S. Joseph. 2008. Using poultry litter biochars as soil amendments. Aust. J. Soil Res. 46:437-444.

Chardon, W., O. Oenema, P. Del Castilho, R. Vriesema, J. Japenga, and D. Blaauw. 1997. Organic phosphorus in solutions and leachates from soils treated with animal slurries. J. Environ. Qual. 26:372-378.

Chardon, W., G. Aalderink, and C. Van der Salm. 2007. Phosphorus leaching from cow manure patches on soil columns. J. Environ. Qual. 36:17-22.

Dai, X., D. Vietor, F. Hons, T. Provin, R. White, T. Boutton, and C. Munster. 2009. Effect of composted biosolids on soil organic carbon storage during establishment of transplanted sod. Hortscience 44:503-507.

DeLaune, P., P. Moore Jr, and J. Lemunyon. 2006. Effect of chemical and microbial amendment on phosphorus runoff from composted poultry litter. J. Environ. Qual. 35:1291-1296.

Donald, R., D. Anderson, and J. Stewart. 1993. Potential role of dissolved organic carbon in phosphorus transport in forested soils. Soil Sci. Soc. Am. J. 57:1611-1618.

Eaton, A., and M. Franson. 2005. Standard methods for the examination of water & wastewater. Amer Public Health Assn., Washington, D.C.

Elliott, H., G. O'Connor, P. Lu, and S. Brinton. 2002. Influence of water treatment residuals on phosphorus solubility and leaching. J. Environ. Qual. 31:1362-1369.

Evanylo, G., C. Sherony, J. Spargo, D. Starner, M. Brosius, and K. Haering. 2008. Soil and water environmental effects of fertilizer-, manure-, and compost-based fertility practices in an organic vegetable cropping system. Agriculture, Ecosystems & Environment 127:50-58.

Evers, G. 2002. Ryegrass-bermudagrass production and nutrient uptake when combining nitrogen fertilizer with broiler litter. Agron. J. 94:905-910.

Flavel, T., and D. Murphy. 2006. Carbon and nitrogen mineralization rates after application of organic amendments to soil. J. Environ. Qual. 35:183-193.

Franzluebbers, A. 1999. Potential c and n mineralization and microbial biomass from intact and increasingly disturbed soils of varying texture. Soil Biology and Biochemistry 31:1083-1090.

Gale, E., D. Sullivan, C. Cogger, A. Bary, D. Hemphill, and E. Myhre. 2006. Estimating plant-available nitrogen release from manures, composts, and specialty products. J. Environ. Qual. 35:2321-2332.

Gaskin, J., C. Steiner, K. Harris, K. Das, and B. Bibens. 2008. Effect of low-temperature pyrolysis conditions on biochar for agricultural use. Transactions of the ASABE 51:2061-2069.

Gaskin, J.W., R.A. Speir, K. Harris, K.C. Das, R.D. Lee, L.A. Morris, and D.S. Fisher. 2010. Effect of peanut hull and pine chip biochar on soil nutrients, corn nutrient status, and yield. Agron. J. 102:623-633.

Green, V., and D. Stott. 2001. Polyacrylamide: A review of the use, effectiveness, and cost of a soil erosion control amendment, 10th International Soil Conservation Organization, Purdue University and the USDA-ARS National Soil Erosion Research Laboratory. pp. 1-6.

Gregorich, E., P. Rochette, S. McGuire, B. Liang, and R. Lessard. 1998. Soluble organic carbon and carbon dioxide fluxes in maize fields receiving spring-applied manure. J. Environ. Qual. 27:209-214.

Guardado, I., O. Urrutia, and J. Garcia-Mina. 2007. Size distribution, complexing capacity, and stability of phosphate- metal- humic complexes. J. Agric. Food Chem 55:408-413.

Haney, R., A. Franzluebbers, E. Porter, F. Hons, and D. Zuberer. 2004. Soil carbon and nitrogen mineralization: Influence of drying temperature. Soil Sci. Soc. Am. J. 68:489-492.

Hansen, J., B. Cade-Menun, and D. Strawn. 2004. Phosphorus speciation in manure-amended alkaline soils. J. Environ. Qual. 33:1521-1527.

Havlin, J., and P. Soltanpour. 1980. A nitric acid plant tissue digest method for use with inductively coupled plasma spectrometry. Commun. Soil Sci. Plant Anal. 11:969-980.

Hay, F., D. Vietor, C. Munster, R. White, and T. Provin. 2007. Leaching loss of no 3-n and dissolved p from manure and fertilizer during turfgrass establishment. Plant Soil 296:1-17.

Heggenstaller, A., R. Anex, M. Liebman, D. Sundberg, and L. Gibson. 2008. Productivity and nutrient dynamics in bioenergy double-cropping systems. Agron. J. 100:1740-1748.

Helton, T., T. Butler, M. McFarland, F. Hons, S. Mukhtar, and J. Muir. 2008. Effects of dairy manure compost and supplemental inorganic fertilizer on coastal bermudagrass. Agron. J. 100:924-930.

Hons, F., R. Moresco, R. Wiedenfeld, and J. Cothren. 1986. Applied nitrogen and phosphorus effects on yield and nutrient uptake by high-energy sorghum produced for grain and biomass. Agron. J. 78:1069-1078.

Huang, X., and M. Shenker. 2004. Water-soluble and solid-state speciation of phosphorus in stabilized sewage sludge. J. Environ. Qual. 33:1895-1903.

Ivanova, L., D. Richards, and D. Smallman. 2008. Assessment of the anaerobic biodegradation potential of msw. Waste and Resource Management 161:167-180.

Jackson, B., R. Wright, and M. Alley. 2009. Comparison of fertilizer nitrogen availability, nitrogen immobilization, substrate carbon dioxide efflux, and nutrient leaching in peat-lite, pine bark, and pine tree substrates. Hortscience 44:781-790.

Jiao, Y., J. Whalen, and W. Hendershot. 2007. Phosphate sorption and release in a sandy-loam soil as influenced by fertilizer sources. Soil Sci. Soc. Am. J. 71:118-124.

Johnson, G., J. Davis, Y. Qian, and K. Doesken. 2006. Topdressing turf with composted manure improves soil quality and protects water quality. Soil Sci. Soc. Am. J. 70:2114-2121.

Johnson, J., D. Reicosky, B. Sharratt, M. Lindstrom, W. Voorhees, and L. Carpenter-Boggs. 2004. Characterization of soil amended with the by-product of corn stover fermentation. Soil Sci. Soc. Am. J. 68:139-147.

Kaboneka, S., W.E. Sabbe, and A. Mauromoustakos. 1997. Carbon decomposition kinetics and nitrogen mineralization from corn, soybean, and wheat residues. Commun. Soil Sci. Plant Anal. 28:1359 - 1373.

Ketterings, Q., J. Cherney, G. Godwin, T. Kilcer, P. Barney, and S. Beer. 2007. Nitrogen management of brown midrib sorghum x sudangrass in the northeastern USA. Agron. J. 99:1345-1351.

Kirchmann, H. 1991. Carbonic and nitrogen mineralization of fresh, aerobic and anaerobic animal manures during incubation with soil. Swedish Journal of Agricultural Research (Sweden).

Kleinman, P., A. Sharpley, B. Moyer, and G. Elwinger. 2002a. Effect of mineral and manure phosphorus sources on runoff phosphorus. J. Environ. Qual. 31:2026-2033.

Kleinman, P., A. Sharpley, A. Wolf, D. Beegle, and P. Moore Jr. 2002b. Measuring water-extractable phosphorus in manure as an indicator of phosphorus in runoff. Soil Sci. Soc. Am. J. 66:2009-2015.

Kleinman, P., and A. Sharpley. 2003. Effect of broadcast manure on runoff phosphorus concentrations over successive rainfall events. J. Environ. Qual. 32:1072-1081.

Kleinman, P., M. Srinivasan, A. Sharpley, and W. Gburek. 2005. Phosphorus leaching through intact soil columns before and after poultry manure application. Soil Sci. 170:153-166.

Kleinman, P., D. Sullivan, A. Wolf, R. Brandt, Z. Dou, H. Elliott, J. Kovar, A. Leytem, R. Maguire, and P. Moore. 2007. Selection of a water-extractable phosphorus test for manures and biosolids as an indicator of runoff loss potential. J. Environ. Qual. 36:1357-1367.

Laird, D., R. Brown, J. Amonette, and J. Lehmann. 2009. Review of the pyrolysis platform for coproducing bio-oil and biochar. Biofuels, Bioproducts and Biorefining 3:547-562.

Lehmann, J., J. Pereira da Silva, C. Steiner, T. Nehls, W. Zech, and B. Glaser. 2003. Nutrient availability and leaching in an archaeological anthrosol and a ferralsol of the central amazon basin: Fertilizer, manure and charcoal amendments. Plant Soil 249:343-357.

Lehmann, J., J. Gaunt, and M. Rondon. 2006. Bio-char sequestration in terrestrial ecosystems: A review. Mitigation and Adaptation Strategies for Global Change 11:395-419.

Lehmann, J. 2007. Bio-energy in the black. Frontiers in Ecology and the Environment 5:381-387.

Leytem, A., J. Sims, and F. Coale. 2004. Determination of phosphorus source coefficients for organic phosphorus sources: Laboratory studies. J. Environ. Qual. 33:380-388.

Liang, B., J. Lehmann, D. Solomon, J. Kinyangi, J. Grossman, B. O'Neill, J.O. Skjemstad, J. Thies, F.J. Luizao, J. Petersen, and E.G. Neves. 2006. Black carbon increases cation exchange capacity in soils. Soil Sci. Soc. Am. J. 70:1719-1730.

Loschinkohl, C., and M. Boehm. 2001. Composted biosolids incorporation improves turfgrass establishment on disturbed urban soil and reduces leaf rust severity. Hortscience 36:790-794.

Maguire, R., and J. Sims. 2002. Soil testing to predict phosphorus leaching. J. Environ. Qual. 31:1601-1609.

Malecki-Brown, L., J. White, and K. Reddy. 2007. Soil biogeochemical characteristics influenced by alum application in a municipal wastewater treatment wetland. J. Environ. Qual. 36:1904-1913.

McCoy, E. 1998. Sand and organic amendment influences on soil physical properties related to turf establishment. Agron. J. 90:411-419.

McDowell, R., and A. Sharpley. 2001. Soil phosphorus fractions in solution: Influence of fertiliser and manure, filtration and method of determination. Chemosphere 45:737-748.

McGeehan, S., and D. Naylor. 1988. Automated instrumental analysis of carbon and nitrogen in plant and soil samples. Commun. Soil Sci. Plant Anal. 19:493-505.

McMichael, B., and R. Lascano. 2003. Laboratory evaluation of a commercial dielectric soil water sensor. Vadose Zone Journal 2:650-658.

Mehlich, A. 1984. Mehlich 3 soil test extractant: A modification of mehlich 2 extractant. Commun. Soil Sci. Plant Anal. 15:1409-1416.

Miller, J., E. Olson, D. Chanasyk, B. Beasley, F. Larney, and B. Olson. 2006. Phosphorus and nitrogen in rainfall simulation runoff after fresh and composted beef cattle manure application. J. Environ. Qual. 35:1279-1290.

Misselbrook, T., B. Pain, A. Stone, and D. Scholefield. 1995. Nutrient run-off following application of livestock wastes to grassland. Environ. Pollut. 88:51-56.

Moore Jr, P., T. Daniel, and D. Edwards. 2000. Reducing phosphorus runoff and inhibiting ammonia loss from poultry manure with aluminum sulfate. J. Environ. Qual 29:37-49.

Moore Jr, P., and D. Edwards. 2007. Long-term effects of poultry litter, alum-treated litter, and ammonium nitrate on phosphorus availability in soils. J. Environ. Qual. 36:163-174.

Mukhtar, S., L. Lazenby, and S. Rahman. 2007. Evaluation of a synthetic tube dewatering system for animal waste pollution control. Applied Engineering in Agriculture 23:669-675.

Mulvaney, C. 1996. Nitrogen-inorganic forms, Methods of Soil Analysis. Part 3 - chemical methods, Soil Science Society of America, Urbana, IL USA. pp. 1123-1184.

Munster, C., J. Hanzlik, D. Vietor, R. White, and A. McFarland. 2004. Assessment of manure phosphorus export through turfgrass sod production in erath county, Texas. J. Environ. Manag. 73:111-116.

Novak, J.M., W.J. Busscher, D.L. Laird, M. Ahmedna, D.W. Watts, and M.A.S. Niandou. 2009. Impact of biochar amendment on fertility of a southeastern coastal plain soil. Soil Sci. 174:105-112.

O'Connor, G., H. Elliott, N. Basta, R. Bastian, G. Pierzynski, R. Sims, and J. Smith Jr. 2005. Sustainable land application: An overview. J. Environ. Qual. 34:7-17.

Oehl, F., E. Frossard, A. Fliessbach, D. Dubois, and A. Oberson. 2004. Basal organic phosphorus mineralization in soils under different farming systems. Soil Biology and Biochemistry 36:667-675.

Penn, C., and R. Bryant. 2008. Phosphorus solubility in response to acidification of dairy manure amended soils. Soil Sci. Soc. Am. J. 72:238-243.

Pesta, G. 2006. Anaerobic digestion of organic residues and wastes. Utilization of By-Products and Treatment of Waste in the Food Industry 1:53-72.

Pierzynski, G. 2000. Methods of phosphorus analysis for soils, sediments, residuals, and waters. Southern Cooperative Series Bulletin 396:1-110.

Pote, D.H., T.C. Daniel, P.A. Moore, Jr., D.J. Nichols, A.N. Sharpley, and D.R. Edwards. 1996. Relating extractable soil phosphorus to phosphorus losses in runoff. Soil Sci. Soc. Am. J. 60:855-859.

Royer, I., D. Angers, M. Chantigny, R. Simard, and D. Cluis. 2007. Dissolved organic carbon in runoff and tile-drain water under corn and forage fertilized with hog manure. J. Environ. Qual. 36:855-863.

Sanderson, M., and R. Jones. 1997. Forage yields, nutrient uptake, soil chemical changes, and nitrogen volatilization from bermudagrass treated with dairy manure. J. Prod. Agric. 10:266-271.

Schnell, R.W., D.M. Vietor, R.H. White, T.L. Provin, and C.L. Munster. 2009. Effects of composted biosolids and nitrogen on turfgrass establishment, sod properties, and nutrient export at harvest. Hortscience 44:1746-1750.

Schnell, R.W., D.M. Vietor, C.L. Munster, R.H. White, and T.L. Provin. 2010. Effect of turfgrass establishment practices and composted biosolids on water quality. J. Environ. Qual. 39:697-705.

Sharpley, A., S. Smith, O. Jones, W. Berg, and G. Coleman. 1992. The transport of bioavailable phosphorus in agricultural runoff. J. Environ. Qual. 21:30-35.

Sharpley, A., and B. Moyer. 2000. Phosphorus forms in manure and compost and their release during simulated rainfall. J. Environ. Qual. 29:1462-1469.

Shigaki, F., A. Sharpley, and L. Prochnow. 2006. Source-related transport of phosphorus in surface runoff. J. Environ. Qual. 35:2229-2235.

Sims, G., T. Ellsworth, and R. Mulvaney. 1995. Microscale determination of inorganic nitrogen in water and soil extracts. Commun. Soil Sci. Plant Anal. 26:303-316.

Sistani, K., A. Adeli, H. Tewolde, and G. Brink. 2008. Broiler chicken litter application timing effect on coastal bermudagrass in southeastern us. Nutr. Cycl. Agroecosyst. 81:49-57.

Smit, A., P. Bindraban, J. Schrˆder, J. Conijn, and H. van der Meer. 2009. Phosphorus in agriculture: Global resources, trends and developments. Plant Research International. 282: 1-36.

Steiner, C., W. Teixeira, J. Lehmann, T. Nehls, J. de MacÍdo, W. Blum, and W. Zech. 2007. Long term effects of manure, charcoal and mineral fertilization on crop production and fertility on a highly weathered central amazonian upland soil. Plant Soil 291:275-290.

Tarkalson, D., and R. Mikkelsen. 2004. Runoff phosphorus losses as related to phosphorus source, application method, and application rate on a piedmont soil. J. Environ. Qual. 33:1424-1430.

Tesfamariam, E., J. Annandale, J. Steyn, and R. Stirzaker. 2009. Exporting large volumes of municipal sewage sludge through turfgrass sod production. J. Environ. Qual. 38:1320-1328.

Tilman, D., K. Cassman, P. Matson, R. Naylor, and S. Polasky. 2002. Agricultural sustainability and intensive production practices. Nature 418:671-677.

Tisdale, S., W. Nelson, J. Beaton, and J. Havlin. 1993. Micronutrients and other beneficial elements in soils and fertilizers. Soil Fertility and Fertilizers 1:1-624.

Tognetti, C., M. Mazzarino, and F. Laos. 2008. Compost of municipal organic waste: Effects of different management practices on degradability and nutrient release capacity. Soil Biology and Biochemistry 40:2290-2296.

Toor, G., L. Condron, H. Di, K. Cameron, and B. Cade-Menun. 2003. Characterization of organic phosphorus in leachate from a grassland soil. Soil Biology and Biochemistry 35:1317-1323.

Torbert, H., K. King, and R. Harmel. 2005. Impact of soil amendments on reducing phosphorus losses from runoff in sod. J. Environ. Qual. 34:1415-1421.

Tryon, E. 1948. Effect of charcoal on certain physical, chemical, and biological properties of forest soils. Ecol. Monogr. 18:81-115.

Turner, B., M. Kay, and D. Westermann. 2004. Colloidal phosphorus in surface runoff and water extracts from semiarid soils of the western united states. J. Environ. Qual. 33:1464-1472.

Turner, B., and A. Leytem. 2004. Phosphorus compounds in sequential extracts of animal manures: Chemical speciation and a novel fractionation procedure. Environ. Sci. Technol 38:6101-6108.

Van Kessel, J., J. Reeves III, and J. Meisinger. 2000. Nitrogen and carbon mineralization of potential manure components. J. Environ. Qual. 29:1669-1677.

Van Zwieten, L., S. Kimber, S. Morris, K. Chan, A. Downie, J. Rust, S. Joseph, and A. Cowie. 2010. Effects of biochar from slow pyrolysis of papermill waste on agronomic performance and soil fertility. Plant Soil 327:235-246.

Vietor, D., T. Provin, R. White, and C. Munster. 2004. Runoff losses of phosphorus and nitrogen imported in sod or composted manure for turf establishment. J. Environ. Qual. 33:358-366.

Vietor, D.M., E.N. Griffith, R.H. White, T.L. Provin, J.P. Muir, and J.C. Read. 2002. Export of manure phosphorus and nitrogen in turfgrass sod. J. Environ. Qual. 31:1731-1738.

Vietor, D.M., R.W. Schnell, C.L. Munster, T.L. Provin, and R.H. White. 2010a. Biosolid and alum effects on runoff losses during turfgrass establishment. Bioresour. Technol. 101:3246-3252.

Vietor, D.M., R.W. Schnell, T.L. Provin, R.H. White, and C.L. Munster. 2010b. Effect of alum treatments on turfgrass coverage and runoff losses during establishment. Hortscience 45:119-124.

Vogel, K., J. Pedersen, S. Masterson, and J. Toy. 1999. Evaluation of a filter bag system for ndf, adf, and ivdmd forage analysis. Crop Sci. 39:276-279.

Wang, F., D. Couillard, J. Auclair, and P. Campbell. 1998. Effects of alum-treated waste water sludge on barley growth. Water, Air, & Soil Pollution 108:33-49.

Whalen, J. 2002. Calcium co-amendments modify extractable orthophosphate levels in fresh and composted cattle manure. Water, Air, & Soil Pollution 141:105-124.

Woodard, K., L. Sollenberger, L. Sweat, D. Graetz, S. Rymph, and Y. Joo. 2007. Five year-round forage systems in a dairy effluent sprayfield: Phosphorus removal. J. Environ. Qual. 36:175-183.

Worley, J., T. Bass, and P. Vendrell. 2008. Use of geotextile tubes with chemical amendments to dewater dairy lagoon solids. Bioresour. Technol. 99:4451-4459.

Wright, A., T. Provin, F. Hons, D. Zuberer, and R. White. 2008. Compost impacts on dissolved organic carbon and available nitrogen and phosphorus in turfgrass soil. Waste Manag. 28:1057-1063.

APPENDIX A

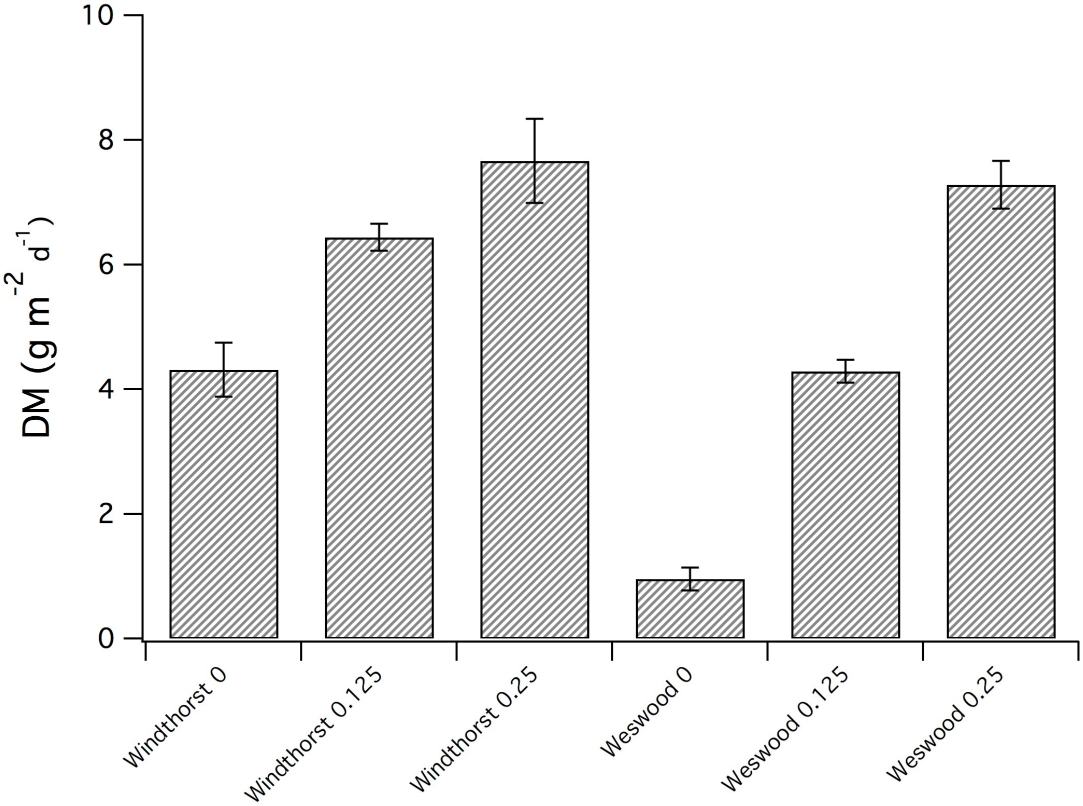
A-1. Total Organic Carbon (TOC), Total Nitrogen (TN), Total Phosphorus (TP), Mehlich-3 P (M3P), Water Extractable P (WEP) Mehlich-3 Na (M3Na) and extractable NO_3-N concentration of soil and Geotube® residual solids (GRS) before mixing and packing into columns.

	pH	TOC	TN	TP	M3P	WEP	M3Na	NO_3-N
		---------g kg^{-1}-----		-----------------------mg kg^{-1}-------------------				
Windthorst	6.4	3.0	0.5	211	12.5	4.1	138	45.8
Weswood	8.3	12.1	0.4	285	16.9	4.6	132	1.9
GRS	5.7	221	20.7	1586	-	18.6		740

A-2. Mean soil pH, bulk density, water content, total organic carbon (SOC), N (TN), P (TP) Mehlich-3 Na (M3Na) concentration (0-10 cm depth) 90 d after incorporation of increasing rates of Geotube® residual solids (GRS). The standard error of the mean is given in parenthesis below mean.

Soil	GRS rate	pH	Density	Soil water	SOC	TN	TP	M3Na
	$m^3\ m^{-3}$		$g\ cm^{-3}$	$kg\ kg^{-1}$	$g\ kg^{-1}$	----------------mg kg^{-1}	------------------	
Windthorst	0	8.6	1.32	0.22	3.3	413	43	529
		(0.05)	(0.11)	(0.02)	(0.1)	(13)	(2.9)	(63)
Windthorst	0.125	8.3	1.26	0.21	11.5	1213	239	488
		(0.06)	(0.09)	(0.02)	(0.5)	(48)	(8.9)	(32)
Windthorst	0.25	8.1	1.14	0.12	22.3	2050	594	524
		(0.06)	(0.09)	(0.02)	(1.0)	(82)	(108)	(30)
Weswood	0	8.7	1.31	0.31	4.3	475	272	609
		(0.09)	(0.09)	(0.02)	(0.2)	(31)	(10.6)	(79)
Weswood	0.125	8.6	1.20	0.28	11.5	1150	432	713
		(0.07)	(0.08)	(0.03)	(0.2)	(19)	(20.6)	(79)
Weswood	0.25	8.3	1.11	0.26	21.3	1913	683	627
		(0.11)	(0.07)	(0.02)	(0.6)	(44)	(25.9)	(52)
Soil		**	ns	****	ns	ns	****	**
GRS rate		****	*	***	****	****	****	ns
Soil*GRS rate		ns	ns	*	ns	ns	*	ns

* = P < 0.05; ** = P < 0.01; *** = P < 0.005; **** = P < 0.001

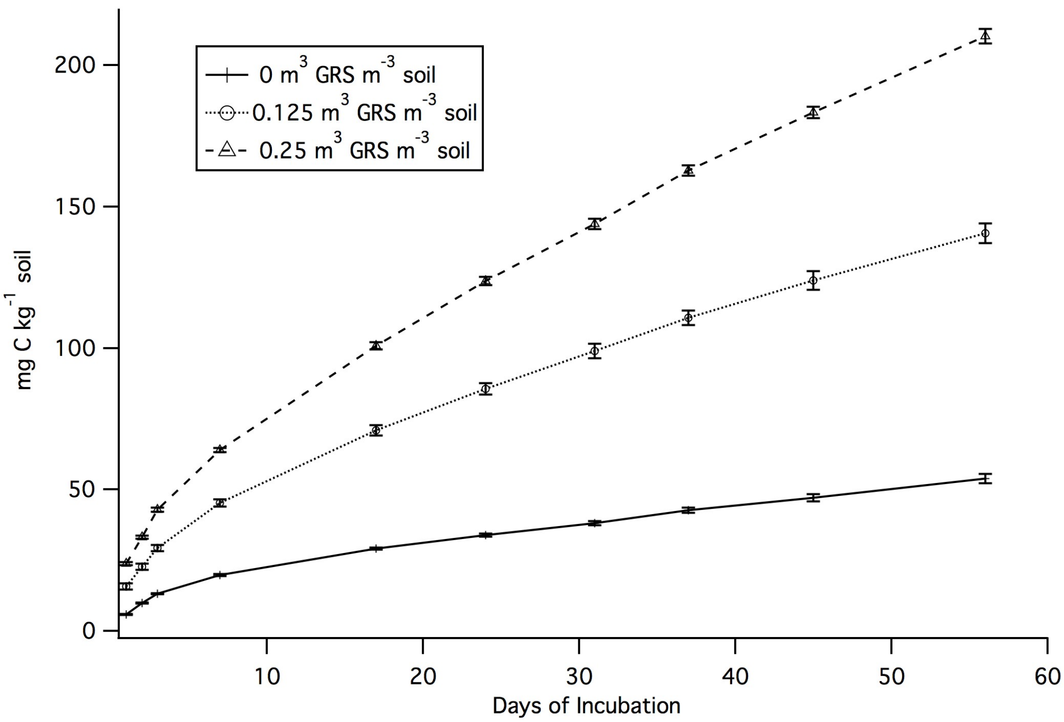


A-3. Average daily dry matter (DM) production for turfgrass grown in Windthorst and Weswood soils amended with increasing rates of Geotube® residual solids (GRS). Error bars indicate the standard error of the mean.

A-4. Mean concentrations of Mehlich-3 P (M3P), water extractable P (WEP), NO_3-N and NH_4N within 0- to 10-cm depth 90 d after increasing Geotube® residual solids (GRS) rates were mixed with two soil types in column lysimeters. The standard error of the mean is given in parenthesis below mean.

Soil	GRS rate	M3P	WEP	NO_3-N	NH_4-N
	$m^3\ m^{-3}$	-------------------------------	mg kg^{-1}	-----------------------------	
Windthorst	0	16 (1.4)	1.9 (0.3)	17.9 (4.5)	12 (2.5)
Windthorst	0.125	95 (3.6)	13.7 (1.6)	23.5 (5.7)	21 (5.0)
Windthorst	0.25	173 (5.8)	19.7 (3.1)	26.0 (5.7)	35 (9.3)
Weswood	0	21 (1.3)	2.5 (0.3)	12.7 (3.4)	9.9 (2.3)
Weswood	0.125	79 (3.3)	15.2 (1.0)	23.3 (4.3)	19 (4.9)
Weswood	0.25	151 (8.1)	16.4 (1.5)	26.4 (4.5)	33 (8.2)
Soil		ns	ns	ns	ns
GRS rate		****	****	ns	****
Soil*GRS rate		ns	ns	ns	ns

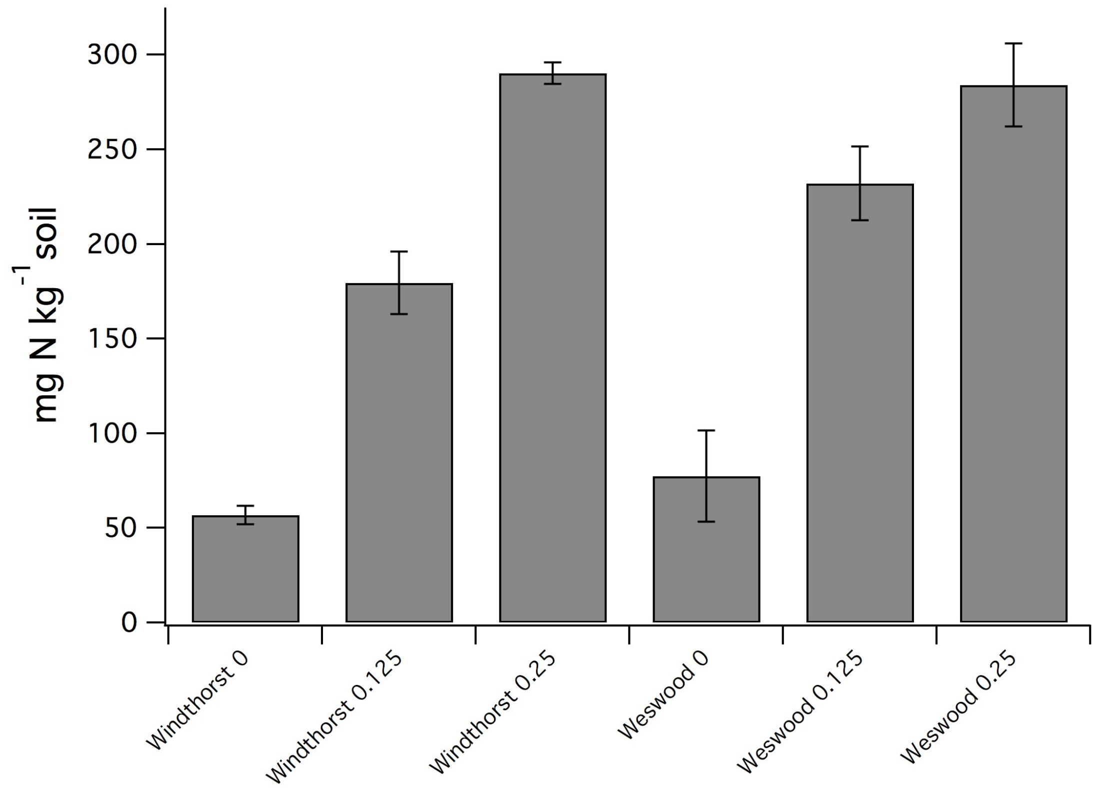
* = P < 0.05; ** = P < 0.01; *** = P < 0.005; **** = P < 0.001



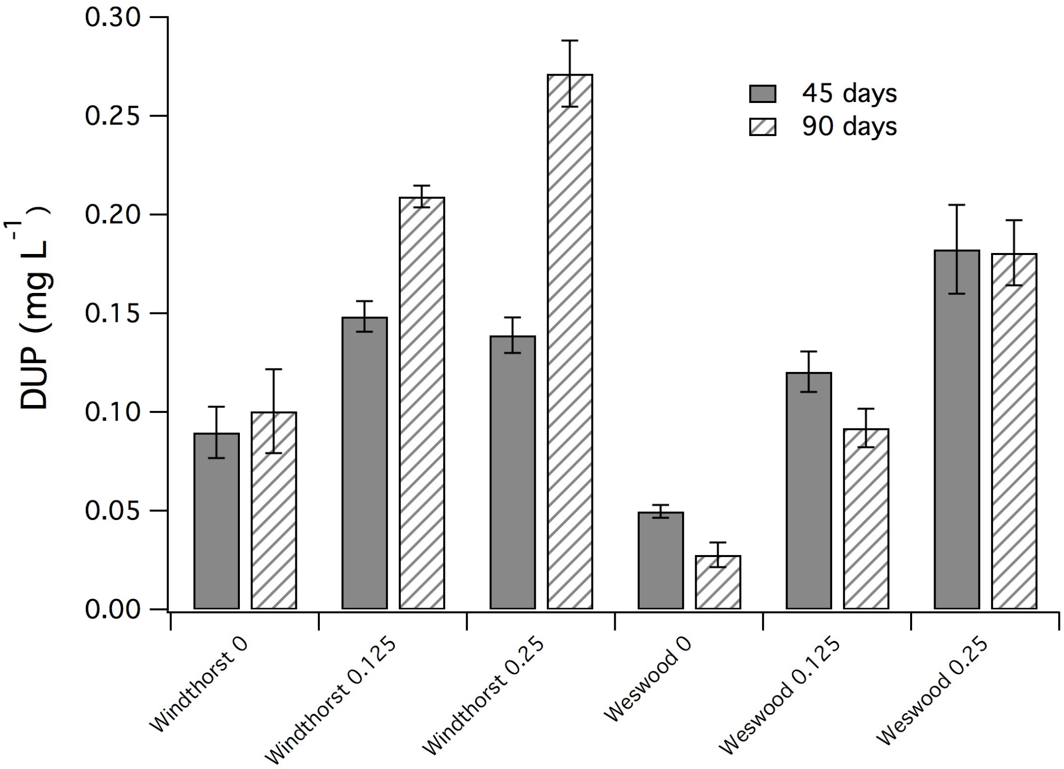
A-5. Mean value of cumulative carbon evolved from Windthorst and Weswood soils with increasing rates of Geotube® residual solids (GRS) over a 56-d incubation. Error bars indicate the standard error of the mean.

A-6. Results of modeling percent C remaining over a 56-d incubation for Windthorst and Weswood soils with increasing rates of Geotube® residual solids (GRS) using a four-parameter double exponent model ($y = F*\exp^{(-kf * days)} + S*\exp^{(-ks * days)}$) . The intercepts for the fast (F) and slow (S) pool and decay rates for the respective pools are given. The rate constant (k) is in days.

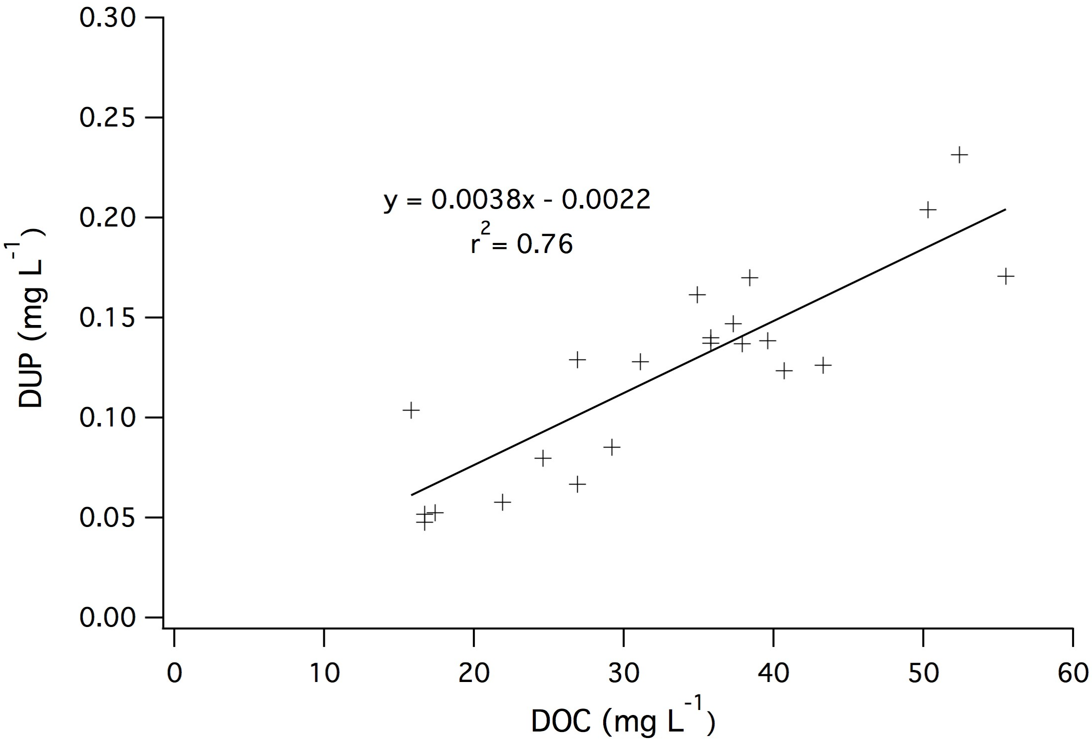
		Fast Pool		Slow Pool	
Treatment	R^2	Intercept (F)	k_f	Intercept (S)	k_s
Windthorst 0.125	0.997	0.49	0.067	99.43	0.00013
Windthorst 0.25	0.999	0.28	0.089	99.62	0.00017
Weswood 0.125	0.992	0.38	0.093	99.43	0.00022
Weswood 0.25	0.987	0.28	0.103	99.57	0.00019



A-7. Total nitrogen (NH_4-N + NO_3-N) mineralized for Windthorst and Weswood soils with and without Geotube® residual solids (GRS) over a 56-day incubation. Error bars indicate the standard error of the mean.



A-8. Concentration of dissolved un-reactive P (DUP) in leachate collected at 45 and 90 days for Windthorst and Weswood soils amended with increasing rates of Geotube® residual solids (GRS). Error bars indicate the standard error of the mean.



A-9. Relationship between concentration of DOC and DUP in leachate at 45 days after planting turfgrass in Windthorst and Weswood soils amended with increasing rates of Geotube® residual solids (GRS).

A-10. Concentration of NO_3-N in leachate at 45 and 90 days after planting of turfgrass in Windthorst and Weswood soils amended with increasing rates of Geotube® residual solids (GRS).

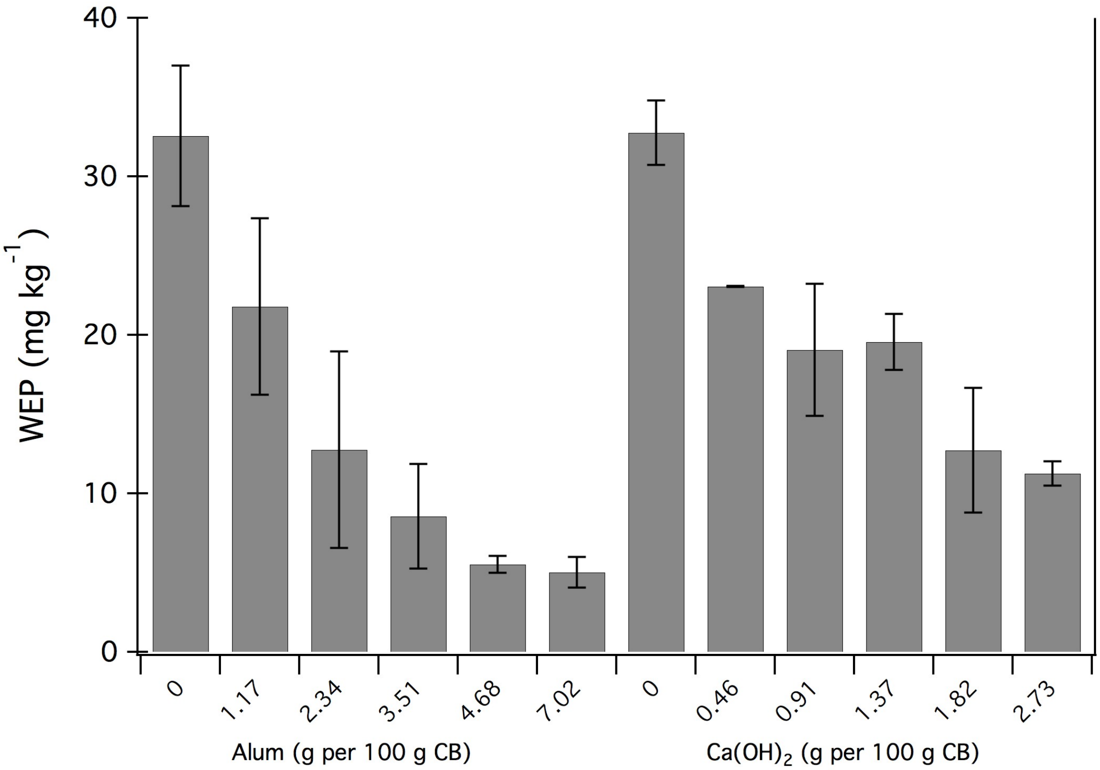
Soil	GRS rate	45 d	90 d
	$m^3\ m^{-3}$	---------------------mg L^{-1}	---------------
Windthorst	0	86	0.3
Windthorst	0.125	161	1.2
Windthorst	0.25	232	64.5
Weswood	0	0.02	0.1
Weswood	0.125	78	0.1
Weswood	0.25	73	1.0
Soil		**	*
GRS rate		*	*
Soil*GRS rate		ns	*

* = P < 0.05; ** = P < 0.01; *** = P < 0.005; **** = P < 0.001

APPENDIX B

B-1. Analysis of chemical properties of composted biosolids (CB) and Burleson and Branyon soil type before soil treatments were imposed.

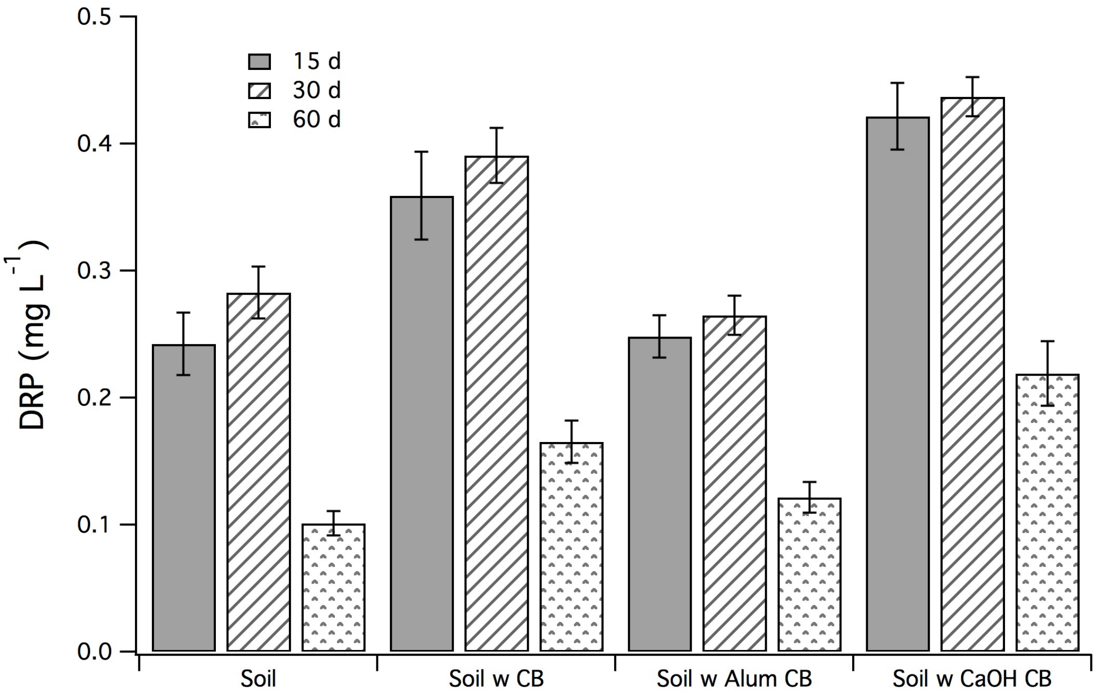
	pH	Total N	Total P	M3P	WEP	Organic C
		----------- g kg^{-1}----------		----------- mg kg^{-1}------- ---		g kg^{-1}
Burleson	6.2	1.71	0.47	97	8.16	22.8
Branyon	7.8	0.98	0.64	87	7.52	11.6
CB (std error)		21.0 (0.68)	9.7 (0.38)		34.5 (2.5)	229 (8.4)



B-2. Concentration of water extractable P (WEP) for increasing concentrations of Alum or $Ca(OH)_2$ in CB. Error bars indicate the standard error of the mean.

B-3. The soil bulk density and soil concentrations of organic carbon (SOC), water extractable P (WEP), Mehlich-3 P (M3P) and total P (TP). Soil types were analyzed separately for soil pH. The means representing main effects (soil type and treatments) were compared for remaining soil properties. Means followed by the same letter within columns indicate no significant difference (LSD, P = 0.05).

	pH	Bulk Density	SOC	WEP	M3P	TP
		g cm^{-3}	g kg^{-1}	------------------mg kg^{-1}	-----------------	
Branyon soil	7.9 A	1.23	11.1	7.52	93	625
Branyon + CB	7.6 B	1.08	27.2	13.97	229	1105
Branyon,+ CB w/ Alum	7.5 B	0.99	27.8	4.21	231	1080
Branyon + CB w/ $Ca(OH)_2$	7.9 A	1.01	26.1	9.68	237	1060
Burleson soil	6.4 c	1.34	17.4	8.16	91	438
Burleson + CB	7.0 b	1.10	35.1	19.98	325	1046
Burleson + CB w/ Alum	6.8 b	1.10	32.4	6.04	295	907
Burleson + CB w/ $Ca(OH)_2$	7.2 a	1.10	30.2	13.93	279	916
<u>Soil Type</u>						
Branyon		1.08 B	23.1 B	8.85 B	197 B	967 A
Burleson		1.15 A	28.8 A	12.03 A	247 A	827 B
<u>Treatment</u>						
No CB		1.29 a	14.2 c	7.84 c	92 b	531 c
CB		1.03 b	31.2 a	16.97 a	277 a	1076 a
CB w/ Alum		1.05 b	30.1 ab	5.12 c	263 a	993 b
CB w/ $Ca(OH)_2$		1.09 b	28.2 b	11.81 b	258 a	988 b
	$P > F$	$P > F$	$P > F$	$P > F$	$P > F$	$P > F$
Soil Type	0.000	0.001	0.000	0.000	0.000	0.000
Treatment	0.000	0.000	0.000	0.008	0.000	0.000
Soil*Treatment	0.000	0.417	0.476	0.311	0.022	0.134

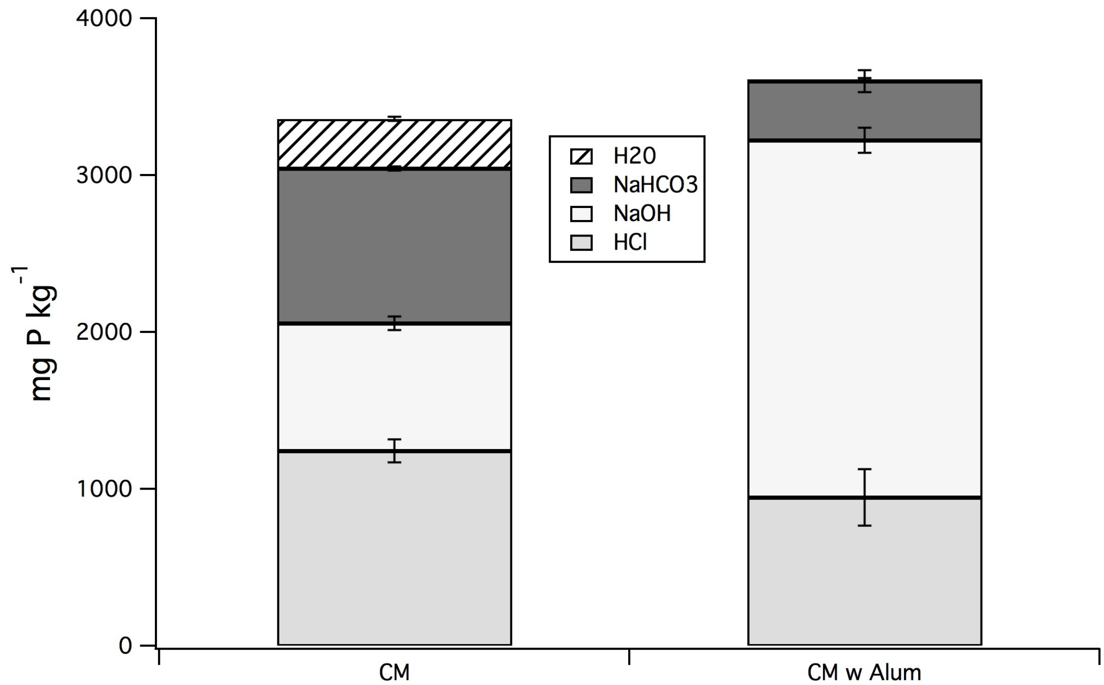


B-4. Mean concentration of dissolved reactive P (DRP) in runoff water at 15, 30 and 60 days after planting Tifway Bermudagrass in Burleson and Branyon soils with and without chemically treated and un-treated composted biosolids (CB). Error bars indicate the standard error of the mean for Burleson and Branyon soils combined.

APPENDIX C

C-1. Total N (TN), total P (TP), total K (TK), organic C (OC), water extractable P (WEP), neutral detergent fiber (NDF), acid detergent fiber (ADF), acid detergent lignin-dry matter (ADL_{dm}), and acid detergent lignin-organic matter (ADL_{om}) of MS (manure solids) with and without Alum before soil treatments were installed. Standard error of mean (std error) given below mean values.

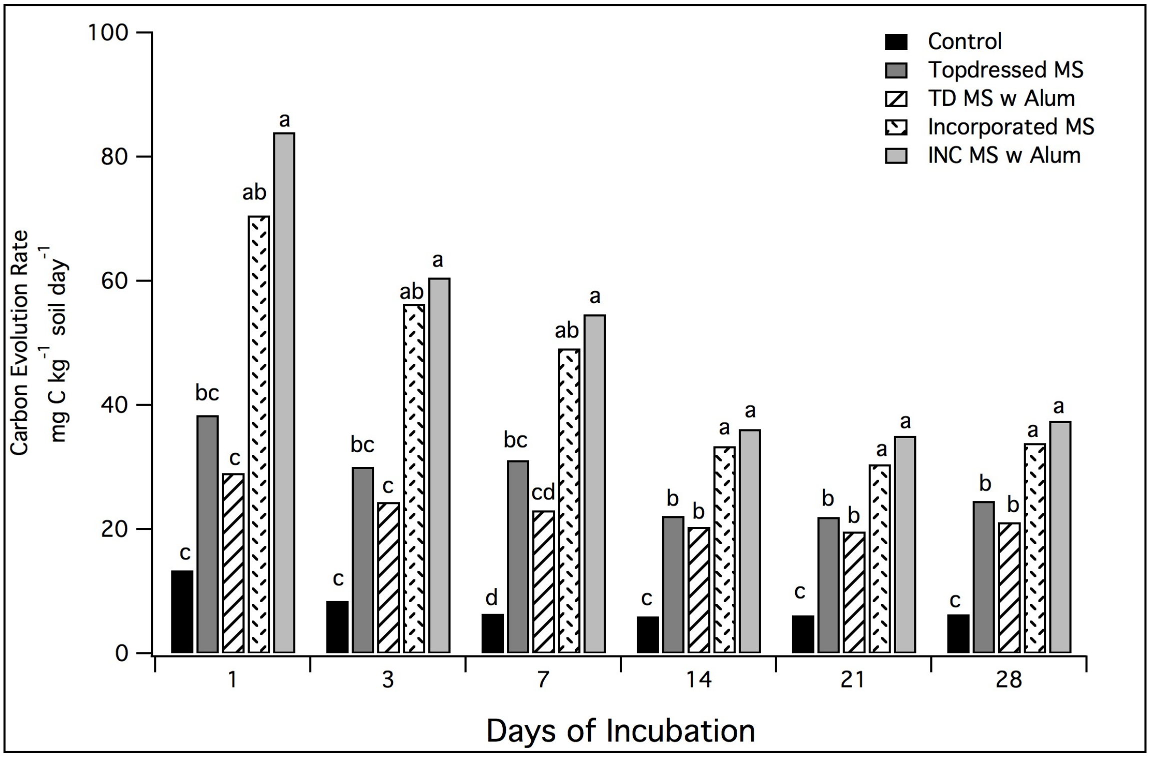
	TN	TP	TK	OC	WEP	NDF	ADF	ADL_{dm}	ADL_{om}
	------------------g kg^{-1}			----------------	mg kg^{-1}	-------------------g kg^{-1}			--------------------
MS	14.8	5.7	8.2	306	278	615	492	386	196
(std error)	(0.48)	(0.53)	(0.20)	(7.8)	(4.43)				
MS with Alum	13.1	6.1	10.1	300	13.6				
(std error)	(0.15)	(1.28)	(2.00)	(3.8)	(3.93)				



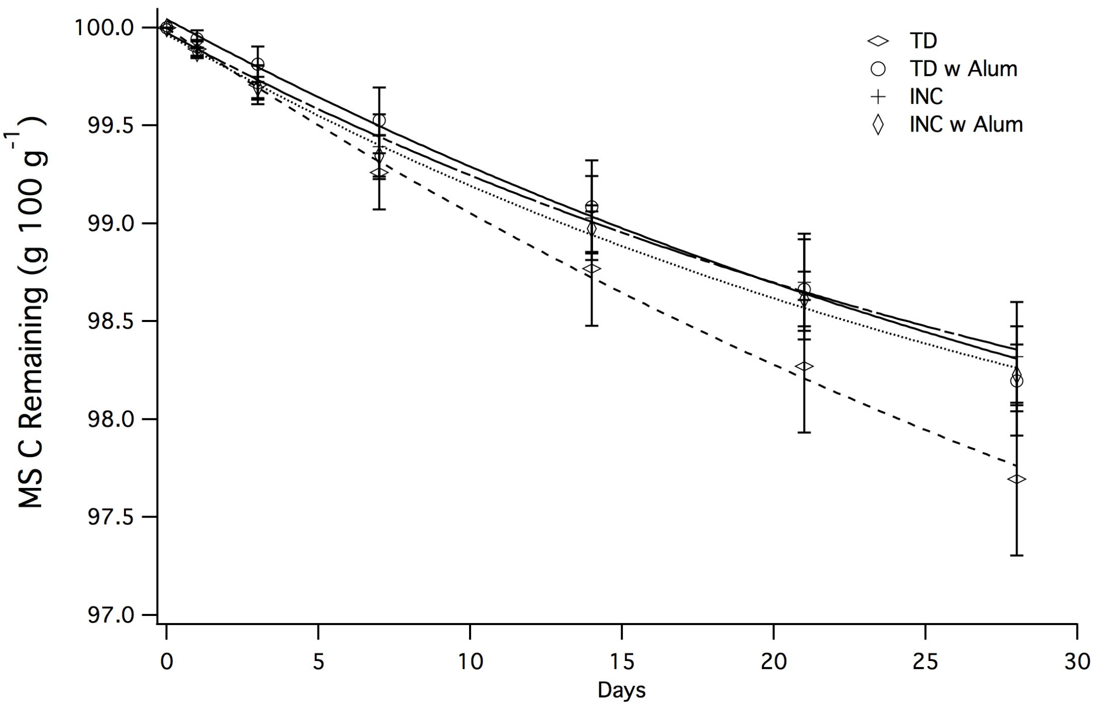
C-2. The amount of P (mg kg^{-1}) sequentially extracted from manure solids (MS) with and without Alum in water, $NaHCO_3$, NaOH, and HCl. Error bars indicate the standard error of the mean.

C-3. Concentration of soil organic C (SOC), total N (TN), Mehlich-3 extractable P (M3P) in soil for field plots amended with MS (manure solids) with and without Alum. In May of 2008, all treatments were sampled from 0- to 5- cm depth. For October of 2008 and 2009, Tifway was sampled below sod harvest depth (2- to 5- cm) and Tifton 85 was sampled from 0- to 5- cm depth. Sampling dates and crop species were analyzed separately if interaction between crop and treatment was significant. Mean values followed by the same letter within column blocks were not significantly differnet (P = 0.05).

	SOC			TN			M3P		
	g kg^{-1}			mg kg^{-1}			mg kg^{-1}		
Treatments	May 2008	Oct 2008	Oct 2009	May 2008	Oct 2008	Oct 2009	May 2008	Oct 2008	Oct 2009
Tifway - Control	7.1	8.7 c	5.8 c	797	1094 c	766 d	218	165 b	169 b
Tifway - Topdressed MS	16.6	8.2 c	6.0 bc	1088	1060 c	839 cd	290	165 b	160 b
Tifway - TD MS w Alum	21.1	10.3 c	8.5 b	1396	1253 c	1056 ab	342	185 b	149 b
Tifway - Incorporated MS	38.0	16.9 b	7.5 bc	2097	1714 b	969 bc	503	318 a	226 a
Tifway - INC MS w Alum	29.4	25.3 a	12.4 a	1734	2203 a	1196 a	380	311 a	265 a
Tifton 85 - Control	6.6	9.3 C	10.3 C	727	1072 C	1001 C	241	134 C	116 B
Tifton 85 - Topdressed MS	21.6	19.6 B	36.2 A	1357	1761 B	3535 A	365	259 B	328 A
Tifton 85 - TD MS w Alum	22.5	17.9 B	26.7 AB	1367	1771 B	2338 B	331	229 B	338 A
Tifton 85 - Incorporated MS	42.7	27.1 A	18.2 BC	2214	2385 A	1706 BC	475	338 A	281 A
Tifton 85 - INC MS w Alum	37.7	29.8 A	20.4 BC	2022	2316 A	2098 B	408	306 A	282 A
Soil Treatment Average									
Control	6.9 C			762 C			230 C		
Topdressed MS	19.1 B			1222 B			327 B		
TD MS w Alum	21.8 B			1381 B			336 B		
Incorporated MS	40.3 A			2155 A			489 A		
INC MS w Alum	33.6 A			1878 A			394 B		



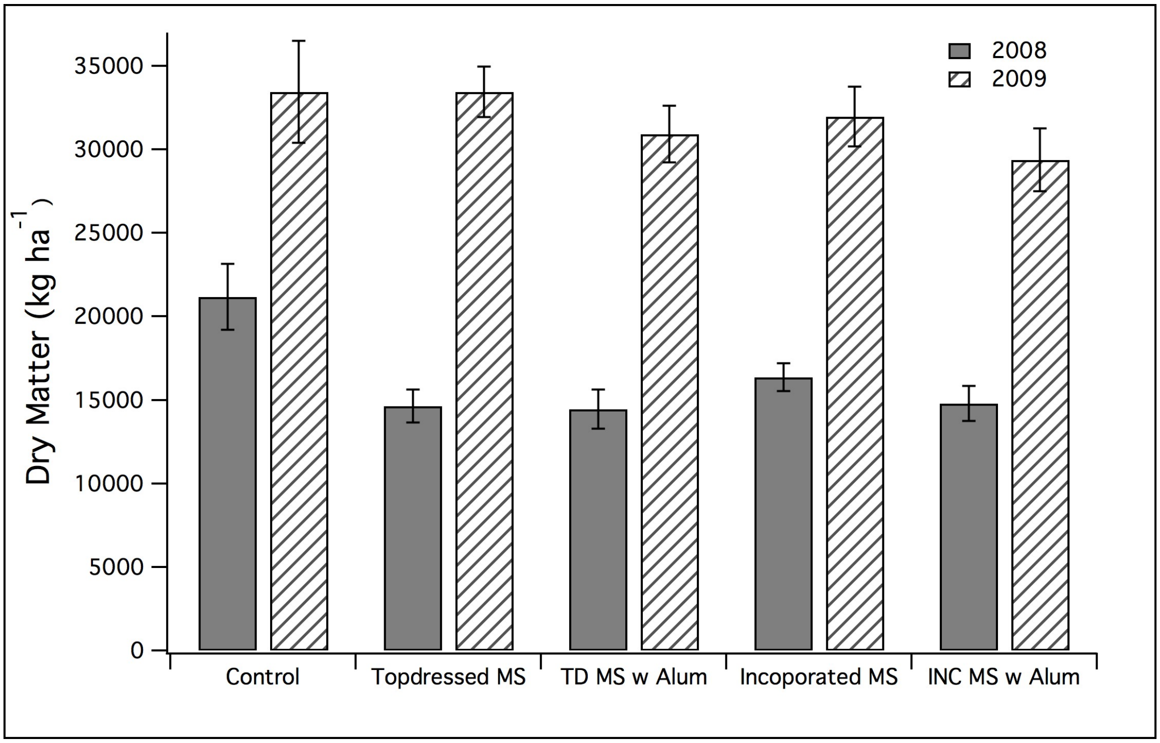
C-4. Carbon (CO_2-C) evolution rate measured under laboratory conditions at 1, 3, 7, 14, 21 and 28 days after cores were collected from Tifway sod and Tifton 85 forage field plots. Treatments comprised topdressed and incorporated manure solids (MS) with or without Alum plus control soil. Bars labeled with the same letter within each sampling date indicate no significant difference between soil treatments (P = 0.05).



C-5. The proportion of C in manure solids (MS) remaining for soil cores sampled from field plots of Tifway sod and Tifton 85 forage and incubated under laboratory conditions for 28 days. Cores were sampled 2 weeks after top-dressing (TD) or incorporating (INC) MS with or without Alum. The proportion of MS source of C remaining on each sampling date was fitted to a four-parameter double exponent decay model. Error bars indicate the standard error of the mean.

C-6. Regression analysis of four-parameter double exponential models of proportion of C in manure solids (MS) remaining for topdressed (TD) or incorporated (INC) MS with or without Alum (Al). Model parameters were derived from means of four replications of cores sampled from field plots of Tifway sod and Tifton 85 forage, which were combined.

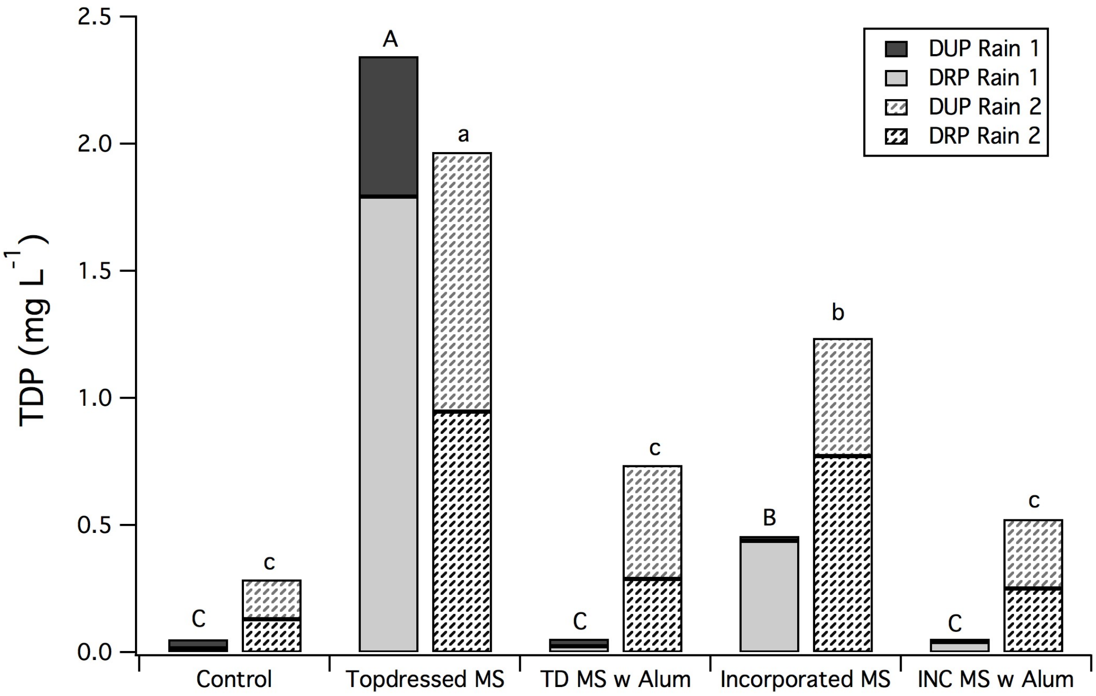
		Fast Pool			Slow Pool		
	R^2	Intercept	k_f ^{days}	$t_{1/2}$ $^{(days)}$	Intercept	k_f ^{days}	$t_{1/2}$ $^{(yrs)}$
TD	1.00	0.214	0.254	3.94	99.797	0.00075	3.64
TDAl	0.99	0.026	0.206	4.85	99.981	0.00064	4.28
INC	0.99	0.298	0.246	4.07	99.708	0.00050	5.52
INCAl	0.99	0.319	0.263	3.80	99.684	0.00052	5.25



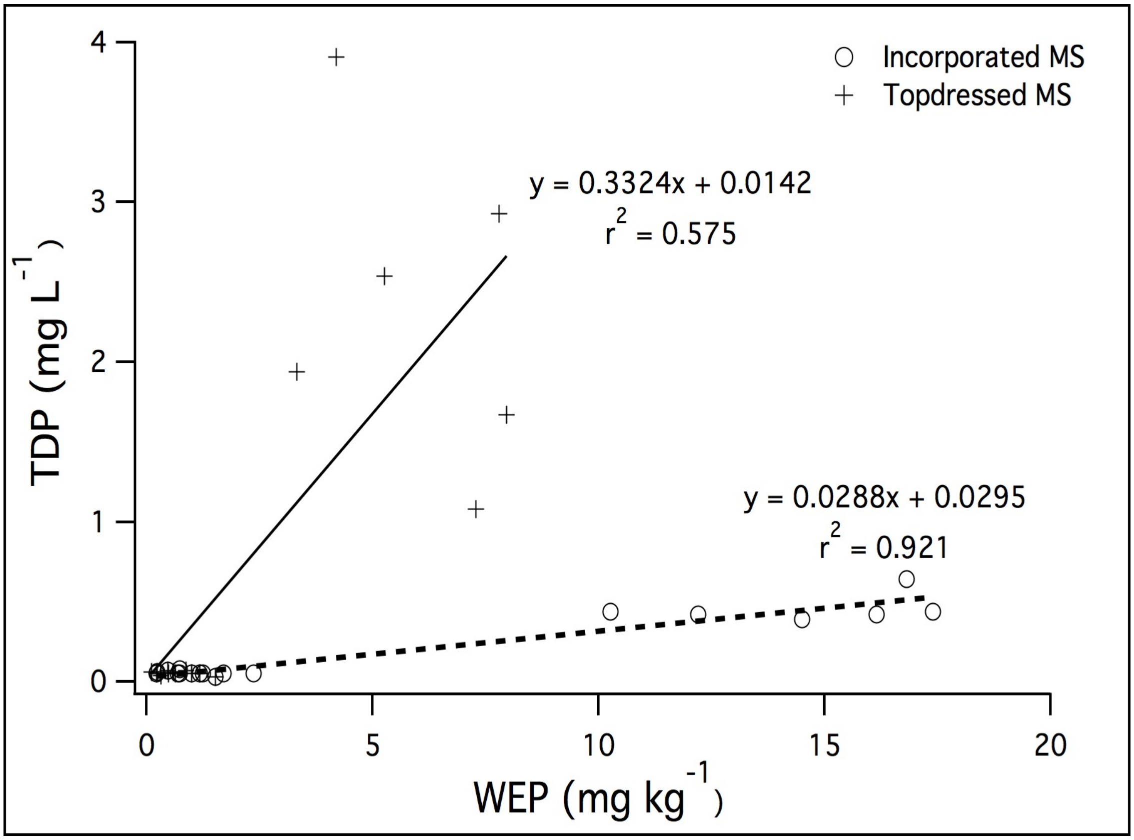
C-7. Total dry matter production of Tifton 85 bermudagrass under field conditions during 2008 and 2009 for topdressed (TD) or incorporated (INC) manure solids (MS) with or without Alum. Error bars indicate the standard error of the mean.

C-8. Total N (TN), total P (TP), total K (TK), soil organic C (SOC), Mehlich-3 extractable P (M3P), Mehlich-3 extractable K (M3K), water extractable P (WEP) of soil packed into box lysimeters amended with top-dressed (TD) or incorporated (INC) manure solids (MS) with or without Alum. Data were combined over treatments applied to Tifway and Tifton 85 bermudagrasses. Means followed by the same letter within columns indicates no significant difference between soil treatments (P = 0.05).

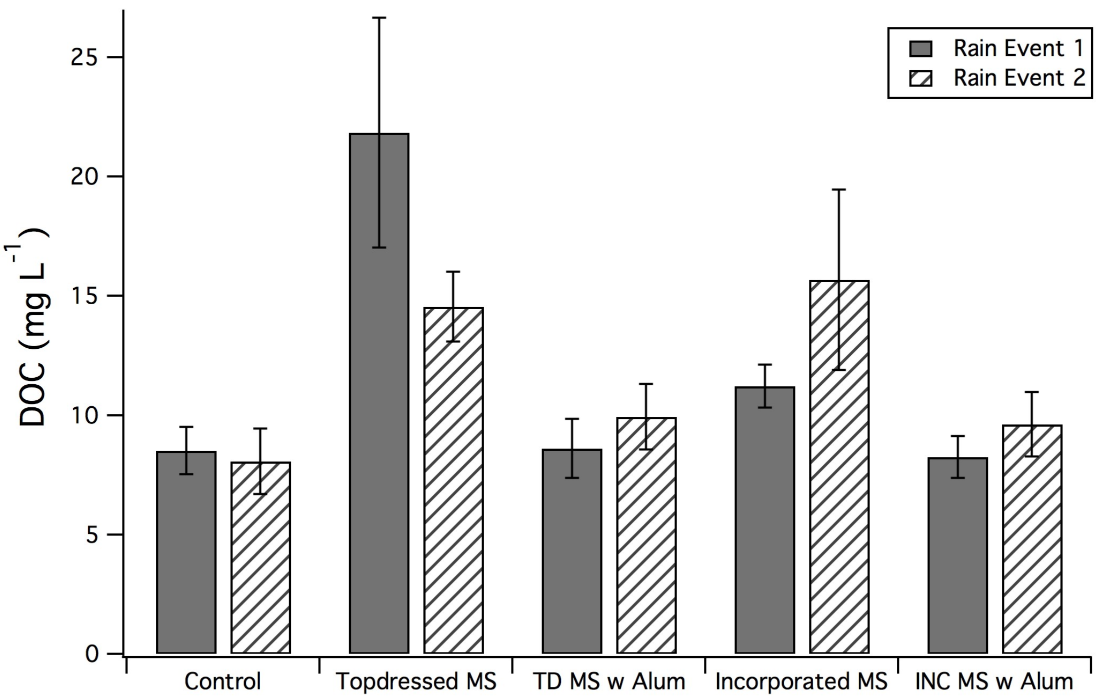
	pH	TN	TP	TK	SOC	M3P	M3K	WEP
		---------------mg kg^{-1}-------------			g kg^{-1}	---------------mg kg^{-1}-------------		
Control	6.8 B	835 C	42 C	313 C	2.4 C	12 C	93 B	1.23 C
Topdressed MS	7.2 B	1558 CB	170 BC	422 BC	13.8 BC	79 B	158 B	7.69 B
TD MS w Alum	5.8 C	1879 B	254 B	410 BC	17.8 AB	97 B	135 B	0.70 C
Incorporated MS	7.7 A	2933 A	440 A	723 A	29.3 A	176 A	379 A	15.82 A
INC MS w Alum	6.2 C	2824 A	465 A	609 AB	30.5 A	164 A	247 AB	1.77 C



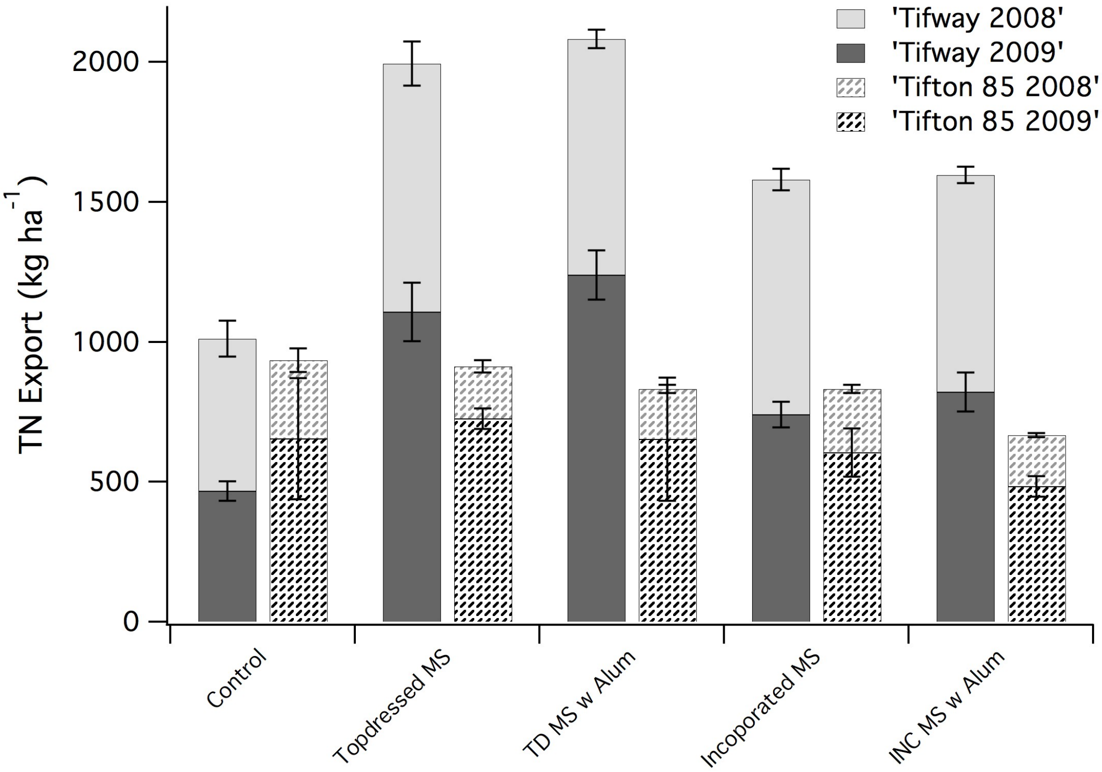
C-9. Concentration of total dissolved P (TDP) in runoff water collected during establishment of Tifway and Tifton 85 bermudagrass after top-dressing (TD) or incorporating (INC) manure solids (MS) in a sandy loam soil. Runoff concentration of TDP is the sum of concentrations of dissolved reactive P (DRP) and dissolved un-reactive P (DUP). Data from Tifway and Tifton 85 were combined, but two simulated rain events were analyzed separately. Bars labeled with the same letter within each rain event indicate no significant difference in TDP between soil treatments (P = 0.05).



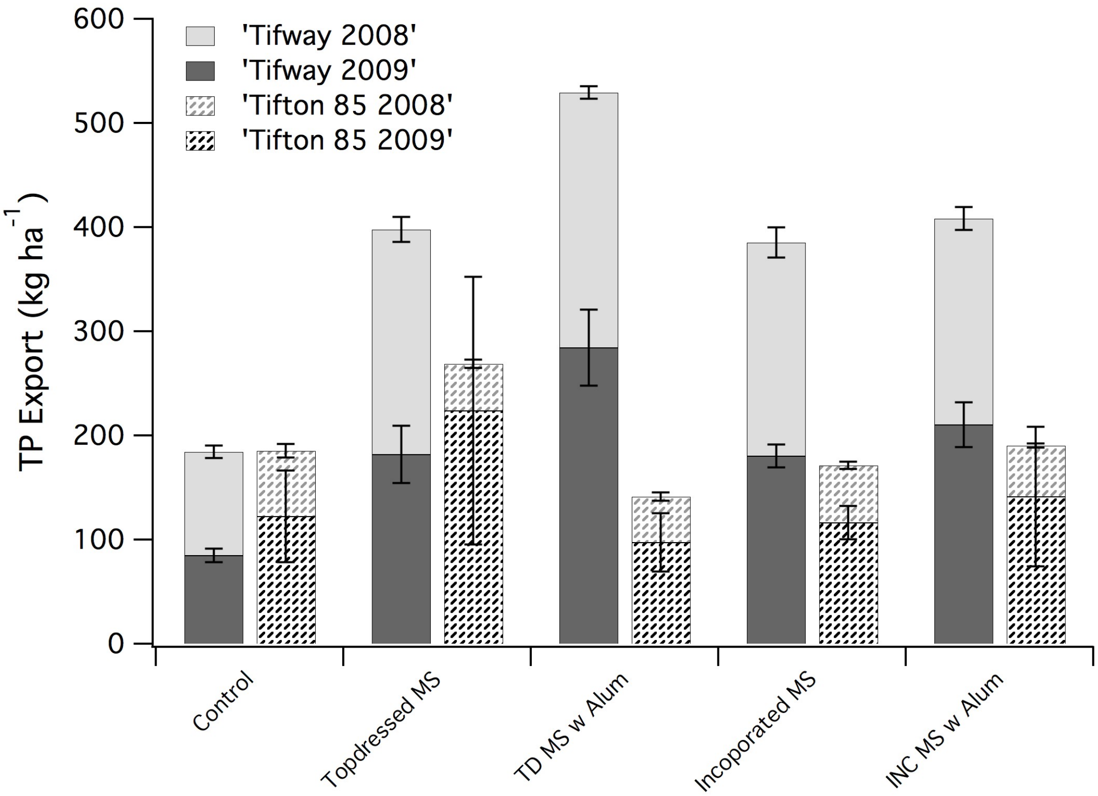
C-10. Relationship between concentrations of soil water extractable P (WEP) and total dissolved P (TDP) in runoff during the first rain event after planting Tifway and Tifton 85 bermudagrasses and top-dressing or incorporating manure solids (MS) in a sandy loam soil.



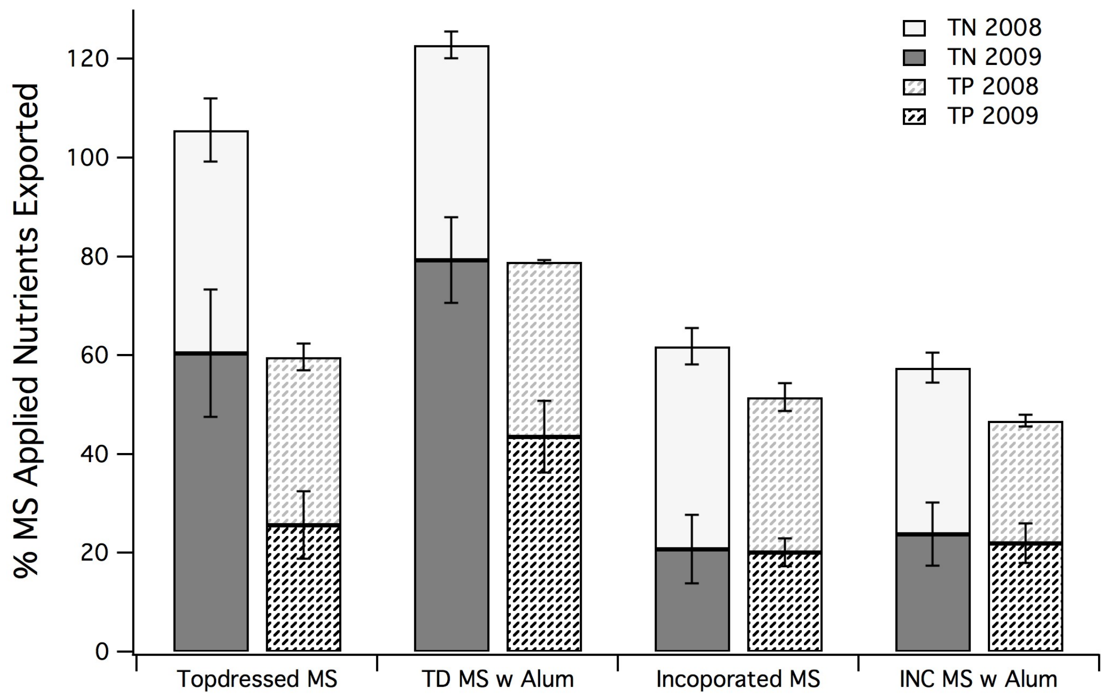
C-11. Mean concentration of dissolved organic carbon (DOC) in runoff during establishment of Tifway and Tifton 85 bermudagrass after top-dressing (TD) or incorporating (INC) manure solids (MS) with or without Alum. Data for bermudagrass species were combined for each rain event. Error bars indicate the standard error of the mean.



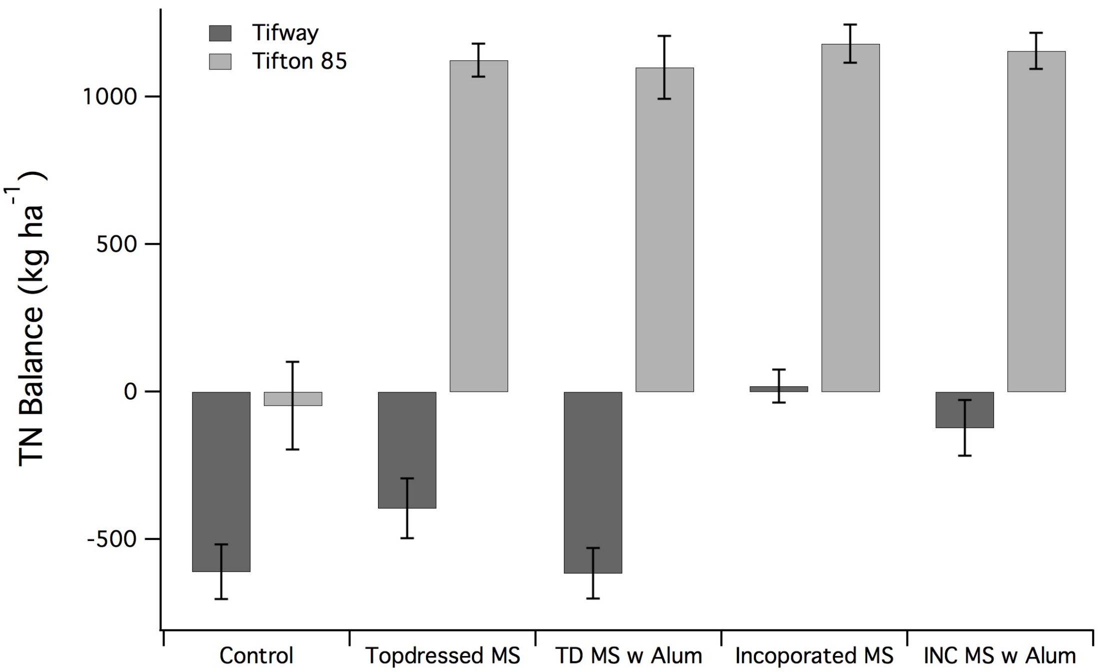
C-12. Amount (kg ha^{-1}) of TN exported in annual sod harvests of Tifway sod in 2008 and 2009 and three forage harvests during 2008,and four forage harvests during 2009 for Tifton 85. Treatments comprised top-dressed (TD) or incorporated (INC) manure solids (MS) with or without Alum and control soil. The error bars indicate the standard error of the mean for each year.



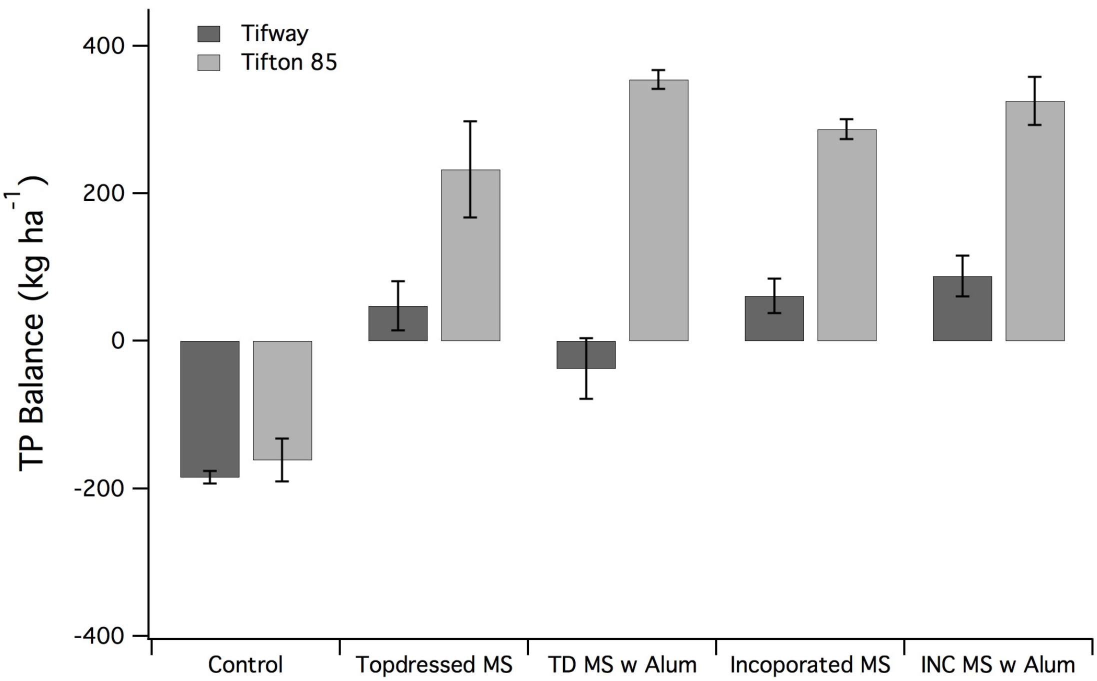
C-13. Amount (kg ha^{-1}) of TP exported in annual sod harvests of Tifway sod in 2008 and 2009 and three forage harvests during 2008,and four forage harvests during 2009 for Tifton 85. Treatments comprised top-dressed (TD) or incorporated (INC) manure solids (MS) with or without Alum and control soil. The error bars indicate the standard error of the mean for each year.



C-14. Proportion of TN and TP in applied manure solids (MS) that were exported in Tifway sod harvests during 2008 and 2009. Treatments comprised top-dressed (TD) or incorporated (INC) manure solids (MS) with or without Alum. Error bars indicate the standard error of the mean.



C-15. Balance of total N (TN) for Tifway sod and Tifton 85 forage production over two years. Treatments comprised top-dressed (TD) or incorporated (INC) manure solids (MS) with or without Alum and control soil. Error bars indicate the standard error of the mean.



C-16. Balance of total P (TP) for Tifway sod and Tifton 85 forage production over two years. Treatments comprised top-dressed (TD) or incorporated (INC) manure solids (MS) with or without Alum and control soil. Error bars indicate the standard error of the mean

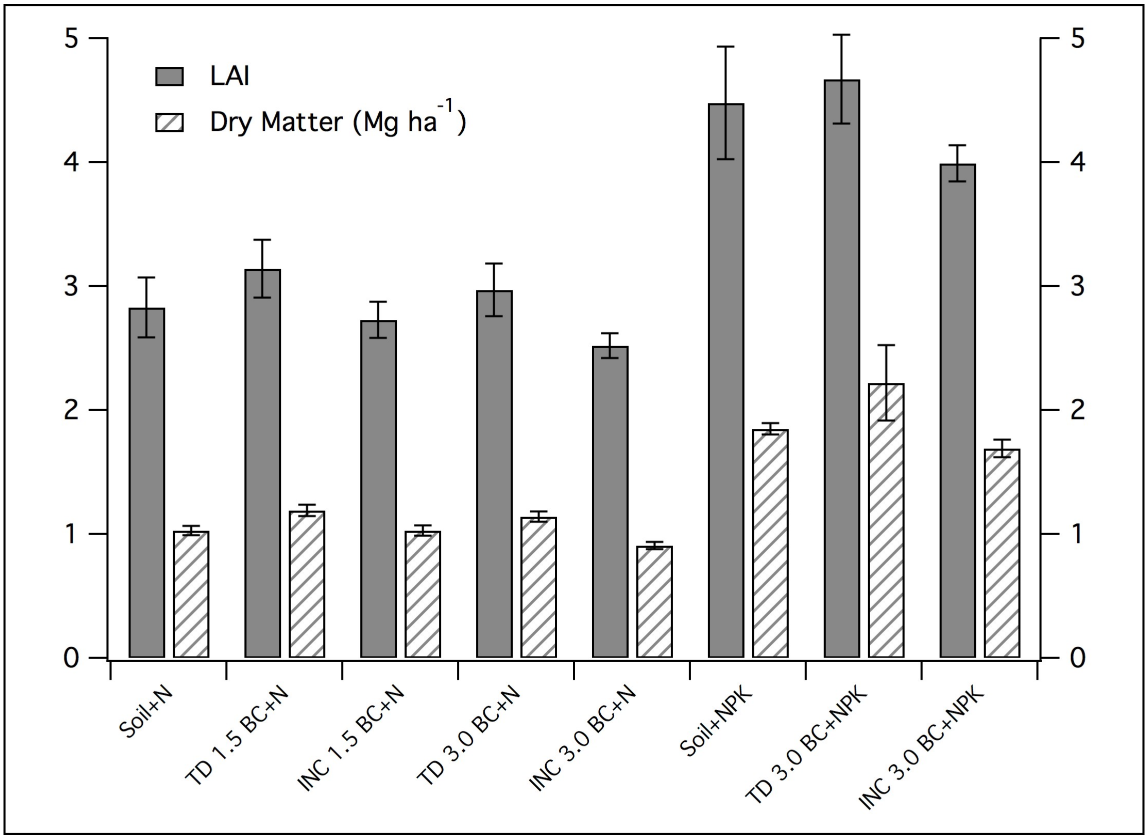
APPENDIX D

D-1. Soil pH and concentrations of total N (TN), total P (TP), total K (TK), organic carbon (OC), NO_3-N, Mehilich-3 P (M3P), Mehilich-3 K (M3K), and water extractable P (WEP) in sorghum feedstock, biochar or soil before soil treatments were imposed. Nutrients conserved in biochar (mass output / mass input) after pyrolysis of sorghum biomass are given as percentages of TN, TP, and TK.

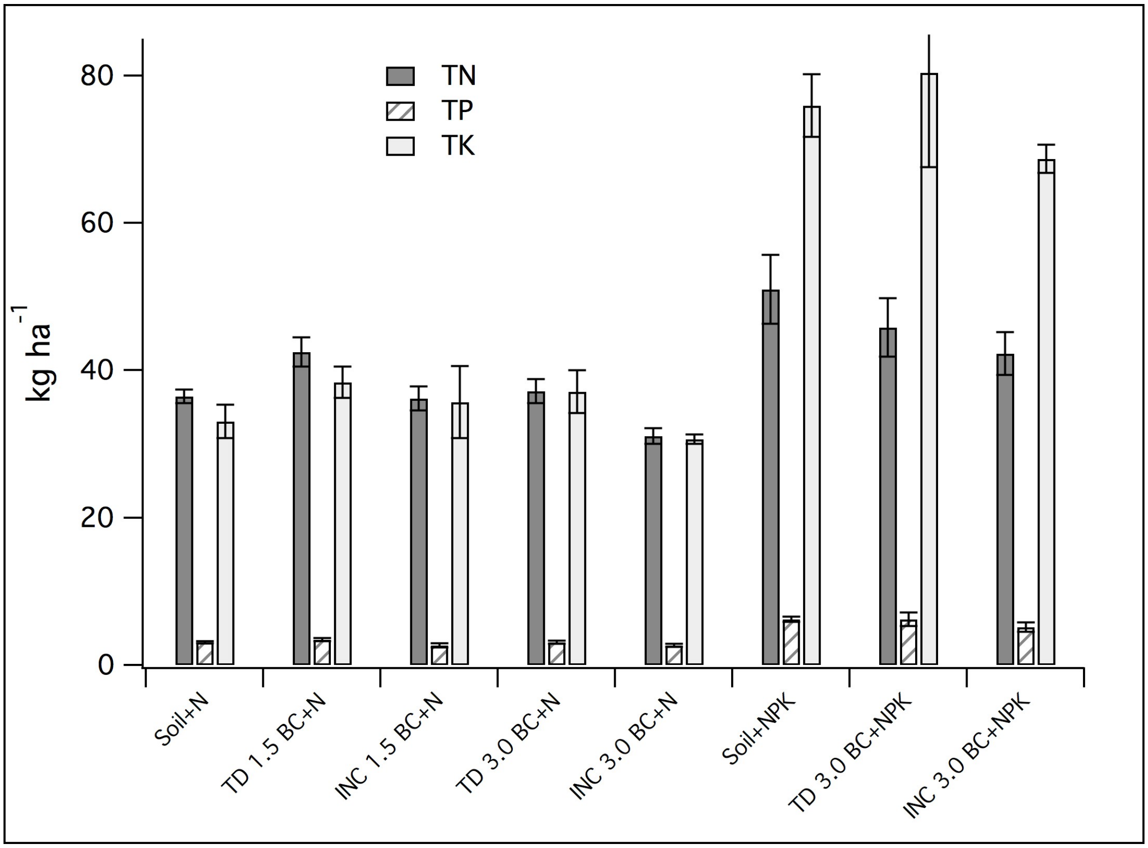
	pH	TN	TP	TK	OC	NO_3	M3P	M3K	WEP
		-------------mg kg^{-1}	----------		g kg^{-1}	------------mg kg^{-1}	----------	----------	
Feedstock		6880	1400	1210					
Biochar		7433	2340	4140	564.5				99.5
(std err.)		(186)	(250)	(1030)	(22.1)				(9.65)
% Conserved		22%	34%	6.8%					
Soil	6.4	570	69.6	428	2.5	17	10	95	1.1

D-2. Soil pH; bulk density (ρ_b); and concentrations of total N, P and K (TN, TP, TK); organic carbon (SOC); soil-test NO_3-N and Mehlich-3 P and K (M3P, M3K); and water-extractable P (0 to 5 cm depth) after the final rain event. Treatments comprised control soil (Soil + N) and soil amended with inorganic fertilizer (NPK) and/or top-dressed (TD) or incorporated (INC) biochar (BC) at 1.5-Mg or 3.0-Mg rates.

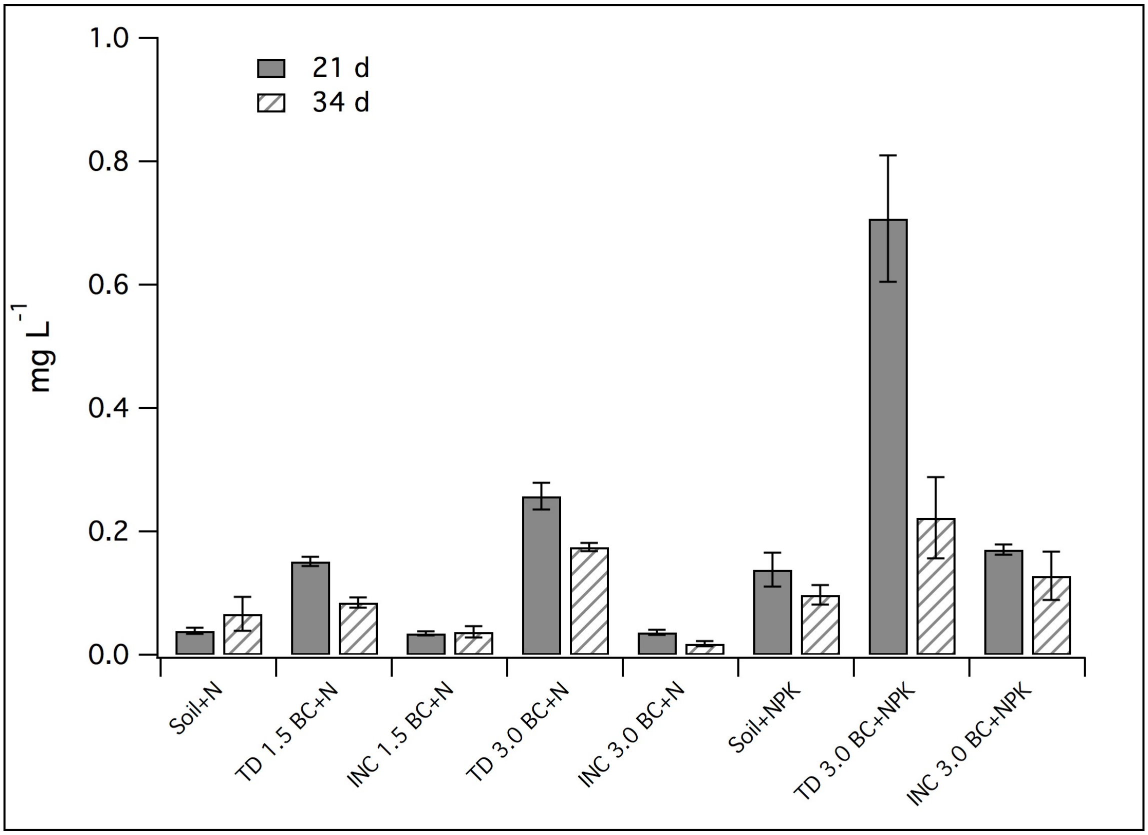
	pH	ρ_b	TN	TP	TK	SOC	NO_3	M3P	M3K	WEP
Treatment		g cm^{-3}	--------------mg kg^{-1}-------------			g kg^{-1}	---------------------mg kg^{-1}-------------------			
Soil+N	6.43	1.57	761	72	356	2.03	3.7	11.7	74	1.19
TD 1.5 BC+N	6.50	1.24	650	72	377	2.40	4.3	13.0	104	1.35
INC 1.5 BC+N	6.53	1.37	660	71	345	2.63	3.7	13.3	88	1.25
TD 3.0 BC+N	6.77	1.29	588	79	428	2.70	6.3	13.7	116	1.17
INC 3.0 BC+N	6.63	1.39	702	78	393	2.40	3.7	12.3	93	1.07
Soil+NPK	6.23	1.39	935	95	374	2.27	3.0	28.3	102	4.78
TD 3.0 BC+NPK	6.70	1.27	766	91	426	2.70	2.7	23.7	132	3.63
INC 3.0 BC+NPK	6.63	1.44	734	90	417	3.20	3.3	20.0	119	2.27
Pr > *F*	.045	.149	.706	.004	.132	.019	.002	.000	.000	.000



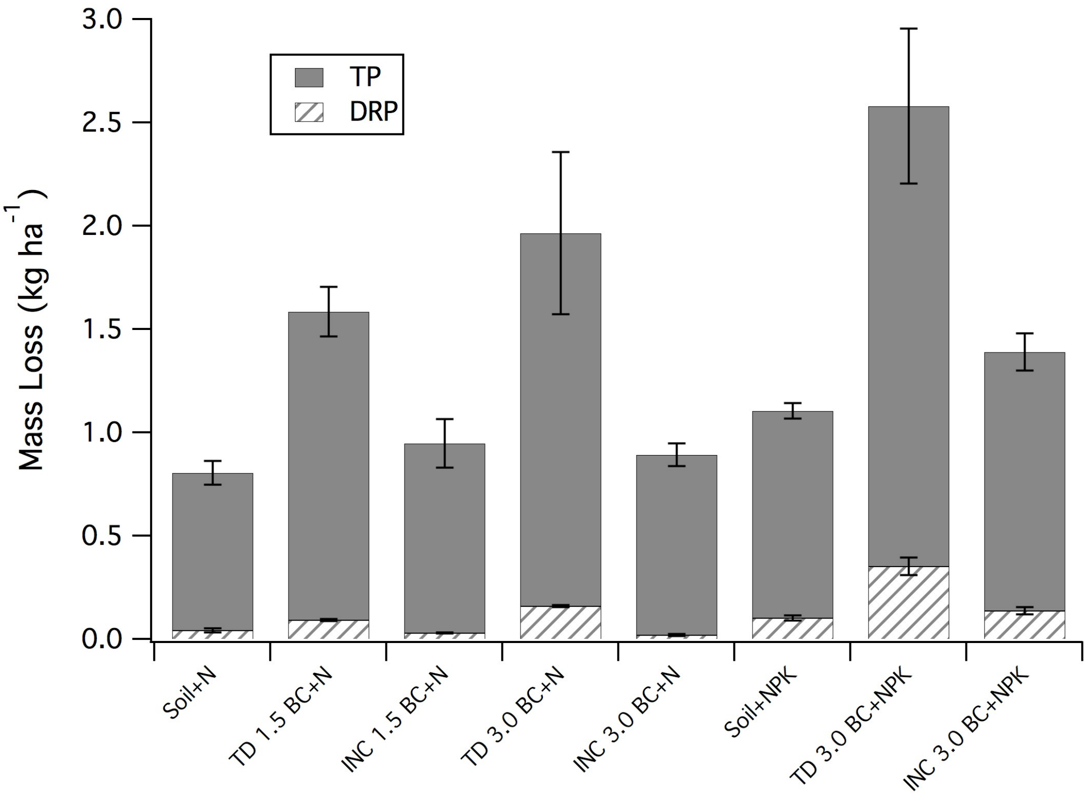
D-3. The mean leaf area index (LAI) and dry matter production (Mg ha^{-1}) with and without top-dressed (TD) or incorporated (INC) biochar (BC), applied at 1.5 or 3.0 Mg ha^{-1}, and with or without supplemental N, P, and K fertilizer nutrients (NPK) over a 45 day period. Error bars indicate the standard error of the mean.



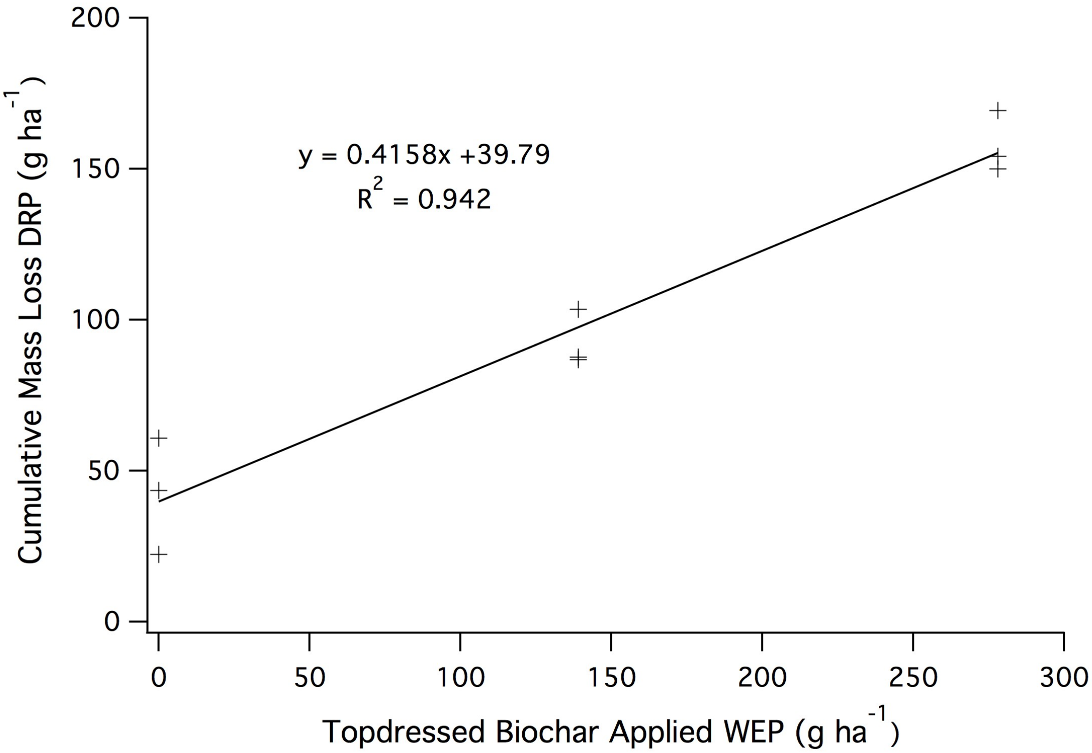
D-4. Content (kg ha^{-1}) of total N (TN), total P (TP), and total K (TK) in sorghum above-ground biomass after 45-day establishment period. Treatments comprised soil with and without N, P, and K fertilizer (NPK) or top-dressed (TD) or incorporated (INC) biochar (BC) at rates of 1.5 or 3 Mg ha^{-1}.



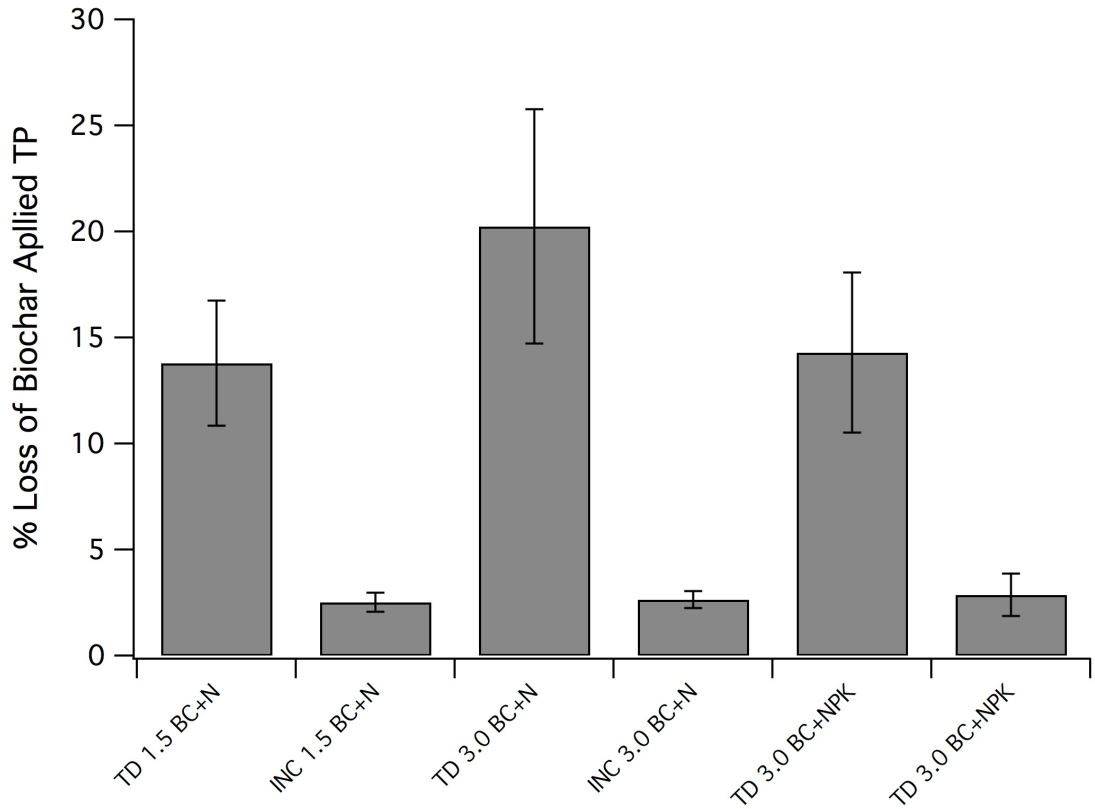
D-5. Mean concentrations of dissolved reactive P (DRP) in runoff at 21 and 34 d after planting sorghum. Treatments comprised soil with and without N, P, and K fertilizer (NPK) or top-dressed (TD) or incorporated (INC) biochar (BC) at rates of 1.5 or 3 Mg ha^{-1}. Error bars indicate the standard error of the mean.



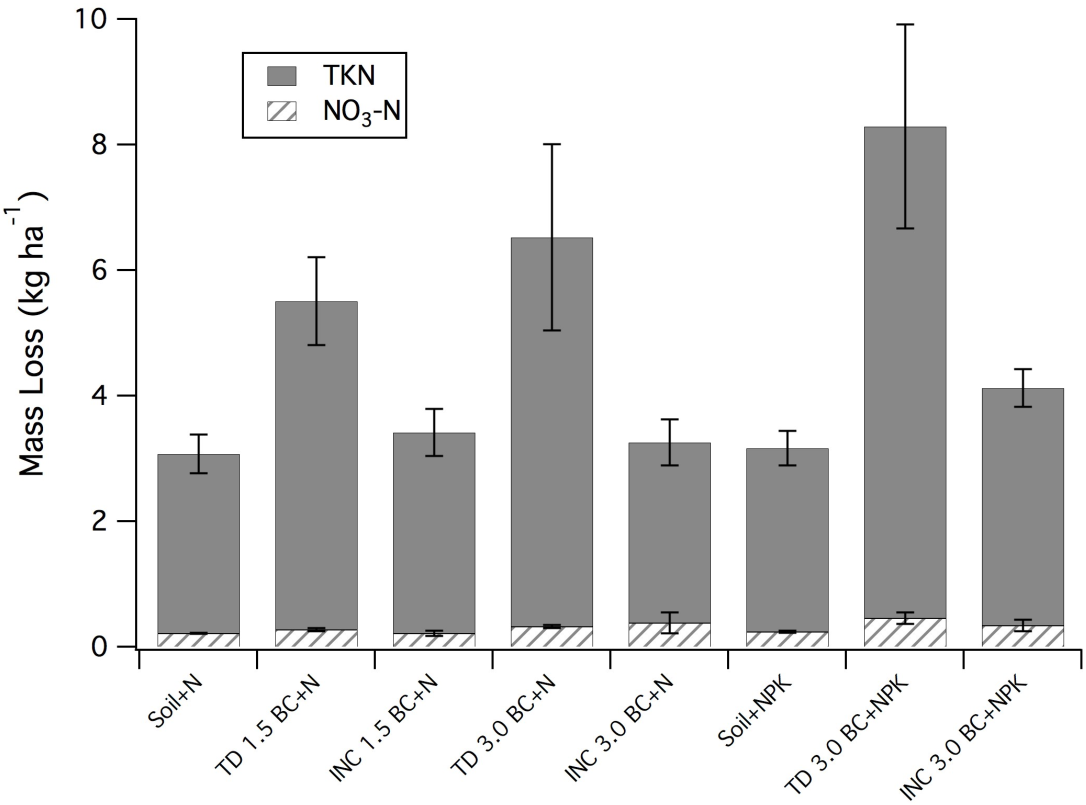
D-6. Cumulative mass loss (kg ha^{-1}) of total P (TP) and dissolved reactive P (DRP) in runoff for rain events at 21 and 34 days after planting sorghum. Treatments comprised soil with and without N, P, and K fertilizer (NPK) or top-dressed (TD) or incorporated (INC) biochar (BC) at rates of 1.5 or 3 Mg ha^{-1}. Error bars indicate the standard error of the mean.



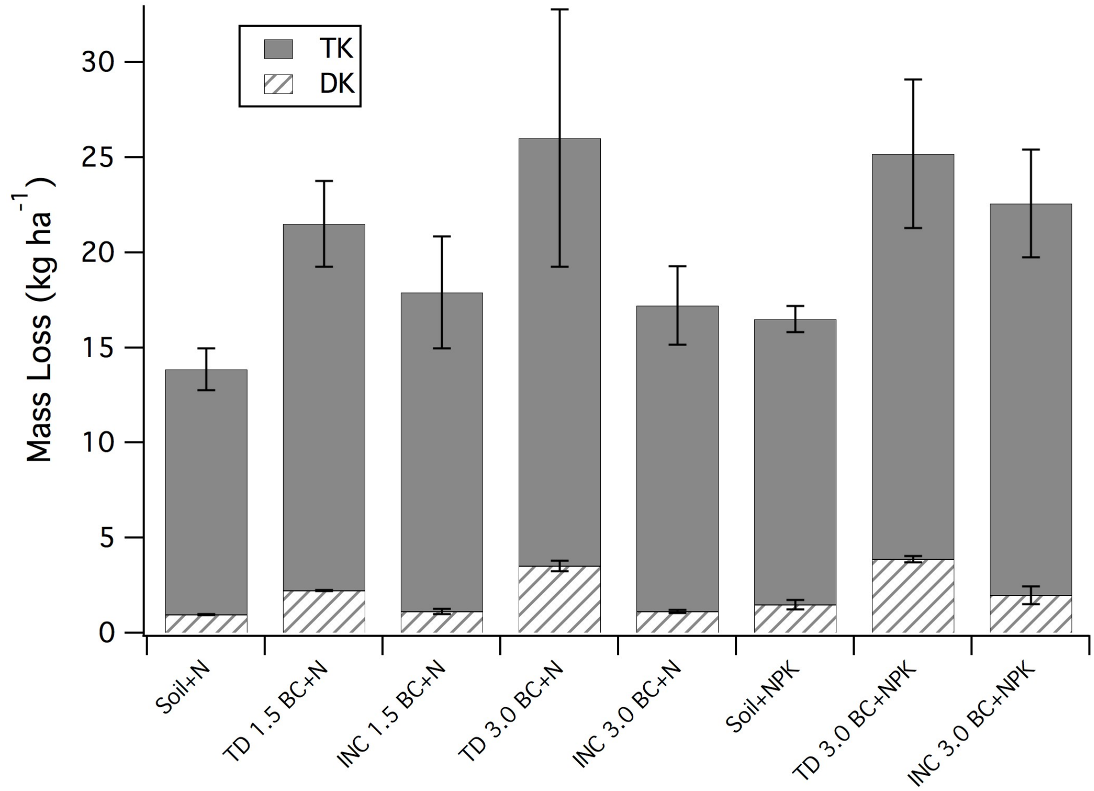
D-7. The relationship between the amount (g ha^{-1}) of water extractable P (WEP) applied with biochar and the mass loss (g ha^{-1}) of dissolved reactive P (DRP) in runoff for soils amended with and without topdressed biochar and supplemental nitrogen fertilizer.



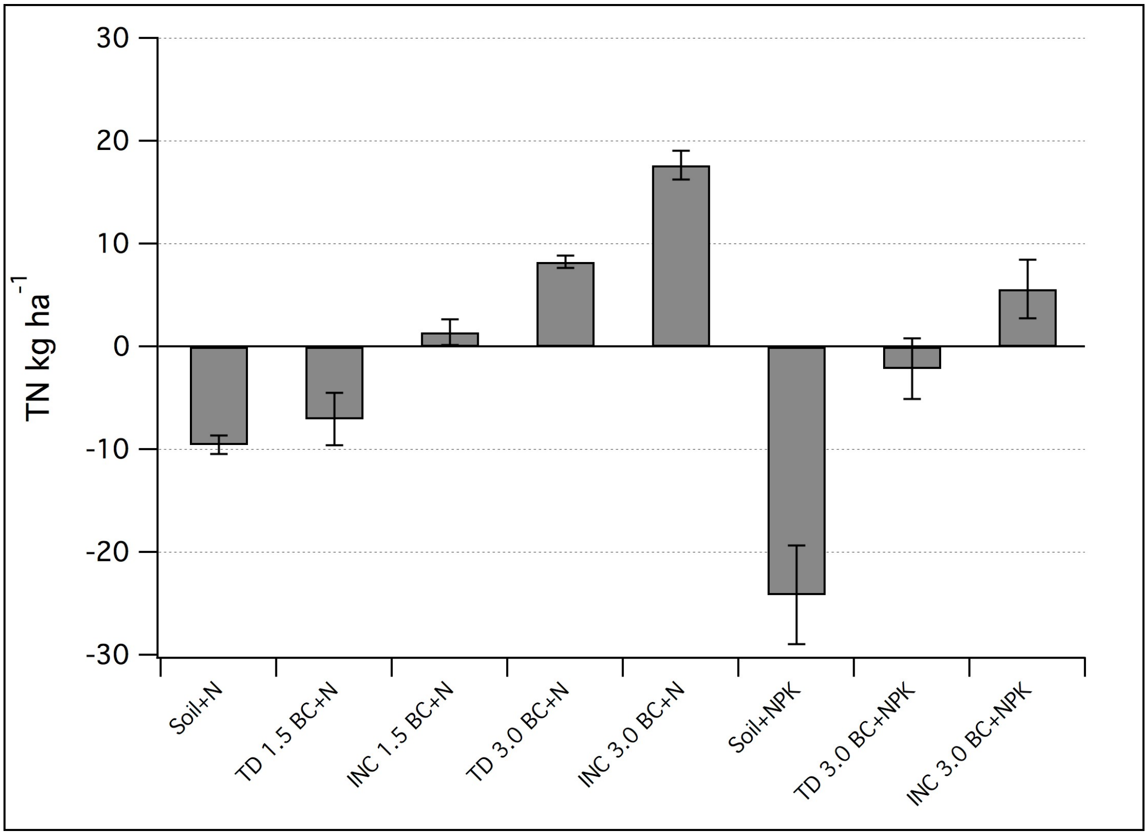
D-8. The percentage of total P (TP) in applied biochar that was lost in runoff over two rain events. Treatments comprised soil with and without N, P, and K fertilizer (NPK) and top-dressed (TD) or incorporated (INC) biochar (BC) at rates of 1.5 or 3 Mg ha^{-1}. Error bars indicate the standard error of the mean.



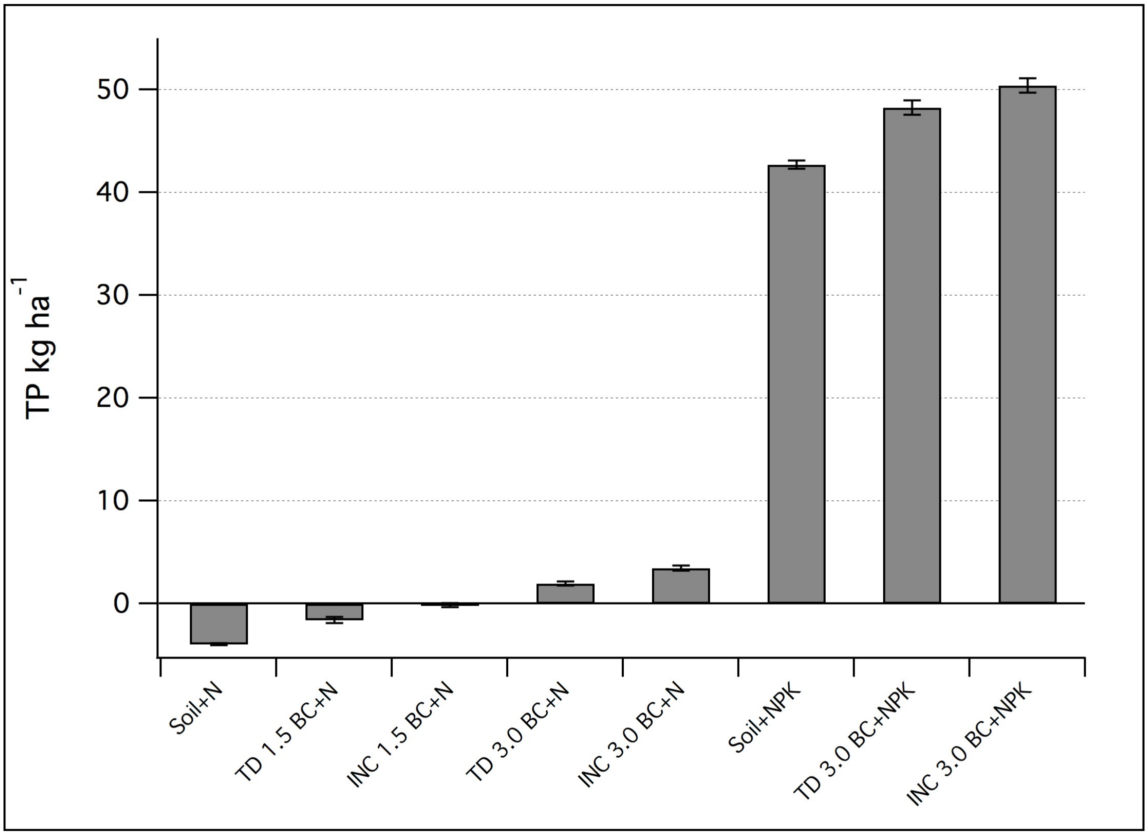
D-9. The cumulative mass loss of total Kjeldahl nitrogen (TKN) and NO_3-N in runoff over two rain events. Treatments comprised soil with and without N, P, and K fertilizer (NPK) and top-dressed (TD) or incorporated (INC) biochar (BC) at rates of 1.5 or 3 Mg ha^{-1}. The error bars indicate the standard error of the mean.



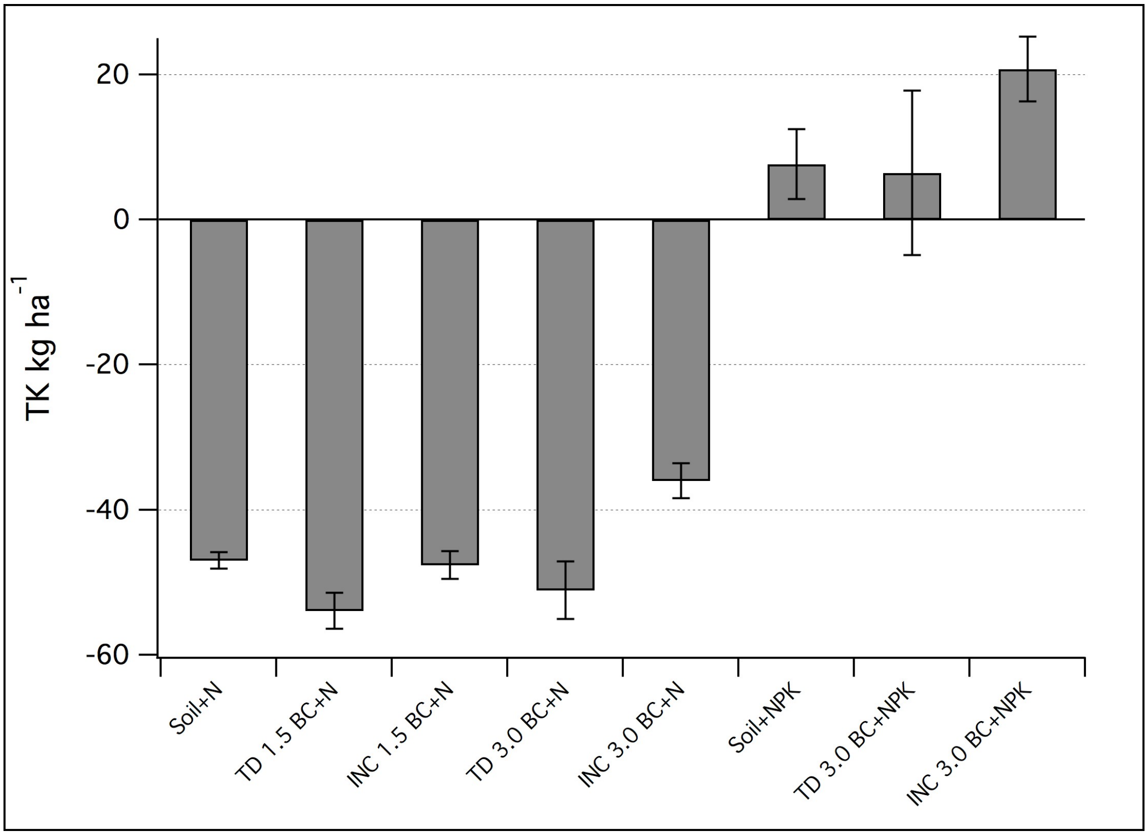
D-10. The cumulative mass loss of total K (TK) and dissolved K (DK) in runoff over two rain events. Treatments comprised soil with and without N, P, and K fertilizer (NPK) and top-dressed (TD) or incorporated (INC) biochar (BC) at rates of 1.5 or 3 Mg ha^{-1}. The error bars indicate the standard error of the mean.



D-11. The mass balance (total N inputs – total N outputs) of N in soil during sorghum establishment in box lysimeters. Treatments comprised soil with and without N, P, and K fertilizer (NPK) and top-dressed (TD) or incorporated (INC) biochar (BC) at rates of 1.5 or 3 Mg ha^{-1}. The error bars indicate the standard error of the mean.



D-12. The mass balance (total P inputs – total P outputs) of P in soil during sorghum establishment in box lysimeters. Treatments comprised soil with and without N, P, and K fertilizer (NPK) and top-dressed (TD) or incorporated (INC) biochar (BC) at rates of 1.5 or 3 Mg ha^{-1}. The error bars indicate the standard error of the mean.



D-13. The mass balance (total K inputs – total K outputs) of in soil during sorghum establishment in box lysimeters. Treatments comprised soil with and without N, P, and K fertilizer (NPK) and top-dressed (TD) or incorporated (INC) biochar (BC) at rates of 1.5 or 3 Mg ha^{-1}. The error bars indicate the standard error of the mean.

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Publications:

Schnell, R., D. Vietor, C. Munster, R. White, and T. Provin. 2009. Effects of Composted Biosolids and Nitrogen on Turfgrass Establishment and Sod Properties at Harvest. HortScience 44 (6): 1746-1750.

Schnell, R., D. Vietor, C. Munster, R. White, and T. Provin. 2010. Effect of Turfgrass Establishment Practices and Composted Biosolids on Water Quality. J. Environ. Qual. 39:1-9.

Proposals:
Schnell, R., D. Vietor, C. Munster and R. White. 2007. Chemically-Treated Composted Biosolids Enhance Water Conservation and Quality on Urban Landscapes. $5,000. TWRI Graduate Research Grants Program.

Schnell, R., D. Vietor, C. Munster T. Provin and R. White. 2007. Cropping Systems for Sustainable Nutrient Management and Dairy Production. $10,000. Southern SARE Graduate Student Grant Program.