Crop residue management effects on crop production, greenhouse gases emissions, and soil quality in the Mid-Atlantic USA

Martín Leonardo Battaglia

Dissertation submitted to the faculty of the Virginia Polytechnic Institute and State University in partial fulfillment of the requirements for the degree of

Doctor of Philosophy
In
Crop & Soil Environmental Sciences

Wade Thomason
John Herschel Fike
Gregory Evanylo
Gordon Groover

9/21/2018
Blacksburg, Virginia

Keywords: corn stover, wheat straw, crop residue retention, soil quality, greenhouse gases, theoretical ethanol yield, theoretical ethanol potential, nutrient uptake and concentration

Crop residue management effects on crop production, greenhouse gases emissions, and soil quality in the Mid-Atlantic USA

Martín Leonardo Battaglia

ABSTRACT

Cellulosic biomass-to-bioenergy systems can provide environmental and economic benefits to modern societies, reducing the dependence on fossil-fuels and greenhouse gas emissions while simultaneously improving rural economies. Corn (*Zea mays* L.) stover and wheat straw (*Triticum aestivum* L.) residues have particular promise given these crops are widely grown and their cellulosic fractions present a captured resource as a co-product of grain production. However, concerns exist about residue removal effects on soil health, greenhouse gases emissions and subsequent crop productivity. The carbon footprint and the crop yield productivity and soil health responses resulting from the removal of crop residues has been studied extensively over the last 20 years, but this research has been largely conducted in the Corn Belt. To investigate the impact of crop residue removal in the Mid-Atlantic USA, combinations of corn stover (0, 3.33, 6.66 and 10 Mg ha^{-1}) and wheat straw (0, 1.0, 2.0, and 3.0 $Mgha^{-1}$) were soil applied in a corn-wheat/soybean (*Glycine max* L. Merr.) rotation in Virginia's Coastal Plain (New Kent, NK). Corn stover (0, 3.33, 6.66, 10 and 20 Mg ha^{-1}) was applied in a continuous corn cropping system in the Ridge/Valley province (Blacksburg, BB). For each system, residues were applied following grain harvest over two production cycles. Each experiment was conducted as a randomized complete design with four replications. In the year following application, retaining approximately 50% (5 Mg ha^{-1}) or more of the stover produced increased CH_4 and N_2O emissions from the loamy soil in BB about 25%. Maximum CH_4 and N_2O fluxes

(4.16 and 5.94 mg m^{-2} day^{-1}, respectively) occurred with 200% (20 Mg ha^{-1}) retention rates. Two cycles of corn stover management in BB, and one cycle of corn stover or wheat straw management in the sandy loam soils in NK did not affect fluxes of any gas. In 2017, total retained residues affected CH_4 and N_2O fluxes following V4 nitrogen fertilization, but these results were not biologically meaningful. Two cycles of crop residue management, with retention rates up to 20 Mg ha^{-1} of corn stover in BB and up to 13 Mg ha^{-1} of corn stover and wheat straw in NK, had no effect on total nitrogen (TN) and carbon (TC), CN ratios, bulk density (BD), soil pH, field capacity, permanent wilting point, plant available water and water aggregate stability across soil depths and aggregate sizes in Virginia. Residue retention did not affect grain, crop residue yields or nutrient uptake in either system in this short-term study. Feedstock quality was largely unaffected by treatment, although total retention rates around 70% maximized sulfur concentration of the 0-10-cm wheat straw in NK. Theoretical ethanol potential (TEP) and theoretical ethanol yield were not affected by residue rates in NK1 or NK2. At BB, over a 2 yr period, TEP of the 0-30-cm stover portion was minimized with retention rates around 30%. Neither residue harvest (nor residue application) had negative effects on short-term biomass yield in grain-based cropping systems in the Coastal Plain or Ridge/Valley regions of Virginia. This study is one of the first to assess residue removal in the Mid-Atlantic USA and is the first study to investigate the impacts that managing more than one crop residue in a multi-crop system. Longer-term research of this type may be warranted both to determine the consequences of residue management and to start building a regionally-specific body of knowledge about these practices.

Crop residue management effects on crop production, greenhouse gases emissions, and soil quality in the Mid-Atlantic USA

Martín Leonardo Battaglia

GENERAL AUDIENCE ABSTRACT

Over the last decade, strategic economic and environmental concerns have increased interest in the use of crop residues as sustainable, renewable sources for bioenergy and bio-products. Most of the work investigating the sustainability of residue removal has occurred in the US Corn Belt, where corn stover and wheat straw (the part of the plant that is not grain) supplies are abundant. Although the research data from the Corn Belt provide guarded optimism about residue harvest systems in the Midwest, it is not suitable to extrapolate these results to the South because of differences in soils, climate, and cropping systems. Cooler, humid conditions can sustain higher levels of soil organic matter, lessening but not eliminating concerns about stover removal. Current research from the Midwest region suggests routine stover harvest – within limits – can be sustainable. The development of new bioenergy and bioproduct industries in the Southeast region is leading to a growing expectation that regional cropping systems will supply the millions of tons of biomass needed for these new businesses. However, few data are available regarding sustainable crop residue harvest from the Southeast. Sustainable levels of residue removal may be quite low given regional soil and climatic conditions, and the effects of residue removal on soil health parameters and greenhouse gas emissions remain to be defined. The purpose of this project was to determine the amount of corn stover and wheat straw can sustainably be harvested from Virginia's grain-based cropping systems without reducing plant productivity or soil quality or increasing GHG emissions. This research

generated regionally relevant information on the impacts of crop residue removal to help determine whether harvesting wheat straw and corn stover can be a sustainable practice for the region's cropping systems. In a first stage, short term impacts of residue removal on soil quality and greenhouse gases were measured in Blacksburg and New Kent, VA, over the period 2015-2017.

DEDICATION

To my parents, because their hard work made possible what I become today. To my beloved kids, Valentino and Paris, and my wife, the reasons behind all reasons in my life. I wish this humble work can be a source of inspiration to you sometime in your future lives.

ACKNOWLEDGMENTS

To the committee members in my dissertation, Dr. Gordon Groover and Dr. Gregory Evanylo, and my two co-advisors in this fantastic journey of learning and life, Dr. Wade Thomason and Dr. John Fike for their constant insights to improve my professional work. Your time, patience, wisdom and friendship has been an invaluable gift I will always celebrate and treasure with all my heart for the rest of my life.

TABLE OF CONTENTS

CHAPTER 1

Overall Introduction

1. STATEMENT OF PROBLEM, RATIONALE AND JUSTIFICATION

The purpose of this project is to determine the amount of corn stover and wheat straw can sustainably be harvested from Virginia's grain-based cropping systems without reducing plant productivity or soil quality or increasing GHG emissions. Use of crop residues for bioenergy has grown over the last decade as a result of government policies to create alternative energy sources and secure US fuel supplies (EISA-EPA, 2007), but there is concern about the ability to sustainably harvest these materials. Most of the work investigating the sustainability of residue removal has occurred in the US Corn Belt, given the large available supply of biomass. However, such results are not readily extrapolated to the Southeast due to the differences in soil resources and climate.

Data from studies in the Midwest suggest that 25 and 50% of the available biomass in the Corn Belt can be sustainably harvested (Morachan et al., 1972; Glassner et al., 1998; Nelson, 2002; Blanco-Canqui et al., 2006b; Graham et al., 2007; Blanco-Canqui and Lal, 2009; Khanna and Paulson, 2016). Currently more than 90% of the corn stover produced in the United States remains in the field, uncollected (Glassner et al., 1998 cited by Kim and Dale, 2004). However, the harvestable percentage varies widely based on factors such as soil type, soil organic matter (SOM), and topography, and any residue removal system must guard against long term losses in soil fertility and grain yield stability (Blanco-Canqui et al., 2006a; Blanco-Canqui and Lal, 2009; Sindelar, 2012). Although the research data generated in the US Corn Belt area indicates that residue harvest can be sustainably harvested within certain limits in the Midwest, these results cannot be used as a decision-making tool for farmers and prospective companies in the Southeast region of the US due to differences in soils, climate, and cropping systems between regions. Only one study has specifically considered the potential supply of residues in the Southeast (Gonzalez et al., 2011). However, that research focused on potential supply and the supply chain and provided no specific information to guide producers and industry about sustainable harvest levels for the region.

The need to fill this information gap is particularly pressing given the development of new industries in the region. Generating information on crop residue removal's environmental impacts will be an important first step in determining the sustainability of these practices for the region's cropping systems. While development of a cellulosic biofuel industry in Virginia is still in the feasibility stage, our research will help determine the effects of residue management on soil quality parameters such as nutrient balance, bulk density, aggregate stability, carbon and nitrogen pools and gas fluxes, and water holding capacity and plant available water. This research will provide an initial basis for understanding best practices for crop residue management in terms of soil quality and nutrient stewardship and the overall carbon foot-print in the South's typical grain-based cropping systems.

2. PROJECT RELEVANCE TO SUSTAINABLE AGRICULTURE

Residue harvest from annual cropping systems has the capacity to both increase and diversify farm income, and, depending on the end products may reduce fossil fuel use and greenhouse gas fluxes to the atmosphere (Wilhelm et al., 2004). Crop residues from corn, wheat and other grains are to some extent "subsidized" by the primary (grain) production system. As well, their harvest and use in bio-products systems reduces potential land conversion that would be required for dedicated biomass production systems. If biomass is converted to fuels such as ethanol from corn stover, this has the potential to greatly reduce life-cycle greenhouse gas emissions (90 to 103%) relative to petroleum gasoline (Wang et al., 2012). Residue harvest may also impact emissions at the field level. Corn stover removal has had negligible effects on soil N_2O emissions in some studies (e.g., Baker et al., 2014), while reducing emission up to 7% with moderate removal rates in others (Jin et al., 2014). In both of these studies, however, moderate levels of stover harvest reduced CO_2 fluxes with intermediate corn stover removal.

Development of biomass to bio-products industries has potential to stimulate farm economies and rural communities (Wilhelm et al., 2004) by providing extra income for additional farm products that currently are oxidized and lost from the system. In some cases, removing residues from the field can help to reduce insects and diseases (Wilhelm et al., 2004) and may result in shorter emergence intervals for corn (Wilhelm et al., 1986; Vetsch and Randall, 2002). Finally, co-products derived from bio-products and biofuels synthesis can be used to manufacture fertilizers and soil amendments that support soil fertility and improve SOM levels (Johnson et al., 2004; Wilhelm et al., 2004). Although these benefits have been observed in Midwestern stover harvest systems, the challenge is to determine the level of sustainable residue removal for Southeastern cropping systems.

3. STRUCTURE OF THIS DISSERTATION

Chapter 1 briefly states the purpose of this project which is to determine how much corn stover and wheat straw can sustainably be harvested from Virginia's grain-based cropping systems without reducing plant productivity or soil quality or increasing GHG emissions. Following, the justification of this research is presented in the light of the a) increased interest in the use of crop residues for extended uses as a result of government policies to create alternative energy sources and secure US fuel supplies, b) the lack of related information generated in the Mid-Atlantic USA at the regional scale, and c) the complete lack of information at the national and global scale on the impacts that managing more than one crop residue in a multi-crop system may have in our agroecosystems. Finally, the project relevance to sustainable agriculture is briefly discussed by providing with some examples of reported benefits calculated from simulation models and measured in field studies conducted in the US in the last 35 years.

Chapter 2 presents an extensive literature review regarding the impacts that corn stover and wheat straw removal or retention had on chemical and physical parameters of soil quality, greenhouse gases emissions, and plant productivity in field studies conducted in the US and abroad over the last 50 years.

Field studies are presented regarding corn stover and wheat straw residue management effects on greenhouse gases emissions (Chapter 3), soil quality (Chapter 4) and plant productivity (Chapter 5). Those field experiments were conducted over the period 2015-2017 in the Coastal Plain and Ridge/Valley physiographic regions.

4. LITERATURE CITED

Baker, J.M., J. Fassbinder, and J.A. Lamb. 2014. The impact of corn stover removal on N2O emission and soil respiration: an investigation with automated chambers. BioEnergy Res. 7: 503-508.

Blanco-Canqui, H., and R. Lal. 2009. Corn stover removal for expanded uses reduces soil fertility and structural stability. Soil Sci. Soc. Am. J. 73: 418-426.

Blanco-Canqui, H., R. Lal, R.C. Izaurralde, and L.B. Owens. 2006a. Rapid changes in soil carbon and structural properties due to stover removal from no-till corn plots. Soil Sc. 171: 468-482.

Blanco-Canqui, H., R. Lal, W.M. Post, and L.B. Owens. 2006b. Changes in long-term no-till corn growth and yield under different rates of stover mulch. Agron J. 98:1128-1136.

EISA-EPA. 2007. Summary of the Energy Independence and Security Act. Public Law 110-140. US Environmental Protection Agency. At: https://www.epa.gov/laws-regulations/summary-energy-independence-and-security-act (last updated: 8/2/2016; accessed: 9/26/2018). US Environmental Protection Agency.

Glassner, D., J. Hettenhaus, and T. Schechinger. 1998. Corn stover collection project: a pilot for establishing infrastructure for agricultural residue and other crop collection for biomass processing to ethanol. *In:* Proc. Bioenergy 1998 Conference, 4–8 October, 1998, Madison, WI, pp. 1100–1110.

Gonzalez, R., R. Phillips, D. Saloni, H. Jameel, R. Abt, A. Pirraglia, and J. Wright. 2011. Biomass to energy in the southern United States: supply chain and delivered cost. Bioresources 6: 2954-2976.

Graham, R.L., R. Nelson, J. Sheenan, R.D. Perlack, and L.L. Wright. 2007. Current and potential U.S. corn stover supplies. Agron. J. 99: 1-11.

Jin, V.L., J.M. Baker, J.M.F. Johnson, D.L. Karlen, R.M. Lehman, S.L. Osborne, T.J. Sauer, D.E. Stott, G.E. Varvel, R.T. Venterea, M.R. Schmer, and B.J. Wienhold. 2014. Soil greenhouse gas emissions in response to corn stover removal and tillage management across the US Corn Belt. BioEnergy Res. 7: 517-527.

Johnson, J.M.F., D.C. Reicosky, B. Sharratt, M.J. Lindstrom, W.B. 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.

Khanna, M., and N. Paulson. 2016. To harvest stover or not: Is it worth it? Department of Agricultural and Consumer Economics. University of Illinois, Urbana-Champaign. Farmdoc daily 6:32.

Kim, S., and B.E. Dale. 2004. Global potential bioethanol production from wasted crops and crop residues. Biomass and Bioenergy 26: 361-375.

Morachan, Y.B., W.C. Moldenhauer, and W.E. Larson. 1972. Effects of increasing amounts of organic residues on continuous corn: I. Yields and soil physical properties. Agron. J. 64:199-203.

Nelson, R.G. 2002. Resource assessment and removal analysis for corn stover and wheat straw in the eastern and Midwestern United States: rainfall and wind-induced soil erosion methodology. Biomass Bioenergy 22: 349-363.

Sindelar, A.J. 2012. Stover, tillage, and Nitrogen management in continuous corn for grain, ethanol, and soil carbon. Ph.D. diss., University of Minnesota, Minneapolis, MN.

Vetsch, J.A., and G.W. Randall. 2002. Corn production as affected by tillage system and starter fertilizer. Agron. J. 94:532–540.

Wang, M., J. Han, J.B. Dunn, H. Cai and A. Elgowainy. 2012. Well-to-wheels energy use and greenhouse gas emissions of ethanol from corn, sugarcane and cellulosic biomass for US use. IOP Publishing. Environ. Res. Lett. 7 (2012) 045905: 13pp. At: stacks.iop.org/ERL/7/045905 (accessed: 9/26/2018).

Wilhelm, W.W., J.W. Doran, and J.F. Power. 1986. Corn and soybean yield response to crop residue management under no-tillage production systems. Agron. J. 78:184-189.

Wilhelm, W.W., J.M. Johnson, J.L. Hatfield, W.B. Voorhees, and D.R. Linden. 2004. Crop and soil productivity response to corn residue removal: a literature review. Agron. J. 96: 1-17.

CHAPTER 2

Impacts of corn stover and wheat straw removal on crop performance and productivity, soil health and greenhouse gases emissions: a literature review

1. ABSTRACT

Crop residue harvest for biofuel production may provide society with environmental and economic benefits. However, to sustainably utilize these materials will require better understanding of residue harvest's impacts on agricultural productivity and soil health. Numerous studies have investigated residue removal effects on crop production, soil quality and greenhouse gas emissions, but none have evaluated these parameters together. Changes in soil organic fractions and nutrients depended largely on the amount of residue returned, soil depth and texture, slope and tillage. Reductions in organic fractions occurred primarily with complete stover removal, in the top 15 to 30-cm in fine-textured-soils. Extractable soil potassium (K^+) and total nitrogen (N) were more sensitive to stover removal than other nutrient. Soil loss, and water and nutrient runoff declined as residue levels increased up to around a 70% retention threshold. Amounts of surface residues were directly related with soil available water and inversely related to soil temperature. Corn (*Zea mays* L.) stover removal at rates of ≥50% decreased soil aggregate stability in half of the studies reviewed. Stover management effects on soil bulk density varied widely, with increases and decreases occurring by soil layer and residue and tillage management. Reductions in CO_2 and N_2O fluxes typically occurred following complete residue removal. Applying urea to soils with high levels (9.5 Mg ha^{-1}) of low-quality feedstocks reduced N_2O emissions in coarse but increased it in fine-textured soils compared with fertilizer alone. Adding more than 8 Mg ha^{-1} of wheat straw

resulted in greatest CO_2 and N_2O emissions, regardless of N rates. Applying wheat straw typically increased CH_4 emissions. In years when water was not limiting in rainfed or irrigated studies, corn and wheat removal rates ≥90% produced similar or greater grain yield than no removal in most studies. Conversely, when water was limiting, corn grain yield decreased up to 21% with stover removal ≥ 90% in some studies. Regardless of available water, most studies found little evidence that residue management had long-term impacts on grain yield. Longer-term research would be needed to conclusively discern the environmental and production impacts of residue harvest for different climates, soils and management practices.

2. INTRODUCTION

Over the last decade, strategic, economic and environmental concerns have spurred increased interest in the use of crop residues as a renewable source of energy in the US and globally. Although many crops are good candidates for biofuels, corn and wheat have received the greatest attention because of low cost, immediate availability, and relatively concentrated areas of production in the main agricultural regions (U.S. Department of Energy, 2011). In this scenario, corn stover is expected to play a central role in the national goal to decrease foreign dependency on fossil fuels (Sindelar, 2012). Modelling efforts suggest total US corn stover produced could supply between 90 and 196 million Mg biomass yr^{-1} (Walsh et al., 2000; Gallagher et al., 2003; Graham et al., 2007) and straw produced from wheat and barley around 71 million Mg yr^{-1} (Tarkalson et al., 2009). The 2011 U.S. Billion-Ton Update (U.S. Department of Energy, 2011) estimated a baseline scenario of total corn stover supply between 98 and 109 million Mg yr^{-1} in year 2022, and 117 and 127 million Mg yr^{-1} in 2030 with corn stover prices between \$55 and 65 Mg^{-1} of dry matter. With the same prices, but in a high-yield scenario that assumes a 1% annual growth in crop yields, corn stover supplies are estimated to be between 207 and 246 million Mg yr^{-1} for the period 2022-2030. Assessments at a state level have also been conducted for corn in US Corn Belt states including Iowa (Sheehan et al., 2004) and Minnesota (Sheaffer et al., 2010).

Second generation biofuels from cellulosic feedstocks may help to address concerns about the use of grain crops for fuel. This potential to lose food grains to fuel production has been characterized as the "food vs. fuels" scenario (Tenenbaum, 2008; Thompson, 2012). Using residues from annual cropping systems may both increase and diversify farm income, and, depending on the end products could reduce fossil fuel use and greenhouse gas fluxes to the atmosphere (Wilhelm et al., 2004). Because the grain enterprise to some extent "subsidizes" stover and straw, crop residue production costs are limited. As well, harvesting the residues in addition to their grains reduces need for land conversion that might be associated with dedicated energy crop production. Converting biomass to fuels such as ethanol or butanol has the potential to greatly (90 to 103%) reduce life-cycle greenhouse gas emissions relative to petroleum gasoline (Wang et al., 2012). At the field level, complete corn stover removal (compared with no stover removal) reduced CO_2 and N_2O, two major greenhouse gases and air pollutants, up to 4% and 7% respectively (Baker et al., 2014; Jin et al., 2014). Moreover, co-products of bio-products and biofuels synthesis can be used to develop organic fertilizers and soil amendments (Johnson et al., 2004; Wilhelm et al., 2004). Residue removal may also reduce insect and disease pressures (Wilhelm et al., 2004).

Although residue harvest may provide several benefits, concerns have arisen about potential negative impacts because crop residues act as both sinks and sources of soil carbon that provide important ecosystem services to humankind and the environment. Crop residues contribute to agricultural productivity by reducing soil erosion and enhancing soil structure and other physical properties (Wilhelm et al., 2007; Raffa et al., 2014) through their positive effects on soil organic carbon (SOC) stocks, nutrient availability, bulk density, water holding capacity, and water infiltration (Barber, 1979; Magdoff et al., 1997; Franzluebbers, 2002; Anderson-Teixera et al., 2009; Blanco-Canqui and Lal, 2009; Kenney, 2011; Sindelar, 2012) among others.

Historically, farmers in the US either have left all corn residues in the field or have collected part or the whole plant for animal feed or bedding. Any early estimate of available U.S. stover supplies indicated that more than 90% of these residues were left in the field (Glassner et al., 1998). While the increased use of crop residues for fuel and products is changing these historical trends, it has also created concerns about their sustainable harvest. Most of the work investigating the sustainability of residue removal has occurred in the US Corn Belt. Those studies suggest that between 25 and 60% of the available biomass in the Corn Belt can be sustainably harvested (Blanco-Canqui et al., 2006b; Glassner et al., 1998; Graham et al., 2007; Khanna and Paulson, 2016; Nelson, 2002). This wide range in harvestable percentage is based on factors such as soil type, SOM, and topography and any residue removal system must guard against long-term losses in soil fertility and stability (Blanco-Canqui et al., 2006a; Blanco-Canqui and Lal, 2009; Sindelar, 2012).

Numerous studies have considered the impacts of corn stover and wheat straw management on grain yield, greenhouse gas emissions, soil C and N pools, nutrient balances, and soil physical parameters such as aggregate stability, bulk density, and available water. However, none have evaluated these parameters together. The objective of this review is to summarize the current knowledge about the effects of corn stover and wheat straw management on: a) SOM, SOC, and particulate organic matter (POM); b) nutrient balances, soil pH, and cation exchange capacity (CEC); c) soil erosion and water quality; d) available water, aggregate stability and bulk density;

e) greenhouse gas emissions; corn and wheat grain yield; f) other potential impacts, over the last 50 years.

3. CROP RESIDUE MANAGEMENT EFFECTS ON SOIL QUALITY PARAMETERS

Among environmental considerations for biomass harvest systems, whether based on annual or perennial species), one of the most important is the effect on soil quality parameters such as SOM and SOC, nutrient balances, soil pH, aggregate stability and water holding capacity.

3.1. CROP RESIDUE MANAGEMENT EFFECTS ON SOIL ORGANIC MATTER, SOIL ORGANIC CARBON AND PARTICULATE ORGANIC MATTER

Soil organic matter and SOC are the chemical indicators most widely studied in experiments to determine the effects of crop residue removal. Both SOM and SOC are highly influenced by the amount of crop residue returned to the system (Karlen et al., 1994; Huggins et al., 1998; Benjamin et al., 2008; Kendall et al., 2015). Barber (1979) studied the effect of returning 0, 100 and 200% corn stover, and fallow for 6 yr followed by 5 yr of 100% corn stover return in continuous corn for 11 yr on a silt loam soil in Indiana. In the 0 to 15-cm depth, SOM was greatest (about 3.4%) with the 200% stover return after the sixth and eleventh yr. Returning 0% and 100% corn stover decreased SOM in the 0 to 15-cm soil depth by an average of 18% and 11% when compared with 200% return. Differences between the 0% and 100% return rate were only apparent at the end of the eleventh year, when no stover return decreased SOM by 10% relative to the 100% return rate. These results likely also reflect the contribution of the crop root system in SOC maintenance in the previous years.

Response to stover removal often varies by depth and tillage system. Removing all corn stover reduced SOC between 21% and 34% through a soil profile (measured in 20-cm increments to 60-cm depth) in a silty clay loam under long -term no-till management. Removal did not reduce SOC under conventional tillage at any depth, but SOC values for complete stover retention treatment under conventional tillage were 15% to 35% lower than similar treatments under no-till. Highest SOC values for complete stover retention under no-till across all comparisons likely explains the differences between stover management treatments in no-till, a situation that did not occur under conventional tillage (Dendooven et al., 2012). Regardless of tillage system, SOC decreased in surface layers when no stover was returned (14% and 4% in the 0 to 5- and 5 to 15-cm depths, respectively), but did not differ in deeper layers (15 to 30- and 30 to 60-cm) of a clay loam soil complex in Minnesota (Sindelar, 2012). In a silt loam soil with 10% slope in Ohio, SOC decreased up to 27% in the 10 to 20-cm depth when removal rates were ≥75%. In silt loam and a clay loam soils with limited (less than 2%) slopes, different removal rates did not affect SOC in the 10 to 20-cm depth. In all three soils, removal rates ≥75% decreased SOC between 20 and 30% in the surface (top 10 cm) layer (Blanco-Canqui and Lal, 2009).

Blanco-Canqui et al. (2006a) studied continuous no-till on silt loam clay-loam soils in Ohio. They observed that removal rates as low as 1.25 Mg ha^{-1} reduced SOC and degraded soil structure after just one year in coarse-textured soils but had little to no impact in fine-textured soil. Liang et al. (1998) found that retention of residue C was 81% to 175% higher and its turnover slower, in clay than in coarse-textured soils in a 12 yr experiment with continuous corn in Canada. Although this may partially explain the lack of short-term changes in the SOC levels in clay loam soils observed by Blanco-Canqui et al. (2006a) and Johnson et al. (2013), it conflicts with the changes reported by Sindelar (2012) for similar soils. According to Blanco-Canqui et al. (2006a), soils where short-term changes in the SOC levels are less prone to occur may have reached an equilibrium state that buffers them against changes when stover is added or removed (Blanco-Canqui et al., 2006a). Baseline SOC levels for the 0 to 5-cm depth were much smaller (i.e., 2.5 g kg^{-1} soil) in the experiment conducted by Sindelar (2012) compared with values measured by the other two authors (i.e., 20-30 g kg^{-1} soil).

The POM fraction is composed of fine plant and microbial residues in the early stages of humification (Bernard et al., 1996; Carter, 2002). This fraction is highly responsive to changes in C inputs (Gregorich and Janzen, 1996) and management (Cambardella and Elliot, 1992; Sequeira and Alley, 2011). The POM fraction has an estimated turnover time between one and eight yr (Carter, 2000) and can represent up to 45% of the active SOM pool (Carter et al., 1998).

Johnson et al. (2013) investigated the response of POM and SOC to corn stover return rates (full, ~ 7.8 Mg ha^{-1} yr^{-1}; moderate, ~ 3.8 Mg ha^{-1} yr^{-1}; and low ~ 1.5 Mg ha^{-1} yr^{-1}) and tillage [(chisel-tilled; well-established (10 yr old); and newly-established (0 yr old) no-till)] at two depths (0-5 and 5-10-cm) in a clay loam soil in Minnesota. For most comparisons, POM did not differ with stover return rate, regardless of tillage system (Johnson et al. 2013). Among comparisons, POM was decreased only at the 0 to 5-cm in long-term no-till plots with low stover returns (i.e., 1 comparison out of 18). Johnson et al. (2013) concluded that changes in SOC and POM levels in response to stover management may take longer than 3 years to be detected in clay loam soils and that POM levels in long-term no-till plots can be reduced in three or fewer cycles of stover treatments if only low levels of stover are returned.

In contrast, Sindelar (2012) was able to measure short-term changes in SOM and POM in response to residue removal in a clay loam soil complex in Minnesota. In their study, POM was more responsive than SOC to stover management, and changes were observed to depths of 30 cm.

Sindelar (2012) concluded that stover removal in a continuous corn rotation can negatively affect SOM and POM within 3 years in fine-textured soils.

Others also have reported decreased SOM and SOC with corn residue removal (Dolan et al., 2006; Moebious-Clune et al., 2008; Kim et al. 2009; Kenney, 2011; Baker et al., 2014). Retention of crop residues, on the other hand, have resulted in positive (Clapp et al., 2000) to little or no effect (Johnson and Chamber, 1996; Nicholson et al., 1997) on these parameters. Perhaps the smaller carbon contribution of surface (compared with root) residue to POM and SOC pools renders less evident the occurrence of changes with varying rates of surface stover removal. In a simulated no-till experiment, Gale and Cambardella (1998) demonstrated that 66% of the ^{14}C in surface residue had been respired as CO_2 after 360 d of decomposition, while 11% remained as surface residue and only 16% was in the soil. In contrast, 56% of the root-derived ^{14}C in the soil evolved as CO_2, and 42% remained in the soil after the incubation period. Large (500–2000 μm) and small (53–500 μm) POM fractions contained 11 to 16% of the initial root-derived ^{14}C, but less than 3% of the initial surface residue-derived ^{14}C. These trends agree with findings from Larson et al. (1972) in Iowa and Barber (1979) in Indiana, who reported greater root (23% and 18%) than aboveground C (18% and 8-11%) entering the SOM pool.

Negative impacts of wheat straw removal on SOC and SOM have been less evident. Working in a 14-year study in a silty clay loam soil and furrow irrigation in Texas, Undersander and Reiger (1985) did not find a difference in SOM in the 0 to 15- and 15 to 30-cm depths when comparing nil and full wheat straw removal rates. Regardless of wheat straw management, Bordovsky et al. (1999) reported SOC increases in the 0 to 7.5-cm depth over an 11-year study with continuous irrigated wheat in a fine sandy loam soil in Texas. However, these increases occurred more rapidly when the straw was not removed from the soil surface. In Iran, Bahrani et al. (2002) did not observe a reduction in the SOC in the top 30-cm of soil after removing all straw from the soil surface. Curtin and Fraser (2003) reported no differences in SOC in the 0 to 15-cm soil depth in a silt loam soil in New Zealand for treatments that included incorporation, burning or removal of aboveground wheat straw.

Although no field studies on the impact of residue removal on soil quality parameters has been conducted in Virginia, some authors studied the impact of other practices in these parameters. Working in the Coastal Plain region of Virginia with sandy loam soils under continuous no-till, Spargo et al. (2012) found that total soil organic C and N increased linearly with time under continuous no-till in the 0-2.5 and 2.5-7.5 cm depths. No changes in C and N pools were found for the 7.5-15 cm depth. Similarly, linear increases in POM-C and POM-N fractions were reported with time under continuous no-till for the 0-2.5 depth, with no change for the 2.5-7.5 and 7.5-15 cm depth increments. Sequeira and Alley (2011), working in the Valley/Ridge province of Virginia studied the short-term effects of different management practices on different soil C POM-and N pools in the top 15-cm soil . Factors were arranged as a split-split plot design with crop rotation as a main plot [3 levels: continuous corn silage; corn silage in the first year, alfalfa (*Medicago sativa* L.) in the second year, and alfalfa in the third year; or corn grain in the first year, soybean in the second year, and corn silage in the third year, tillage as sub-plot [2 levels: reduced tillage (RT) or no-till (NT)] and cover crop management as sub-subplots [3 levels: rye (*Secale cereale* L.) cover crop remaining on the field after being killed at the early-boot stage, rye cover crop harvested from the field at the early-boot stage, or no cover crop planted]. Soil organic N fractions, both in the bulk soil and POM fraction, were not affected by any combination of factors under analysis, whereas organic C in both pools were only affected by cover crop management. Overall, both soil organic pools had significantly more C when the rye cover crop was left on the field when compared with harvesting after chemical killing. Bulk SOC was similar for the RK and NC treatments, while POM-C values were higher for the first treatment.

3.2. CROP RESIDUE MANAGEMENT EFFECTS ON NUTRIENT BALANCES, pH and CATION EXCHANGE CAPACITY

An extensive body of literature is available on the effect of SOM on chemical soil parameters. Blanco-Canqui and Lal (2009) found that only complete stover removal decreased the amount of total N in the soil, but this response depended largely on the soil texture considered. Increasing the amount of returned residue may increase N immobilization which may require additional application of fertilizer N (Power and Doran, 1988). With complete residue removal, total N was negatively affected in the silt loam but not the clay loam soil, and trends were more noticeable in the 0 to 10-cm than in the 10 to 20-cm depth. Similarly, Karlen et al. (1994) found lowest nitrate values in the 0 to 2.5 and 2.5 to 7.5-cm depths in a silt loam soil in Wisconsin following complete corn stover removal compared with 100% and 200% stove retention rates.

Available soil P appears much less affected by crop residue management, irrespective of soil texture. Blanco-Canqui and Lal (2009) reported large reductions (40%) in soil P only with 100% stover removal in the surface 10 cm of a silt loam soil (Blanco-Canqui and Lal, 2009). Available P at different soil depths was not affected by residue management in studies conducted in a sandy loam soil in South Carolina (Karlen et al., 1984) and a silt loam soil in Wisconsin (Karlen et al., 1994).

Soil K, on the other hand, responded with greater overall variability than available P to stover management and regardless of soil texture. At 0 to 10-cm depth, stover removal rates of 75 and 100% reduced extractable K+ in silt loams with 2% slope or 10% slopes and a clay loam with

<1% slope (Blanco-Canqui and Lal, 2009). Similar findings were reported by Morachan et al. (1972), who studied stover return rates in a silty clay loam soil in Iowa. They observed 16% and a 53% reduction in soil extractable K+ when no stover was returned, when compared with return rates of 4 and 16 Mg^{-1} over an 11-year period, respectively. Karlen et al. (1984) found that harvesting 66% and 90% of the corn residues for 2 yr significantly reduced the soil extractable K levels in the 5 to 20-cm depth increment of an Ap horizon of a sandy loam soil in South Carolina, but these removal rates had no effect on K in the Ap (0 to 5 cm), E (20 to 40 cm) or Bt (40 to 90 cm) horizons. In this study, soil extractable Ca, Mg and Mn levels were unresponsive to different stover removal rates in most comparisons (Karlen et al., 1984). In another study, Ca^{+2} and Mg^{+2} and CEC only decreased with 100% stover removal rates on sloping soils (10%) in the surface (0 to 10 cm) horizon (Blanco-Canqui and Lal, 2009). With CEC, decreases of up to 10% were observed in one silt loam soil. In most comparisons, soil pH did not change with different rates of stover removal at any soil depth, similar to reports from Karlen et al. (1984). Soil pH only increased in two individual comparisons for complete stover removal, one at each soil depth (Blanco-Canqui and Lal, 2009). Similarly, Morachan et al. (1972) working in Iowa with five corn stover return rates (0, 2, 4, 8 and 16 Mg ha^{-1}), reported a significantly lower pH of 4.8 for the 16 Mg ha^{-1} stover application treatment versus pH 5.3 when no stover was returned. The authors suggested that the low pH resulting from high rates of stover retention may have caused an Al^{+3}-induced Ca^{+2} deficiency in the plant.

In summary, parameters such as soil Ca^{+2}, Mg^{+2}, available PO_4^{3-}, NO_3^-, and CEC seem less prone to change than extractable K+ and total soil N with stover management. The greatest differences typically occur with stover removal rates close to 100% and at the 0 to 10-cm depths.

However, these responses can also be highly dependent upon slope, soil texture and depth, as previously discussed.

3.3. CROP RESIDUE MANAGEMENT EFFECTS ON SOIL EROSION and WATER QUALITY

Protection against potential soil erosion, with its concomitant effects on soil and water quality, is one of the major worldwide concerns related with the extraction of crop residues for alternative uses (McAloon et al., 2000; Mann et al., 2002; Nelson, 2002; Wilhem et al., 2004; Andrews, 2006), but few experiments have studied these parameters together under different residue management schemes.

Lindstrom (1986) conducted experiments in a loam soil in Minnesota and a silty clay loam soil in South Dakota to determine the relationship between water runoff and soil loss as a result of changes in the amount of corn residue remaining on surface. Residue levels were calculated with the USLE (Universal Soil Loss Equation) model based on estimated retained residue needed (Y; 2240 kg ha^{-1} for the loam and 1680 kg ha^{-1} for the silty clay loam soil) to control soil loss at the soil loss tolerance (T). Residue was harvested to establish levels of 0.5 Y, Y and 2Y at each location (i.e., 1120, 2240 and 4480 kg ha^{-1}; 840, 1680, and 3360 kg ha^{-1} for the loam and silty clay loam soils, respectively). Lindstrom (1986) found decreasing water runoff and soil loss with increasing amounts of residue left on the soil surface up to approximately 70% retention rates (i.e., ~ 30% removal). Retention rates above 70% had no further reductions in runoff or soil loss. Moreover, the rainfall energy needed to start the runoff process was higher when residue was left on surface (Lindstrom et al., 1984). Residue management can also impact the quality of the water runoff. Grande et al. (2005) found that total P and dissolved reactive P in the runoff were inversely related to the amount of residue remaining on surface, although their concentrations were unaffected by retained residue level.

3.4. CROP RESIDUE MANAGEMENT EFFECTS ON AVAILABLE WATER, AGGREGATE STABILITY AND BULK DENSITY

In a 4-year study in a silty clay loam soil in Nebraska with continuous corn and 0, 50, 100 or 150% stover return, Wilhelm et al. (1986) found that returning all stover increased soil available water at planting (i.e., water stored between -0.03 and -1.50 MPa in the depth increment 0-1.8 m) by 25% and 13% when compared with 0 and 50% return rates. Moreover, an increase of 6 mm in the soil available water around planting was calculated for each extra Mg ha^{-1} of residue returned. Increased corn stover return rates significantly reduced soil temperatures (measured at 50-mm depth throughout the crop cycle), with positive effects on soil water conservation, similar to reports from Power et al. (1986) in Nebraska and Blanco-Canqui et al. (2006b) in Ohio. When added to the model, soil temperature and available water accounted for 80 and 90% of the total variation in corn grain and residue yields, respectively. Corn stover returned to the system can also have positive impacts on soil available water later in the crop growth cycle. Doran et al. (1984) measured increases in soil available water greater than 100% around the critical period of yield determination in corn when 100% of the stover was retained relative to 0% retention in a 3-year study in Nebraska.

The interrelationship between SOM formation, stabilization and turnover with biological activity and aggregate dynamics have been studied since the early 1900s (Six et al., 2004). Soil aggregate stability is an important indicator of the cohesive forces maintaining the soil particles together against the effect of disruptive forces like water, wind and mechanical management

(Amézketa, 1999; Six et al., 2004). Stover removal rates ≥ 50% reduced water aggregate-stability in some studies (Bordovsky et al., 1999; Blanco-Canqui and Lal, 2009), but had little impact in others (Karlen et al., 1994; Hammerbeck et.al., 2012). In the long-term study conducted by Bordovsky et at. (1999) in Texas, microaggregation values were 15% and 19% higher when residue was retained, both in non-irrigated (27.1 vs. 23.5 g kg^{-1}) and under irrigated conditions (32.3 vs. 27.1 g kg^{-1}). Karlen et al. (1994) found no difference between the percentage of water stable aggregates from treatments that retained 0% and 100% corn stover in the previous 10 yr in Wisconsin. Conversely, treatments that retained 200% stover rates in a 10 yr period increased the amount of water stable aggregates by an average of 38% compared with the 0 and 100% retention. Working in a corn-soybean rotation in a silty clay loam soil in South Dakota, Hammerbeck et.al. (2012) reported a 40% increase in the water aggregate stability for aggregate sizes between 0.84 and 2.0 mm when no residue removal compared with stover removal rates >4.0 Mg ha^{-1}. However, stover management did not affect water aggregate stability for other aggregate sizes (i.e., 2.0-6.4, 6.4-19.2, and >19.2 mm) in this study.

Evidence for the impact of different corn stover management on soil bulk density (BD) is conflicting. Working in a silt loam soil in Minnesota, Clapp et al. (2000) found decreases in BD at the end of the thirteenth yr in the 0 to 5-cm depth with 100% corn stover retained in no-till, but not in moldboard or chisel-plow tillage systems. However, 100% stover retention increased the BD for the 20 to 40-cm depth for all tillage system. Similar results were found at the end of the twenty-second yr, when BD had decreased by 6% at the 0 to 5 and 5 to 10-cm depths but increased by 5% in the 30 to 45-cm depth when 100% of the stover was retained (Dolan et al., 2006). These results agree with the findings from Sindelar (2012), where 100% stover retention reduced BD by 0.26 and 0.14 g cm^{-3} in the 0 to 5 and 5 to 15-cm depths. Conversely, 0% and 100% stover retention did not change the BD in any year in the first 20-cm in Québec (Dam et al., 2005) or 50-cm of soil in Iowa (Karlen et al., 1994).

4. CROP RESIDUE MANAGEMENT EFFECTS ON GREENHOUSE GAS EMISSIONS

One of the main goals of proposed bioenergy production systems is the mitigation of the projected changes of the climate at a global scale (Baker et al., 2014). However, crop residue removal can have detrimental consequences through its effects on soil processes that may increase the production of greenhouse gases, especially N_2O (Carter et al., 2002; Baker et al., 2014). Furthermore, some authors have recently stated that the release of N_2O resulting from biofuel production may counterbalance the reduction in global warming resulting from fossil fuel displacement (Carter et al., 2002; Crutzen et al., 2008).

4.1. CO_2, CH_4 AND N_2O FLUXES UNDER DIFFERENT CORN RESIDUE MANAGEMENT

Carbon dioxide (CO_2) from combustion of fossil fuels is by far the largest contributor to global warming and is expected to continue so in the future (Houghton, 2007). Produced in considerable less amount than CO_2, nitrous oxide (NO_2) is a long-lived GHG and a major contributor to climate change (Gentile et al., 2008). Nitrous oxide has a global warming power (GWP) between 265 and 310 times that of CO_2 over 100-yr (GGWG, 2010; EPA, 2017). Methane (CH_4), has a shorter lifetime than CO_2 but an estimated GWP 28-36 greater than that of CO_2 over 100 yr (EPA, 2017).

As part of a USDA multi-location research project, the impact of 0 and 100% corn stover removal rates on N_2O and CO_2 fluxes were measured in a 2-yr study in Minnesota (Baker et al., 2014). Baker et al. (2014) concluded that full stover removal may have little impact on N_2O soil fluxes. Moreover, full stover removal reduced the soil CO_2 flux by up to 10%, but this reduction did not offset the C removed from the system compared with zero or intermediate removal rates, thus implying a net loss of C from the system. Working on the same project, Jin et al. (2014) summarized soil GHG emissions data from nine corn production systems under different corn stover management . Overall, stover removal reduced CO_2 emissions by 4%, similar to reports from Baker et al. (2014). Additionally, Jin et al. (2014) reported decreases in N_2O of 7% relative to no removal. Jin et al. (2014) concluded that lower GHG emissions in response to stover removal might imply a confounded effect of lower C and N inputs, and microclimatic differences related to spatial changes in soil cover. In agreement with this, complete stover removal reduced CO_2 and N_2O fluxes by 11% and 36%, regardless of tillage system, in a silty clay loam soil in Mexico. In this study, neither tillage system nor residue management affected the CH_4 fluxes from soil (Dendooven et al., 2012). Similarly, Abalos et al. (2013) reported a 51% decrease in the N_2O fluxes with no stover application, compared with stover applied at ~ 10.5 Mg ha^{-1} in a clay loam soil in Spain. Different than Dendooven et al. (2012), Baker et al. (2014) and Jin et al. (2014), however, corn stover removal did not affect the CO_2 fluxes from the soil in the Abalos et al. (2013) study.

Incorporation of crop residues resulted in mixed effects on N_2O emissions, and responses were strongly influenced by soil texture and feedstock quality. Gentile et al. (2008) measured a reduction of fertilizer-derived N_2O emissions when urea (120 kg N ha^{-1}) was applied with a low-quality corn feedstock (42% C, 1.3% N, C:N ratio of 31, 3.1% lignin, 1.1% Polyphenols) at ~ 9.5 Mg ha^{-1} in two coarse textured soils from Zimbabwe, likely explained by an increased in the N immobilization from the fertilizer. Conversely, the interactive effects following application of urea and corn stover increased the N_2O losses in two fine-textured soils from Ghana and Kenya, compared with fertilizer alone. In this case, increases in N_2O fluxes in the urea + stover treatment were explained by increases in both the N_2O fluxes from the fertilizer and the soil N pool. According to authors, this response may imply that denitrification was the main factor controlling N_2O fluxes in fine-textured soils where addition of residue rapidly depletes O_2 levels through increased microbial activity (Tiedje et al., 1984). The addition of high quality residues with low C:N ratio increased N_2O emissions for all soil textures (Gentile et al., 2008). Similarly, Huang et al. (2004) reported increases in both N_2O and C_2O after incorporation of crop residues in a 21 d incubation study with a fine-textured silty clay soil, independent of the type of residue utilized. The degree of this response was quantitatively dependent on the C:N ratio of the residues applied (C:N range: 8-118; 57 and 63 for corn stover and wheat straw, respectively), with fluxes of both gases negatively correlated ($r > 0.78$ in all cases) with C:N ratio (Huang et al., 2004), similar to recent reports from Lin et al. (2013) and Shan and Yan (2013).

4.2. CO_2, CH_4 AND N_2O FLUXES UNDER DIFFERENT WHEAT RESIDUE MANAGEMENT

Removal of wheat straw residues also has the potential to reduce the flux of greenhouse gasses from soils. Lenka and Lal (2013) studied the effect of three wheat straw addition [S_0 (0), S_8 (8) and S_{16} (16 Mg ha^{-1} yr^{-1}) and two fertilization rates [F_0 (0) and F_1 (244 Kg N ha^{-1} yr^{-1})] on CO_2, N_2O and CH_4 fluxes during a 15-year no-till residue management study in Ohio. No crop was grown during the 15 yr where treatments were imposed, and residues were applied as baled air-dried wheat straw from external sources. A significant interaction of wheat straw and fertilizer was found for all three gas fluxes. Diurnal CO_2 and N_2O fluxes were lowest for 0 and 8 Mg ha^{-1} yr^{-1} for both fertilization rates (i.e., S_0F_0, S_0F_1, S_8F_0, and S_8F_1), averaging 1.587 g CO_2 m^{-2} d^{-1} and 0.510 mg N_2O m^{-2} d^{-1}, respectively. Treatment $S_{16}F_0$ increased CO_2 and N_2O fluxes by 30% and 52% (2.059 g CO_2 m^{-2} d^{-1} and 0.774 mg N_2O m^{-2} d^{-1}, respectively). Treatment $S_{16}F_1$ resulted in maximum fluxes of 2.306 CO_2 m^{-2} d^{-1} and 1.020 mg N_2O m^{-2} d^{-1}, a 45% and 100% increase from minimum values found when 0 and 8 Mg ha^{-1} yr^{-1} of wheat straw was added. Soils that received no wheat straw over a 15-year period had a net CH_4, uptake under both fertilization rates, with values of -2.390 and -2.790 mg CH_4 m^{-2} d^{-1}, that were not statistically different between them. Over the same period, incorporation of wheat straw at rates of 8 and 16 Mg ha^{-1} yr^{-1}, with and without fertilizer, resulted in net CH_4 emission (range: 0.108 to 3.153 mg CH_4 m^{-2} d^{-1}) with greatest numerical values (not significantly different among them) when fertilizer was applied (Lenka and Lal, 2013).

5. EFFECTS OF RESIDUE MANAGEMENT ON GRAIN YIELD

One of the first field studies seeking to understand the impact of crop residue management in grain yield was conducted by Morachan et al. (1972) in Iowa. In this study, five stover return rates (i.e., 0, 2, 4, 8 and 16 Mg ha^{-1} yr^{-1}) were applied over 13 yr in continuous corn on a silty clay loam soil. Overall, during the first 9 yr, there was no grain yield differences among treatments. However, significant grain yield reductions occurred for most comparisons over the latter yr of the experiment (1963-1966), as more stover was returned to the soils. Morachan et al. (1972) proposed two potential explanations for these observations: first, high stover return rates were associated with lower soil pH values, which may have caused an Al^{+3}-induced Ca^{+2} deficiency in the plant; second, measured increments in leaf K^+/Ca^{+2} and K^+/Ca^{+2} + Mg^{+2} balances (resulting from additions of K^+-rich residues) may have caused severe cation imbalances in the crop plants, with consequent poor plant growth and grain yield penalties. Four years after termination of field treatments, maximum grain yields were observed in plots that formerly received 8 and 16 Mg ha^{-1} of stover (Morachan et al., 1972), a response that may have reflected the higher nutrient status of plots with high residue returns since the area was not fertilized following field study completion (Larson et al., 1972).

In a 3-year study in no-till clay-loam soils in Nebraska with corn stover return rates of 0, 50, 100 and 150% Doran et al. (1984) observed a 3-yr grain yield decrease of 21% when no stover was returned, compared with maximum yields achieved with 100 and 150% return rates. Variations in grain yields were explained, in part, by 52% and 59% reductions in available water in June and July in plots without stover in the third experimental year. At this time, corn stands were at tasseling and silking (Abendroth et al., 2011), critical stages for grain yield determination when the presence of abiotic or biotic limitations or both can severely decrease grain yields (Robins and Domingo, 1953; Claassen and Shaw, 1970; Hall et al., 1981; NeSmith and Ritchie, 1992).

Wilhelm et al. (1986) continued with the work started by Doran et al. (1984) in the subsequent 4 years (1980-1983), replacing the corn-sorghum [*Sorghum bicolor* (L.) Moenchl]-soybean [*Glycine max* (L.) Merr.] rotation with no-till continuous corn. Grain yields during the first and the fourth year were irresponsive to treatments, likely due to the mixed effects of above-average air temperatures and below-average precipitation during the growth cycle. However, for each Mg ha^{-1} of residue applied in the range from 0 to 8 Mg ha^{-1}, grain yields increased by 0.32 Mg ha^{-1} in the second year of the study and 0.26 Mg ha^{-1} in the third (Wilhelm et al., 1986).

Karlen et al. (1984) researched the effects of three stover removal rates under conservation tillage (i.e., 0, 66, or 90%) on grain yields on a sandy loam soil in South Carolina over a 3-yr. In non-irrigated plots, removing up to 90% of the stover did not reduce grain yields in the first, decreased grain yields the second, and increased grain yields the final year compared to 0% removal. Under irrigation, removing 66% or 90% of the stover supported greater yields than 0% removal in year one, but had no effect the following two yr. Karlen et al. (1984) suggested that lower soil coverage with the high removal rate treatment may have exacerbated the water deficits in a dry year like the second one, which in turn could explain the grain yield reductions observed with 90% removal that year. However, when moisture was generally not limiting, harvesting crop residues had no impact on grain yields in non-irrigated conditions. More recently, similar results were reported by Linden et al. (2000) working with corn in a silt loam in east-central Minnesota. When all the stover was removed compared with 100% return, corn grain yields decreased by 18% in dry yr over a 12-yr period, although more years showed this response under chisel tillage compared with moldboard and no-till. Corn grain yields did not differ by residue management when available water was not limiting in all 24 tillage-year comparisons. Similar results were found with different wheat straw treatments under irrigation. Working in a fine sandy loam soil in the Texas Rolling Plains, Bordovsky et al. (1998) observed a 6% grain yield increase in irrigated wheat after complete compared with no straw removal over an 8 yr period, likely the result of more uniform plant stands with less surface residue. Over the same period, regardless of the water status during crop cycle, complete residue removal in dryland conditions did not affect wheat yield during any year compared with no residue removal (Bordovsky et al., 1998).

Crop residue management strategies can have long-term residual effects on crop yield potential, although research results vary. Power et al. (1998) found that returning 150% of the total stover had the greatest impact over a 10 yr period, increasing final grain yields by 16% compared with complete removal. However, long-term differences between 0 and 100% return were not significant. Moreover, these effects were not affected by time and other management practices such as tillage, N fertilization or cover crops. More likely they reflect long-term changes in soil properties, microbial activity and soil N mineralization rates. Conversely, Dam et al. (2005) found no long-term residue effect on either corn grain or dry matter yields working with a factorial arrangement of crop residue (2 levels: without and with residue) and tillage (3 levels: no-till, reduced tillage and conservation tillage) in Canada. In his experiment, significant tillage × residue interaction for both grain and corn dry matter yields was only found in 2 out of 12 yr. When differences occurred, treatments with no residue retained were among the maximum yields in all comparisons.

Blanco-Canqui et al. (2006b) evaluated the impacts of six corn stover return rates on continuous no-till corn yields on two silt loam and one clay-loam soil over 2-yr in Ohio. Regardless of soil texture, there was no grain yield differences among treatments during the first year. In the second year, grain yields among treatments did not differ for silt loam and clay-loam soils with less than 2% slope. Grain yields from a silt loam with ~10% slopes and more yr under no-till, were 21% lower in treatments with 0 and 25% compared to treatments with 50, 75, 100 and 200% return rates . Authors attributed these effects to lower soil moisture content and increased soil temperatures when little or no stover was returned to soils with pronounced slopes. Similar results were reported by Power et al. (1986) who found soil temperatures up to 7°C lower with full corn stover retention compared with complete removal, presumably due to the reduction of radiation being intercepted and absorbed in bare soils. This study suggests that up to 50% of the existing stover residues after grain harvest could be potentially removed without affecting grain yields in the short term (≤2 yr), although these responses may be highly dependent on type of soil, topography and management history.

6. OTHER POTENTIAL IMPACTS OF RESIDUE REMOVAL

In some cases, removing residues from the field can help to reduce insect and disease pressure (Wilhelm et al., 2004) and may result in faster emergence for corn (Wilhelm et al., 1986; Swan et al., 1987; Vetsch and Randall, 2002), especially in regions with a short and cool spring season.

Dam et al. (2005) observed slower spring corn emergence, ranging from 14 to 63%, in no-till + no stover removal, compared with no-till + complete stover removal and conventional tillage with or without stover removal. The authors concluded that the slowest emergence rates found under no-till with no stover removal in central Canada could be explained by high surface residue cover and lower soil temperatures (Dam et al., 2005). Similar findings were observed by Swan et al. (1987) working in Wisconsin and Minnesota with different tillage systems and a wide range of surface residue coverage. In three sites-years, the number of growing degree days (GDD) from planting to V6 growth stage (Abendroth et al., 2011) linearly decreased as soil cover increased. In two sites-years, for each unit increase in the percent of residue cover, the soil GDD decreased by 0.82 units, and by 0.53 units in the third significant location [slope (b) of the linear regression soil GDD = a + b (% cover)]. At the same time, Swan et al. (1987) observed an increase in the air GDD required to achieve both 80% emergence and the V6 growth stage with increases in the percent residue cover. Each unit increase in percent cover increased the air GDD requirement to complete the planting-emergence and the planting-V6 stage between 0.18 and 0.51, and 0.51 and 0.81 units, respectively. Similarly, Schneider and Gupta (1985) found that corn emergence occurred more rapidly in the treatments with least corn residue left on surface. In another experiment, Swan et al. (1994) found that 200% corn stover retained rates decreased plant density at harvest by 5% and increased grain moisture by 4% in two silt loam soils in Wisconsin over a 7-yr period, compared with 100% retained rates.

7. SUMMARY

Biofuel production from crop residues has the potential to provide with numerous environmental and economic benefits to modern societies. However, residues contribute to agricultural productivity and have profound impact in soil health and quality. Depending on factors such as soil type, SOM, and topography, between 25 and 60% of the available crop residues in the Corn Belt could be sustainably harvested. Although numerous studies regarding the impacts of corn stover and wheat straw management on grain yield, greenhouse gas emissions, soil C and N pools, nutrient balances, aggregate stability, bulk density, and available water, among other parameters, are available in the literature, none have evaluated these parameters together. The present review addresses this need and provides with an empirical foundation for the related work that has been conducted in Virginia and is presented in the following chapters.

8. OBJECTIVES and HYPOTHESIS

The <u>overall objectives</u> for these experiments are as follows:

1. To asses/evaluate the short-term impacts of various crop residue removal rates from common mid-Atlantic cropping systems in terms of subsequent:
   i. Grain and total biomass productivity.

ii. Total carbon and nitrogen (TC, TN, respectively), pH, bulk density, water holding capacity (WHC), permanent wilting point (PWP), plant available water (PAW) and water aggregate stability.

iii. Carbon dioxide (CO_2), methane (CH_4), nitrous oxide (N_2O), and global warming potential (GWP) emissions.

2. To calculate theoretical ethanol yields on a per area basis for corn and wheat in typical Mid-Atlantic USA cropping systems and compare/contrast this data with available data produced in other regions.

The *overall hypotheses* for these experiments are as follows:

I. In the short term, retaining 3.33 Mg residues ha^{-1} yr^{-1} or less in CC the rotation in southwestern Virginia will result in lower grain, plant biomass and ethanol yields in subsequent years and will detrimentally affect total C and N, bulk density, CO2, N2O and CH4 fluxes bulk density, and other chemical and physical parameters of soil quality. The change in these parameters will be measured at the 0-2.5 cm depth.

II. In the short term, treatments retaining 5 Mg corn stover and/or wheat straw ha^{-1} yr^{-1} or less in eastern Virginia will result in lower grain, plant biomass and ethanol yields in subsequent years and will detrimentally affect total C and N, bulk density, CO2, N2O and CH4 fluxes bulk density, and other chemical and physical parameters of soil quality. The greatest change in these parameters will be measured at the 0-2.5 cm depth.

9. LITERATURE CITED

Abalos, D., A. Sanz-Cobena, L. Garcia-Torres, J.W. van Groenigen, and A. Vallejo. 2013. Role of maize stover incorporation on nitrogen oxide emissions in a non-irrigated Mediterranean barley field. Plant Soil 364: 357-371.

Abendroth, L.J., R.W. Elmore, M.J. Boyer, and S.K. Marlay. 2011. Corn growth and development. PMR 1009. Iowa State University Extension, Ames, Iowa.

Amézketa, E. 1999. Soil Aggregate Stability: A Review, J. Sustain. Agric. 14: 83-151.

Anderson-Teixeira, K. J., Davis, S. C., Masters, M., and Delucia, E.H. 2009. Changes in soil organic carbon under biofuel crops. GCB Bioenergy 1: 75–96.

Andrews, S.S. 2006. Crop Residue Removal for Biomass Energy Production:

Effects on Soils and Recommendations. USDA-Natural Resource Conservation Service.At: https://www.nrcs.usda.gov/Internet/FSE_DOCUMENTS/nrcs142p2_053255.pdf (accessed: 10/12/2018)

Bahrani, M.J., M. Kheradnam, Y. Emam, H. Ghadiri, and M.T. Assad. 2002. Effect of tillage methods on wheat yield and yield components in continuous wheat cropping. Exper. Agric. 38: 389-395.

Baker, J.M., J. Fassbinder, and J.A. Lamb. 2014. The impact of corn stover removal on N2O emission and soil respiration: an investigation with automated chambers. BioEnergy Res. 7: 503-508.

Barber, S.A. 1979. Corn residue management and soil organic matter. Agron. J. 71: 625-627.

Benjamin, J.G., A.D. Halvorson, D.C. Nielsen, and M.M. Mikha. 2008. Crop management effects on crop residue production and changes in soil organic matter in the central Great Plains. Agron. J. 102:990-997.

Bernard, E., C. Chenu, J. Balesdent, P. Puget, and D. Arrouays. 1996. Fate of particulate organic matter in soil aggregates during cultivation. Eur. J. Soil Sci. 47: 495–503.

Blanco-Canqui, H., and R. Lal. 2009. Corn stover removal for expanded uses reduces soil fertility and structural stability. Soil Sci. Soc. Am. J. 73: 418-426.

Blanco-Canqui, H., R. Lal, R.C. Izaurralde, and L.B. Owens. 2006a. Rapid changes in soil carbon and structural properties due to stover removal from no-till corn plots. Soil Sc. 171: 468-482.

Blanco-Canqui, H., R. Lal, W.M. Post, and L.B. Owens. 2006b. Changes in long-term no-till corn growth and yield under different rates of stover mulch. Agron J. 98:1128-1136.

Bordovsky, D.G., M. Choudhary, C.J. Gerard. 1998. Tillage Effects on Grain Sorghum and Wheat Yields in the Texas Rolling Plains. Agron. J. 90: 638-643.

Bordovsky, D.G., M. Choudhary, and C.J. Gerard. 1999. Effect of tillage, cropping, and residue management on soil properties in the Texas Rolling Plains. Soil Sci. 164 (5): 331-340.

Cambardella, C.A., and E.T. Elliot. 1992. Particulate soil organic matter changes across a grassland cultivation sequence. Soil Sci. Soc. Am. J. 56: 777–783.

Carter, M.R. 2002. Soil Quality for Sustainable Land Management: Organic Matter and Aggregation Interactions that Maintain Soil Functions. Agron. J. 94: 38-47.

Carter, M.R., E.G. Gregorich, D.A. Angers, R.G. Donald, and M.A. Bolinder. 1998. Organic C and N storage, and organic C fractions in adjacent cultivated and forested soils of eastern Canada. Soil Tillage Res. 47:253–261.

Clapp, C.E., R.R. Allmaras, M.F. Layese, D.R. Linden, and R.H. Dowdy. 2000. Soil organic carbon and 13-C abundance as related to tillage, crop residue, and nitrogen fertilizer under continuous corn management in Minnesota. Soil Tillage Res. 55: 127-142.

Claassen, M.M., and R.H. Shaw. 1970. Water deficits effects on corn. II. Grain components. Agron. J. 62: 652-655.

Crutzen, P.J., A.R. Mosier, K.A. Smith, and W. Winiwater. 2008. N2O release from agro-biofuel production negates global warming reduction by replacing fossil fuels. Atmos. Chem. Phys. 8: 389–395.

Curtin, D., and P.M. Fraser. 2003. Soil organic matter as influenced by straw management practices and inclusion of grass and clover seed crops in cereal rotations. Aust J Soil Res. 41 (1): 65-106.

Dam, R.F., B.B. Mehdi, M.S.E. Burgess, C.A. Madramootoo, G.R. Mehuys, and I.R. Callum. 2005. Soil bulk density and crop yield under eleven consecutive years of corn with different tillage and residue practices in a sandy loam soil in central Canada. Soil Tillage Res. 84 (1): 41-53.

Dendooven, L., L. Patiño-Zúñiga, N. Verhulst, M. Luna-Guido, R. Marsch, and B. Govaerts. 2012. Global warming potential of agricultural systems with contrasting tillage and residue management in the central highlands of Mexico. Agric. Ecosyst. Environ. 152: 50-58.

Dolan, M.S., C.E. Clapp, R.R. Almaras, J.M. Baker, and J.A.E. Molina. 2006. Soil organic carbon and nitrogen in a Minnesota soil as related to tillage, residue and nitrogen management. Soil and Tillage Res. 89: 221-231.

Doran, J.W., W.W. Wilhelm, and J.F. Power. 1984. Crop residue removal and soil productivity with no-till corn, sorghum, and soybean. Soil Sci. Soc. Am. J. 48:640-645.

EPA. 2017. Greenhouse Gas Emissions. Understanding Global Warming Potentials. United States Environmental Protection Agency. At: https://www.epa.gov/ghgemissions/understanding-global-warming-potentials (updated: 02/14/17; accessed: 10/12/2018).

Franzluebbers, A.J. 2002. Water infiltration and soil structure related to organic matter and its stratification with depth. Soil Tillage Res. 66:197-205.

Gale, W.J., and C.A. Cambardella. 1998. Carbon Dynamics of Surface Residue– and Root-derived Organic Matter under Simulated No-till. Soil Sci. Soc. Am. J. 64:190-195.

Gallaher, P., M. Dikemann, J. Fritz, E. Wailes, W. Gauther, and H, Shapouri. 2003. Biomass from crop residues: cost and supply estimates. USDA, Office of the Chief Economist, Energy Policy and New Uses. Agricultural economic report 819. USDA, Washington, DC. At: http://ageconsearch.umn.edu/bitstream/34063/1/ae030819.pdf (accessed 10/12/2018).

Gentile, R., B. Vanlauwe, P. Chivenge, and J. Six. 2008. Interactive effects from combining fertilizer and organic residue inputs on nitrogen transformations. Soil Biol. Biochem. 40: 2375–2384.

GGWG. 2010. Agriculture's role in greenhouse gas emissions & capture. Greenhouse Gas Working Group Rep. ASA, CSSA, and SSSA, Madison, WI.

Glassner, D., J. Hettenhaus, and T. Schechinger. 1998. Corn stover collection project: a pilot for establishing infrastructure for agricultural residue and other crop collection for biomass processing to ethanol. In: Proc. Bioenergy 1998 Conference, 4–8 October, 1998, Madison, WI, pp. 1100–1110.

Graham, R.L., R. Nelson, J. Sheenan, R.D. Perlack, and L.L. Wright. 2007. Current and potential U.S. corn stover supplies. Agron. J. 99: 1-11.

Grande, J.D., K. Karthikeyan, P.S. Miller, and J.M. Powell. 2005. Corn residue level and manure application timing effects on phosphorus losses in runoff. J. Environ. Qual. 34: 1620-1631.

Gregorich, E.G., and H.H. Janzen. 1996. Storage of soil carbon in the light fraction and macroorganic matter. p. 167–190. In M.R. Carter and B.A. Stewart (ed.) Structure and organic matter storage in agricultural soils. Lewis Publ., CRC Press, Boca Raton, FL.

Hall, A.J., J.H. Lemcoff, and N. Trapani. 1981. Water stress before and during flowering in maize and its effects on yield, its components, and their determinants. Maydica 26: 19–38.

Hammerbeck, A.L., S.J. Stetson, S.L. Osborne, T.E. Schumacher, and J.L. Pikul, Jr. 2012. Corn Residue Removal Impact on Soil Aggregates in a No-Till Corn/Soybean Rotation. Soil Sci. Soc. Am. J. 76: 1390-1398.

Houghton, R. A. 2007. Balancing the global carbon budget. Annu. Rev. Earth Planet. Sci. 35: 313 – 347.

Huang, Y., J. Zou, X. Zheng, Y. Wang, and X. Xu. 2004. Nitrous oxide emissions as influenced by amendment of plant residues with different C:N ratios. Soil Biol. Biochem. 36: 973-981.

Huggins, D.R., C.E. Clapp, R.R. Allmaras, J.A. Lamb, and M.F. Layese. 1998. Carbon dynamics in corn-soybean sequences as estimated from natural carbon-13 abundance. Soil Sci. Soc. Am. J. 62:195-203.

Jin, V.L., J.M. Baker, J.M.F. Johnson, D.L. Karlen, R.M. Lehman, S.L. Osborne, T.J. Sauer, D.E. Stott, G.E. Varvel, R.T. Venterea, M.R. Schmer, and B.J. Wienhold. 2014. Soil greenhouse gas emissions in response to corn stover removal and tillage management across the US Corn Belt. BioEnergy Res. 7: 517-527.

Johnson, J.M.F., V. Acosta-Martinez, C.A. Cambardella, and N.W. Barbour. 2013. Crop and soil responses to using corn stover as a bioenergy feedstock: observations from the Northern US Corn Belt. Agric. J. 3: 72-89.

Johnson, J.M.F., D.C. Reicosky, B. Sharratt, M.J. Lindstrom, W.B. 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.

Johnson, P.A., and B.J. Chamber. 1996. Effects of husbandry on soil organic matter. Soil Use Manage. 13: 102-103.

Karlen, D.L., P.G. Hunt, and R.B. Campbell. 1984. Crop residue removal effects on corn yield and fertility of a Norfolk sandy loam. Soil Sc. Soc. Am. J. 48:868-872.

Karlen, D.L., N.C. Wollenhaupt, D.C. Erbach, E.C. Berry, J.B. Swan, N.S. Eash, and J.L. Jordahl. 1994. Crop residue effects on soil quality following 10-years of no-till corn. Soil Tillage Res. 31:149-167.

Kendall, J.R.A., D.S. Long, H.P. Collins, F.J. Pierce, A. Chatterjee, J.L. Smith, and S.L. Young. 2015. Soil carbon dynamics of transition to Pacific Northwest cellulosic ethanol feedstock production. Soil Sci. Soc. Am. J. 79:272-281.

Kenney, I.T. 2011. Regional assessment of short-term impacts of corn stover removal for bioenergy on soil quality and crop production. MSc. thesis, Kansas State University, Manhattan, KS.

Khanna, M., and N. Paulson. To Harvest Stover or Not: Is it Worth it? Farmdoc daily (6):32. Department of Agricultural and Consumer Economics, University of Illinois at Urbana-Champaign, February 18, 2016.

Kim, S., and B.E. Dale. 2004. Global potential bioethanol production from wasted crops and crop residues. Biomass and Bioenergy 26: 361-375.

Kim, S., B.E. Dale, and R. Jenkins. 2009. Life cycle assessment of corn grain and corn stover in the United States. Int. J. Life Cycle Assess. 14: 160-174.

Larson, W.E., Y.B. Morachan, C.E. Clapp, and W.H. Pierre. 1972. Effects of increasing amounts of organic residues on continuous corn: II. Organic carbon, nitrogen, phosphorus, and sulfur. Agron. J. 64:204–209.

Lenka, N.K., and R. Lal. 2013. Soil aggregation and greenhouse gas flux after 15 years of wheat straw and fertilizer management in a no-till system. Soil Tillage Res. 126: 78-89.

Liang, B.C., E.G. Gregorich, A.F. MacKenzie, M. Schnitzer, R.P. Voroney, C.M. Monreal, and R.P. Beyaert. 1998. Retention and turnover of corn residue carbon in some eastern Canadian soils. Soil Sci. Soc. Am. J. 62: 1361-1366.

Lin, S., J. Iqbal, R. Hu, M. Shaaban, J. Cai, and X. Chen. 2013. Nitrous oxide emissions from yellow brown soil as affected by incorporation of crop residues with different carbon-to-nitrogen ratios: a case study in central China. Arch. Environ. Contam. Toxicol. 65:183–192.

Linden, D.R., C.E. Clapp, and R.H. Dowdy. 2000. Long-term corn grain and stover yields as a function of tillage and residue removal in east central Minnesota. Soil Tillage Res. 56:167-174.

Lindstrom, M.J. 1986. Effects of residue harvesting on water runoff, soil erosion and nutrient loss. Agric. Ecosyst. Environ. 16: 103-112.

Lindstrom, M.J., Voorhees, W.B. and Onstad, C.A. 1984. Tillage systems and residue cover effects on infiltration in northwestern Corn Belt soils. Journal of Soil and Water Conservation Jan.-Feb.: 64-68.

Magdoff, F., L. Lanyon, and B. Liebhardt. 1997. Nutrient cycling, transformations, and flows: Implications for a more sustainable agriculture. Adv. Agron. 60:2-68.

Mann, L., Tolbert, V. and Cushman, J. 2002. Potential environmental effects of corn (Zea mays L.) stover removal with emphasis on soil organic matter and erosion. Agric. Ecosyst. Environ. 89: 149-166.

McAloon, A., F. Taylor, W. Yee, K. Ibsen, and R. Wooley. 2000. Determining the cost of producing ethanol from cornstarch and lignocellulosic feedstocks. Tech. Rep. NREL/TP-580-28893. Natl. Renewable Energy Lab., Golden, CO. At: https://www.nrel.gov/docs/fy01osti/28893.pdf (accessed: 10/12/2018).

Moebious-Clune, B.N., H.M. van Es, O.J. Idowo, R.R. Schindelbeck, D.J. Moebious-Clune, D.W. Wolfe, G.S. Abawi, J.E. Thies, B.K. Gugino, and R. Lucey. 2008. Long-term effects of harvesting maize stover and tillage on soil quality. Soil Sc. Soc. Am. J. 72: 960-969.

Morachan, Y.B., W.C. Moldenhauer, and W.E. Larson. 1972. Effects of increasing amounts of organic residues on continuous corn: I. Yields and soil physical properties. Agron. J. 64:199-203.

Nelson, R.G. 2002. Resource assessment and removal analysis for corn stover and wheat straw in the Eastern and Midwestern United States – rainfall and wind-induced soil erosion methodology. Biomass Bioenerg. 22:349-363.

NeSmith, D.S., and J.T. Ritchie. 1992. Short- and long-term responses of corn to a pre-anthesis soil water deficit. Agron. J. 84: 107–113.

Nicholson, F.A., B.J. Chambers, A.R. Mills, and P.J. Strachan. 1997. Effects of repeated straw incorporation on crop fertilizer nitrogen requirements, soil mineral nitrogen and nitrate leaching losses. Soil Use Management 13: 136-142.

Power, J.F. and Doran, J.W. 1988. Role of crop residue management in nitrogen cycling and use. In Cropping Strategies for Efficient Use of Water and Nitrogen. Ed. Hargrove et al., pp 101–113. ASA Spec. Publ. 51, ASA CSA and SSSA, Madison, WI

Power, J.F., J.W. Doran, and W.W. Wilhelm. 1986. Crop residue effects on soil environment and dryland maize and soybean production. Soil Tillage Res. 8:101-111.

Power, J.F., P.T. Koerner, J.W. Doran, and W.W. Wilhelm. 1998. Residual effects of crop residues on grain production and selected soil properties. Soil Sci. Soc. AM. J. 62:133-1397.

Raffa, D.W., A. Bogdanski, O. Dubois, and P. Tittonell. 2014. Take it or Leave it? Towards a decision support tool on sustainable crop residue use. Part 1: Soil management. Environment and natural resources management working paper No. 61 Energy. 110 p. Food and Agriculture Organization of the United Nations. Rome, Italy. At: http://www.fao.org/3/a-i4120e.pdf (accessed: 10/12/2018).

Robins, J.S., and C.E. Domingo. 1953. Some effects of severe soil moisture deficits at specific growth stages in corn. Agron. J. 45: 618–621.

Schneider, E. C., and S. C. Gupta. 1985. Corn Emergence as Influenced by Soil Temperature, Matric Potential, and Aggregate Size Distribution1. Soil Sci. Soc. Am. J. 49:415-422.

Sequeira, C.H., and M.M. Alley. 2011. Soil organic matter fractions as indices of soil quality changes. Soil Sci. Soc. Am. J. 75: 1766-1773.

Sequeira, C.H., M.M. Alley, and B.P. Jones. 2011. Evaluation of potentially labile soil organic carbon and nitrogen fractionation procedures. Soil Biol. Biochem. 43: 438–444.

Shan, J., and X. Yan. 2013. Effects of crop residue returning on nitrous oxide emissions in agricultural soils. Atmos. Environ. 71:170–175.

Sheaffer, C., J. Lamb and Rosen C. 2010. Corn stover: ethanol production and nutrient uptake. At: http://www.mncorn.org/sites/mncorn.org/files/research/final-reports/201211/B6-CornStoverEthanolProduction_2.pdf (accessed: 10/12/2018).

Sheehan, J., A. Aden, K. Paustian, K. Killian, J. Brenner, M. Walsh, and R. Nelson. 2004. Energy and environmental aspects of using corn stover for fuel ethanol. J. Ind. Ecol. 7: 117-146.

Sindelar, A.J. 2012. Stover, tillage, and nitrogen management in continuous corn for grain, ethanol, and soil carbon. Ph.D. diss., University of Minnesota, Minneapolis, MN.

Six, J., H. Bossuyt, S. Degryze, and K. Denef. 2004. A history of research on the link between (micro)aggregates, soil biota, and soil organic matter dynamics. Soil Tillage Res. 79:7–31.

Spargo, J.T., M.A. Cavigelli, M.M. Alley, J.E. Maul, J.S. Buyer, C.H. Sequeira and R.F. Follet. 2012. Changes in soil organic carbon and nitrogen fractions with duration of no-tillage management. Soil Sci. Soc. Am. J. 76: 1624-1633.

Swan, J.B., R.L. Higgs, T.B. Bailey, N.C. Wollenhaupt, W.L. Paulson, and A.E. Peterson. 1994. Surface Residue and In-Row Treatment Effects on Long-Term No-Tillage Continuous Corn. Agron. J. 86: 711-718.

Swan, J.B., E.C. Schneider, J.F. Moncrief, W.H. Paulson, and A.E. Peterson. 1987. Estimating Corn Growth, Yield, and Grain Moisture from Air Growing Degree Days and Residue Cover. Agron. J. 79:53-60.

Tarkalson, D.D., B. Brown, H. Kok, and D.L. Bjorneberg. 2009. Impact of Removing Straw from Wheat and Barley Fields: A Literature Review. Better crops 93 (3): 17-19.

Tenenbaum, D.J. 2008. Food vs. Fuel: Diversion of Crops Could Cause More Hunger. Environ. Health Persperct. 116(6): A254–A257.

Thompson, P.B. 2012. The Agricultural Ethics of Biofuels: The Food vs. Fuel Debate. Agriculture J. 2: 339-358.

Tiedje, J.M., Sexstone, A.J., Parkin, T.B., Revsbech, N.P., Shelton, D.R., 1984. Anaerobic processes in soil. Plant and Soil 76, 197–212.

Undersander, D.J., and C. Reiger. 1985. Effect of wheat residue management on continuous production of irrigated winter wheat. Agron J. 77: 508-511.

U.S. Department of Energy. 2011. U.S. Billion-Ton Update: Biomass Supply for a Bioenergy and Bioproducts Industry. R.D. Perlack and B.J. Stokes (Leads), ORNL/TM-2011/224. Oak Ridge National Laboratory, Oak Ridge, TN. 227p. At: http://www.eesi.org/files/billion_ton_update.pdf (accessed: 10/12/2018).

Vetsch, J.A., and G.W. Randall. 2002. Corn production as affected by tillage system and starter fertilizer. Agron. J. 94:532–540.

Walsh, M.E., R.L. Perlack, A. Turhollow, D.G. de la Torre Ugarte, D.A. Becker, R.L. Graham, S.E. Slinsky, and D.E. Ray. 2000. Biomass feedstock availability in the United States: 1999 state level analysis. Oak Ridge National Lab., Oak Ridge, TN. At: https://www.nrc.gov/docs/ML0719/ML071930137.pdf (accessed 10/12/2018).

Wang, M., J. Han, J.B. Dunn, H. Cai, and A. Elgowainy. 2012. Well-to-wheels energy use and greenhouse gas emissions of ethanol from corn, sugarcane and cellulosic biomass for US use. Environ. Res. Lett. 7. doi:10.1088/1748-9326/7/4/045905

Wilhelm, W.W., J.W. Doran, and J.F. Power. 1986. Corn and soybean yield response to crop residue management under no-tillage production systems. Agron. J. 78:184-189.

Wilhelm, W.W., J.M. Johnson, J.L. Hatfield, W.B. Voorhees, and D.R. Linden. 2004. Crop and soil productivity response to corn residue removal: a literature review. Agron. J. 96: 1-17.

Wilhelm, W.W., J.M.F. Johnson, D.L. Karlen, and D.T. Lightle. 2007. Corn stover to sustain soil organic carbon further constrains biomass supply. Agron. J. 99: 1665–1667.

CHAPTER 3

Crop residue management effects on greenhouse emissions in the Mid-Atlantic USA

1. ABSTRACT

Bioenergy production systems have potential to reduce greenhouse gas (GHG) emissions. The carbon footprint resulting from crop residue management has been studied extensively, largely in the Corn Belt. However, effects of harvesting more than one crop residue from a multi-crop system on GHG emissions have not been reported. This study compared GHG emissions in response to various residue retention rates. Measures were collected at specific times during the growing season in two contrasting cropping systems in the Mid-Atlantic USA. Methane (CH_4), nitrous oxide (N_2O), and carbon dioxide (CO_2) emissions were measured in response to five corn stover application rates (0,3.33,6.66,10, and 20 Mg ha^{-1}) in a continuous corn grain (*Zea mays* L.) rotation following corn-planting in 2016 and 2017 in Blacksburg, VA. Two other experiments were initiated during the wheat (June 2015, NK1) and corn (September 2015, NK2) phases of a corn–wheat (*Triticum aestivum* L.)-soybean (*Glycine max* L. Merr.) rotation in New Kent, VA. Here, the impact of retaining either corn stover (0,3.33,6.66 and 10 Mg ha^{-1}) or wheat straw (0,1,2 and 3 Mg ha^{-1}) in the previous year, or the combined effect of both after two years was measured. Soybean residue management was not considered. In the year following application, retaining approximately 50% (5 Mg ha^{-1}) or more of the stover produced increased CH_4 and N_2O emissions from the loamy soil in Blacksburg about 25%. Maximum CH_4 and N_2O fluxes (4.16 and 5.94 mg m^{-2} day^{-1}, respectively) occurred with 200% (20 Mg ha^{-1}) retention rates. Two cycles of corn stover management in Blacksburg, and one cycle of corn stover or wheat straw management in the sandy

loam soils in both experiments in New Kent did not affect measured fluxes of any gas. In 2017, total corn and wheat retained residues affected only CH_4 and N_2O fluxes following V4 nitrogen fertilization, but these results were not biologically meaningful. This is the first study to investigate the effects of crop residue on GHG emissions in a multi-crop system common in humid temperate zones. Longer-term monitoring studies are warranted to understand crop residue management effects on GHG emissions in these systems.

2. INTRODUCTION

Global greenhouse gasses emissions have increased by 75% since the 1970s (Smith et al., 2014). In a recent report, agriculture, together with forestry, anthropogenic land-use (i.e., wetlands and settlements) and other land (i.e., bare soil, rock and ice) accounted for about 24% of the total emissions (mainly comprised of CH_4 and N_2O) and was only surpassed by the energy production sector (around 34% of the total emissions) (Smith et al., 2014). From this total, estimated contributions from agriculture (5.0 to 5.8 Gt CO_2 eq yr^{-1}) and land-use change (i.e., changes in forest and other woody biomass stocks, forest and grassland conversion to pasture or cropland, and abandonment of croplands, pastures or other managed lands that regrow into their prior natural grassland or forest condition; IPCC, 2000) were similar (4.3 to 5.5 Gt CO_2 eq yr^{-1}) in the period 2000 to 2010. In this context, lignocellulosic bioenergy systems that utilize crop residues could play an important role in mitigating the increases in anthropogenic GHG emissions (Wilhelm et al., 2004; Macedo et al., 2015; Souza et al., 2017). Farmers could implement these practices, and interest in lignocellulosic ethanol production from crop residues has been recently on the rise (Golden et al., 2015; Zabed et al., 2016), as these residues are available in considerable amount (Singh et al., 2010) and do not compete with food production (Tenenbaum, 2008; Thompson, 2012). Moreover, crop residues are to some extent "subsidized" by the primary grain production, so there is no need for land conversion when considering biomass harvest. However, this potential largely depends on the development of sustainable and efficient bioenergy systems (Chum et al., 2011), and uncertainties regarding these topics still remain (Smith et al., 2014).

In residue-based bioenergy production systems, the potential impact of residue removal on soil processes and emissions of GHG, especially N_2O, is of particular interest and concern (Carter et al., 2002; Baker et al., 2014). Increased accumulation of crop residues in the soil surface, which increases the supply of readily available C and N for microbial activity (Aulakh et al. 2001; Huang et al. 2004), increases potential C oxidation, nitrification and denitrification rates in soils (Ball et al., 1999; Fontaine et al. 2004; Stehfest and Bouwman, 2006; Ball et al., 2008; Derrien et al. 2014; Badagliacca et al., 2017). As a result, most field studies indicate a reduction in the total GHG emissions following crop residue removal from agricultural systems (Dendooven et al., 2012; Abalos et al., 2013; Lenka and Lal, 2013; Baker et al., 2014; Jin et al., 2014).

Because of the scope of the systems and the number of variables involved, much of the analysis of GHG emissions has been conducted via modeling. For example, all soil nitrogen-related emissions, when modeled with DayCent (Parton et al. 1988; Hartman et al., 2016) were significantly reduced with corn stover removal (Kim and Dale, 2005; Kim et al., 2009). Simulations of vehicular emissions with the GREET model (Argonne National Laboratory, 2017) indicate that ethanol from corn stover can reduce GHG emissions by 90–103% relative to petroleum gasoline (Wang et al., 2012), a much higher reduction when compared with ethanol production from corn grain (Wang et al., 2011). Incorporating crop residues has increased soil N_2O emissions in some studies (Huang et al., 2004; Gentile et al., 2008; Lin et al., 2013) but had no significant effect on N_2O fluxes across 112 studies in a recent meta-analysis (Shan and Yan, 2013).

However, some authors have stated that the release of N_2O resulting from converting crop residues to biofuels may counterbalance the reduction in global warming resulting from fossil fuel displacement (Carter et al., 2002; Crutzen et al., 2008). However, full life cycle assessments would be more appropriate, as these analyses do not consider the fossil fuels needed for biomass, fertilizer and pesticide production, and the use of co-products from the bioenergy industry (Crutzen et al., 2008).

Although crop residue removal for biofuel production has the potential to reduce GHG emissions from soil or through fuel consumption and use, and despite extensive study worldwide, to date no data on this have been generated in the temperate humid Mid-Atlantic USA. Generating information on the impact of crop residue removal on GHG emissions is crucial to assess its environmental sustainability in the region and a prerequisite if bioethanol is to become a viable substitute for the fossil fuels intended to be displaced (Hill et al., 2006). Thus, the objectives of this study were to document short-term changes in soil CO_2, CH_4 and N_2O emissions in response to various levels of residue removal from continuous corn (CC) grain and corn-wheat/soybean rotations in Virginia.

3. MATERIALS AND METHODS

Three field experiments assessing the impact of crop residue retention were conducted over three years, from 2015 through 2017, in two physiographic regions in Virginia.

3.1. Experiment 1

The experiment was conducted in Blacksburg, Virginia (BB), in the Valley and Ridge physiographic region, on a Unison and Braddock loam soil (fine, mixed semiactive, mesic hapludults) (NRCS, 2018). The experiment utilized a RCBD with five levels of corn stover retention: 0, 3.33, 6.66, 10, and 20 Mg ha^{-1} corn stover retained in a continuous corn (CC) grain rotation (Table 1). Four replications were established at BB resulting in a total of 20 experimental units (EU), with 5 EU per replication. Each EU was 4.6-m wide and 9.1-m long. Residue retention rates were calculated based on reported average grain yields from Virginia in the period 2011-2017 (USDA-NASS, 2018) and calculated corn stover yields, based on an assumed harvest index (HI) of 0.45. The treatments retaining 0, 3.33, 6.66, 10 and 20 Mg dry matter ha^{-1} corresponded to 0, 33, 66, 100 and 200% of the total stover produced, respectively.

3.2. Experiment 2 and 3

Two experiments were conducted in New Kent, Virginia (NK) in the Coastal Plain region. Experiments were initiated at different crop phases of the same corn – wheat / soybean (CWS) rotation., with one set of residue return treatments initiated in the wheat phase of the rotation in June 2015 (NK1) and another with identical treatments, initiated after corn harvest in September 2015 (NK2). Soils in NK1 and NK2 were both classified as Altavista sandy loam (fine-loamy, mixed, semiactive, thermic Aquic Hapludults) (NRCS, 2018). Each experiment at NK utilized a RCBD with a factorial arrangement of sixteen treatments resulting from the combination of four residue retained rates each for corn (0, 3.33, 6.66 and 10 Mg ha^{-1}) and wheat (0, 1, 2 and 3 Mg ha^{-1}) (Table 1). Soybean residues were not harvested. Total annual retained residues from both corn and wheat were also calculated (Table 1). Four replications were established, resulting in 64 EU per experiment, with 16 EU per replication. Each EU 4.6-m wide by 9.1-m long, and residue retention rates both for corn and wheat were calculated as in Experiment 1 based on a HI of 0.45. Treatments retaining 0, 3.33, 6.66 and 10 Mg dry corn stover ha^{-1}, and 0, 1, 2, and 3 Mg dry wheat straw ha^{-1} corresponding to approximately 0, 33, 66 and 100% of the total stover and straw produced respectively.

Management applied to both Blacksburg and New Kent locations, including nutrient and pests control, fertility, plant density, and planting and harvesting dates, followed best recommendations practices to optimize corn grain yields for southwestern Virginia (CC rotation), and corn, wheat and soybean grain yields for eastern Virginia (CWS rotation), according to Virginia Cooperative Extension Recommendations (Brann et al., 2009). Detailed information on the management applied to these experiments can be seen in Table 2 (CC rotation, Blacksburg) and 3 (CWS rotation, New Kent).

3.3. GHG measurements

Measurement of CO_2, CH_4 and N_2O fluxes was performed by the non-static manual chamber method. In 2015, six to eight aluminum chambers (12.5-cm tall by 15-cm diameter) were placed randomly within non-trafficked inter-rows across the experimental area for baseline GHG measurements. Initial chamber installation for baseline GHG measures followed corn stover treatment allocation in September (NK2) and October (BB), and after wheat straw allocation in June (NK1). At least six months after treatment allocation, measurements in 2016 and 2017 were taken following corn and wheat planting and harvesting and corn nitrogen fertilization at V4 stage (Abendroth et al., 2011) in all four replications of each study. A timeline chart with the information of each component for both rotations and the gas sampling protocol for the different studies is shown in Figure 1.

For each sampling event, metal chambers were field installed four to six days before GHG collection. This schedule allowed the collection system to reach a new equilibrium state, ensuring proper soil-to-chamber contact. Chambers were hammered into the soil to a depth of 7.5-cm, leaving 5 cm of the chamber above ground level and open for normal soil-atmosphere dynamic exchange until gas sampling (Figure 2). For chamber-to-chamber headspace volume corrections, headspace height measurements were taken at four different points within the open space of each chamber. At sampling time, each chamber was covered with a plastic cap. The cap was fitted with two rubber septum stoppers, the first to allow gas collection and the second to avoid vacuum development during sampling. Capping times were recorded, and 30-ml gas samples collected one hour after the recorded capping time for each chamber. All sampling occurred between 1600 and 1800 h. Gas samples were then transferred to 30-ml pre-evacuated borosilicate glass vials assembled with butyl stoppers and aluminum seals (Figure 2). Before sampling, glass vials had been evacuated using a vacuum pressure pump by flushing alternate cycles of N_2 air (3 cycles) followed by CO_2-free air (4 cycles) for 3 minutes each time, for a total flushing time of 21 minutes per glass vial. Following gas sampling, all chambers were removed from the field plots until the next sampling time. Gas samples were analyzed for CO_2, CH_4 and N_2O relative concentrations with a GC-2014 Gas Chromatograph (Shimadzu Scientific Instruments, Columbia, MD). Chromatograph calibration followed the procedures of Lenka and Lad (2013). Briefly, a thermal conductivity detector and a flame ionization detector, both with a dual flow rate differential system, were utilized for CO_2 and CH_4 concentration analysis, respectively. Helium was used as the carrier gas, while hydrogen and hydrocarbon-free air were used as the flame gases for CH_4 detection. Nitrous oxide concentration was analyzed by electron capture detector with a fixed system using ^{63}Ni370MBq as radiation source and argon/methane mixture as the carrier gas. Detector temperature was set at 350 °C in all cases. Low, medium, and high standards concentrations were used for each gas (N_2O: 0.37, 1.00, and 5 ppm; CH_4: 1.00, 5.00, and 10 ppm; CO_2: 370, 1750, and

5000 ppm). Standards were run in duplicate to calibrate the chromatograph prior to each gas analysis.

3.4. GHG flux calculation

i. *Moles (n) of CO_2, CH_4 and N_2O per milliliter:*

$$n = \frac{10^{-3} * P}{R * T}$$ [Equation 1]

where n is in moles ml^{-1}; 10^{-3} = conversion factor (l ml^{-1}); $P = 1$ is the standard atmospheric pressure in atmospheres (atm); $R = 0.0821$ l atm K^{-1} mol^{-1} is the universal gas constant; and $T = 298$ K is the standard temperature in Kelvin degrees.

ii. *CO_2, CH_4 and N_2O gas mass per milliliter:*

$$G_m = n * C * M_w * 10^3$$ [Equation 2]

where G_m is the gas mass in mg ml^{-1}; C = gas concentration from chromatograph reading, as volume fraction; Mw = molecular weight of CO_2, CH_4 and N_2O (i.e., 44.0095; 16.0425; and 44.0128, respectively), in g mole^{-1}; and 10^3 = conversion factor (mg g^{-1}).

iii. *Total CO_2, CH_4 and N_2O gas mass per chamber:*

$$TG_M = G_m * V_c$$ [Equation 3]

where TG_M is the total gas mass per chamber, in mg; and Vc = volume of the chamber, in ml.

iv. *Flux of CO_2, CH_4 and N_2O gas flux per unit area and time:*

$$q = \frac{TG_M}{A * T} * K$$ [Equation 4]

where q is the flux of the gas in mg m^{-2} day^{-1}; A is the cross-sectional area of the chamber, in m^2; T = 1 is the total time the chamber was capped with a lid, in hours; and K = 24 is the conversion factor from hours to day. Because of the size of the flux, CO_2 flux is expressed in units of g m^{-2} day^{-1}.

3.5. Weather data

Weather data for the NK sites were obtained from the National Oceanic and Atmospheric Administration's (NOAA) West Point, VA, weather station from the National Climate Data Center (*https://www.ncdc.noaa.gov/cdo-web/search*). Total accumulated rainfall (mm), and daily average air temperatures (°C) were collected from beginning of May through end of October at New Kent, VA (Figure 3). Weather data obtained from the Kentland Farm NOAA weather station were used for specific comparisons but are not presented.

3.6. Data analysis

Linear regression analysis was conducted to examine the relationship between the flux of CO_2, CH_4 and N_2O following specific management practices for the treatment rates of residue retention using the PROC REG procedure in SAS version 9.4 (SAS Institute, 2014). The quadratic or linear models were selected based on R^2 values and model p-values. Regression equations for each gas flux as a function of either corn stover, wheat straw or the total summation of both were developed. Minimum, maximum and mean for each comparison were also calculated (Table 4). In three particular comparisons within a location (i.e., 2016 vs. 2017, BB; NK1 vs. NK2, 2016; and May vs. October 2017, NK2) analysis of variance GLIMMIX procedure in SAS version 9.4 (SAS Institute, 2014) was conducted to test the average effects of residue retained treatments on gas fluxes with significance set at $P \leq 0.10$. At Blacksburg, five corn stover retained rates (0, 3.33, 6.66, 10 and 20 Mg ha^{-1}) were considered as the treatment for the model effects both in 2016 and 2017. At New Kent, treatments applied in 2015 were the treatments considered in the analysis of GHG emissions in 2016. These included four stover retained rates applied at NK2: 0, 3.33, 6.66 and 10 Mg ha^{-1}, and four straw retained rates applied at NK1: 0, 1, 2 and 3 Mg ha^{-1}. In 2017, the total retained residues that resulted from the factorial combination of four corn stover treatments allocated in 2015 and four wheat straw treatments allocated in 2016 in NK2 were considered in the model. The INFLUENCE statement was used to assess for the presence of outliers.

4. RESULTS and DISCUSSION

4.1. Data analysis

Linear quadratic models were significant in four out of eighteen comparisons (Table 4). When differences were significant, CH_4 and N_2O, but not the CO_2 fluxes were affected by different rates of residue retention (Table 4). Multiple linear regression models were not significant ($p>0.10$) in any comparison. As a result, the parameters for the quadratic model, the higher order model in our comparisons, are presented in Table 4, with a, b, and c being numerical coefficients different than zero in the equation $Y = ax^2 + bx + c$, where Y represents the estimated CH_4, N_2O (in mg m^{-2} d^{-}) or CO_2 (in g m^{-2} d^{-1}) gas flux and x a particular residue retained treatment (in Mg ha^{-1}). When the quadratic models were significant, the derivative of the curve was used to calculate the inflection point as $x' = - b / 2a$, where x' represents either the maximum or minimum point of the quadratic curve across the range of responses.

4.2. Effect of corn stover retained rates on GHG emissions

Average CO_2, CH_4, N_2O post-corn planting measured in June differed between 2016 and 2017 in BB ($p \leq 0.0016$). Soil moisture content and soil temperature can have a profound impact on the rates of C mineralization, nitrification, denitrification and microbial respiration from soil, thus impacting the resulting GHG fluxes from soil (Davidson et al., 1998; Ball et al., 1999; Borken et al., 2003; Smith et al., 2003; Ball et al., 2008; Wei et al., 2014; Badagliacca et al., 2017). In 2017, mean CO_2, CH_4, N_2O fluxes across treatments were 43%, 23%, and 26% greater ($p \leq 0.001$) than similar measurements taken in 2016 (Table 4), likely due to the direct effect of greater soil moisture content (65% more precipitation within one week previous to readings) and indirect effect of greater air temperature (around 2°C more at reading time) on soil temperature in 2017. Lenka and Lal (2013), found greater CO_2 and N_2O fluxes with increases in both soil and air temperature. Similarly, increased anaerobic conditions resulting from greater soil moisture content resulted in greater CH_4 and N_2O fluxes in others experiments (Mosier et al., 2004; Lenka and Lal, 2013). However, conditions favoring greater gas flux emissions may have also generated more variability in the field samples collected in 2017, when standard errors for the calculated gas fluxes were, on average, a 13% greater than values calculated in 2016. In 2017, between around 4 and 12% of the total variability in gas flux emissions was explained by the fit quadratic models. In 2016, the resulting models explained between 26 and 52% of the total variability in the gas flux emissions (Table 4).

Rates of corn stover retained (i.e., 0, 3.33, 6.66, 10 and 20 Mg ha^{-1}) did not affect CO_2 emissions following corn planting in a continuous corn rotation at BB in 2016 ($p = 0.195$) or 2017 ($p = 0.494$). Corn stover retained rates affected the CH_4 and N_2O fluxes in 2016 ($p = 0.048; p = 0.051$, respectively) (Figure 4) but not in 2017 ($p = 0.363; p = 0.467$, respectively) (Table 4).

When differences were significant, calculated minimum points in the regression curve corresponded to corn stover retained rates (x-axis) of 5.07 Mg ha^{-1} for CH_4 and 4.90 Mg ha^{-1} for N_2O, respectively. Predicted CH_4 and N_2O responses at the minimum stover retained rates (y-axis) were 3.36 and 4.76 mg m^{-2} day^{-1} for CH_4 and N_2O, respectively. Increasing the amount of stover retained in the range 0 to about 5 Mg ha^{-1} (50% retained rate) gradually decreased CH_4 and N_2O fluxes, but reductions across the retention rates were lower than 3% in both cases, similar to reports from Baker et al. (2014) and Jin et al. (2014) in a 2 yr study conducted in Minnesota. On the other hand, stover retained rates at or greater than 5 Mg ha^{-1} gradually increased CH_4 and N_2O up to 24% and 25% (4.16 and 5.94 mg m^{-2} day^{-1}) respectively, corresponding to maximum retention rates of 20 Mg ha^{-1} (200% retained rate). In the studies conducted by Jin et al. (2014) and Baker et al. (2014), as well as in other studies (Dendooven et al., 2012; Abalos et al., 2013) comparisons did not include the 50% corn stover retained rate.

Corn stover treatments (i.e., 0, 3.33, 6.66 and 10 Mg ha^{-1}) did not affect any gas flux (CH_4, $p = 0.223$; N_2O, $p = 0.151$; CO_2, $p = 0.114$) emissions measured within 20 d of wheat harvest and soybean planting around mid-July 2016 at NK2. In this sampling time, resulting models explained between 13 and 25% of the total variability in the gas flux emissions (Table 4).

Similar to other reports (Jin et al., 2014; Guzman et al., 2015), CO_2 fluxes represented more than 90% of the total GHG emissions across all comparisons in both studies. The range of CO_2 flux in the loam soils at BB in 2016 and 2017 (3.83 to 8.11 g m^{-2} d^{-1}) more closely aligned with the range (5.70 to 9.51 g m^{-2} d^{-1}) of emissions reported by Wei et al. (2014), who worked with a clay soil. Those authors studied CO_2 emissions in a continuous corn rotation in response to soil moisture and temperature at different landscape positions. In both cases, the ranges of CO_2 flux were greater than values observed in the sandy loam soil at NK2 (2.78 to 6.86 g m^{-2} d^{-1}; Table 4).

At both locations, retaining 0 Mg ha^{-1} of corn stover for one (July, 2016; NK2) and two consecutive years (2017, BB) (Table 4) did not affect the emissions of any gas compared with the retention of 10 Mg ha^{-1} (100% retained rate) of corn stover. Similar results were recently found in by Guzman et al. (2015) in a study conducted under a continuous corn rotation at two locations in Iowa. In this study, retaining 0% or 100% of the corn stover had no effect on cumulative CO_2 emissions during the crop season (i.e., April through October) in 2 out of 3 years in a clay loam soil, and in the CO_2 and N_2O emissions across two years in a silty clay loam soil. These results likely occurred because only small differences in soil temperature and water content across residue treatments were measured in these well-drained soils, which was similar to the defining characteristics for the very deep soils used in our experiments. In other studies, retaining 0% corn stover at harvest reduced emissions of CO_2 by 4 to 11% (Dendooven et al., 2012; Baker et al., 2014; Jin et al., 2014), and N_2O by 7 to 51% (Dendooven et al., 2012; Abalos et al., 2013; Jin et al., 2014) compared with 100% stover retention, but did not affect CH_4 fluxes. In those studies, the cumulative fluxes of the three gases were compared for periods of time longer than a year, with more variable soil water content and temperatures over the period under analysis. Moreover, these studies did not compare the impact of varying levels of stover retention on GHG fluxes, and comparisons were restricted to treatments retaining either 0% or 100% corn stover.

4.3. Effect of wheat straw retained rates on GHG emissions

There was no significant effect on the post-corn harvest CH_4 ($p = 0.304$), N_2O ($p = 0.669$) and CO_2 ($p = 0.718$) emissions taken in mid-October 2016 after one year of differential wheat straw management at NK1 (Table 4). This is in contrast to Lenka and Lal (2013), who worked in a 15-year residue management study with three wheat straw application rates (0, 8, and 16 Mg ha^{-}

^{1} yr^{-1}) and two fertilization rates (0 and 244 kg N ha^{-1} yr^{-1}) under no-till management in Ohio. In their study, the 0 and 8 Mg ha^{-1} yr^{-1} retention rate treatments had lowest CO_2 (1.587 g m^{-2} d^{-1}) and N_2O (0.510 mg m^{-2} d^{-1}) emissions for both fertilization rates. On the other hand, retaining 16 Mg ha^{-1} yr^{-1} of wheat straw combined with N application increased average CO_2 (2.306 g m^{-2} d^{-1}) and N_2O (1.020 mg m^{-2} d^{-1}) fluxes by 45% and 100%, respectively. Incorporating wheat straw at 8 and 16 Mg ha^{-1} yr^{-1} (both with and without fertilizer) resulted in CH_4 emission ranging from 0.108 to 3.153 mg m^{-2} d^{-1} (Lenka and Lal, 2013). The discrepancy with our results is likely explained by a combination of factors in Lenka and Lal (2013) not reproduced in our study, including a) the use of maximum straw retained rates more than 5 times greater than respective rates used in our experiment (16 vs. 3 Mg ha^{-1} yr^{-1}, respectively); b) the likely cumulative effect of 15 yr of wheat straw management, compared with the single year-effect in our study at the time when measurements were collected; c) the use of N as a factor under analysis, which interacted with straw rates for all three gas fluxes, and greatly affected the flux of all gases in their study; and d) the lack of crop grown over the 15 yr where treatments were allocated. The authors also used straw from external sources, a situation that does not reproduce certain conditions found in productive environments under intensive management, like the presence of plant roots and the traffic of machinery for different labors. In spite of this, the range of responses for CH_4 (0.04 to 3.93 vs. 0.11 to 3.15 mg m^{-2} d^{-1}), N_2O (0.02 to 0.92 vs. 0.46 to 1.02 mg m^{-2} d^{-1}), and CO_2 (0.22 to 2.14 g m^{-2} d^{-1} vs. 1.51 to 2.31), in our study were within the range of estimates reported by Lenka and Lal (2013), respectively.

Lowest absolute values for CH_4 (0.04 mg m^{-2} d^{-1}), N_2O (0.02 mg m^{-2} d^{-1}), and CO_2 (0.22 g m^{-2} d^{-1}) across locations and years (Table 4) were measured at this sampling time (October 10, 2016). Following one cycle of residue management and for plots with similar soil series and fertility, CH_4, N_2O and CO_2 emissions across wheat straw retention rates of 0, 33, 66, and 100% in mid-October at NK1 were 58, 92 and 81% lower (*$p < 0.0001$*) than measurements taken in mid-July at NK2 across similar corn retention rates (Table 4). These differences may be explained as a result of both the three-fold lower amount of wheat residue applied relative to corn stover, as well as the lower air temperature around sampling time in the plot where wheat (15.5 °C, October 10, 2016) and corn stover treatments were allocated in the previous year (25.3 °C on July 12, 2016, respectively). The impact of air temperature on soil temperature, a main factor controlling the rates of GHG emissions in different environments, has been extensively cited in the literature (Smith et al., 2003; Dalal and Allen, 2008; Wei et al., 2014; Gritsch et al., 2015). These observations are in agreement with reports from Lenka and Lal (2013) who calculated positive correlations between air (0.71 and 0.64) and soil temperature (0.68 and 0.45) and CO_2 and N_2O fluxes, respectively.

4.4. Combined effect of corn stover and wheat straw retained rates on GHG emissions

Total retained rates resulting from the summation of corn and wheat residue treatments allocated in 2015 and 2016 in NK2 affected both the CH_4 ($p = 0.020$) and the N_2O ($p = 0.049$) but not the CO_2 ($p = 0.135$) (Table 4) fluxes measured after a corn V4 nitrogen fertilization on May 29, 2017. When differences were significant, calculated inflection points in the regression curve corresponded to total residue retained rates (x-axis) of 5.59 Mg ha^{-1} for CH_4 and 5.44 Mg ha^{-1} for N_2O, respectively. Predicted CH_4 and N_2O responses at the maximum total retained rates (y-axis) were 4.30 and 5.79 mg m^{-2} day^{-1} for CH_4 and N_2O, respectively. However, significant differences in both CH_4 and N_2O fluxes across treatments in this sampling time do not seem to be biologically meaningful. The best fitting line for each quadratic model had only slight curvature across the range of predicted values for the residue retained treatments (Figure 5). In fact, the range between the minimum and maximum predicted values across the sixteen treatments for both gas fluxes was less than 10% (Figure 5).

In the same experiment, residue treatments did not have an effect on the CH_4 ($p = 0.138$), N_2O ($p = 0.885$), or CO_2 ($p = 0.682$) fluxes (Table 4) measured following corn harvest on October 16, 2017. Despite of the similarity among treatments, the dispersion of the data at the different sampling times were quite contrasting. While the difference between the absolute maximum and minimum CH_4, N_2O, CO_2, fluxes were less than 10% in all cases at the end of the May sampling time, the differences for similar comparisons were, on average, 24% for the same measurements taken in mid-October. In fact, the magnitude of standard errors for the measurements taken in October represented anywhere between 32 and 54% of the average value for each gas flux. In both cases, gas samples were taken more than 1.5 y after the corn stover treatments allocation in September 2015. However, less than a year elapsed between wheat straw treatment allocation in June 2016 and the post-V4 fertilization sampling in May 2017. Conversely, when samples were taken in mid-October 2017, surface straw residues had been on the soil surface for a longer time (about 1.5 yr; Figure 1), thus exposed to the additional weathering effects of higher temperatures and accumulated rainfall during summer 2017 (Figure 3). As a result of this longer weathering process, intact straw biomass left on the surface was lower in mid-October and may in turn have fostered more interactions with the already decomposed corn residues. This likely created the greater GHG data dispersion in the measurements taken later in the year. Unfortunately, there is a noticeable lack of experiments addressing the combined effects of corn and wheat residues simultaneously removed or retained in a corn-wheat/soybean rotation. To the best of our knowledge, no other similar field experiment has been reported in the literature. Recent evidence from a four-year study conducted in Illinois on silty clay loam and silt loam soils with no residue management applied indicated that maximum CO_2 and N_2O emissions for a corn-wheat/soybean rotation were greater than those emissions from a continuous soybean rotation but similar to emissions from a continuous corn and a corn-soybean rotation (Behnke et al., 2018).

Fertilization with 56 kg N ha^{-1} at the corn V4 stage greatly impacted the size of the GHG fluxes measured at the end of May compared with measurements taken four months later in the corn-wheat/soybean rotation. Lenka and Lal (2013) reported significant increases in CH_4, N_2O, and CO_2 fluxes following the addition of 244 Kg N ha^{-1} yr^{-1} versus no N addition in most comparisons. Based on data from several authors (Bouwman et al., 2002a,b; Stehfest and Bouwman, 2006; Novoa and Tejeda, 2006), De Klein et al. (2007) it is estimated that approximately 1% of the N present in both synthetic and organic fertilizers and in crop residues, would be emitted from soils as N_2O. Additionally, higher soil N availability following N fertilization stimulates residue decomposition (Duiker and Lal, 1999; Jacinthe et al., 2002; Kim and Dale, 2005) and reduces CH_4 oxidation (Lenka and Lal, 2013), resulting in both greater CH_4 and CO_2 emissions. In other cases, N fertilization with (Gentile et al., 2008) or without (McSwiney and Robertson, 2005; Smith et al., 2011) crop residues addition has resulted in significant increases in the N_2O fluxes from the soils. In our study, average CH_4, N_2O, and CO_2 fluxes within 10 days from a V4 nitrogen fertilization in corn were 19, 56 and 50% greater ($p < 0.0001$) than similar comparisons taken at the same plot in mid-October.

5. CONCLUSIONS

The objective of this study was to compare the relative GHG emissions resulting from various residue retained rates in two contrasting cropping systems in the Mid-Atlantic USA. Since absolute determination of the cumulative GHG emissions for a given space of time was not an objective of our study, we deemed appropriate the use of manual instead of automated chambers, which allowed us to use up to 64 chambers at the same time for GHG measurements at both NK location, a number 3 to 4 times greater than the amount of simultaneous automated chambers used in most studies.

Following one year of stover management, retaining approximately 50% (i.e., 5 Mg ha^{-1}) or more of the stover produced in the previous year gradually increased CH_4 and N_2O in the loamy soil in Blacksburg up to 24% and 25%, with maximum CH_4 and N_2O fluxes (4.16 and 5.94 mg m^{-2} day^{-1}, respectively) following 200% retention rates (i.e., 20 Mg ha^{-1}). Two cycles of corn stover management in this location, and one cycle of corn stover or wheat straw management in the sandy loam soils in both experiments at New Kent, did not affect measured fluxes of any gas.

The impact of higher soil and air temperatures, greater soil water contents and increased availability of N following fertilization on soil GHG fluxes was evident across different comparisons in our study, in agreement with an ample body of scientific evidence documenting strong positive correlations between these environmental factors and the size of the GHG fluxes from agricultural soils. The total time that aboveground residues undergo chemical decomposition also seems to be an important environmental control to the final GHG pool size and the range of measurable responses. The combined effect of corn and wheat residues simultaneously managed for extended uses had an impact in the CH_4 and N_2O fluxes following a corn V4 nitrogen fertilization at NK2, but these results were not biologically meaningful. In the other 4 comparisons, corn stover and wheat straw residues retained over a 2 yr period did not have an effect on GHG in our experiments. Strikingly, no previous studies have addressed this question, in spite of the extensive adoption of the corn-wheat/soybean rotation by farmers across the US and global humid temperate regions. To date, an overwhelming amount of studies have addressed the impact of 0, 50 and 100% corn stover removal or retained rates on the CO_2, CH_4 and N_2O fluxes from soils under a continuous corn rotation scheme. To a lesser degree, a similar approach has been used in wheat. While valuable information has been generated from these efforts, the continuous corn or wheat approach does not truly address the inherent complexity of current agricultural systems. Our study is a first effort aiming to understand the impacts that crop residue management in complex second-generation biofuel systems have on GHG emissions. More studies of this type are warranted to start building a body of knowledge, absent at this time, in this sense.

6. LITERATURE CITED

Abendroth, L.J., R.W. Elmore, M.J. Boyer, and S.K. Marlay. 2011. Corn growth and development. PMR 1009. Iowa State University Extension, Ames, Iowa.

Abalos, D., A. Sanz-Cobena, L. Garcia-Torres, J.W. van Groenigen, and A. Vallejo. 2013. Role of maize stover incorporation on nitrogen oxide emissions in a non-irrigated Mediterranean barley field. Plant Soil 364: 357-371.

Argonne National Laboratory. 2017. GREET Model. *At:* https://greet.es.anl.gov/ (accessed: 10/15/2018).

Aulakh, M.S., T.S. Khera, J.W. Doran, and K.F. Bronson. 2001. Denitrification, N_2O and CO_2 fluxes in rice-wheat cropping system as affected by crop residues, fertilizer N and legume green manure. Biol Fertil Soils 34:375–389. doi: 10.1007/s003740100420

Badagliacca, G., P. Ruisi, R.M. Rees, and S. Saia. 2017. An assessment of factors controlling N_2O and CO_2 emissions from crop residues using different measurement approaches. Biol. Fertil. Soils 53: 547–561. doi.org/10.1007/s00374-017-1195-z

Baker, J.M., J. Fassbinder, and J.A. Lamb. 2014. The impact of corn stover removal on N_2O emission and soil respiration: an investigation with automated chambers. BioEnergy Res. 7: 503-508.

Ball, B.C., I. Crichton, and G.W. Horgan. 2008. Dynamics of upward and downward N_2O and CO_2 fluxes in ploughed or no-tilled soils in relation to water-filled pore space, compaction and crop presence. Soil Till. Res. 101: 20-30.

Ball, B.C., A. Scott, and J.P. Parker. 1999. Field N_2O, CO_2 and CH_4 fluxes in relation to tillage, compaction and soil quality. Soil Till. Res. 53: 29-39.

Behnke, G.D., S.M. Zuber, C.M. Pittelkow, E.D. Nafziger, and M.B. Villamil. 2018. Long-term crop rotation and tillage effects on soil greenhouse gas emissions and crop production in Illinois, USA. Agric Ecosyst Environ. 261: 62-70

Borken, W., E.A. Davidson, K. Savage, J. Gaudinski, and S.E. Trumbore. 2003. Drying and Wetting Effects on Carbon Dioxide Release from Organic Horizons. Soil Sci. Soc. Am. J. 67:1888-1896

Bouwman, A.F., L.J.M. Boumans, and N.H. Batjes. 2002a. Emissions of N_2O and NO from fertilised fields: Summary of available measurement data. Glob. Biogeochem. Cycles 16(4), 1058, doi: 10.1029/2001GB001811, 2002.

Bouwman, A.F., L.J.M. Boumans, and N.H. Batjes. 2002b. Modeling global annual N_2O and NO emissions from fertilised fields. Glob. Biogeochem. Cycles, 16(4), 1080, doi: 10.1029/2001GB001812, 2002.

Brann, D.E., D.L. Holshouser, and G.L. Mullins. Agronomy Handbook 2000. Virginia Tech Cooperative Extension. Publication 424-100. 134 p.

Carter, M.R. 2002. Soil Quality for Sustainable Land Management: Organic Matter and Aggregation Interactions that Maintain Soil Functions. Agron. J. 94: 38-47.

Chum H., A. Faaij, J. Moreira, G. Berndes, P. Dhamija, et al. 2011. Bioenergy. *In:* IPCC Special Report on Renewable Energy Sources and Climate Change Mitigation [O. Edenhofer, R. Pichs-Madruga, Y. Sokona, K. Seyboth, P. Matschoss, S. Kadner, T. Zwickel, P.

Eickemeier, G. Hansen, S. Schlömer, C. von Stechow (eds)], Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, pp. 209 – 332.

Crutzen, P.J., A.R. Mosier, K.A. Smith, and W. Winiwater. 2008. N_2O release from agro-biofuel production negates global warming reduction by replacing fossil fuels. Atmos. Chem. Phys. 8: 389–395.

Dalal, R.C., and D.E. Allen. 2008. Greenhouse gas fluxes from natural ecosystems. Aust. J. Bot. 56: 369-407

Davidson, E.A., E. Belk, and R.D. Boone. 1998. Soil water content and temperature as independent or confounded factors controlling soil respiration in a temperate mixed hardwood forest. Global Change Biol. 4: 217-227

De Klein, C., R.S.A. Novoa, S. Ogle, K.A. Smith, P. Rochette, T.C. Wirth, B.G. McConkey, A. Mosier, and K. Rypdal. 2006. N_2O Emissions from Managed Soils, and CO_2 Emissions from Lime and Urea Application. 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Vol. 4. Agriculture, Forestry and Other Land Use

Del Grosso, S.J., W.J. Parton, A.R. Mosier, M.D. Hartman, J. Brenner, D.S. Ojima, and D.S. Schimel. 2001. Simulated interaction of carbon dynamics and nitrogen trace gas fluxes using the DAYCENT model. *In:* Modeling carbon and nitrogen dynamics for soil management. M.J. Shaffer, L. Ma, and S. Hansen (Eds), pp. 303-332.

Dendooven, L., L. Patiño-Zúñiga, N. Verhulst, M. Luna-Guido, R. Marsch, and B. Govaerts. 2012. Global warming potential of agricultural systems with contrasting tillage and residue management in the central highlands of Mexico. Agric. Ecosyst. Environ. 152: 50-58.

Derrien, D., C. Plain, P.E. Courty, L. Gelhaye, T.C.W. Moerdijk-Poortvliet, F. Thomas, A. Versini, B. Zeller, L.S. Koutika, H.T.S. Boschker, and D. Epron. 2014. Does the addition of labile substrate destabilise old soil organic matter? Soil Biol. Biochem. 76:149–160. doi: 10.1016/j.soilbio.2014.04.030

Duiker, S.W., and R. Lal. 1999. Crop residue and tillage effects on carbon sequestration in a Luvisol in central Ohio. Soil Till. Res. 52: 73-81

Fontaine, S., G. Bardoux, D. Benest, B. Verdier, A. Mariotti, and L. Abbadie. 2004. Mechanisms of the priming effect in a savannah soil amended with cellulose. Soil Sci. Soc. Am. J. 68:125. doi: 10.2136/sssaj2004.0125

Gentile, R., B. Vanlauwe, P. Chivenge, and J. Six. 2008. Interactive effects from combining fertilizer and organic residue inputs on nitrogen transformations. Soil Biol. Biochem. 40: 2375–2384.

Golden, J.S., R.B. Handfield, J. Daystar, and T.E. McConnell. 2015. An Economic Impact Analysis of the U.S. Biobased Products Industry: A Report to the Congress of the United States of America. A Joint Publication of the Duke Center for Sustainability & Commerce and the Supply Chain Resource Cooperative at North Carolina State University. *At:* https://biopreferred.gov/BPResources/files/EconomicReport_6_12_2015.pdf (accessed: 10/15/2018).

Gritsch, C., M. Zimmermann, and S. Zechmeister-Boltenstern. 2015. Interdependencies between temperature and moisture sensitivities of CO2 emissions in European land ecosystems. Biogeosciences 12: 5981–5993

Guzman, J., M. Al-Kaisi, and T. Parkin. 2015. Greenhouse Gas Emissions Dynamics as Influenced by Corn Residue Removal in Continuous Corn System. Soil Sci. Soc. Am. J. 79:612–625

Hartman, M.D., W.J. Parton, S.F Del Grosso, M. Easter, J. Hendryx, T. Hilinski, R. Kelly, C.A. Keough, K. Killian, S. Lutz, E. Marx, R. McKeown, S. Ogle, D.S. Ojima, K. Paustian, A. Swan, and S. Williams. 2016. The Daily Century Ecosystem, Soil Organic Matter, Nutrient Cycling, Nitrogen Trace Gas, and Methane Model. NREL-Colorado State University.At: http://warnercnr.colostate.edu/lam/shared/25hFJQJu/DayCent_Manual_07.29.201/6.pdf (accessed: 10/15/2018).

Hill, J., E. Nelson, D. Tilman, S. Polasky, and D. Tiffany. 2006. Environmental, economic, and energetic costs and benefits of biodiesel and ethanol biofuels. Proc Natl Acad Sci U S A 103: 11206-11210

Huang, Y., J. Zou, X. Zheng, Y. Wang, and X. Xu. 2004. Nitrous oxide emissions as influenced by amendment of plant residues with different C:N ratios. Soil Biol. Biochem. 36: 973-981. doi: 10.1016/j.soilbio.2004.02.009

IPCC. 2000. Implications of different definitions and generic issues . Land use, land-use change and forestry. *In:* R.T. Watson et al., editors, Cambridge University Press, UK. pp 375.

Jacinthe, P.A., R. Lal, and J.M. Kimble. 2002. Carbon budget and seasonal carbon dioxide emission from a central Ohio Luvisol as influenced by wheat residue amendment. Soil Tillage Res 67: 147–157.

Jin, V.L., J.M. Baker, J.M.F. Johnson, D.L. Karlen, R.M. Lehman, S.L. Osborne, T.J. Sauer, D.E. Stott, G.E. Varvel, R.T. Venterea, M.R. Schmer, and B.J. Wienhold. 2014. Soil greenhouse gas emissions in response to corn stover removal and tillage management across the US Corn Belt. BioEnergy Res. 7: 517-527.

Kim, S., and B.E. Dale. 2005. Life cycle assessment of various cropping systems utilized for producing biofuels: Bioethanol and biodiesel. Biomass and Bioenergy 29: 426–439.

Kim, S., B.E. Dale, and R. Jenkins. 2009. Life cycle assessment of corn grain and corn stover in the United States. Int. J. Life Cycle Assess. 14: 160-174.

Lenka, N.K., and R. Lal. 2013. Soil aggregation and greenhouse gas flux after 15 years of wheat straw and fertilizer management in a no-till system. Soil Tillage Res. 126: 78-89.

Lin, S., J. Iqbal, R. Hu, M. Shaaban, J. Cai, and X. Chen. 2013. Nitrous oxide emissions from yellow brown soil as affected by incorporation of crop residues with different carbon-to-nitrogen ratios: a case study in central China. Arch. Environ. Contam. Toxicol. 65:183–192.

Macedo, I.C., A.M. Nassar, A.L. Cowie, J.E.A. Seabra, L. Marelli, M. Otto, M.Q. Wang, and W.E. Tyner. 2015. Greenhouse Gas Emissions from Bioenergy. *In*: G.M. Souza et al., editors, Bioenergy & Sustainability: Bridging the Gaps, 72, SCOPE, Paris. France. pp. 582-617.

McSwiney, C.P., and G.P. Robertson. 2005. Nonlinear response of N_2O flux to incremental fertilizer addition in a continuous maize (Zea mays L.) cropping system. Global Change Biol. 11: 1712-1719

Mosier, A., R. Wassmann, L. Verchot, J. King, and C. Palm. 2004. Methane and nitrogen oxide fluxes in tropical agricultural soils, sources, sinks and mechanisms. Environment, Development and Sustainability 6: 11–49.

Novoa, R., and H.R. Tejeda, H.R. 2006. Evaluation of the N_2O emissions from N in plant residues as affected by environmental and management factors. Nutr. Cycl. Agroecosyst. 75: 29-46.

NRCS. 2018. Natural Resource Conservation Service. Web Soil Survey. *At:* http://websoilsurvey.nrcs.usda.gov/app/HomePage.htm (accessed: 10/15/2018).

Parton, W.J., J.W.B. Stewart, and C.V. Cole. 1988. Dynamics of C, N, P and S in grassland soils: a model. Biogeochemistry 5: 109-131.

SAS Institute. 2014. SAS 9.4 user's guide. SAS Inst., Cary, NC.

Shan, J., and X. Yan. 2013. Effects of crop residue returning on nitrous oxide emissions in agricultural soils. Atmos. Environ. 71:170–175.

Singh, A., D. Pant, N.E. Korres, A. Nizami, S. Prasad, and J.D. Murphy. 2010. Key issues in life cycle assessment of ethanol production from lignocellulosic biomass: Challenges and perspectives. Bioresour. Technol. 101: 5003–5012

Smith P., M. Bustamante, H. Ahammad, H. Clark, H. Dong, E. A. Elsiddig, et al. 2014. Agriculture, Forestry and Other Land Use (AFOLU). *In:* Climate Change 2014: Mitigation of Climate Change. Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change, O. Edenhofer et al., editors, Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Souza, G.M., M.V.R. Ballester, C.H. de Brito Cruz, H. Chum, B. Dale, V.H. Dale, E.C.M. Fernandes, T. Foust, A. Karp, L. Lynd, R.M. Filho, A. Milanez, F. Nigro, P. Osseweijer, L.M. Verdade, R.L. Victoria, and L. Van der Wielen. 2017. The role of bioenergy in a climate-changing world. Environ. Dev. 23: 57-64.

Stehfest, E., and L. Bouwman. 2006. N_2O and NO emission from agricultural fields and soils under natural vegetation: summarizing available measurement data and modeling of global annual emissions. Nutr. Cycl. Agroecosyst., 74: 207-228.

Tenenbaum, D.J. 2008. Food vs. Fuel: Diversion of Crops Could Cause More Hunger. Environ. Health Persperct. 116: A254–A257.

Thompson, P.B. 2012. The Agricultural Ethics of Biofuels: The Food vs. Fuel Debate. Agriculture J. 2: 339-358.

USDA-NASS. 2018. Quick stats Lite (Beta). *At:* http://www.nass.usda.gov/Quick_Stats/Lite/index.php (accessed: 10/15/2018).

Wang, M., J. Han, J.B. Dunn, H. Cai, and A. Elgowainy. 2012. Well-to-wheels energy use and greenhouse gas emissions of ethanol from corn, sugarcane and cellulosic biomass for US use. Environ. Res. Lett. 7. doi:10.1088/1748-9326/7/4/045905

Wang, M.Q, J. Han J, Z. Haq, W.E. Tyner, M. Wu, and A. Elgowainy. 2011. Energy and greenhouse gas emission effects of corn and cellulosic ethanol with technology improvements and land use changes. Biomass Bioenergy 35: 1885–1896.

Wei, S., X. Zhang, N.B. McLaughlin, A. Liang, S. Jia, X. Chen, and X. Chen. 2014. Effect of soil temperature and soil moisture on CO2 flux from eroded landscape positions on black soil in Northeast China. Soil Tillage Res. 144: 119-125.

Wilhelm, W.W., J.M. Johnson, J.L. Hatfield, W.B. Voorhees, and D.R. Linden. 2004. Crop and soil productivity response to corn residue removal: a literature review. Agron. J. 96: 1-17.

Zabed, H., J.N. Sahu, A.N. Boyce, and G. Faruq. 2016. Fuel ethanol production from lignocellulosic biomass: An overview on feedstocks and technological approaches. Renew. Sust. Energ. Rev. 66: 751-774

7. TABLES AND FIGURES

Table 1. Residue sources and application rates for a field-based greenhouse gas emissions study in Blacksburg and New Kent, VA.

Location	Treatment	Corn stover	Wheat straw	Total residue†
		Mg ha^{-1}		
Blacksburg	1	0.00	-	-
	2	3.33	-	-
	3	6.66	-	-
	4	10.00	-	-
	5	20.00	-	-
New Kent (NK1 and NK2)	1	0.00	0.00	0.00
	2	0.00	1.00	1.00
	3	0.00	2.00	2.00
	4	0.00	3.00	3.00
	5	3.33	0.00	3.33
	6	3.33	1.00	4.33
	7	3.33	2.00	5.33
	8	3.33	3.00	6.33
	9	6.66	0.00	6.66
	10	6.66	1.00	7.66
	11	6.66	2.00	8.66
	12	6.66	3.00	9.66
	13	10.00	0.00	10.00
	14	10.00	1.00	11.00
	15	10.00	2.00	12.00
	16	10.00	3.00	13.00

† Total retained residues here refer to the summation of corn stover and wheat straw residue treatments allocated in 2015 and 2016. Soybean residues, even when part of the rotation, are not considered here.

Table 2. Detail of the management practices applied to continuous corn in the period 2015-2017 at Kentland Farm, Blacksburg, VA

	2015	2016	2017
Fertilizer application date	4-May		
		4-Jul	19-Jun
$N–P_2O_5–K_2O$, kg ha^{-1}	43-43-43		
		56-0-0	56-0-0
Seeding date	3-May	9-May	11-May
Tillage	No-till	No-till	No-till
Genotype	P1498HR	P1498HR	P1498HR
Seed rate, seeds ha^{-1}	69189	69189	69189
Seed rate, kg ha^{-1}	-	-	-
Row width, cm	76	76	76
Herbicides date	4-May (pre-emerg); 10-June (post-emerg)	9-May (all); 24-May (barley silage)	2-May
Pre-emergent			
2,4 D (Dimethylamine salt) ¶		0.498 kg a.i. ha^{-1}	0.498 kg a.i. ha^{-1}
Bicep II Magnum (Atrazine+S-Metolachlor) ƛ	0.573 kg a.i. ha^{-1} + 0.351 kg a.i. ha^{-1}	0.717 kg a.i. ha^{-1} + 0.439 kg a.i. ha^{-1}	0.717 kg a.i. ha^{-1} + 0.439 kg a.i. ha^{-1}
Atrazine 4L (Atrazine) ‡	0.235 kg a.i. ha^{-1}		
Glystar plus (Glyphosate) §	0.919 kg a.i. ha^{-1}	0.316 kg a.i. ha^{-1}	0.316 kg a.i. ha^{-1}
Post-emergent			
Glystar plus (Glyphosate) §	0.460 kg a.i. ha^{-1}	0.316 kg a.i. ha^{-1}	
Insecticide date	None	None	None
Fungicide date	None	None	None
Irrigation management	None	None	None
Harvest date	8-Oct	Not harvested	20-Oct

¶ (46.8%) 2,4-Dichlorophenoxyacetic acid

ƛ Atrazine (33.0%) 2-chloro-4-ethylamino-6-isopropylamino-striazine; S-Metolachlor (26.1%) Acetamide, 2-chloro-N-(2-ethyl-6-methylphenyl)-N-(2-methoxy-1-methylethyl],(S)

‡ Atrazine (41.9%) 2-chloro-4-ethylamino-6-isopropylamino-striazine

§ Glyphosate (41.0%) (N-(phosphonomethyl)glycine)

Table 3. Detail of the management practices performed in NK1 in the period 2014-2017 at Lanexa, New Kent, VA*

	2014	2015		2016		2017
	Wheat	Wheat (cont.)	Soybean	Corn	Wheat	Wheat (cont.)
Fertilizer application date	5-Nov	19-Mar	-		2-Nov	15-Mar
		2-Apr				5-Apr
N–P_2O_5–K_2O, kg ha^{-1}	40-60-80	67-0-0	-		40-60-80	67-0-0
		67-0-0	-			67-0-0
Seeding date	28-Oct	-	23-Jun	15-Apr	25-Oct	-
Tillage	No-till	No-till	No-till	No-till	No-till	No-till
Genotype	USG 3404	-	Asgrow 5332	Channel 197-31	USG 3404	USG 3404
Seed rate, seeds ha^{-1}	-	-	346,000	59,300		
Seed rate, kg ha^{-1}	0	-	-	-	135	-
Row width, cm	19	19	19	76	19	19
Herbicides			-			
Pre-emergent				15-Apr		
Atrazine 4L (Atrazine) ẞ	-	-	-	0.235 kg a.i. ha^{-1}	-	-
Glystar plus (Glyphosate) ∑	-	-	-	0.919 kg a.i. ha^{-1}	-	-
Post-emergent	23-Nov	1-Apr	-	15-May	21-Nov	3-Apr
Metribuzin 75 (Metribuzin) ¥	0.039 kg a.i. ha^{-1}	-	-	-	0.039 kg a.i. ha^{-1}	-
Prowl H2O (Pendimethalin) ₤	0.309 kg a.i. ha^{-1}	-	-	-	0.309 kg a.i. ha^{-1}	-
Glystar plus (Glyphosate) ∑	-	-	-	0.460 kg a.i. ha^{-1}	-	-
Harmony SG (Thifensulfuron-methyl) ¶	-	0.092 kg a.i. ha^{-1}	-	-	-	0.092 kg a.i. ha^{-1}
Insecticide date	23-Nov	1-Apr	-	-	23-Nov	1-Apr
Baythroid XL (ß-cyfluthrin) Ω	0.002 kg a.i. ha^{-1}	-	-	-	0.002 kg a.i. ha^{-1}	-
Warrior T (Lambda-cyhalothrin) †	-	0.001 kg a.i. ha^{-1}	-	-	-	0.001 kg a.i. ha^{-1}
Fungicide date	-	1-Apr; 14-May	-		-	2-Apr; 11-May
QuiltXcel (Azoxystrobin + Propiconazole) ƛ	-	0.022 + 0.017 kg a.i. ha^{-1}		-	-	0.022 + 0.017 kg a.i. ha^{-1}
Prosaro (Prothioconazole + Tebuconazole) §	-	0.020 + 0.020 kg a.i. ha^{-1}		-	-	0.020 + 0.020 kg a.i. ha^{-1}
Growth regulator date	None	1-Apr	-	-	-	3-Apr
Palisade EC ‡	-	0.002 kg a.i. ha^{-1}	-	-	-	0.002 kg a.i. ha^{-1}
Irrigation management	None	None		None		None
Harvest date	-	22-Jun		6-Sep	-	22-Jun

*Plot 2, initiated after corn harvest in September 2015, had similar treatments than Plot 1.

ꟹ Atrazine (33.0%) 2-chloro-4-ethylamino-6-isopropylamino-striazine; S-Metolachlor (26.1%) Acetamide, 2-chloro-N-(2-ethyl-6-methylphenyl)-N-(2-methoxy-1-methylethyl],(S)

∑ Glyphosate (41.0%) (N-(phosphonomethyl)glycine)

¥ Metribuzin (75%): 4-Amino-6-(1,1-dimethylethyl)-3-(methylthio)-1,2,4-triazin-5 (4H)-one

₤ Pendimethalin (38.%): N-(1 -ethylpropyl)-3,4-dimethyl-2,6-dinitrobenzenamine

¶ Thifensulfuron-methyl (50%): Methyl 3-[[[[(4-methoxy-6-methyl-1,3,5-triazin-2-yl) amino]carbonyl]amino]sulfonyl]-2-thiophenecarboxylate

Ω ß-cyfluthrin (12.7%): Cyano(4-fluoro-3-phenoxyphenyl)methyl-3-(2,2-dichloro-ethenyl)-2,2-dimethyl-cyclopropanecarboxylate

† Lambda-cyhalothrin (11.4%) : [1α(S*),3α(Z)]-(±)-cyano-(3-phenoxyphenyl)methyl-3-(2-chloro-3,3,3-trifluoro-1-propenyl)-2,2-dimethylcyclopropanecarboxylate

ƛ Azoxystrobin (13.5%): Methyl (E)-2-{2-[6-(2-cyanophenoxy)pyrimidin-4-yloxy]phenyl}-3-methoxyacrylate ; Propiconazole (11.7%): 1-[[2-(2,4-dichlorophenyl)-4-propyl-1,3-dioxolan-2-yl]methyl]-1H-1,2,4-triazole

§ Prothioconazole (19%): 2-[2-(1-Chlorocyclopropyl)-3-(2-chlorophenyl)-2-hydroxypropyl]-1, 2-dihydro-3H-1, 2,4-triazole-3-thione ; Tebuconazole (19%): alpha-[2-(4-chlorophenyl)ethyl]-alpha-(1, 1-dimethylethyl)-1H-1, 2,4-triazole-1-ethanol

‡ Trinexapac-ethyl (12.0%): Cyclohexanecarboxylic acid, 4-(cyclopropylhydroxymethylene)- 3,5-dioxo-, ethyl ester(95266-40-3)

Table 4. Quadratic regression parameters, R square coefficient, p-value, minimum, maximum and mean response values for the flux of CH_4 (mg m^{-2} day^{-1}), N_2O (mg m^{-2} day^{-1}), and CO_2 (g m^{-2} day^{-1}) in 2016 and 2017 in Blacksburg and New Kent, Virginia.

Blacksburg

Year †		Parameter a	b	c	R-sq	Pr>t	Minimum	Maximum	Mean
2016	CH_4, mg m^{-2} day^{-1}	0.0036	-0.0365	3.45	0.514	0.048	2.74	4.52	3.55
	N_2O, mg m^{-2} day^{-1}	0.0052	-0.0510	4.89	0.516	0.051	3.91	6.52	5.06
	CO_2, g m^{-2} day^{-1}	0.0048	-0.0556	4.69	0.258	0.195	3.83	6.40	4.78
2017	CH_4, mg m^{-2} day^{-1}	-0.0011	0.0413	4.18	0.121	0.363	3.42	5.01	4.38
	N_2O, mg m^{-2} day^{-1}	-0.0025	0.0612	6.19	0.042	0.467	5.10	7.51	6.39
	CO_2, g m^{-2} day^{-1}	-0.0015	0.0523	6.60	0.067	0.494	5.59	8.11	6.86

New Kent, experiment 1

Year ‡		Parameter a	b	c	R-sq	Pr>t	Minimum	Maximum	Mean
2016	CH_4, mg m^{-2} day^{-1}	-0.2239	0.5448	1.34	0.110	0.304	0.04	3.93	1.71
	N_2O, mg m^{-2} day^{-1}	0.0600	-0.1371	0.96	0.020	0.669	0.02	0.92	0.41
	CO_2, g m^{-2} day^{-1}	0.0095	-0.5457	3.62	0.020	0.718	0.22	2.14	0.99

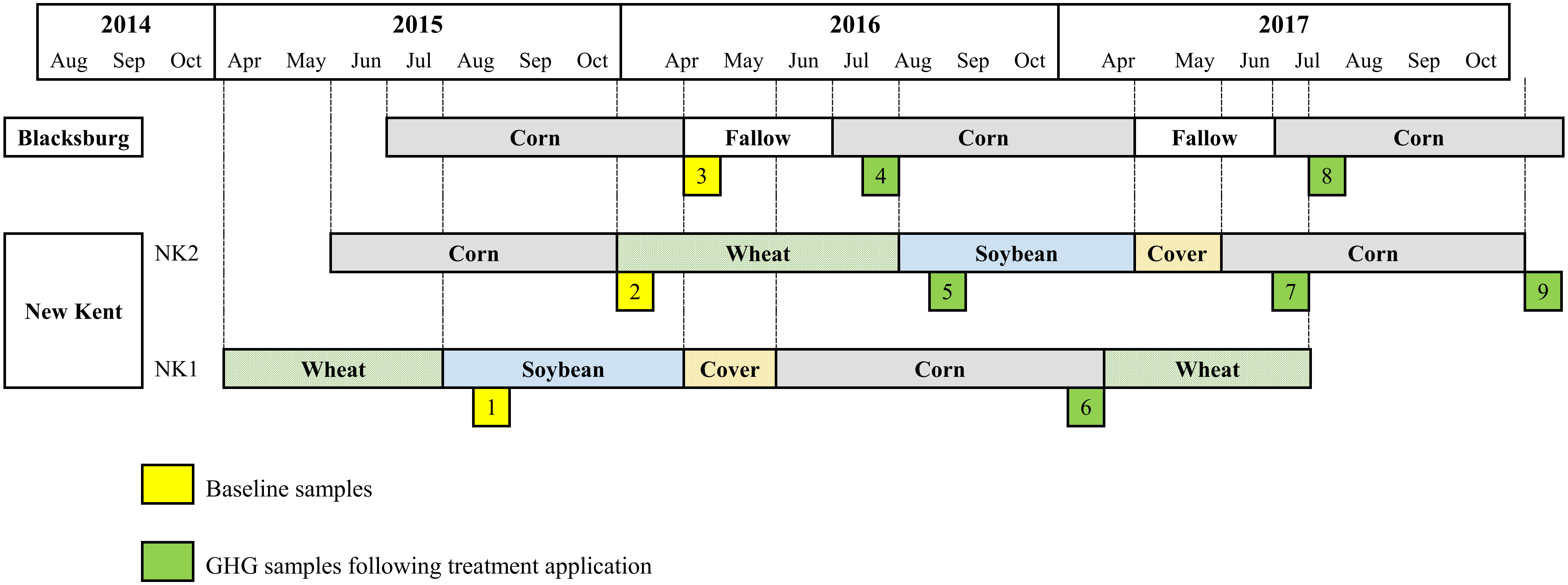
New Kent, experiment 2

Year ⁋		Parameter a	b	c	R-sq	Pr>t	Minimum	Maximum	Mean
July, 2016	CH_4, mg m^{-2} day^{-1}	0.0152	-0.1506	4.02	0.132	0.223	3.17	5.06	4.07
	N_2O, mg m^{-2} day^{-1}	0.0286	-0.2667	5.44	0.202	0.151	3.15	6.91	5.42
	CO_2, g m^{-2} day^{-1}	0.0375	-0.3377	5.23	0.249	0.114	2.78	6.86	5.27
May, 2017	CH_4, mg m^{-2} day^{-1}	-0.0080	0.0911	4.04	0.107	0.020	3.57	5.22	4.17
	N_2O, mg m^{-2} day^{-1}	-0.0096	0.1044	5.51	0.090	0.049	4.48	7.30	5.65
	CO_2, g m^{-2} day^{-1}	-0.0081	0.0780	5.97	0.074	0.135	5.19	7.78	6.02
October, 2017	CH_4, mg m^{-2} day^{-1}	-0.0258	0.4350	2.17	0.087	0.138	0.22	7.48	3.51
	N_2O, mg m^{-2} day^{-1}	0.0028	-0.0364	3.69	0.001	0.885	0.67	7.55	3.62
	CO_2, g m^{-2} day^{-1}	-0.0099	0.1942	3.28	0.012	0.682	0.43	10.06	4.01

† Measurements taken following corn planting on June 7, 2016 and June 12, 2017, corresponding to sampling times 4 and 8 in Figure 1, respectively.

‡ Measurements taken following corn harvest on October 16, 2016, corresponding to sampling time 6 in Figure 1.

⁋ Measurements taken following wheat harvest/soybean planting, N fertilization at V4 in corn, and corn harvest on July 12, 2016, May 29, 2017, and October 16, 2017, respectively. Dates corresponding to sampling times 5, 7 and 9 in Figure 1, respectively.



Experiment	Sampling time	Date	Management
NK1	1	15-Jul-15	Wheat harvest / soybean planting
NK2	2	1-Oct-15	Corn harvest - Baseline
BB	3	15-Oct-15	Corn harvest - Baseline
BB	4	7-Jun-16	Post corn planting
NK2	5	12-Jul-16	Wheat harvest / soybean planting
NK1	6	10-Oct-16	Corn harvest
NK2	7	29-May-17	Post-V4 N corn fertilization
BB	8	12-Jun-17	Post corn planting
NK2	9	16-Oct-17	Corn harvest

Figure 1. Chronological chart with the detail of the full crop rotations and the gas sampling protocol for Blacksburg and New Kent (NK1 and NK2) sites in Virginia (top); information including sequential sampling order, effective date of sampling and main crop management milestone at the time of gas sampling for each study (bottom).

Figure 2. Metal ring after field installation at the pre-sampling stage (left); following gas sampling from metal chamber, samples were evacuated into a 30-ml glass vial for posterior gas concentration analysis at the laboratory level (right).

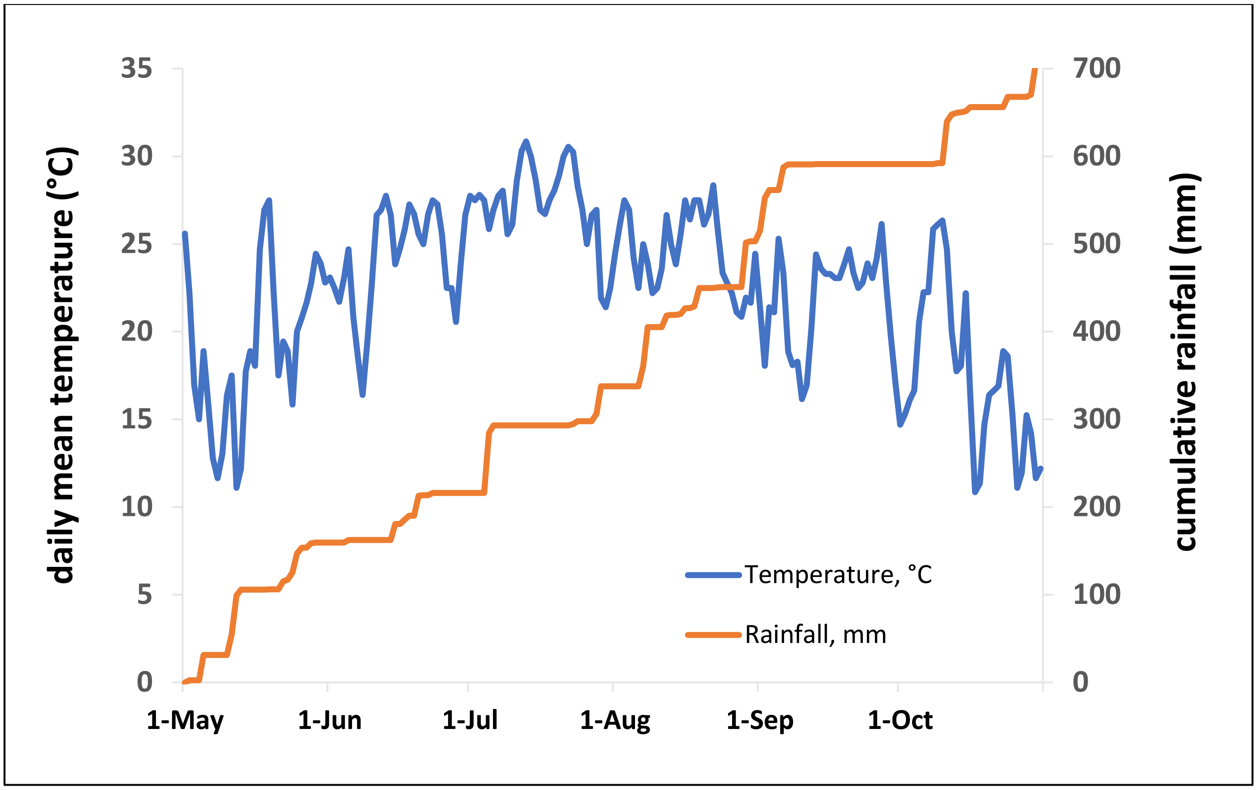


Figure 3. Total accumulated rainfall (in mm) and daily mean air temperature (in °C) for the period May through October 2017, in New Kent, VA. Source: National Oceanic and Atmospheric Administration, National Centers for Environmental Information (*https://www.ncdc.noaa.gov*).

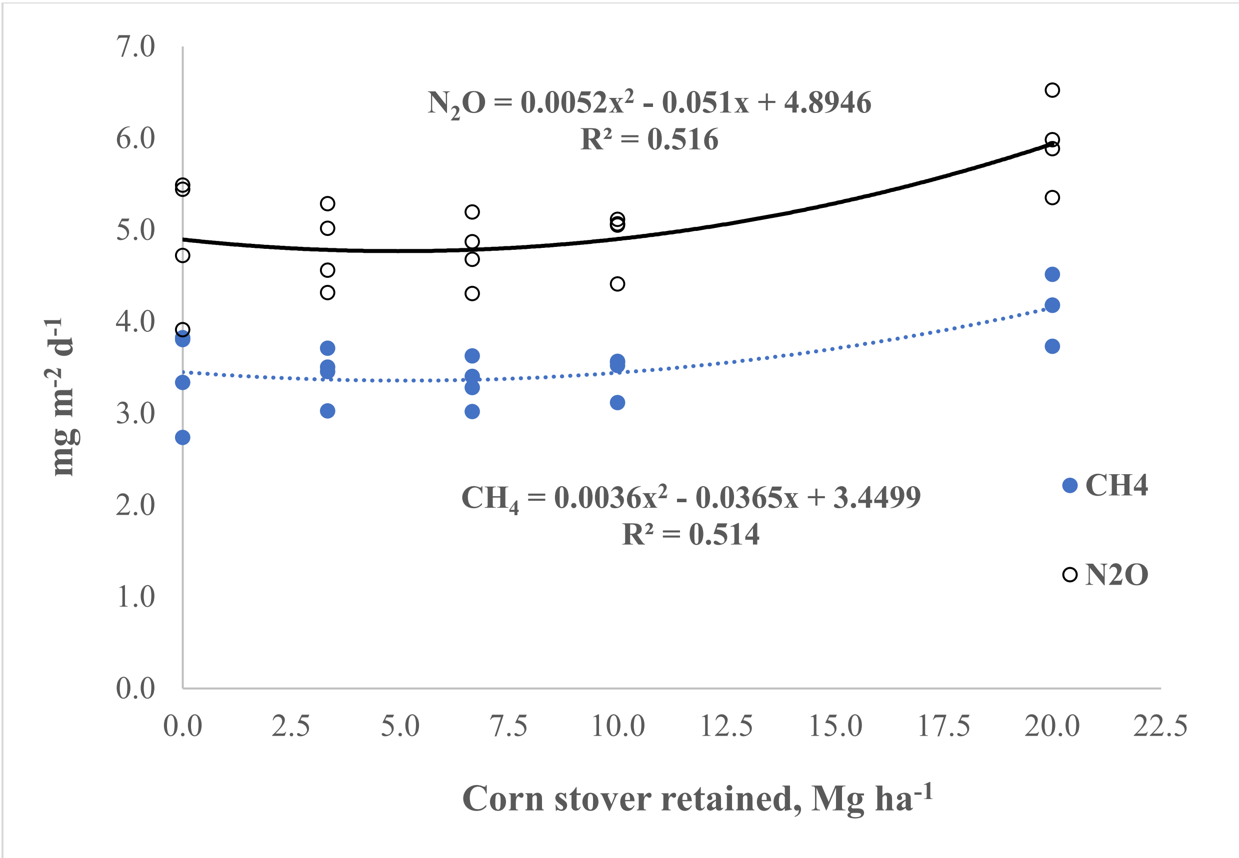


Figure 4. Flux of CH_4 and N_2O (mg m^{-2} day^{-1}) under five corn stover retained rates (in Mg dry matter ha^{-1}) following corn planting in a continuous corn rotation in 2016 in Blacksburg (BB), Virginia. Polynomial best fit line, quadratic equation and R-square are presented for each gas.

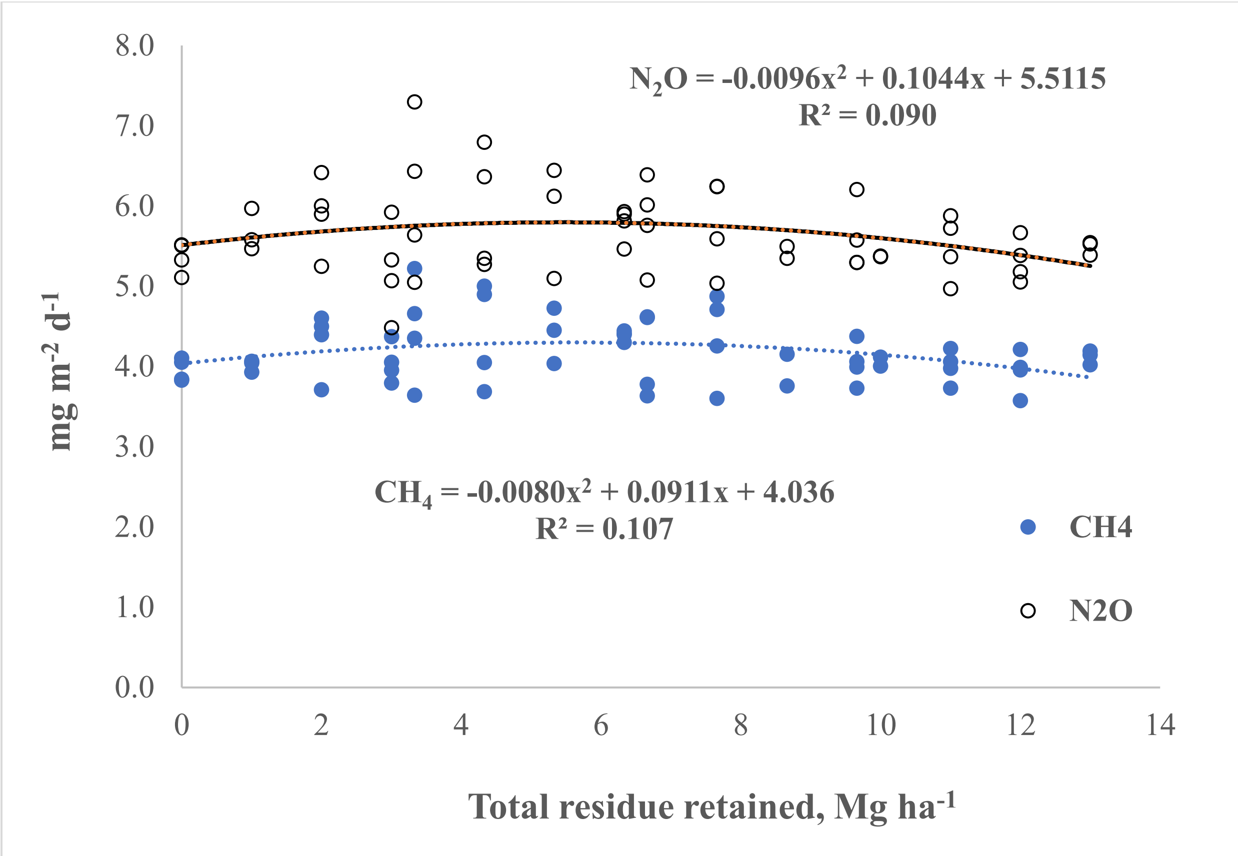


Figure 5. Flux of CH_4 and N_2O (mg m^{-2} day^{-1}) in a factorial design of sixteen total residue retained rates (in Mg dry matter ha^{-1}) from corn and wheat following a V4 nitrogen fertilization in corn in a corn - wheat/soybean rotation in 2017 in New Kent, experiment 2 (NK2), Virginia. Polynomial best fit line, quadratic equation and R-square are presented for each gas.

CHAPTER 4

Crop residue management effects on parameters of soil quality

1. ABSTRACT

The use of crop residues for biofuel production has the potential to provide with environmental and economic benefits to modern societies. Because of the profound impacts that crop residues have on agricultural productivity and soil health and quality, a sustainable utilization of these residues is required. Thus, one of the most important environmental effects to consider when removing biomass, either from annual or perennial species, is the effect on soil quality. Variable responses to residue removal (or retention) have been reported, but most such research has occurred in the Midwest and Great Plains. Little information is available on the effects of residue management for energy production in the Southeast. Thus, we determined crop yield and quality response to a range of biomass retention rates in grain cropping systems. Combinations of corn (*Zea mays* L.) stover (0, 3.33, 6.66 and 10 Mg ha^{-1}) and wheat (*Triticum aestivum* L.) straw (0, 1.0, 2.0, and 3.0 $Mgha^{-1}$) were soil applied in a corn-wheat/soybean (*Glycine max* L. Merr.) rotation in Virginia's Coastal Plain. Corn stover (0, 3.33, 6.66, 10 and 20 Mg ha^{-1}) was applied in a continuous corn cropping system in the Ridge/Valley province. For each system, residues were applied following grain harvest over two production cycles. Each experiment was conducted as a randomized complete design with four replications. Two cycles of crop residue management, with retention rates up to 20 Mg ha^{-1} of corn stover in Blacksburg and up to 13 Mg ha^{-1} of corn stover and wheat straw in New Kent, had no effect on total nitrogen (TN) and carbon (TC) concentrations,

CN ratios, bulk density (BD), soil pH, field capacity, permanent wilting point, plant available water and water aggregate stability across soil depths and aggregate sizes in Virginia. In one situation when residue management affected BD (0-2.5-cm depth, NK1) differences across the sixteen total retained residues treatments were less than 5%., thus rendering them not biologically or environmentally meaningful Overall, results of this study did not show any short-term impact resulting from various rates of crop residue retention in Virginia cropping systems. These incipient negative impacts resulting from very low rates of residue return warrant further studies to corroborate if these results are to be found following long-term scenarios of crop residue management.

2. INTRODUCTION

Over the last decade, economic and environmental concerns have spurred increased interest in the use of crop residues as sustainable, renewable sources for bioenergy and bio-products. Among other advantages, biofuels from lignocellulosic feedstocks have the potential to reduce reliance on imported fossil fuels and the greenhouse gases emissions, while fostering the farm economy and the development of rural communities (Hettenhaus et al., 2000; Wilhelm et al., 2004; Searchinger et al., 2008; Naik et al., 2010; Greenwell et al., 2012; Balan, 2014). However, the balance between stover removed and carbon returned to the fields must be considered due to potential increased erosion, decreased soil organic matter content and detrimental impacts in other soil quality parameters with stover removal (Karlen et al., 1994; Huggins et al., 1998; Wilhelm at el., 2004; Blanco-Canqui et al., 2006a; Benjamin et al., 2008; Kendall et al., 2015). Moreover, crop residues can act both sinks and sources of soil carbon that provide ecosystem services, and contribute to agricultural productivity by enhancing soil structure and stability (Wilhelm et al., 2007; Raffa et al., 2014) through their positive effects on soil organic carbon (SOC) and nitrogen (SON) stocks, nutrient availability, bulk density, water holding capacity, and water infiltration (Barber, 1979; Magdoff et al., 1997; Franzluebbers, 2002; Anderson-Teixera et al., 2009; Blanco-Canqui and Lal, 2009; Kenney, 2011; Sindelar, 2012).

As a result, not all of the stover produced will be available for use as a biofuel. Based on negative effects of corn stover removal on physical and chemical indicators of soil health such as soil organic matter (SOM) and SOC, particulate organic matter (POM), total soil carbon (TC) and nitrogen (TN), bulk density, water aggregate stability (WAS), among others, McAloon et al. (2000), Nelson (2002), Blanco-Canqui et al. (2006b) and Blanco-Canqui and Lal (2009) suggested that no more than 30% of corn crop residues can sustainably be removed in the US Corn Belt area. Other less conservative assessments indicate that 30 to 60% stover removal could be sustainably harvested in US (Morachan et al., 1972; Glassner et al., 1998; Kadam and McMillan, 2003; Graham et al., 2007; Nelson, 2002). Although the research data from the U.S. Corn Belt provide guarded optimism about residue harvest systems in the Midwest, such practices may not be appropriate to other geophysical regions of the country due to differences in soils, climate, and cropping systems. Recent interest from both farmers and bioenergy companies are growing the biomass industry in the southeastern U.S. with an expectation that cropping systems will supply biomass needed to sustain these new businesses. However, little information is available regarding sustainable crop residue harvest from the Southeast and the effects of residue removal on soil health parameters and greenhouse gas emissions remain to be defined. Only one study has specifically studied the potential supply and the supply chain of residues in the Southeast (Gonzalez et al., 2011), but this information provides no specific information to guide producers and industry about sustainable harvest levels for the region. The purpose of this research was to generate regionally relevant information on the short-term impacts of crop residue management on soil quality to help determine whether harvesting wheat straw and corn stover can be a sustainable practice for the region's cropping systems. The objectives of this experiment were to assess/evaluate the short-term impacts of various crop residue removal rates from common mid-Atlantic cropping systems in TC and TN, soil pH, bulk density, field moisture capacity (FC), permanent wilting point (WP) and WAS.

3. MATERIALS and METHODS

Three field experiments assessing the impact of crop residue removal were conducted over three years, from 2015 through 2017, in two physiographic regions in Virginia.

3.1. Experiment 1

The experiment was conducted Blacksburg, Virginia (BB), in the physiographic province known as Valley and Ridge. The Valley and Ridge region is characterized by the presence of sedimentary rocks including limestone, dolomite, and shale (Virginia Department of Environmental Quality, 2018). Soils in the experimental site are very deep, well drained, moderately permeable and with high available water in the profile, classified as Unison and Braddock loams (fine, mixed semiactive, mesic hapludults), formed from granite, gneiss, schist, sandstone, quartzite, and shale alluvium and colluvium parent materials, with slopes ranging from 2 to 7% (NRCS, 2018).

The experiment utilized a RCBD with five levels of corn stover retention: 0, 3.33, 6.66, 10, and 20 Mg ha^{-1} corn stover retained in a continuous corn (CC) grain rotation (Table 1). Four replications were established at BB resulting in a total of 20 experimental units (EU), with 5 EU per replication. Each EU was 4.6-m wide and 9.1-m long. Residue retention rates were calculated based on reported average grain yields of 8.7 Mg ha^{-1} in Virginia for the period 2011-2017 (USDA-NASS, 2018) and calculated corn stover yields, assuming a harvest index (HI) of 0.45. The treatments retaining 0, 3.33, 6.66, 10 and 20 Mg dry matter ha^{-1} corresponded to 0, 33, 66, 100 and 200% of the total stover produced, respectively (Table 1). A detailed timeline chart with the information of each component for both crop rotations and the proposed soil sampling framework time for this location is shown in Figure 1.

3.2. Experiment 2 and 3

Two experiments were conducted in New Kent, Virginia (NK) in the Coastal Plain region. The Coastal Plain region is composed mostly of unconsolidated deposits such as sand, clay, gravel and shell strata. Experiments were initiated at different crop phases of the same corn – wheat / soybean (CWS) rotation (Figure 1), with one set of residue return treatments initiated in the wheat phase of the rotation in June 2015 (NK1) and another with identical treatments, initiated after corn harvest in September 2015 (NK2). Soils in NK1 and NK2 are very deep, well drained, and with high available water in profile, and were both classified as Altavista sandy loam (fine-loamy, mixed, semiactive, thermic Aquic Hapludults) (NRCS, 2018). Each experiment at NK utilized a RCBD with a factorial arrangement of sixteen treatments resulting from the combination of four residue retained rates each for corn (0, 3.33, 6.66 and 10 Mg ha^{-1}) and wheat (0, 1, 2 and 3 Mg ha^{-1}),) with soybean residues completely retained in the C-W/S rotation (Table 2). Total annual retained residues from both corn and wheat were also calculated (Table 2). Four replications were established, resulting in 64 EU per experiment, with 16 EU per replication. Each EU 4.6-m wide by 9.1-m long, and residue retention rates both for corn and wheat were calculated as in Experiment

1 based on a HI of 0.45. Treatments retaining 0, 3.33, 6.66 and 10 Mg dry corn stover ha^{-1}, and 0, 1, 2, and 3 Mg dry wheat straw ha^{-1} (Table 2) corresponding to approximately 0, 33, 66 and 100% of the total stover and straw produced respectively.

Management applied to both Blacksburg and New Kent locations, including nutrient and pests control, fertility, plant density, and planting and harvesting dates, followed best recommendations practices to optimize corn grain yields for southwestern Virginia (CC rotation), and corn, wheat and soybean grain yields for eastern Virginia (CWS rotation), according to Virginia Cooperative Extension Recommendations (Brann et al., 2009).

3.3. Soil sampling and soil quality parameters determination

Baseline soil samples were taken in 2015 on a whole-plot bases at both locations, immediately after allocating the wheat straw treatments in NK1 (mid-June), and the corn stover treatments in NK2 (mid-September) and BB location (mid-October) (yellow boxes at the bottom of each timeline rotation in Figure 1). At this time, six and eleven intact soil cores were taken at three randomly chosen sites at the BB location (i.e., about half the size of each experiment in NK) and at each experiment in the NK location, respectively, from non-trafficked interrows, to a depth of 15-cm.

Soil samples were taken with a bulk density sampler with 5-cm diameter. Baseline soil samples were used to determine baseline routine soil analysis, soil bulk density, TC and TN. Two out of the six 0 to 15-cm depth cores were left intact, air-dried and submitted to the Virginia Tech Soil Testing Lab (https://www.soiltest.vt.edu/) for baseline routine soil analysis determination (Table 3). Briefly, nutrients available for plant uptake were extracted with Mehlich 1 solution (0.05N HCl in 0.025N H2SO4) (Mehlich, 1953) using a 1:5 vol:vol soil to extractant ratio and analyzed using ICP-AES analysis. Water pH was determined in a 1:1 (vol/vol) ratio of deionized water:soil (Thomas, 1996). Cation Exchange Capacity (CEC) was estimated by summation of the Mehlich 1 extractable non-acid generating cations (Ca, Mg and K), plus the acidity estimated from the Mehlich soil-buffer pH after conversion of all analytical results to meq/100 cm3 or cmol(+)/kg. The remaining four cores were separated in place into 0 to 2.5, 2.5 to 7.5, and 7.5 to 15-cm depth increments as suggested in a prior study conducted in Virginia by Spargo et al. (2012). Samples were air-dried and later used for determination of bulk density, TC and TN. Bulk density was determined for each soil depth increment as the weight of the intact air-dried soil over the corresponding depth volume. Small soils aliquots were oven dried (105° C, 24 h) to obtain an air-to-oven dry mass correction factor, in order to express values in an oven-dried basis (Arshad et al., 1996). Following, air-dry samples were gently crushed and passed through a 2-mm sieve to determine, per each soil depth increment, the TC and TN contents. Total soil organic C and total N were determined in duplicate after dry combustion of soil subsamples (between 1 to 1.5-g) that were first ground to a powder with an automatic mortar and pestle machine for 3 minutes and then analyzed through a dry combustion process using a VarioMax CNS macro elemental analyzer (Elementar, Mt. Laurel, NJ).

Final soil samples were taken in mid-June at NK1 (post-wheat harvest), and between mid-September and beginning of October in 2017 in NK2 and BB (post-corn grain harvest; red boxes in Figure 1). As per baseline soil samples, these samples were used for bulk density, TC and TN determination. Additionally, these samples were used for soil pH, C mineralization rates, water holding capacity and water aggregate stability determination. At sampling time, three intact soil cores were taken to a depth of 15-cm at each EU in the four replications, and each core was separated in place into 0 to 2.5, 2.5 to 7.5, and 7.5 to 15-cm depth increments and each depth was stored in a paper bag. For water stable aggregate stability measurements, a fourth soil sample was taken at each EU to a depth of 5-cm and stored in a fourth paper bag. Soil cores were then air-dried until constant mass weight and bulk density was determined at each the 0-2.5, 2.5-7.5, and 7.5-15-cm depths. Air-dried samples for each depth increment were then gently crushed to pass a 2-mm sieve for TC and TN determination, as previously described.

Soil pH was determined as previously described, for the soil depth increment 0 to 7.5-cm. Since this depth increment was not originally collected at the field level, samples were artificially created by compositing a proportional 1/3 and 2/3 portion from the 0 to 2.5-cm and 2.5 to 7.5-cm subsamples taken at each EU, respectively.

Water holding capacity [i.e., total amount of water a soil can hold at field moisture capacity (FC)] was determined for the 0 to 7.5-cm depth increment in pressure chambers set at a pressure of -0.033 MPa (Kirkham, 2014) in a 1500F2, 15 bar pressure plate extractor (Soil moisture equipment corp., Santa Barbara, CA). Protocol for sample preparation followed the same procedure than that used for soil pH samples. Following sampling preparation, soil samples were moistened and placed in chambers for 7 days. Recovered soil samples were then weighed and FC (%) calculated. With this information, permanent wilting point (WP; -1.5 MPa) (Kirkham, 2014) was calculated through chilled mirror technology from intact soil cores with a WP4 water potential meter (Decagon Devices, Inc., Pullman, WA). Briefly, three soil subsamples were taken from each soil sample and placed into plastic cups. A known amount of water was added, and then plastic cups were closed with caps. One of the subsamples was kept dry, in order to cover the FC-WP range. Following, samples were left to equilibrate for a week and then run through the WP4C device for water potential determinations. With the water content and water potential information for different subsamples, WP at -15 bars was then interpolated. Finally, plant available plant (PAW) was calculated as the difference between the FC and WP.

Water aggregate stability was measured in 0 to 5-cm soil samples, following a modified procedure from Six et al. (1998). Briefly, air-dried soil samples were gently crushed to pass a 4-mm sieve and representative samples containing all sizes from ~4mm were collected. Small soil aliquots were weighed and oven-dried (105°C, 24 h) for air-to-oven mass corrections (GWC, %). Following, 50-g soil samples were poured through two sieves (2000 µm on top of a 250 µm) on top of a solid pan and submerged in deionized water for 5 minutes. The entire stack of sieves was moved then up and down (approximate stroke length of 3-cm) about 50 times in 2 minutes. Aggregates remaining in the 2000 µm sieve were then washed off with deionized water, separated from roots and small rocks and collected and weighed. Soil aggregates were oven-dried overnight at 55°C and dry weights recorded. The same procedure was applied to the other two fractions (i.e., >250 and <250 µm). Material contained in the soil slurry in the pan (i.e., <250 µm) was washed onto a 53 µm sieve. Then, five g of the oven-dried aggregates for the >2000-, 2000 to 250- and 250 to 53-µm aggregates were collected to determine the sand content (%) by hydrometer method. Finally, sand-free aggregates weight was calculated. E.g., for a sample with 12% sand, and 15.0 g of 250 to 2000 µm size, calculations were as follows:

Sand-free aggregate weight = 15.0 * (1-12/100)) = 15.0 x 0.88 = 13.2 g of sand-free soil in the size range 250 to 2000 µm

3.4. Weather data

Weather data for the BB and NK sites were obtained from the National Oceanic and Atmospheric Administration’s (NOAA) Kentland Farm and West Point, VA, weather stations from the National Climate Data Center (*https://www.ncdc.noaa.gov/cdo-web/search*), respectively. Total accumulated rainfall (mm), and daily average air temperatures (°C) were collected from beginning of May through end of October at Blacksburg and New Kent, VA. The 30-yr average (1981-2010) rainfall and air temperature are also presented (Figure 2).

3.5. Data analysis

Linear regression analysis was conducted to examine the relationship between crop residue management on parameters of soil quality in 2017 using the PROC REG procedure in SAS version 9.4 (SAS Institute, 2014). Rates of crop residue return, yrs and locations were considered fixed effects, while blocks were considered random effects. The quadratic or linear models were selected based on R^2 values and model p-values. Regression equations for each soil quality parameter as a function of either corn stover (BB), or total summation of corn and wheat residues applied in previous years in both NK experiments were developed. Minimum, maximum and mean for each comparison were also calculated (Table 4, 5 and 6). At Blacksburg, five corn stover retained rates (0, 3.33, 6.66, 10 and 20 Mg ha^{-1}) (Table 1) applied in the previous two years were considered as the treatment for the model effects in 2017. Total retained residues were considered in the model at both experiment in NK location (Table 2). In NK1, total retained residues in 2017 resulted from the factorial combination of four wheat straw treatments allocated in 2015 and four corn stover treatments allocated in 2016. In NK2, total retained residues in 2017 resulted from the combination of four corn stover treatments allocated in 2015 and four wheat straw treatments allocated in 2016. Normality of data was assessed with the UNIVARIATE procedure. Prior to analysis, assumptions of equal variances for each group were visually checked by plotting the studentized residuals against predicted values. Across response variables and years, assumptions of homoscedasticity and approximation to normal distribution were met for all comparisons. Presence of outliers in Y was assessed with the INFLUENCE procedure of SAS.

4. RESULTS AND DISCUSSION

4.1. Weather data

Blacksburg and New Kent locations are located in a region classified as humid subtropical, according to the Köppen climate classification (Kottek et al., 2006). Annual long-term average (30-yr period, 1981-2010) precipitations and average air temperatures are typically higher in New Kent (1153 mm and 15.6 °C) compared with Blacksburg (1039 mm and 12.2 °C). The three years under study were wetter and slightly hotter in most comparisons than an average year at both locations. In New Kent, accumulated rains were 23, 43, and 13% greater than average for the period May-October in 2015, 2016, and 2017, respectively (Figure 2B and D). As a result of this, the remarkable rates of biomass decomposition that occur in this region of the US leave negligible amounts of undecomposed biomass from previous crops after 1 yr in the field (Thomason, personal communication), resulting in a robust estimation of the total corn and wheat residues produced in the previous season. Over the same subperiod, accumulated precipitations across years in BB (range: 615-679 mm) were 10% to 21% greater (Figure 2A) and air temperatures up to 1.6°C higher (Figure 2C) than the 30-year average. Although rates of decomposition in Blacksburg are expected to be lower compared to those occurring in New Kent, these conditions over the length of the study resulted in negligible amount of undecomposed material following 1 yr in the field.

4.2. Effect of corn stover and wheat straw retention on total soil nitrogen and carbon concentrations, CN ratio and bulk density

The retention of 0, 3.33, 6.66, 10.00, and 20.00 Mg ha^{-1} of corn stover over a two years period did not affect total soil nitrogen (TN) ($p>0.76$) or carbon (TC) ($p>0.44$), or the CN ratio ($p>0.26$) at any soil depth in the continuous corn rotation in Blacksburg (Table 4). Similarly, total retained residues resulting from the application of wheat straw and corn stover rates in NK1, and corn stover and wheat straw rates in NK2 in the period 2015/2016, respectively, did not affect TN ($p>0.50$) or TC ($p>0.51$), or the CN ratio ($p>0.21$) at any soil depth in the corn-wheat/soybean rotations in New Kent (Table 4). Maskina et at. (1993) measured the residual effects of previous no-till residue rates over an 8 yr period in a silty clay loam soil in Nebraska. During the first 3 yr (1978-1980), residue rates of 0, 50, 100 and 150% were retained for each crop in the rotation corn, sorghum and soybean. In the second 5 yr (1980-1984), sorghum treatment was omitted, and continuous corn and soybean were grown in two separate blocks in the same experimental area. Same annual retention rates were applied for each crop in this period. Residues applied ranged from 0 to 15 and 8 Mg ha^{-1} yr^{-1} for corn and soybean, respectively (Power et al., 1986), and average quantity of both crop residues varied from 0 to about 6 Mg ha^{-1} yr^{-1} (150% rate) (Maskina et al., 1993). Two and three yr after residue treatment were discontinued, soil samples were taken at the 0-30 cm and 0-7.5 cm depths, respectively, and composed each yr across corn and soybean blocks. Similar to our short-term results in New Kent, the addition of 100% crop residue rates or less over an 8 yr period did not affect TN and soil organic carbon (SOC; TC was not measured in this experiment) in the study conducted by Maskina et al. (1993). However, retaining more than 100% residue had a different long-term effect in TN and SOC, a situation that we did not observe in the short-term with corn retention rates of up to 200% in Blacksburg. In the Maskina et al. (1993)

study, 8 yr residual effect of 150% retention rates increased TN by 16% at the 0-30, and TN and SOC by 12% and 14% at the 0-7.5-cm depth, respectively, compared with 0 and 50% rates ($p<0.10$). Moreover, long-term organic matter levels only decreased by about 10% at the 0-7.5-cm depth when no residue was applied over the 8 yr period compared with 150%, but not with 50% and 100% retention rates (Maskina et al., 1993). Although the impact of crop residue management on these parameters has not been previously studied in Virginia, Spargo et al. (2012) studied the impact of time under no-till in soil organic C and N in sandy loam soils in the Coastal Plain region of Virginia. Similar to results from the long-term study conducted by Maskina et al. (1993), total soil organic C and N increased linearly with time under continuous no-till in the 0-2.5 and 2.5-7.5, but not in the 7.5-15-cm depths in Virginia (Spargo et al., 2012).

Bulk density (BD) was not affected by corn stover ($p>0.19$) or total residue retention rates ($p>0.51$) at any soil depth in Blacksburg or NK2, respectively. In NK1, total residues retained affected BD in the 0-2.5 ($p = 0.097$) (Table 4; Figure 3) but not in the 2.5-7.5 ($p = 0.796$) or the 7.5-20-cm ($p = 0.611$) depths (Table 4). When differences were significant at the 0-2.5-cm depth in NK1, the calculated minimum point in the regression curve corresponded to a total residue retained rate (x-axis) of 7.79 Mg ha^{-1} (approximately 60% retention rate). Predicted BD response (y-axis) at this minimum total retained rate in 2017 was 1.52 g cm^{-3}. As a result, BD at the 0-2.5-cm depth gradually decreased with increasing residue retention up to total retention rates around 60%, to further increase when more than 60% of the total residue was retained (Figure 3). However, significant differences in BD do not seem to be biologically or environmentally meaningful in the top 2.5-cm soil in NK1. The best fitting line for the quadratic model had a seemingly flat shape across the range of predicted values for the total retained treatments (Figure 3). In fact, the range between the minimum and maximum predicted values across the sixteen treatments was less than 5% (Figure 3). Similar to our results in loams (BB) and sandy loam (NK) soils, retention of up to 150% residue over an 8 yr period did not affect BD in the upper 7.5-cm of silty clay loam soils in measurements taken two (Maskina et al., 1993) and nine yr (Power et al., 1998) after residue treatment completion in studies conducted in Nebraska. In other studies, 100% corn stover retention did not affect BD in the first 20-cm in Québec (Dam et al., 2005) or 50-cm of soil in Iowa (Karlen et al., 1994), but reduced BD in the first 15-cm, compared with 0% retention, in other short (Sindelar, 2012) and long-term studies (Clapp et al., 2000; Dolan et al., 2006).

4.3. Effect of corn stover and wheat straw retention on field moisture capacity, permanent wilting point, plant available water and soil pH

Field moisture capacity (FC) and permanent wilting point (PW) were not affected ($p>0.14$) by corn stover or total residue retention rates at the 0-7.5-cm soil depth at any location (Table 5). As a result, the plant available water (PAW) between matric potentials of -1.5 and -0.033 MPa Mpa (WP and FC, respectively), was not affected ($p>0.75$) by different residue management at any location (Table 5). In the study conducted by Maskina et al. (1993), the residual effect of different crop retention rates over an 8 yr period did not affect ($p>0.10$) FC values measured at the 0-7.5-cm depth two yr following residue treatment completion. Karlen et al. (1994) did not find differences in PAW between 0 and 100% corn stover retention rates over a 10 yr period in a silt loam soil in Wisconsin. In this study, retaining 150% of the stover increased PAW by 11% compared with the 0% retention rate, but was not different than 100% retention rate. Different to our results, Wilhelm et al. (1986) found that returning 100 and 150% of the corn stover over a 4 yr period increased PAW at planting in a silty clay loam by 28% and 15% when compared with 0 and 50% return rates. However, different techniques used to determine PAW may likely explain, at least in part, some of the differences seen in each case. Measurements in the Wilhelm et al. (1986) study were determined for the 0 to 1.8-m depth increment and by means of the use of a neutron-scatter technique, compared to our determinations through soil sampling and lab techniques for the 0-7.5-cm soil depth. Moreover, differences in PAW are highly influenced by soil texture (Easton and Bock, 2016), which may in turn explain the differences between the moderate coarse and medium texture soils in our experiments, and the moderately fine silty clay loam soil in Wilhelm et al. (1986). Regardless of these differences in PAW, several authors reported on the positive impacts that corn stover returned had on soil available water (i.e., total available water at a given time minus WP) (Doran et al., 1984; Power et al., 1986; Blanco-Canqui et al., 2006b).

Retaining up to 20.00 Mg ha^{-1} of corn stover for two consecutive yr in Blacksburg or up to 13 Mg ha^{-1} of corn stover and wheat straw over a two yr period at both experiments in New Kent, did not impact the soil pH at the 0 to 7.5-cm depth (p> 0.45) (Table 5). Power et al. (1998) did not measured changes in soil pH (p.0.10) resulting from retention of up to 150% residue over an 8 yr period at neither the 0-7.5, 7.5-15, or 15-30-cm soil depths nine yr after residue treatment completion in Nebraska. In short-term studies with three or less cycles of residue management, soil pH did not change (p>0.05) in most comparisons in sandy loam soil in South Carolina Karlen et al. (1984) and in silt loams soils with 2 and 10% slopes, and clay loam soils with less than 1% slope in Ohio (Blanco-Canqui and Lal, 2009).

4.4. Effect of corn stover and wheat straw retention on water aggregate stability

Water aggregate stability (WAS) measured as the free-sand dry weight for the 0 to 5 cm depth was not affected by corn stover (BB; p>0.26) or total residue retention rates (NK; p>0.11) at any aggregate size (Table 6). In the Karlen et al. (1994) study, residue management had also no effect in the WAS when ≤ 100% of the corn stover was retained over a 10 yr period. In this study, WAS only increased by 43 and 31% when 200% of the corn stover was returned over 10 yr, compared with 0 and 100% retention rates, respectively. Working in a corn-soybean rotation in a silty clay loam soil in South Dakota, Hammerbeck et.al. (2012) reported a 40% increase in the WAS only for aggregate sizes between 0.84 and 2.0 mm when 100% of the stover was left on surface, compared with retention rates ≤ 50%. However, stover management did not affect WAS for other aggregate sizes (i.e., 2.0-6.4, 6.4-19.2, and >19.2 mm) in this study. In other cases, soil texture and climatic conditions may result in WAS being more susceptible to changes in residue management. In the long-term study conducted by Bordovsky et at. (1999) in coarse sandy soils in Texas, microaggregation values were 15% and 19% higher when residue was retained, both in non-irrigated (27.1 vs. 23.5 g kg^{-1}) and under irrigated conditions (32.3 vs. 27.1 g kg^{-1}).

5. CONCLUSIONS

Weather conditions at both the Blacksburg and New Kent locations were characterized by equally distributed, abundant precipitations and average air temperatures above average over the 3 years when studies were conducted. These conditions were conducive to optimum plant growth and resulted in high rates of crop residue decomposition applied in previous years. In our studies, systems were most likely at a state of equilibrium following several years of continuous crop residue incorporation and best management practices, including no-till systems to prevent wind and water erosion, adequate fertilization, maintenance of soil organic matter and soil structure. Under these conditions, a portion of the crop residue produced in current cropping systems could be sustainably harvested for extended uses (Karlen et al., 2011a) without negative impacts in parameters of soil quality. Two cycles of crop residue management in Virginia, with retention rates up to 20 Mg ha^{-1} of corn stover in Blacksburg and up to 13 Mg ha^{-1} of corn stover and wheat straw in New Kent, had no effect on parameters of soil quality like TN, TC, CN ratios, bulk density, soil pH, field capacity, permanent wilting point, plant available water and water aggregate stability across different soil depths and aggregate sizes. In the single situation when residue management affected bulk density at the 0-2.5-cm depth in NK1, differences in BD were not biologically or environmentally meaningful, as these differences across the sixteen total retained residues treatments were less than 5%. In all cases, retaining none of the residues produced in Virginia cropping systems did not result in negative short-term effects in our studies. These incipient negative impacts resulting from very low rates of residue return warrant further studies to corroborate if these results are to be found in the long-term.

6. LITERATURE CITED

Balan, V. 2014. Current Challenges in Commercially Producing Biofuels from Lignocellulosic Biomass. ISRN Biotechnology vol. 2014, Article ID 463074, 31 pgs. https://doi.org/10.1155/2014/463074

Benjamin, J.G., A.D. Halvorson, D.C. Nielsen, and M.M. Mikha. 2008. Crop management effects on crop residue production and changes in soil organic matter in the central Great Plains. Agron. J. 102:990-997.

Blanco-Canqui, H., and R. Lal. 2009. Corn stover removal for expanded uses reduces soil fertility and structural stability. Soil Sci. Soc. Am. J. 73: 418-426.

Blanco-Canqui, H., R. Lal, W.M. Post, R.C. Izaurralde, and L.B. Owens. 2006a. Rapid changes in soil carbon and structural properties due to stover removal from no-till corn plots. Soil Sc. 171: 468-482.

Blanco-Canqui, H., R. Lal, W.M. Post, and L.B. Owens. 2006b. Changes in long-term no-till corn growth and yield under different rates of stover mulch. Agron J. 98:1128-1136.

Brann, D.E., D.L. Holshouser, and G.L. Mullins. Agronomy Handbook 2000. Virginia Tech Cooperative Extension. Publication 424-100. 134 p.

Easton, Z.M., and E. Bock. 2016. Soil and Soil Water Relationships. Virginia Cooperative Extension. Publication BSE-194P. At: https://ext.vt.edu/content/dam/ext_vt_edu/topics/agriculture/water/documents/Soil-and-Soil-Water-Relationships.pdf (accessed: 10/26/2018).

Franzluebbers, A.J., and R.L. Haney. 2018. Evaluation of Soil Processing Conditions on Mineralizable C and N across a Textural Gradient.

Franzluebbers, A.J., R.L. Haney, C.W. Honeycutt, H.H. Schomberg, and F.M. Hons. 2000. Flush of carbon dioxide following rewetting of dried soil relates to active organic pools. Soil Sci. Soc. Am. J. 64:613–623.

Glassner, D., J. Hettenhaus, and T. Schechinger. 1998. Corn stover collection project: a pilot for establishing infrastructure for agricultural residue and other crop collection for biomass processing to ethanol. In: Proc. Bioenergy 1998 Conference, 4–8 October, 1998, Madison, WI, pp. 1100–1110.

Graham, R.L., R. Nelson, J. Sheenan, R.D. Perlack, and L.L. Wright. 2007. Current and potential U.S. corn stover supplies. Agron. J. 99: 1-11.

Greenwell, H.C. M. Loyd-Evans, and C. Wenner. 2012. Biofuels, science and society. Interface Focus (3) 1: 1–4.

Hettenhaus, J.R., R. Wooley, and A. Wiselogel. 2000. Biomass commercialization prospects in the next 2 to 5 years: Biomass colloquies 2000. NREL/ACO-9-29-039-01. Natl. Renewable Energy Lab., Golden, CO. At: http://infohouse.p2ric.org/ref/40/39078.pdf (posted: 10/10/2000; accessed: 9/14/2018).

Huggins, D.R., C.E. Clapp, R.R. Allmaras, J.A. Lamb, and M.F. Layese. 1998. Carbon dynamics in corn-soybean sequences as estimated from natural carbon-13 abundance. Soil Sci. Soc. Am. J. 62:195-203.

Kadam, K., and J. McMillan. 2003. Availability of corn stover as a sustainable feedstock for bioethanol production. Bioresour. Technol. 88:17–25.

Karlen, D.L., S.J. Birrell, and J.R. Hess. 2011a. A five-year assessment of corn stover harvest in Central Iowa, USA. Soil Tillage Res. 115–116:47–55.

Karlen, D.L., N.C. Wollenhaupt, D.C. Erbach, E.C. Berry, J.B. Swan, N.S. Eash, and J.L. Jordahl. 1994. Crop residue effects on soil quality following 10-years of no-till corn. Soil Tillage Res. 31:149-167.

Kendall, J.R.A., D.S. Long, H.P. Collins, F.J. Pierce, A. Chatterjee, J.L. Smith, and S.L. Young. 2015. Soil Carbon Dynamics of Transition to Pacific Northwest Cellulosic Ethanol Feedstock Production. Soil Sci. Soc. Am. J. 79:272-281.

Kirkham, M.B. 2014. Field Capacity, Wilting Point, Available Water, and the Nonlimiting Water Range. In: M.B. Kirkham (ed). Principles of Soil and Plant Water Relations. Academic Press. P. 153-170.

Maskina, M.S., J.F. Power, J.W. Doran, and W.W. Wilhelm. 1993. Residual Effects of No-Till Crop Residues on Corn Yield and Nitrogen Uptake. Soil Sci. Soc. Am. J. 57:1555-1560

Mehlich, A. 1953. Determination of P, Ca, Mg, K, Na, and NH4. North Carolina Soil Test Division (Mimeo). Raleigh, NC.

McAloon, A., F. Taylor, W. Yee, K. Ibsen, and R. Wooley. 2000. Determining the cost of producing ethanol from cornstarch and lignocellulosic feedstocks. Tech. Rep. NREL/TP-580-28893. Natl. Renewable Energy Lab., Golden, CO. At: https://www.nrel.gov/docs/fy01osti/28893.pdf (accessed: 9/14/2018).

Morachan, Y.B., W.C. Moldenhauer, and W.E. Larson. 1972. Effects of increasing amounts of organic residues on continuous corn: I. Yields and soil physical properties. Agron. J. 64:199-203.

Naik, S.N., V.G. Vaibhav, P.K. Rout, and A.K. Dalai. 2010. Production of first and second generation biofuels: A comprehensive review. Renew. Sust. Energ. Rev. 14: 578-597.

Nelson, R.G. 2002. Resource assessment and removal analysis for corn stover and wheat straw in the Eastern and Midwestern United States – rainfall and wind-induced soil erosion methodology. Biomass Bioenerg. 22:349-363.

NRCS. 2018. Natural Resource Conservation Service. Web Soil Survey. At: https://websoilsurvey.sc.egov.usda.gov/App/HomePage.htm (updated: 8/21/2014; accessed: 9/14/2018).

Searchinger, T., R. Heimlich, R. Houghton, F. Dong, A. Elobeid, J. Fabiosa, S. Tokgoz, D. Hayes, and T. Yu. 2008. Use of U.S. croplands for biofuels increases greenhouse gases through emissions from land use change. Science 319: 1157–1268.

Spargo, J.T., M.A. Cavigelli, M.M. Alley, J.E. Maul, J.S. Buyer, C.H. Sequeira and R.F. Follet. 2012. Changes in soil organic carbon and nitrogen fractions with duration of no-tillage management. Soil Sci. Soc. Am. J. 76: 1624-1633.

Thomas, G.W. 1996. Soil pH and soil acidity. In D.L. Sparks (ed). Methods of soil analysis. Part 3. SSSA Book Ser.5. SSSA, Madison, WI. p. 475-490.

Virginia Department of Environmental Quality. 2018. Physiographic Provinces of Virginia.At:https://www.deq.virginia.gov/Programs/Water/WaterSupplyWaterQuantity/GroundwaterProtectionSteeringCommittee/PhysiographicProvincesofVirginia.aspx (accessed: 9/14/2018).

7. TABLES AND FIGURES

Table 1. Residue sources and application rates for a residue management field-based experiment in a continuous corn rotation in Blacksburg, VA.

Treatment	Crop	Stover retained	Crop	Straw retained
		Mg ha^{-1}		Mg ha^{-1}
1	Grain Corn	0.00	none	-
2	Grain Corn	3.33	none	-
3	Grain Corn	6.66	none	-
4	Grain Corn	10.00	none	-
5	Grain Corn	20.00	none	-

Table 2. Residue sources and application rates for residue management field-based experiments NK1 and NK2 in a corn-wheat/soybean rotation in New Kent, VA.

Treatment	Corn stover†	Wheat straw‡	Total residue⁋
	Mg ha^{-1}		
1	0.00	0.00	0.00
2	0.00	1.00	1.00
3	0.00	2.00	2.00
4	0.00	3.00	3.00
5	3.33	0.00	3.33
6	3.33	1.00	4.33
7	3.33	2.00	5.33
8	3.33	3.00	6.33
9	6.66	0.00	6.66
10	6.66	1.00	7.66
11	6.66	2.00	8.66
12	6.66	3.00	9.66
13	10.00	0.00	10.00
14	10.00	1.00	11.00
15	10.00	2.00	12.00
16	10.00	3.00	13.00

† Corn stover retained rates either applied in NK2 in September 2015 or in NK1 in September 2016.
‡Wheat straw retained rates either applied in NK1 in June 2015 or in NK2 in June 2016.
⁋ Total residue resulting from the summation of corn stover and wheat straw residues applied in a 2 yr period. Soybean residues, even when part of the rotation, were not considered here.

Table 3. Values for selected soil routine analysis on baseline samples (0-15 cm depth) taken previously to residue treatment in the experiments conducted in Blacksburg (BB) and New Kent (NK1 and NK2), VA.

	pH †	P	K	Ca	Mg	Zn	Mn	Cu	Fe	B	CEC	Acidity	Base Sat	Ca Sat	Mg Sat	K Sat	P	K	Ca	Mg
			mg / kg soil ‡								Cmol(+)/kg ⁋	%					Rating €			
Location																				
BB	6.0	43.0	66.5	492.5	97.0	1.3	26.6	0.9	14.2	0.2	4.9	28.8	71.2	51.2	16.6	3.4	H	M	M	H+
NK1	6.2	78.7	113.3	548.0	92.7	1.6	6.7	0.1	16.3	0.3	4.1	9.1	90.9	65.3	18.5	7.2	VH	H	M	H
NK2	5.5	16.3	64.7	368.0	62.3	1.0	11.1	0.6	18.4	0.1	3.6	30.0	70.0	51.0	14.4	4.6	M+	M	M-	M+

† Water pH was determined in a 1:1 (vol/vol) ratio of deionized water:soil (Thomas, 1996).
‡ Nutrients available for plant uptake were extracted with Mehlich 1 solution (0.05N HCl in 0.025N H_2SO_4) (Mehlich, 1953) using a 1:5 vol:vol soil to extractant ratio and analyzed using ICP-AES analysis.
⁋ Cation Exchange Capacity (CEC) is estimated by summation of the Mehlich 1 extractable non-acid generating cations (Ca, Mg and K), plus the acidity estimated from the Mehlich soil-buffer pH after conversion of all analytical results to meq/100 cm3 or cmol(+)/kg.
€ Rating levels based on soil test recommendations for Virginia; Virginia Cooperative Extension (2017). Letters M, H, and VH stand for medium, high and very high extractable levels of either P, K, Ca or Mg. Each category is subdivided in 3 subcategories to account for differences within a soil nutrient level availability (i.e., M-, M, and M+ stand for medium low, medium average and medium high extractable levels).

Table 4. Quadratic regression parameters, R square coefficient of determination, p-value, minimum, maximum and mean response values for total carbon (TC) and nitrogen (TN) (%), CN ratio, and bulk density (g cm^{-3}) measured at the 0-2.5, 2.5-7.5 and 7.5-20 cm soil depths in 2017 in Blacksburg and New Kent (NK1 and NK2), Virginia.

Blacksburg

Soil indicator	Depth, cm	Parameter a	Parameter b	Parameter c	R-sq	Pr>t	Min	Max	Mean
TN, %	0-2.5	-3.2E-06	0.0002	0.129	0.035	0.894	0.12	0.15	0.13
	2.5-5.0	1.9E-05	-0.0002	0.094	0.031	0.765	0.08	0.11	0.09
	7.5-20.0	-1.2E-07	-4.0E-05	0.074	0.003	0.998	0.07	0.09	0.07
TC, %	0-2.5	-0.0005	0.0210	1.320	0.400	0.444	1.22	1.67	1.43
	2.5-5.0	0.0001	0.0002	0.943	0.162	0.698	0.87	1.05	0.96
	7.5-20.0	-0.0002	0.0052	0.775	0.024	0.718	0.70	0.95	0.79
CN Ratio	0-2.5	-0.0047	0.1599	10.170	0.389	0.262	9.47	12.82	10.92
	2.5-5.0	-0.0010	0.0264	10.079	0.005	0.866	8.74	11.28	10.18
	7.5-20.0	-0.0026	0.0750	10.530	0.138	0.408	9.76	11.93	10.84
Bulk density (g cm^{-3})	0-2.5	0.0004	-0.0079	1.571	0.143	0.191	1.48	1.64	1.55
	2.5-5.0	0.0001	-0.0018	1.650	0.061	0.549	1.59	1.70	1.65
	7.5-20.0	0.0001	-0.0016	1.603	0.088	0.425	1.57	1.64	1.60

New Kent 1

Soil indicator	Depth, cm	Parameter a	Parameter b	Parameter c	R-sq	Pr>t	Min	Max	Mean
TN, %	0-2.5	-3.8E-05	0.0018	0.111	0.069	0.843	0.08	0.16	0.12
	2.5-5.0	-2.9E-05	0.0009	0.081	0.042	0.799	0.06	0.11	0.09
	7.5-20.0	-6.0E-05	0.0010	0.046	0.023	0.511	0.03	0.07	0.05
TC, %	0-2.5	-0.0003	0.0163	1.230	0.051	0.915	0.88	1.80	1.32
	2.5-5.0	0.0005	0.0001	0.872	0.042	0.713	0.67	1.22	0.90
	7.5-20.0	-0.0006	0.0113	0.542	0.021	0.608	0.34	0.93	0.58
CN Ratio	0-2.5	0.0017	-0.0274	11.060	0.007	0.650	9.94	11.62	10.98
	2.5-5.0	0.0073	-0.0898	10.730	0.036	0.212	8.52	11.70	10.57
	7.5-20.0	0.0038	-0.0523	12.070	0.003	0.705	10.40	15.27	11.95
Bulk density (g cm^{-3})	0-2.5	0.0012	-0.0187	1.590	0.077	0.097	1.16	1.62	1.54
	2.5-5.0	0.0001	-0.0010	1.580	0.003	0.796	1.44	1.66	1.58
	7.5-20.0	0.0004	-0.0091	1.660	0.064	0.611	1.47	1.81	1.62

New Kent 2

Soil indicator	Depth, cm	Parameter a	Parameter b	Parameter c	R-sq	Pr>t	Min	Max	Mean
TN, %	0-2.5	-0.0001	0.0034	0.150	0.084	0.507	0.08	0.21	0.16
	2.5-5.0	-5.8E-05	0.0015	0.100	0.045	0.683	0.07	0.13	0.10
	7.5-20.0	2.1E-05	0.0003	0.054	0.093	0.778	0.04	0.07	0.06
TC, %	0-2.5	-0.0015	0.0409	1.620	0.131	0.517	0.95	2.37	1.80
	2.5-5.0	-0.0003	0.0117	1.010	0.054	0.816	0.79	1.31	1.06
	7.5-20.0	0.0002	0.0024	0.581	0.110	0.730	0.50	0.74	0.61
CN Ratio	0-2.5	0.0001	0.0258	10.810	0.060	0.976	10.15	11.94	10.98
	2.5-5.0	0.0023	-0.0266	10.360	0.010	0.544	9.76	11.29	10.32
	7.5-20.0	1.8E-05	-0.0081	10.860	0.004	0.998	9.52	11.98	10.81
Bulk density (g cm^{-3})	0-2.5	-0.0005	0.0061	1.470	0.012	0.513	1.33	1.59	1.48
	2.5-5.0	-0.0003	0.0053	1.510	0.018	0.554	1.29	1.59	1.52
	7.5-20.0	0.0004	-0.0033	1.600	0.024	0.524	1.47	1.70	1.60

Table 5. Quadratic regression parameters, R square coefficient of determination, p-value, minimum (min), maximum (max) and mean response values for field capacity, wilting point and plant available water (kg kg^{-1}), and soil pH measured at the 0 to 7.5 cm soil depth in 2017 in Blacksburg and New Kent (NK1 and NK2), Virginia.

Blacksburg

Soil indicator	Parameter a	Parameter b	Parameter c	R-sq	Pr>t	Min	Max	Mean
Field capacity, kg kg^{-1}	2.2E-05	-1.3E-05	0.198	0.095	0.749	0.19	0.22	0.20
Wilting point, kg kg^{-1}	4.2E-05	-0.0008	0.049	0.175	0.147	0.04	0.05	0.05
Plant available water, kg kg^{-1}	-1.9E-07	0.0008	0.150	0.081	0.754	0.14	0.17	0.15
Soil pH	0.0012	-0.0072	4.920	0.100	0.664	4.24	5.95	4.99

New Kent 1

Soil indicator	Parameter a	Parameter b	Parameter c	R-sq	Pr>t	Min	Max	Mean
Field capacity, kg kg^{-1}	0.0002	-0.0007	0.124	0.059	0.510	0.10	0.18	0.13
Wilting point, kg kg^{-1}	0.0002	-0.0013	0.045	0.119	0.148	0.02	0.07	0.05
Plant available water, kg kg^{-1}	0.0000	0.0006	0.079	0.006	0.984	0.03	0.13	0.08
Soil pH	-0.0014	0.0165	5.270	0.018	0.733	4.72	7.02	5.30

New Kent 2

Soil indicator	Parameter a	Parameter b	Parameter c	R-sq	Pr>t	Min	Max	Mean
Field capacity, kg kg^{-1}	-0.0001	0.0018	0.141	0.006	0.620	0.09	0.19	0.14
Wilting point, kg kg^{-1}	0.0001	0.0009	0.029	0.023	0.379	0.02	0.04	0.03
Plant available water, kg kg^{-1}	-0.0001	0.0008	0.112	0.002	0.804	0.05	0.17	0.11
Soil pH	-0.0040	0.0560	5.560	0.014	0.451	4.55	6.72	5.69

Table 6. Quadratic regression parameters, R square coefficient of determination, p-value, minimum (min), maximum (max) and mean response values for water aggregate stability measured as the free-sand dry weight (DW, in g) per each aggregate size (in µm) for the 0 to 5 cm depth in 2017 in Blacksburg and New Kent (NK1 and NK2), Virginia.

Blacksburg

Soil indicator	Aggregate size, µm	Parameter a	Parameter b	Parameter c	R-sq	Pr>t	Min	Max	Mean
Free sand DW, g	>2000	-0.0040	0.0768	1.200	0.111	0.264	0.53	2.41	1.36
	<2000 to > 250	-0.0082	0.1424	12.180	0.100	0.329	10.17	15.12	12.41
	<250 to >53	0.0077	-0.1195	8.030	0.116	0.359	5.33	9.89	7.93

New Kent 1

Soil indicator	Aggregate size, µm	Parameter a	Parameter b	Parameter c	R-sq	Pr>t	Min	Max	Mean
Free sand DW, g	>2000	0.0085	-0.0997	2.700	0.006	0.643	0.18	7.08	2.55
	<2000 to > 250	-0.0107	0.1454	10.470	0.023	0.318	8.25	12.45	10.80
	<250 to >53	-0.0003	-0.0180	5.380	0.003	0.984	1.64	8.34	5.24

New Kent 2

Soil indicator	Aggregate size, µm	Parameter a	Parameter b	Parameter c	R-sq	Pr>t	Min	Max	Mean
Free sand DW, g	>2000	-0.0183	0.2905	1.470	0.069	0.262	0.54	5.59	2.31
	<2000 to > 250	-0.0165	0.2077	6.880	0.042	0.275	5.43	9.75	7.29
	<250 to >53	0.0281	-0.4202	7.570	0.108	0.110	4.23	8.93	6.45

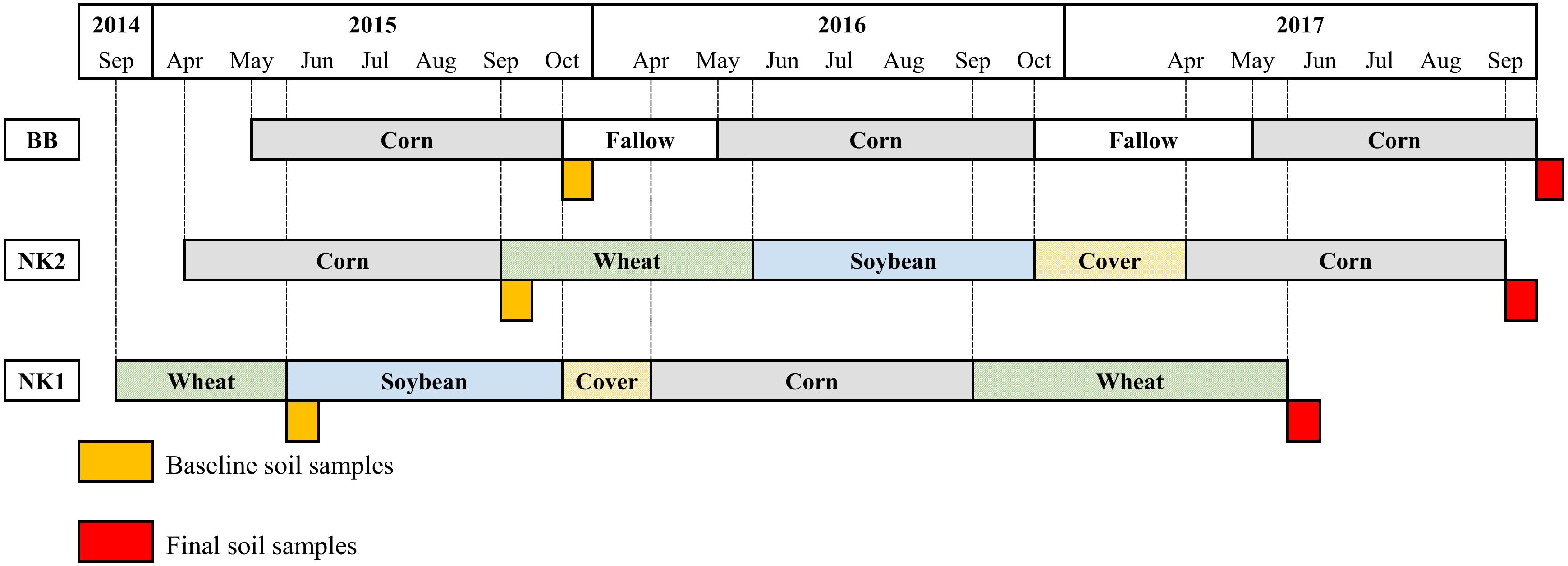


Figure 1. Chronological chart with the detail of the full crop rotations and soil sampling framework in Blacksburg (BB) and New Kent (NK1 and NK2), VA.

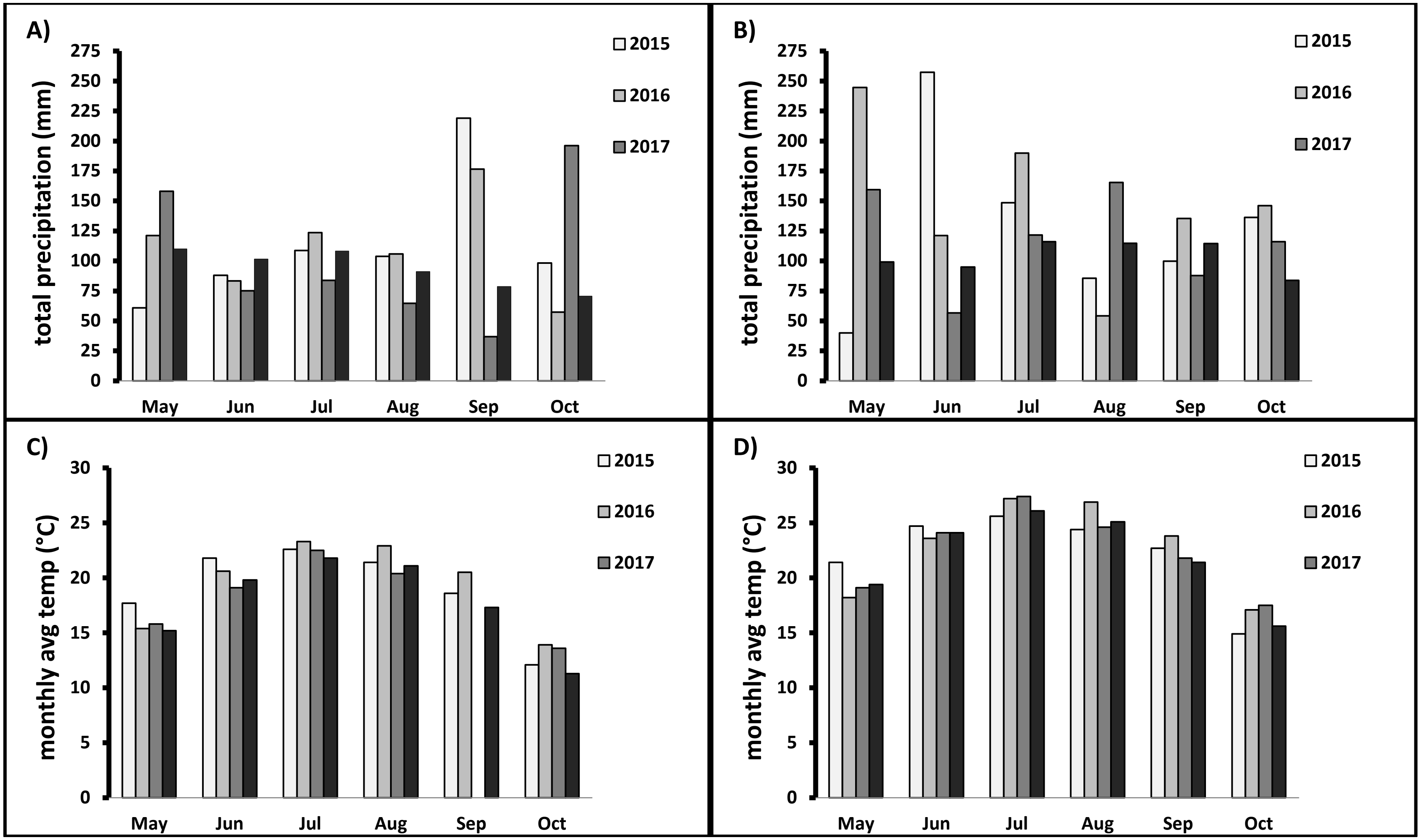


Figure 2. Total monthly and 30-yr monthly averages rainfall (mm), and air temperatures (°C) from May through October during the period 2015-2017 in Blacksburg (A and C) and New Kent (B and D), VA, respectively. Source: National Oceanic and Atmospheric Administration, National Centers for Environmental Information (https://www.ncdc.noaa.gov/cdo-web/search).

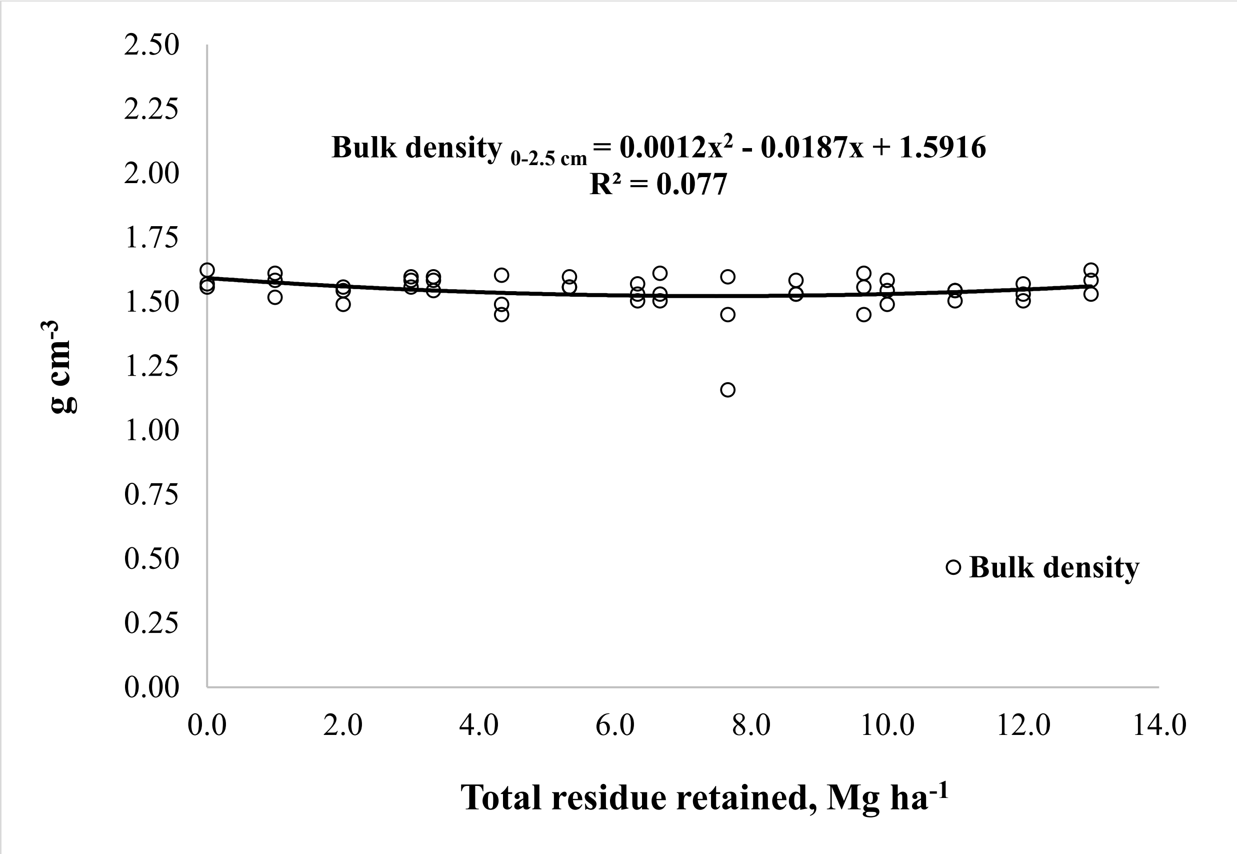


Figure 3. Bulk density (g cm^{-3}) at the 0-2.5-cm soil depth in a factorial design of sixteen total residue retained rates (in Mg dry matter ha^{-1}) from corn and wheat in a corn - wheat/soybean rotation in 2017 in New Kent, experiment 1 (NK1), Virginia. Polynomial best fit line, quadratic equation and R-square value are presented in the figure.

CHAPTER 5

Crop residue removal effects on crop production, feedstock quality and theoretical ethanol production in the mid-Atlantic USA

1. ABSTRACT

Cellulosic biomass-to-bioenergy systems have potential to provide fuels, reduce greenhouse gas emissions, and improve rural economies. Corn (*Zea mays* L.) and wheat (*Triticum aestivum* L.) residues have particular promise given these crops are widely grown and their cellulosic fractions present a captured resource as co-products of grain production. However, concerns exist about residue removal effects on subsequent crop productivity. Variable responses to residue removal (or retention) have been reported, but most such research occurred in the Midwest US. Little information is available on the effects of residue management in the Southeast. Thus, we determined crop yield and quality response to a range of biomass retention rates in grain cropping systems. Combinations of corn stover (0, 3.33, 6.66 and 10 Mg ha^{-1}) and wheat straw (0, 1.0, 2.0, and 3.0 Mg ha^{-1}) were soil applied in a corn-wheat/soybean (*Glycine max* L. Merr.) rotation in Virginia's Coastal Plain. Corn stover (0, 3.33, 6.66, 10 and 20 Mg ha^{-1}) was applied in a continuous corn system in the Ridge/Valley province. For each system, residues were applied following grain harvest over two production cycles. Each experiment was conducted as a randomized complete design with four replications. Weather was generally mild in all year-sites and water and nutrients did not limit crop production. As a result, residue retention rates did not affect grain, crop residue yields or nutrient uptake in either system in this short-term study. Feedstock quality was largely unaffected by treatment, although total retention rates around 70%

maximized sulfur concentration of the 0-10-cm wheat straw in NK1. Theoretical ethanol potential (TEP) and theoretical ethanol yield were not affected by total residue rates in New Kent location. At Blacksburg, over a 2 yr period, minimum TEP of the 0-30-cm corn stover portion corresponded to a stover retention rate of about 30%. Retaining >30% increased TEP of the >30-cm corn stover in this location, likely reflecting the impact of greater retention rates on corn feedstock quality (i.e., > hemicellulose and cellulose contents). Nutrient replacement cost for total N, P, K and S uptake resulting from harvesting the >30-cm portion of the corn stover ranged between $18.3 and 36.9. Similar nutrient replacement cost resulting from harvesting the >10-cm portion of the wheat straw were between $6.1 and 11.8 per ha. Neither residue harvest (nor residue application) had negative effects on short-term biomass yield in grain-based cropping systems in the Coastal Plain or Ridge/Valley regions of Virginia.

2. INTRODUCTION

2.1. Overview

An increased interest in the use of crop residues as a source of lignocellulosic biomass for biofuel production in the US and other regions of the world occurred over the last decade (Golden et al., 2015). Although many crops are good candidates for biofuels, corn has received the greatest attention, followed by wheat. Residues from corn and wheat have the potential to meet this demand because of the abundant aerial biomass produced by corn and to a lesser extent by wheat. These crops are widely grown worldwide and their biomass production is, to some extent, "subsidized" by the primary grain production, characteristics that could favor the use of both crop residues for biofuel production. Using second-generation biofuel feedstocks, the non-edible part of the plant rich in cellulose, hemicellulos and lignin (Golden et al., 2015), to make biofuels may help to address concerns about use of grain crops for fuel production (Searchinger et al., 2008; Tenenbaum, 2008; Thompson, 2012). Sorghum (*Sorghum bicolor* L.) is another crop that has been proposed to this end, due to its high potential biomass production per area basis, and tolerance to drought conditions (Rooney et al., 2007) and saline-alkaline soils (Vasilakoglou et al., 2011). However, sorghum area planted in the world and in the United States is much lower than for corn and wheat. Soybean (*Glycine max* (L.) Merr.) is another valuable crop in most current agricultural rotations, but the scarce aerial biomass produced by this crop and difficulties related with stover harvesting prevent its use as a biofuel feedstock. Other crops such as barley (*Hordeum vulgare* L.) could be potentially used to this end when present in grain rotations replacing winter wheat or after corn for silage.

Historically, farmers in the US have either left all corn and wheat residues in the field or have collected part or the whole plant for animal feed or bedding; by the end of the 1990s, it was estimated than more than 90% of the corn stover produced in the US remains in the field (Glassner et al., 1998). While the increased interest in crop residues for alternative uses has the potential to change these historical trends, second-generation biofuels such as cellulosic ethanol, are still in the demonstration stage (Golden et al., 2015). This interest, however, has also fueled concerns about the sustainable harvest of these residues. Determining sustainable rates of crop residue removal has been a main research goal for agronomists (Mitchell et al., 2016) in the last 40 yrs, and most of this work has occurred in the US Corn Belt. Studies of residue removal impacts on soil health suggest that between 25 and 60% of the available biomass in that region can be sustainably harvested (Glassner et al., 1998; Nelson, 2002; Kadam and McMillan, 2003; Blanco-Canqui et al., 2006b; Graham et al., 2007; Khanna and Paulson, 2016). This wide range in harvestable percentage is based on factors such as soil type, soil organic matter, and topography and any residue removal system must guard against short-term increases in nutrient removal rates (Karlen et al., 2011a, b) and long-term losses in soil fertility and stability (Blanco-Canqui et al., 2006a; Blanco-Canqui and Lal, 2009; Sindelar, 2012).

2.2. Implications of crop residue management in the US and mid-Atlantic USA region

Corn is expected to play a central role in the United States' goal to decrease dependency on fossil fuels (Sindelar, 2012). Likewise, the country can lead the world in the use of corn and be an important player in the use of wheat residue for ethanol production. According to FAOSTAT (2017), world corn grain production was 1,060 million Mg, making it the world's second largest agricultural commodity by quantity in 2016, after sugarcane (*Saccharum officinarum* L.) and before rice (*Oryza sativa* L.) and wheat. With estimated corn grain, soybean and wheat production of 365.1, 112.1 and 59.4 million Mg in 2015-2016 (USDA-NASS, 2018), the United States is the largest corn and soybean producer in the world, and the fourth largest in wheat (FAOSTAT, 2017), with an approximate share of 35, 34 and 8% of total world production, respectively. Mid-Atlantic USA's agricultural economy could also benefit by capturing additional income from these residues. During the period 2011-2017, harvested area for corn, soybean and wheat in Virginia averaged 138,000, 242,000 and 90,000 ha yr^{-1}, with average yields of 8.69, 2.64 and 4.33 Mg ha^{-1}, respectively. For the same period, harvested areas of corn for silage and barley averaged 52,000 and 12,500 ha, with an average yield of 14.57 and 4.20 Mg ha^{-1}, respectively (USDA-NASS, 2018) (Table 1). Finally, annual average crop value to the Commonwealth's economy was about \$2.25 million for barley, \$191.0 million for corn, \$222.2 million for soybean, and \$44.5 million for wheat in 2016-2017 (USDA-NASS, 2018).

Current stover supplies for the US and Virginia would be above 350 and 1.4 million Mg yr^{-1} (USDA-NASS, 2018), respectively, assuming a commonly-used harvest index (HI) value of

0.5 (Linden et al., 2000; Graham et al., 2007; Pennington, 2013). This may or may not prove a suitable assumption. For example, HI in Virginia ranged from 0.16 to 0.75 over several yrs and sites managed without irrigation and under farmer management practices (i.e. 2005, n=36; 2006, n=24; 2012, n=10 and 2014, n=10), with an average of 0.43 across yrs and locations (Thomason, unpublished data). This average value is within the range cited for others (DeLoughery and Crookston, 1979; Andrade and Ferreiro, 1996; Pennington, 2013). The highly variable nature of the HI data collected is in agreement with previous reports from Pennington (2013), who found corn HI values ranging from 0.35-0.79 in Michigan.

Theoretical ethanol potential (TEP), i.e. the amount of ethanol that can be produced per unit of residue utilized, is also important, and can be estimated from parameters such as cellulose, hemicellulose and pentose and hexose sugar concentration (Faga et al., 2010; Goff et al., 2010; Zhang et al., 2010; Vogel et al., 2011). Corn and wheat residues are mainly composed of hemicellulose (15-35%), a heterogeneous polymer containing pentoses, hexoses and uronic acids; cellulose (30-50%), a homogeneous polymer consisting of long-chain glucose monomers; lignin (10-20%), an aromatic rigid polymer resistant to chemical degradation; and ash (up to 15%) (Sun and Tomkinson, 2000; Mielenz, 2001; Gírio et al., 2010; Limayem and Ricke, 2012; Corbin et al., 2015). Current U.S. Department of Energy's TEP estimations for corn stover as a whole are on the order of 472 l Mg^{-1} stover (U.S. Department of Energy, 2014). In the case of winter wheat, predicted TEP has increased about 40% over the past decade and a half, ranging from 288 l Mg^{-1} in the early 2000s (Kerstetter and Lyons, 2001) to a current 406 l Mg^{-1} (Corbin et al., 2015), reflecting probably both the use of better processing and enzymes technologies.

Using 30% and 50% stover removal rates, theoretical values that represent maximum rates to be sustainable harvested under conventional and no-till systems according to Khanna and

Paulson (2016), and considering i) an average corn and wheat grain yield of 8.69 and 4.33 Mg ha^{-1} (period 2011-2017, Table 1) respectively, ii) a conversion rate of 472 l Mg of $stover^{-1}$ for corn and 406 l ethanol Mg^{-1} straw for wheat, and iii) a HI of 0.43 for corn and 0.45 for wheat (Thomason, personal communication), theoretical ethanol yields (TEY) in Virginia under 30% and 50% removal rates scenarios would be 1,631 and 2,719 l ethanol ha^{-1} for corn, and 644 and 1,073 l ethanol ha^{-1} for wheat residues, respectively (Battaglia et al., 2017). These TEY will support an annual ethanol production of up to 375 and 97 million l for corn and wheat residues, considering 2011-2017 average harvested area and 50% residue removal rates. Assuming similar HI, conversion and removal rates typical for winter wheat, harvested barley straw could supply another 20 million l of ethanol yr^{-1}. In summary, ethanol potential from corn and wheat residues in Virginia is around 500 million l yr^{-1} when harvesting half of the crop residues produced each year (Battaglia et al., 2017).

Feedstock from other annual residues, perennial dedicated bioenergy crops, and residues from the wood industry, could also find a niche as biomass sources for a thriving bioenergy industry in Virginia. Dedicated bioenergy crops such as Miscanthus and Switchgrass have a high potential productivity on an area basis, low nutrient requirements compared with row crops and the possibility of thriving in marginal lands not suitable for annual crops (Vogel et al., 2002; Thomason et al., 2004; Parrish and Fike, 2005; Fike et al., 2006a, b; Heaton et al., 2008). Recently, Coffin et al. (2016) identified the Southeast USA as one of the regions in the United States that has the highest potential to produce dedicated bioenergy crop due to high net primary productivity and available water. However, dedicated bioenergy crops are not subsidized by a complementary primary use as is the case with grain crop residues. One decade after the enactment of the Biomass Crop Assistance Program (BCAP), field scale adoption of dedicated bioenergy crops has been scarce in the US, in part due to the farmers unwillingness to grow these crops and the fact that corn is the most immediately available source of cellulosic biomass (Barham et al., 2016).

2.3. Effects of corn and wheat residue management on grain yield

Experiments investigating the impact of different crop residue management on crop grain yield have not been conclusive in the mid to long-term. Recently, Karlen et al. (2014) summarized a multilocation corn stover project from 239 site-yrs examining the effects of 0%, 50% and 100% stover removal rates harvested for 5 to 12 yrs, at 36 sites in seven US states, either under conventional or no-till systems, and continuous corn or corn/soybean rotations on grain yield, as part of the USDA-NIFA Sun Grant Regional Partnership. Compared with control (no-removal), harvesting moderate (3.9 Mg/ha) and high (7.2 Mg ha^{-1}) rates of corn stover resulted in grain yield increases ranging from 0.1 to 1.4 Mg ha^{-1} at 57% and 51% of the sites under each removal system, respectively. Averaged across the 36 locations, moderate and high stover removal rates increased grain yield about 0.3 Mg ha^{-1} compared with control. Removal of different rates of corn and wheat residues had positive, negative or no effects in grain yield between but also within most studies, depending on factors such as length of total study, slopes and texture of soils and water regime management (i.e., non- vs. irrigated conditions). However, an ample majority of these studies have shown little to no impact of residue management on crop yields with 3 or less cycles of residue removal/retention (Doran et al., 1984; Karlen et al., 1984; Power et al., 1986, 1998; Wilhelm et al., 1986; Bordovsky et al., 1998; Linden et al., 2000; Dam et al., 2005; Blanco-Canqui et al., 2006b).

The use of residues from annual cropping systems has the capacity to both increase and diversify farm income (Wilhelm et al., 2004). While many benefits have been observed in

Midwestern residue harvest systems, including reduction of insects and diseases pressure (Wilhelm et al., 2004) and shorter emergence interval in corn (Wilhelm et al., 1986; Vetsch and Randall, 2002), the challenge is to determine how much crop residue can sustainably be removed to achieve these benefits in Mid-Atlantic USA cropping systems. Use of crop residues for bioenergy has grown over the last decade as a result of government policies to create alternative energy sources and secure US fuel supplies (EISA-EPA, 2007), but there is concern about the ability to sustainably harvest these materials. Although the research data from the Corn Belt provide guarded optimism about residue harvest systems in the Midwest, such results are not readily extrapolated to the Mid-Atlantic USA due to the differences in soils and climate.

The purpose of this project was to determine how much corn and wheat residues can sustainably be harvested from Virginia's grain-based cropping systems without reducing plant productivity. While development of a residue-based biofuel industry in the Mid-Atlantic USA is still in the feasibility stage, generating information on the impacts of crop residue removal in grain yield, feedstock quality and biofuel production will be an important first step in determining the sustainability of these practices for the region's cropping systems. The objectives of this study were to:

a) evaluate the short-term impacts of residue retained rates on subsequent grain and biomass productivity in common mid-Atlantic USA cropping systems;

b) derive an optimum agronomical rate (OAR) of residue retained for corn and wheat residue retained in eastern, and continuous corn for grain in western Virginia;

c) estimate nutrient uptake/concentration in stover and total biomass; and

d) calculate theoretical ethanol potential and theoretical ethanol yields for corn and wheat residues based on expected crop yields in Virginia; and

3. MATERIALS and METHODS

Three experiments were conducted over three years, from 2015 through 2017, in two physiographic provinces in Virginia. Following, the detail from the materials and methods utilized in each experiment.

3.1. Experiment 1

The experiment was conducted in Blacksburg, Virginia (BB), in the Valley and Ridge physiographic region, on a Unison and Braddock loam soil (fine, mixed semiactive, mesic hapludults) (NRCS, 2018). The experiment utilized a RCBD with five levels of corn stover retention: 0, 3.33, 6.66, 10, and 20 Mg ha^{-1} corn stover retained in a continuous corn (CC) grain rotation (Table 2). A timeline chart with the information of each component for both rotations is shown in Figure 1. Four replications were established at BB resulting in a total of 20 experimental units (EU), with 5 EU per replication. Each EU was 4.6-m wide and 9.1-m long. Use of a "residue retained" (rather than a "residue removed") approach prevented potential problems resulting from residue scarcity due to poor crop productivity; individual plots with less than desired residue production could utilize residues from plots with the same treatment that produced greater amounts of residue than needed for the treatments. Using absolute mass per area (Mg ha^{-1}) rather than a percentage (%) of crop residues produced guaranteed consistent residue retention rates for plots with the same treatment across yrs. Corn residue retention rates were calculated based on reported average grain yields from Virginia in the period 2011-2017 (USDA-NASS, 2018) and calculated corn stover yields assuming a harvest index (HI) of 0.45. The treatments retaining 0, 3.33, 6.66, 10 and 20 Mg dry matter ha^{-1} (Table 2) corresponded to 0, 33, 66, 100 and 200% of the total stover produced, respectively.

Nutrient application and pest control (Table 3) followed best management practices to optimize corn grain yields for southwestern Virginia (Brann et al., 2009). Corn hybrid 'P1498' was planted at 76-cm row width under no-till, at 69,000 seeds ha^{-1} for corn grain production. Plots were established on May 3, 2015, May 9, 2016, and May 11, 2017. Grain corn was mechanically harvested on October 8 in 2015, and October 20 in 2017. Corn was not harvested in 2016, due to early crop burning with Glyphosate in stands that were not resistant.

Previous to treatment allocation, baseline soil samples were taken across the whole experimental area with a 5-cm diameter bulk density sampler to a depth of 15-cm. Later, samples were submitted to the Virginia Tech Soil Testing Lab (https://www.soiltest.vt.edu/) for baseline routine soil analysis determination (Table 4). Briefly, nutrients available for plant uptake were extracted with Mehlich 1 solution (0.05N HCl in 0.025N H_2SO_4) (Mehlich, 1953) using a 1:5 vol:vol soil to extractant ratio and analyzed using ICP-AES analysis. Based on analysis results, sufficiency ratings for P, K, Ca and Mg were then calculated (Virginia Cooperative Extension, 2017). Water pH was determined in a 1:1 (vol/vol) ratio of deionized water:soil (Thomas, 1996). Cation Exchange Capacity (CEC) is estimated by summation of the Mehlich 1 extractable non-acid generating cations (Ca, Mg and K), plus the acidity estimated from the Mehlich soil-buffer pH after conversion of all analytical results to meq/100 cm3 or cmol(+)/kg.

Baseline corn grain data were collected in 2015 but these data were not used for comparisons, because stover retention treatments in the CC rotation were not imposed until after the corn grain harvest in 2015.

Corn stover in the CC rotation was hand harvested prior to grain harvest with a combine in 2017. All plants in 3.1-m row sections from the second and fifth rows of each plot were cut at the ground level, and ears were discarded to estimate the plant biomass other than grain. Fresh weights for the whole plot were recorded. Following, three complete representative plants were selected at each EU and cut at 30-cm from the base. Fresh weights for the resulting <30- and >30-cm portions were recorded, and these plant fractions were dried at 60°C in a forced air oven to a constant weight. The unharvested portion of the crop left in the field and harvested dry matter yield (both below and above the 30-cm cutting height) were calculated with this information.

Following grain harvest in 2015 and 2016, plots were cut to 10 cm height with a rotary mower. This step was performed to obtain uniform residue distribution and facilitate stover treatment allocation. To assess corn stover availability for residue treatment application, square quadrants of approximately 0.3 m^2 were randomly tossed ten times within the whole plot area, and all biomass inside a quadrant area was collected. Average fresh weights of the quadrat samples were recorded, and biomass yields were estimated based on an assumed corn stover moisture concentration of 7.5%. This value was within those normally found after grain harvesting across a wide range of management practices and locations in Virginia and the basis for distributing biomass for each EU. Biomass subsamples from the quadrates were dried at 60°C to determine actual moisture concentrations and verify stover retention matched targeted application rates. Moisture levels averaged 7.7% in 2015 and 2016, resulting in an applied/theoretical stover retained rates of at least 0.99 in all cases. Total area required to supply the appropriate level of biomass per EU was calculated and stover then was evenly distributed at the prescribed rate. Excess stover was removed from the study area. In 2016, additional stover from other corn plots was added to achieve the desired residue retention rate for the 10 and 20 Mg ha^{-1} treatments.

Oven-dried corn stover subsamples (i.e., <30- and >30-cm portions) for different corn stover retained rates collected in 2017 were ground to pass a 2-mm mesh screen. Ten aliquot samples, including five from each plant portion, were randomly chosen and submitted to Waypoint analytical lab (Richmond, VA) for plant tissue analysis, including N, P, K and S (Gavlak, et al., 1994; Simone et al., 1994; Matejovic, 1995). These values were later used to calibrate regression equations for each nutrient to estimate these parameters for all the samples using near-infrared spectroscopy (NIR). Nutrient uptake and removal from harvested stover were then calculated as the product of plant nutrient concentration and stover mass removed from each system.

Neutral and acid detergent fiber (NDF and ADF, respectively), acid detergent lignin (ADL) and ash content (in g kg^{-1} of biomass in a dry basis) were also estimated with a robust NIR equation calibrated and validated with data from other biomass experiments conducted in Virginia. Total hemicellulose was calculated by subtracting ADF from NDF, and total cellulose as ADF minus ADL and expressed in g kg^{-1} of biomass (Dien et al., 2006). Lignin content corresponds to the ADL fraction. These values were used to estimate TEP and TEY (Goff et al., 2010), as follows:

$$H = [\%\ \text{Cellulose} + (\%\ \text{Hemicellulose} \times 0.07) \times 172.82] \quad \textbf{[1]}$$

$$P = [\%\ \text{Hemicellulose} \times 0.93] \times 176.87 \quad \textbf{[2]}$$

$$\text{TEP (l ethanol Mg}^{-1}) = [H+P] \times 4.17 \quad \textbf{[3]}$$

$$\text{TEY (l ethanol ha}^{-1}) = \text{TEP} \times \text{Biomass yield (Mg ha}^{-1}) \quad \textbf{[4]}$$

where H and P are hexose and pentose carbohydrates, respectively. Theoretical ethanol yield (TEY, in l ethanol ha^{-1}) was calculated as the product between the TEP for each treatment and its respective average biomass production.

Weather data were obtained from the National Oceanic and Atmospheric Administration's Blacksburg, VA, weather station from the National Climate Data Center (*https://www.ncdc.noaa.gov/cdo-web/search*). Total monthly rainfall (mm), and monthly average air temperatures (°C) were collected for the active growing seasons of the different crops grown in this study (i.e., May 2015 through October 2017) at Kentland Farm, Blacksburg, VA and presented with the 30-yr average (1981-2010) rainfall and air temperature (Figure 2).

3.2. Experiment 2 and 3

Two experiments were conducted in New Kent, Virginia (NK) in the Coastal Plain region. Experiments were initiated at different crop phases of the same corn – wheat / soybean (CWS) rotation., with one set of residue return treatments initiated in the wheat phase of the rotation in June 2015 (NK1) and another with identical treatments, initiated after corn harvest in September 2015 (NK2) (Figure 1). Soils in NK1 and NK2 were both classified as Altavista sandy loam (fine-loamy, mixed, semiactive, thermic Aquic Hapludults) (NRCS, 2018). Each experiment at NK utilized a RCBD with a factorial arrangement of sixteen treatments resulting from the combination of four residue retained rates each for corn (0, 3.33, 6.66 and 10 Mg ha^{-1}) and wheat (0, 1, 2 and 3 Mg ha^{-1}) (Table 1) with soybean residues completely retained in the C-W/S rotation. Total annual retained residues from both corn and wheat were also calculated (Table 5). Four replications were established, resulting in 64 EU per experiment, with 16 EU per replication. Each EU 4.6-m wide by 9.1-m long, and residue retention rates both for corn and wheat were calculated as in Experiment 1 based on a HI of 0.45. Treatments retaining 0, 3.33, 6.66 and 10 Mg dry corn stover ha^{-1}(Table 5), and 0, 1, 2, and 3 Mg dry wheat straw ha^{-1} corresponding to approximately 0, 33, 66 and 100% of the total stover and straw produced respectively.

Nutrient application and pest control (Table 6) followed best management practices to optimize corn, wheat and soybean grain yields for eastern Virginia (Brann et al., 2009). Similar management was applied to both NK1 and NK2 experiments. At each respective year, corn hybrid Channel 197-31 was planted at 76-cm row width under no-till, at 60,000 seeds ha^{-1} between April 20 and 25 and harvested between September 1 and 10 on both plots. Wheat variety 'USG 3404' and soybean variety Asgrow 5332 were planted at 19-cm row width. Wheat was planted at 135 kg ha^{-1} between October 20 and 25, and harvested between June 15 and 20 on both plots. Immediately after wheat harvest, soybean was planted at 350,000 seeds ha^{-1} and harvested between October 10 and 15 on both plots (Figure 1). Detailed information on the management applied to experiments in New Kent can be seen in Table 6.

Corn stover and wheat straw samples were hand harvested before grain harvest in 2017. In the case of corn, the procedure was similar to that used at the BB location, resulting in plant portions below and above an approximate cutting height of 30 cm from ground level. Wheat samples were hand harvested at ground level in a 3-m^2 area from 5 rows other than those used for grain harvest (3-m long by 1-m wide). Plants were then cut at 10 cm from the base in order to estimate the unharvested (<10-cm) and the potentially harvested (>10-cm) dry matter yields.

Wheat, corn and soybean crops were mechanically harvested in all cases, but grain yields were not recorded in 2015, as no residue treatments had been allocated at that time. For wheat, spreaders situated on the back of the combine were taken off to create straw windrows across the field. Windrows created in this fashion were approximately 1.5-m wide by 0.5 to 1-m tall. After corn harvest, a rotary mower was used on the entire plot area at a height of 10 cm, in order to obtain a uniform residue distribution and facilitate stover treatment allocation. Stover and straw samples were taken at four spots across the plot area in 2015, and from each EU in 2016 by collecting all residue within a randomly-placed quadrant of approximately 0.3 m^2.

Protocols for whole plot fresh weight measurement, subsample fresh and dry weight determination, and grinding (one residue portion in 2015 and 2016; two in 2017) for both corn and wheat followed similar protocol to that used with corn in the BB study. Ground aliquots were submitted to Waypoint analytical lab (Richmond, VA) for plant tissue analysis and all subsamples were analyzed using NIR to estimate nutrient concentration/uptake, TEP and TEY, similarly than in BB study.

To assess annual wheat straw availability before allocating the residue treatments, six representative straw samples (approximately 1.5-m lengths of windrow) were randomly collected from the different windrows and fresh weights were recorded. To assess annual corn stover availability before residue treatment allocation, square quadrats of approximately 0.3 m^2 were randomly tossed fifteen times within the plot area. All biomass bounded by the quadrat was collected, and fresh weights recorded. Lineal meters of wheat straw and corn stover to retain at each plot-treatment were calculated and residues evenly distributed at the prescribed rate. Excess stover was removed from the study area.

Stover and straw subsamples were collected and dried at 60°C to determine moisture content of the allocated residues and verify application rates. In the field, average moisture of 7.5% for corn stover and around 3% for wheat were assumed. Across yrs, residue moisture content averaged 7.4% in corn and 4.2% in wheat, resulting in applied/theoretical retained rates of 0.99 and 0.96 for corn and wheat, respectively.

Weather data were obtained from the National Oceanic and Atmospheric Administration's West Point, VA, weather station from the National Climate Data Center (*https://www.ncdc.noaa.gov/cdo-web/search*). Total monthly rainfall (mm), and monthly average air temperatures (°C) were collected for the active growing seasons of the different crops grown in this study (i.e., May through October, 2015-2017) New Kent, VA. The 30-yr average (1981-2010) rainfall and air temperature for each month of the yr are also presented (Figure 2).

3.3. Data analysis

Linear regression analysis was conducted to test the effects of crop residue management on a) grain yield and total stover and straw yields in 2016 and 2017; and on b) portions of stover and straw yields (i.e., 0-10 and >10-cm for wheat; 0-30 and >30-cm for corn), nutrient concentration and uptake, theoretical ethanol potential and theoretical ethanol yield in 2017 using the PROC REG procedure in SAS version 9.4 (SAS Institute, 2014). Rates of crop residue return, yrs and locations were considered fixed effects, while blocks were considered random effects. The quadratic or linear models were selected based on R^2 values and model p-values. Regression equations for each parameter of plant productivity as a function of either corn stover (BB) or the total summation of corn and wheat residues applied in previous years in both NK experiments were developed. Minimum, maximum and mean for each comparison were also calculated (Table 7 and 8). At the BB location, five corn stover retained rates (0, 3.33, 6.66, 10 and 20 Mg ha^{-1}) were considered as the treatment for the model effects both in 2016 and 2017 (Table 2). At the NK location, treatments applied in 2015 were the treatments considered in the analysis in 2016. These included four straw retained rates applied at NK-plot 1: 0, 1, 2 and 3 Mg ha^{-1} and four stover retained rates applied at NK-plot 2: 0, 3.33, 6.66 and 10 Mg ha^{-1}. In NK1, total retained residues in 2017 resulted from the factorial combination of four wheat straw treatments allocated in 2015 and four corn stover treatments allocated in 2016. In NK2, total retained residues in 2017 resulted from the combination of four corn stover treatments allocated in 2015 and four wheat straw treatments allocated in 2016 (Table 5; Figure 1) were considered in the model. Normality of data was assessed with the UNIVARIATE procedure. Presence of outliers in Y was assessed with the INFLUENCE procedure of SAS. Additionally, and previous to calibrating the NIR parameter estimations (predicted or P; y-axis) with the plant nutrient analysis generated for a subset of subsamples in a commercial laboratory (observed data or O; x-axis), presence of outliers in X (leverage points) were also assessed with the INFLUENCE procedure of SAS.

4. RESULTS and DISCUSSION

4.1. Weather data

Blacksburg and New Kent locations are located in a region classified as humid subtropical, according to the Köppen climate classification (Kottek et al., 2006). Nonetheless, annual average (30-yr period, 1981-2010) precipitations and average air temperatures are higher in New Kent (1153 mm and 15.6 °C) than Blacksburg location (1039 mm and 12.2 °C), respectively. The three years under study were wetter and slightly hotter in most comparisons than an average year at both locations. In New Kent, accumulated rains were 23, 43, and 13% greater than average for the period May-October in 2015, 2016, and 2017, respectively (Figure 2B and D). As a result of this, the remarkable rates of biomass decomposition that occur in this region of the US leave negligible amounts of undecomposed biomass from previous crops after a yr in the field (Thomason, personal communication), resulting in a robust estimation of the total corn and wheat residues produced in the previous season. Over the same subperiod, accumulated precipitations across years in BB (range: 615-679 mm) were 10% to 21% greater (Figure 2A) and air temperatures up to 1.6°C higher (Figure 2C) than the 30-year average. Although rates of decomposition in Blacksburg are expected to be lower compared to those occurring in New Kent, these conditions over the length of the study resulted in negligible amount of undecomposed material following 1 yr in the field.

4.2. Effect of wheat and corn residue retention on grain yields

The retention of 0, 1, 2 and 3 Mg ha^{-1} of wheat straw in 2015 did not have an impact ($p = 0.892$) in corn grain yields (range 9.28-13.24 Mg ha^{-1}) in 2016 in NK1 (Table 7). Retention of 0, 3.33, 6.66 and 10 Mg ha^{-1} of corn stover in 2015 did not impact (p = 0.182) wheat grain yields (range: 2.03-3.67 Mg ha^{-1}) in 2016 in NK2 (Table 7). The lack of grain yield response following one cycle of crop residue management in our studies is in agreement with findings from most field studies conducted in the last 50 years in the US and abroad (Morachan et al., 1972; Doran et al., 1984; Karlen et al., 1984; Wilhelm et al., 1986; Bordovsky et al., 1998; Power et al., 1998; Linden et al., 2000; Dam et al., 2005; Blanco-Canqui et al., 2006b). In these studies, the comparison between the 0 and 100% rates from either a removal or retention standpoint was always present. Additionally, absolute corn stover return rates of up to 16 Mg ha^{-1} (Morachan et al., 1972), or relative returns of 150% (Doran et al., 1984; Power et al., 1998) and 200% (Blanco-Canqui et al., 2006b) were included each year in some cases. A common denominator in these and our studies is that water was not limiting crop production in the year following first time treatment allocation. In one study, however, corn stover returning rates of 0 and 50% reduced grain yields between 4 and 8% compared with 100 and 150% rates following one cycle of stover management, but these results most likely reflected the residual effect of an earlier 8 yr study with similar corn return rates and different management practices (Powel et al., 1998).

In the third year, two cycles of residue retention, with precipitations above average during the entire crop growth cycle in most comparisons produced similar results both in New Kent and Blacksburg. In 2017, total retained residues resulting from the allocation of wheat and corn residues in 2015 and 2016, respectively, did not affect wheat (p = 0.763) grain yields (range: 1.89-4.66 Mg ha^{-1}) at NK1 (Table 7). Similarly, corn and wheat residues applied in 2015 and 2016, respectively, did not affect corn (p = 0.893) grain yields (range: 6.22-14.29 Mg ha^{-1}) at NK2 (Table 7). In the same year, corn stover treatments applied in the two previous years did not impact (p = 0.794 corn grain yields (range: 9.39-12.11 Mg ha^{-1}) in the continuous corn rotation in Blacksburg

(Table 7). Other studies also showed that two or more cycles of residue management had little no none impact on corn and wheat yields when water was not limiting crop production (Morachan et al., 1972, Karlen et. al., 1984; Bordovsky et al., 1998; Power at al., 1998; Blanco-Canqui et al., 2006b). In some cases where cool soils and not water availability can limit plant growth, increased corn grain yields have been reported with stover removal over a 3-yr period in the Upper US Corn Belt, regardless of tillage system or fertilizer N rate (Sindelar et al., 2012), and over an 11-yr period in Canada (Dam et al., 2005). On the other hand, returning less residues resulted in lower grain yields when water availability limited crop production in some studies. Doran et al. (1984) found that corn stover return rates of 0% reduced grain yields between 11 and 15% compared with 50, 100 and 150% following two cycles of stover management, in a year with precipitations below a 30-yr average in July and August. In another study, 0% corn stover return decreased grain yields by 20, 32 and 42% compared with 50. 100 and 150% return rates, respectively, when total precipitations in June and July were 80% and 10% lower than average values (Power et al., 1986). In both cases, when water deficit occurred in June and/or July corn stands were at tasseling and silking (Abendroth et al., 2011), critical stages for grain yield determination when the presence of abiotic or biotic limitations or both can severely decrease grain yields (Robins and Domingo, 1953; Claassen and Shaw, 1970; Hall et al., 1981; NeSmith and Ritchie, 1992).

4.3. Effect of wheat and corn residue retention on corn stover and wheat straw yields

At NK1, corn stover dry matter harvested in 2016 (range: 5.84-17.61 Mg ha^{-1}) were not impacted ($p = 0.190$) by wheat straw treatments allocated in 2015 (Table 7). Two cycles of residue retention produced similar results than one cycle of retention at both locations. In 2017, total retained residues applied in the previous two years did not affect wheat straw dry matter at the 0-10 ($p = 0.419$; range: 0.49-1.41 Mg ha^{-1}), above 10-cm ($p = 0.636$; range: 2.39-5.48 Mg ha^{-1}), or the whole plant dry matter ($p = 0.396$; range: 3.00-6.88 Mg ha^{-1}) in NK1 (Table 7). At NK2, corn stover yields in 2017 were irresponsive to two cycles of corn and wheat residue management at the 0-30 ($p = 0.828$; range: 1.02-2.11 Mg ha^{-1}), above 30-cm ($p = 0.821$; range: 6.78-10.31 Mg ha^{-1}), or the whole plant level ($p = 0.905$; range: 7.98-11.91 Mg ha^{-1}) . Finally, two cycles of corn stover residues returns of up to 20 Mg ha^{-1} did not have an effect on the corn stover yields at the 0-30 ($p = 0.101$; range: 2.23-3.54 Mg ha^{-1}), above 30-cm ($p = 0.137$; range: 9.00-13.67 Mg ha^{-1}), or the whole plant level ($p = 0.103$; range: 11.62-17.22 Mg ha^{-1}) (Table 7). Growing conditions during the entire crop growth cycle, with appropriate management including N-P-K base fertilization at planting and N re-fertilization during crop season for both corn and wheat (Table 3 and 6), alongside with equally distributed precipitations and average air temperatures above average at both locations (Figure 3A, B, C, and D), were conducive to excellent crop production. Routine soil test analysis taken in 2015 revealed appropriate levels of P, K, Ca and Mg for optimum crop production in all cases, according to Virginia Cooperative Extension (2017) guidelines, with high or very high nutrient levels in half of the cases (Table 4). Moreover, a high base saturation (>50%) was measured in all cases, a condition that likely enhanced K, Mg and Ca availability in our studies (Daniels and Haering, 2015). These circumstances, especially those related to the abundant precipitations, likely explain not only the lack of grain yield but also the corn stover and wheat response to crop residue management in our studies. Other authors have observed similar responses both in short (Doran et al., 1984; Power at al., 1986) and long-term experiments (Dam et al., 2005). In hot and dry years, 0% corn stover return rates decreased corn stover dry matter yields by 30%, 34% and 47% compared with 50%, 100% and 150% return rates in Nebraska (Doran et al., 1984). Wetter than average years can also have a detrimental impact in very cold regions by significantly reducing soil temperatures during early stages of crop development.

Retaining crop residues under a no-till system in Canada reduced corn stover yields in two out of 12-yr when precipitation was above average, especially previous or at the planting time in experiments conducted by Dam et al. (2005) in Canada.

4.4. Corn stover and wheat straw nutrient concentration and uptake

Corn stover treatments applied in 2015 and 2016 at the Blacksburg location did not affect the nitrogen (N) ($p>0.24$), phosphorus (P) ($p>0.22$), or potassium (K) ($p>0.22$) concentration (%) measured either at the basal (0 to 30 cm) or the upper portion (>30 cm) of the corn plant in 2017 (Table 8). Total corn and wheat retained residues in the two previous year only affected the concentration of one nutrient ($p = 0.045$; Table 8) out of sixteen comparisons across experiments and plant portions in New Kent. When differences were significant, calculated maximum point in the regression curve corresponded to total retained residue rate (x-axis) of 9.25 Mg ha^{-1}. Predicted S concentration in wheat straw at the 0-10-com plant portion at this maximum residue retained rate (y-axis) was 0.14%. Increasing the amount of total residue returned to the system in the range 0 to 9.3 Mg ha^{-1} (approximately 72% retained rate) gradually increased the S concentration of the 0-10-cm wheat straw in a sandy soil in NK1. Retaining more than 72% of the wheat straw and corn stover produced in the previous years, gradually decreased the S concentration in this portion of the wheat straw (Figure 3). The N, P, K and S uptake by the portion of the corn plant above 30 cm, and the portion of the wheat plant above 10 cm were not affected ($p> 0.31$ in all cases) by different levels of corn stover or total residue retained rates in any of the eleven comparisons across locations measured in 2017 (Table 8).

4.5. Theoretical ethanol potential and ethanol yields

Total residue retained treatments had no impact on the theoretical ethanol production (TEP) ($p>0.14$) and theoretical ethanol yield (TEY) ($p>0.31$) (Table 8) of the above 30- and 10-cm upper part of the corn stover and wheat straw in New Kent. Corn stover retention rates affected the TEP ($p = 0.010$) but not the TEY ($p = 0.336$) in BB (Table 8). When differences were significant, calculated minimum point in the regression curve corresponded to a corn stover retained rate (x-axis) of 5.97 Mg ha^{-1}. Predicted TEP for the upper part of the corn plant portion in BB in 2017 at the minimum residue retained rate (y-axis) was 116 L Mg^{-1}. Over a 2 yr period, increasing the amount of corn stover returned to the system in the range 0 to 6.0 Mg ha^{-1} (approximately 30% retained rate) gradually decreased the TEP of the 0-30-cm corn stover portion in BB. Retention rates >30%, on the other hand, gradually increased the TEP of this portion of the plant in a loam soil in BB (Figure 4), likely reflecting the impact of greater retention rates on corn feedstock quality (i.e., > hemicellulose and cellulose contents; data not shown).

For corn, TEP ranged between 223 and 358 (mean 297), and between 218 and 291 (mean 256) l of ethanol per Mg of corn stover dry matter in BB and NK2, respectively (Table 8). In a 2010 study in Minnesota, Sindelar et al. (2012) reported TEP ranging between 334 to 417 l for the whole plant portion, a range that is slightly above the corn stover TEP calculated in our studies. In this study, however, the cob portion was included as a component of the whole plant portion, a component that was not included in our comparisons (i.e., the whole ear was removed from the plant before corn stover harvest). In the same study, Sindelar et al. (2012) calculated TEP ranging between 455 to 476 l ethanol Mg^{-1} for the cob portion, a value that is close to current U.S. Department of Energy's TEP estimations of 472 l Mg^{-1} stover for the whole plant corn stover fraction (U.S. Department of Energy, 2014). Cobs are the most efficient stover component for bioethanol production on a weight basis. However, cob represents less than 15% of the total corn stover (Sawyer and Mallarino, 2007). More recently, Corbin et al. (2015) estimated TEP ranging from 362 to 456 l ethanol Mg^{-1} stover (without cobs).

Theoretical ethanol production for the upper part of the wheat fodder biomass were between 264 and 322 (mean 299) l of ethanol per Mg of wheat straw dry matter in NK1 (Table 8), similar to estimations of 288 l Mg^{-1} wheat straw from Kerstetter and Lyons (2001) in the early 2000s, but lower than most recent estimations of 406 l Mg^{-1} wheat straw from Corbin et al. (2015).

Corn ethanol yields were between 2163 and 3698 (mean 3032) l ethanol per ha at BB, and between 1510 and 2752 (mean 2188) l ethanol per ha in NK2 in 2017 (Table 8), reflecting both a lower corn stover biomass production and TEP in most comparisons in the NK2 experiment. Our TEY for corn fell within the range estimated by others. Working in Iowa with different corn stover harvest scenarios, Hoskinson et al. (2007) evaluated the ethanol production in a single year in a production plot with no history of residue management in Iowa. Theoretical ethanol yield for the normal-cut scenario in this study (left approximately 40-cm of stubble in the field after the combine harvest, so the TEY was calculated on the >40-cm portion of the corn stover) was 2258 l ethanol per ha. Our estimates are also in the range reported by Sindelar et al. (2012) in Minnesota (1800 to 2900 l ethanol per ha). In this study, 0% corn stover return rates resulted in greater TEY than 100% retention rates, which is explained by the greater corn stover yield for the comparison with no residue left found in this study and not by a difference in TEP between treatments.

In 2017, wheat ethanol yields were between 705 and 1709 (mean 1196) l of ethanol per Mg of dry matter at NK1 (Table 8), a range of value that includes a recent TEY for wheat of 1055 l ethanol per ha reported by Corbin et al. (2015).

5. CONCLUSIONS

In this short-term study, residue retention rates had no effect on grain, crop residue yields, and nutrient uptake in either corn soybean/wheat or in continuous corn cropping systems. Weather during each growing season was generally mild, and stover retention treatments had minimal effects on subsequent feedstock (crop residue) quality relative to energy production. In the corn – wheat/soybean rotation, increasing the amount of total residue returned from 0 to about 72% retention rate gradually increased the S concentration of the 0-10-cm wheat straw in NK1. Retaining more than 72% of the wheat straw and corn stover produced in the previous years, gradually decreased the S concentration in this portion of the wheat straw (Figure 3). Theoretical ethanol potential and theoretical ethanol yield were not affected by total residue rates in New Kent location. At Blacksburg, over a 2 yr period, increasing the amount of corn stover returned in the range 0 to about 30% retained rate, increased the TEP of the 0-30-cm corn stover portion. Retention rates >30%, on the other hand, gradually decreased the TEP of the >30-cm portion of the corn plant (Figure 4). Nutrient replacement cost for total N, P, K and S uptake resulting from harvesting the >30-cm portion of the corn stover ranged between $18.3 and 36.9. Similar nutrient replacement cost resulting from harvesting the >10-cm portion of the wheat straw were between $6.1 and 11.8 per ha. Under short-term scenarios, neither complete biomass removal nor high levels of biomass application had significant negative effect on crop production parameters in grain-based cropping systems in the Coastal Plain or Ridge/Valley regions of Virginia. The consequences of such treatments in longer-term studies and under more challenging environmental conditions warrant further study if these residue harvesting systems are implemented in Virginia.

6. LITERATURE CITED

Abendroth, L.J., R.W. Elmore, M.J. Boyer, and S.K. Marlay. 2011. Corn growth and development. PMR 1009. Iowa State University Extension, Ames, Iowa.

Andrade, F.H., and M.A. Ferreiro. 1996. Reproductive growth of maize, sunflower and soybean at different source levels during grain filling. Field Crops Res. 48: 155–165.

Barham, B.L., D.F. Mooney, and S.M. Swinton. 2016. Inconvenient Truths about Landowner (Un)Willingness to Grow Dedicated Bioenergy Crops. Choices. Quarter 4. At: http://www.choicesmagazine.org/choices-magazine/submitted-articles/inconvenient-truths-about-landowner-unwillingness-to-grow-dedicated-bioenergy-crops (accessed: 9/14/2018).

Battaglia, M.L., G. Groover, and W.E. Thomason. 2017. Value and implications of corn stover removal from Virginia fields. Virginia Cooperative Extension Publication CSES-180. At: http://pubs.ext.vt.edu/CSES/CSES-180/CSES-180.html (accessed: 9/14/2018).

Battaglia, M.L., G. Groover, and W.E. Thomason. 2018. Harvesting and nutrient replacement costs associated with corn stover removal in Virginia. Virginia Cooperative Extension Publication CSES-229NP. At: https://pubs.ext.vt.edu/content/dam/pubs_ext_vt_edu/CSES/cses-229/CSES-229.pdf (accessed: 9/14/2018).

Blanco-Canqui, H., and R. Lal. 2009. Corn stover removal for expanded uses reduces soil fertility and structural stability. Soil Sci. Soc. Am. J. 73: 418-426.

Blanco-Canqui, H., R. Lal, R.C. Izaurralde, and L.B. Owens. 2006a. Rapid changes in soil carbon and structural properties due to stover removal from no-till corn plots. Soil Sc. 171: 468-482.

Blanco-Canqui, H., R. Lal, W.M. Post, and L.B. Owens. 2006b. Changes in long-term no-till corn growth and yield under different rates of stover mulch. Agron J. 98:1128-1136.

Bordovsky, D.G., M. Choudhary, C.J. Gerard. 1998. Tillage Effects on Grain Sorghum and Wheat Yields in the Texas Rolling Plains. Agron. J. 90: 638-643.

Brann, D.E., D.L. Holshouser, and G.L. Mullins. Agronomy Handbook 2000. Virginia Tech Cooperative Extension. Publication 424-100. 134 p.

Claassen, M.M., and R.H. Shaw. 1970. Water deficits effects on corn. II. Grain components. Agron. J. 62: 652-655.

Coffin, A.W., T.C. Strickland, W.F. Anderson, M.C. Lamb, R.R. Lowrance, and C.M. Smith. 2016. Potential for production of perennial biofuel feedstocks in conservation buffers on the Coastal Plain of Georgia, USA. BioEnergy Res 9: 587-600.

Corbin, K.R., C.S. Byrt, S. Bauer, S. DeBolt, D. Chambers, J.A.M. Holtum, G. Karem, M. Henderson, J. Lahnstein, C.T. Beahan, A. Bacic, G.B. Fincher, N.S. Betts, and R.A. Burton. 2015. Prospecting for Energy-Rich Renewable Raw Materials: Agave Leaf Case Study. PLoS ONE 10(8): e0135382.doi:10.1371/journal.pone.0135382

Dam, R.F., B.B. Mehdi, M.S.E. Burgess, C.A. Madramootoo, G.R. Mehuys, and I.R. Callum. 2005. Soil bulk density and crop yield under eleven consecutive years of corn with different tillage and residue practices in a sandy loam soil in central Canada. Soil Tillage Res. 84: 41-53.

Daniels, W.L. and K.C. Haering. 2015. Concepts of Basic Soil Science. In: K.C. Haering and G.K. Evanylo, editors, The Mid-Atlantic Nutrient Management Handbook. Virginia Cooperative Extension. CSES-122. p.49. At: http://pubs.ext.vt.edu/content/dam/pubs_ext_vt_edu/CSES/CSES-122/CSES-122-pdf.pdf (accessed: 9/14/2018).

DeLoughery, R.L., and R.K. Crookston. 1979. Harvest Index of Corn Affected by Population Density, Maturity Rating, and Environment. Agron. J. 71: 577-58.

Doran, J.W., W.W. Wilhelm, and J.F. Power. 1984. Crop residue removal and soil productivity with no-till corn, sorghum, and soybean. Soil Sci. Soc. Am. J. 48:640-645.

EISA-EPA. 2007. Summary of the Energy Independence and Security Act. Public Law 110-140. US Environmental Protection Agency. At: https://www.epa.gov/laws-regulations/summary-energy-independence-and-security-act (last updated: 8/2/2016; accessed: 9/14/2018).

Faga, B.A., M.R. Wilkins, and I.M. Banat. 2010. Ethanol production through simultaneous saccharification and fermentation of switchgrass using Saccharomyces cerevisiae D5A and thermotolerant Kluyveromyces marxianus IMB strains. Bioresour Technol. 101: 2273-2279.

FAOSTAT. 2017. Food and Agricultural commodities production/Commodities by region. At: http://faostat3.fao.org/browse/rankings/commodities_by_regions/E (accessed 9/14/2018).

Fike, J.H., D.J. Parrish, D.D. Wolf, J.A. Balasko, J.T. Green Jr, M. Rasnake and J.H. Reynolds. 2006a. Long-term potential of Switchgrass for biofuel systems. Biomass and Bioenergy 30: 198-206.

Fike, J.H., D.J. Parrish, D.D. Wolf, J.A. Balasko, J.T. Green Jr, M. Rasnake and J.H. Reynolds. 2006b. Switchgrass production for the upper southeastern USA: influence of cultivar and cutting frequency on biomass yields. Biomass and Bioenergy 30: 207-2013.

Gavlak, R.G, D.A. Horneck, and R.O. Miller. 2013. Plant, Soil and Water reference Methods for the Western Region. WREP 125, 4th Edition. 156 p.

Gírio, F.M., C. Fonseca, F. Carvalheiro, L.C. Duarte, S. Marques, and R. Bogel- Łukasik. 2010. Hemicelluloses for fuel ethanol: A review. Bioresour Technol. 101: 4775-4800.

Glassner, D., J. Hettenhaus, and T. Schechinger. 1998. Corn stover collection project: a pilot for establishing infrastructure for agricultural residue and other crop collection for biomass processing to ethanol. In: Proc. Bioenergy 1998 Conference, 4–8 October, 1998, Madison, WI, pp. 1100–1110.

Goff, B.M., K.J. Moore, S.L. Fales, and E.A. Heaton. 2010. Double-cropping sorghum for biomass. Agron. J. 102: 1586–1592.

Golden, J.S., R.B. Handfield, J. Daystar, and T.E. McConnell. 2015. An Economic Impact Analysis of the U.S. Biobased Products Industry: A Report to the Congress of the United States of America. A Joint Publication of the Duke Center for Sustainability & Commerce and the Supply Chain Resource Cooperative at North Carolina State University. At: https://biopreferred.gov/BPResources/files/EconomicReport_6_12_2015.pdf (accessed: 9/14/2018).

Graham, R.L., R. Nelson, J. Sheenan, R.D. Perlack, and L.L. Wright. 2007. Current and potential U.S. corn stover supplies. Agron. J. 99: 1-11.

Hall, A.J., J.H. Lemcoff, and N. Trapani. 1981. Water stress before and during flowering in maize and its effects on yield, its components, and their determinants. Maydica 26: 19–38.

Heaton, E.A., F.G. Dohleman and S.P. Long. 2008. Meeting US biofuel goals with less land: the potential of Miscanthus. Global Change Biology 14: 2000-2014.

Hoskinson, R.L., D.L. Karlen, S.J. Birrell, C.W. Radtke, and W.W. Wilhelm. 2007. Engineering, nutrient removal, and feedstock conversion evaluations of four corn stover harvest scenarios. Biomass Bioenergy 31: 126-136.

Kadam, K., and J. McMillan. 2003. Availability of corn stover as a sustainable feedstock for bioethanol production. Bioresour. Technol. 88:17–25.

Karlen, D.L., S.J. Birrell, and J.R. Hess. 2011a. A five-year assessment of corn stover harvest in Central Iowa, USA. Soil Tillage Res. 115–116:47–55.

Karlen, D.L., S.J. Birrell, J.M.F. Johnson, S.L. Osborne, T.E. Schumacher, G.E. Varvel, R.B. Ferguson, J.M. Novak, J.R. Fredrick, J.M. Baker, J.A. Lamb, P.R. Adler, G.W. Roth, and E.D. Nafziger. 2014. Multilocation Corn Stover Harvest Effects on Crop Yields and Nutrient Removal. BioEnergy Res. 7: 528-539.

Karlen, D.L., P.G. Hunt, and R.B. Campbell. 1984. Crop residue removal effects on corn yield and fertility of a Norfolk sandy loam. Soil Sc. Soc. Am. J. 48:868-872.

Karlen, D.L., G.E. Varvel, J.M.F. Johnson, J.M. Baker, S.L. Osborne, J.M. Novak, P.R. Adler, G.W. Roth, and S.J. Birrell. 2011b. Monitoring soil quality to assess the sustainability of harvesting corn stover. Agron J 103:288–295.

Kerstetter, J.D. and J.K. Lyons. 2001. Wheat Straw for Ethanol Production in Washington: A Resource, Technical, and Economic Assessment. Washington State University

Cooperative Extension Energy Program. WSUCEEP201084. At: http://pacificbiomass.org/documents/WA_AsssessmentWheatStrawToEthanol_rev.pdf (accessed: 9/14/2018).

Khanna, M., and N. Paulson. 2016. To Harvest Stover or Not: Is it Worth it? Department of Agricultural and Consumer Economic. University of Illinois at Urbana-Champaign. Farmdoc daily 6:32.

Kottek, M., J. Grieser, C. Beck, B. Rudolf, and F. Rubel. 2006. World Map of the Köppen-Geiger climate classification Updated. Meteorol Z. 15: 259-263

Limayem, A., and S.C. Ricke. 2012. Lignocellulosic biomass for bioethanol production: Current perspectives, potential issues and future prospects. Prog. Energy Combust. Sci. 38: 449-467

Linden, D.R., C.E. Clapp, and R.H. Dowdy. 2000. Long-term corn grain and stover yields as a function of tillage and residue removal in east central Minnesota. Soil Tillage Res. 56:167-174.

Matejovic, I. 1995. Total nitrogen in plant material determined by means of dry combustion: A possible alternative to determination by Kjedahl digestion. Commun. Soil. Sci. Plant Anal. 26: 2217-2229.

Mehlich, A. 1953. Determination of P, Ca, Mg, K, Na, and NH4. North Carolina Soil Test Division (Mimeo). Raleigh, NC.

Mielenz, J.R. 2001. Ethanol production from biomass: Technology and commercialization status. Curr Opin Microbiol 4: 324-329

Mitchell, R.B., M.R. Schmer, W.F. Anderson, V. Jin, K.S. Balkcom, J. Kiniry, A. Coffin, and P. White. 2016. Dedicated Energy Crops and Crop Residues for Bioenergy Feedstocks in the Central and Eastern USA. BioEnergy Res. 9: 384-398.

Nelson, R.G. 2002. Resource assessment and removal analysis for corn stover and wheat straw in the eastern and Midwestern United States: rainfall and wind-induced soil erosion methodology. Biomass Bioenergy 22: 349-363.

NeSmith, D.S., and J.T. Ritchie. 1992. Short- and long-term responses of corn to a pre-anthesis soil water deficit. Agron. J. 84: 107–113.

NRCS. 2018. Natural Resource Conservation Service. Web Soil Survey. At: http://websoilsurvey.nrcs.usda.gov/app/HomePage.htm ((updated: 8/21/2014; accessed: 9/14/2018).

Parrish, D.J. and J.H. Fike. 2005. The biology and agronomy of Switchgrass for biofuels. Critical Reviews in Plant Sciences, 24: 423-459.

Pennington, D. 2013. Harvest index: A predictor of corn stover yield. Michigan State University Extension. At: http://msue.anr.msu.edu/news/harvest_index_a_predictor_of_corn_stover_yield (updated: 2/11/13; accessed 9/14/2018).

Power, J.F., J.W. Doran, and W.W. Wilhelm. 1986. Crop residue effects on soil environment and dryland maize and soybean production. Soil Tillage Res. 8:101-111.

Power, J.F., P.T. Koerner, J.W. Doran, and W.W. Wilhelm. 1998. Residual effects of crop residues on grain production and selected soil properties. Soil Sci. Soc. AM. J. 62:133-1397.

Robins, J.S., and C.E. Domingo. 1953. Some effects of severe soil moisture deficits at specific growth stages in corn. Agron. J. 45: 618–621.

Rooney, W.L., J. Blumenthal, B. Bean and J.E. Mullet. 2007. Designing sorghum as a dedicated bioenergy feedstock. Biofuels, Bioproducts and Biorefining. Vol.1: 147-157.

Sawyer, J., and A. Mallarino. 2007. Nutrient removal when harvesting corn stover. Integrated Crop Management News. IC-498(22). At: http://www.ipm.iastate.edu/ipm/icm/2007/8-6/nutrients.html (accessed: 9/14/2018).

Searchinger, T., R. Heimlich, R.A. Houghton, F. Dong, A. Elobeid, J. Fabiosa, S. Tokgoz, D. Hayes, and T. Yu. 2008. Use of U.S. croplands for biofuels increases greenhouse gases through emissions from land use change. Science 319:1238–1240.

Simone, H.A., J.B. Jones, Jr., D.A. Smitties, and C.G. Hussey. 1994. A comparison of analytical methods for nitrogen analysis in plant tissues. Commun. Soil Sci. Plant Anal., 25: 943-954

Sindelar, A.J. 2012. Stover, tillage, and Nitrogen management in continuous corn for grain, ethanol, and soil carbon. Ph.D. diss., University of Minnesota, Minneapolis, MN.

Sun R.C., and T. Tomkinson. 2000. Essential guides for isolation/purification of polysaccharides. In: I.D. Wilson, T.R. Adlard, C.F. Poole, M. Cook (Eds.), Encyclopedia of Separation Science, Academic Press, London, 2000, pp. 4568–4574

Tenenbaum, D.J. 2008. Food vs. Fuel: Diversion of Crops Could Cause More Hunger. Environ. Health Persperct. 116: A254–A257.

Thomas, G.W. 1996. Soil pH and soil acidity. In D.L. Sparks (ed). Methods of soil analysis. Part 3. SSSA Book Ser.5. SSSA, Madison, WI. p. 475-490.

Thomason, W.E., W.R. Raun, G.V. Johnson, C.M. Taliaferro, K.W. Freeman, K.J. Wynn and R.W. Mullen. 2004. Switchgrass response to harvest frequency and time and rate of applied nitrogen. J. Plant Nut. Vol.27: 1199-1226.

Thompson, P.B. 2012. The Agricultural Ethics of Biofuels: The Food vs. Fuel Debate. Agriculture J. 2: 339-358.

USDA-NASS. 2018. Quick stats Lite (Beta). At: http://www.nass.usda.gov/Quick_Stats/Lite/index.php (accessed: 9/14/2018).

U.S. Department of Energy. 2014. Alternative Fuels Data Center. Ethanol Feedstocks. At: http://www.afdc.energy.gov/fuels/ethanol_feedstocks.html (last updated: 03/16/2017; accessed: 9/14/2018).

Vasilakoglou, I., K. Dhima, N. Karagiannidis and T. Gatsis. 2011. Sweet sorghum productivity for biofuels under increased soil salinity and reduced irrigation. Field Crops Res. Vol. 120: 38-46.

Vetsch, J.A., and G.W. Randall. 2002. Corn production as affected by tillage system and starter fertilizer. Agron. J. 94:532–540.

Virginia Cooperative Extension. 2017. Soil test recommendations for Virginia. At: https://www.soiltest.vt.edu/content/dam/soiltest_vt_edu/PDF/recommendation-guidebook.pdf (accessed: 9/14/2018).

Vogel, K.P., B.S. Dien, H.G. Jung, M.D. Casler, S.D. Masterson, and R.B. Mitchell. 2011. Quantifying Actual and Theoretical Ethanol Yields for Switchgrass Strains Using NIRS Analyses. Bioenerg. Res. 4: 96. https://doi.org/10.1007/s12155-010-9104-4

Vogel, K.P., J.J. Brejda, D.T. Walters and D.R. Buxton. 2002. Switchgrass biomass production in the midwest USA: harvest and nitrogen management. Agron. J. 94:413–420.

Wilhelm, W.W., J.W. Doran, and J.F. Power. 1986. Corn and soybean yield response to crop residue management under no-tillage production systems. Agron. J. 78:184-189.

Wilhelm, W.W., J.M. Johnson, J.L. Hatfield, W.B. Voorhees, and D.R. Linden. 2004. Crop and soil productivity response to corn residue removal: a literature review. Agron. J. 96: 1-17.

Zhang, J., D. Chu, J. Huang, Z. Yu, G. Dai, and J. Bao. 2010. Simultaneous Saccharification and Ethanol Fermentation at High Corn Stover Solids Loading in a Helical Stirring Bioreactor. Biotechnol. Bioeng. 105: 718-728.

7. TABLES AND FIGURES

Table 1. Grain corn, corn silage, soybean and winter wheat national and state average yields for the period 2011-2017, in Mg ha^{-1}

	2011	2012	2013	2014	2015	2016	2017	Average '11-'17
Grain corn								
Virginia	7.41 †	6.47	9.67	9.11	10.11	9.29	8.79	8.69
US	9.22	7.73	9.93	10.74	10.58	10.96	11.09	10.04
Corn Silage								
Virginia	12.95	11.77	15.69	15.30	16.48	15.69	14.12	14.57
US	14.44	12.32	14.75	15.77	16.01	15.93	15.61	14.97
Soybean								
Virginia	2.69	2.83	2.59	2.66	2.32	2.42	2.96	2.64
US	2.83	2.69	2.96	3.20	3.23	3.50	3.30	3.10
Winter wheat								
Virginia	4.78	4.31	4.17	4.58	4.44	3.57	4.44	4.33
US	3.10	3.17	3.18	2.87	2.86	3.72	3.38	3.18
Barley								
Virginia	4.73	4.41	4.41	4.25	4.03	3.60	3.93	4.20
US	3.72	3.60	3.84	3.91	3.72	4.19	3.91	3.84

Source: USDA-NASS, 2018. At: https://www.nass.usda.gov/Quick_Stats/Lite/index.php (accessed: 10/19/2018)

† Yields expressed with moisture contents of 0.155, 0.130, 0.135, and 0.145 g water g^{-1} for grain corn, soybean, winter wheat, and barley, respectively. Corn silage is reported at 35% dry matter moisture content.

Table 2. Residue sources and application rates for a residue management field-based experiment in a continuous corn rotation in Blacksburg, VA.

Treatment	Crop	Stover retained	Crop	Straw retained
		Mg ha^{-1}		Mg ha^{-1}
1	Grain Corn	0.00	none	-
2	Grain Corn	3.33	none	-
3	Grain Corn	6.66	none	-
4	Grain Corn	10.00	none	-
5	Grain Corn	20.00	none	-

Table 3. Detail of the management practices applied to continuous corn in the period 2015-2017 at Kentland Farm, Blacksburg, VA

	2015	2016	2017
Fertilizer application date	4-May		
		4-Jul	19-Jun
$N–P_2O_5–K_2O$, kg ha^{-1}	43-43-43		
		56-0-0	56-0-0
Seeding date	3-May	9-May	11-May
Tillage	No-till	No-till	No-till
Genotype	P1498HR	P1498HR	P1498HR
Seed rate, seeds ha^{-1}	69189	69189	69189
Seed rate, kg ha^{-1}	-	-	-
Row width, cm	76	76	76
Herbicides date	4-May (pre-emerg); 10-June (post-emerg)	9-May (all); 24-May (barley silage)	2-May
Pre-emergent			
2,4 D (Dimethylamine salt) ¶		0.498 kg a.i. ha^{-1}	0.498 kg a.i. ha^{-1}
Bicep II Magnum (Atrazine+S-Metolachlor) ƛ	0.573 kg a.i. ha^{-1} + 0.351 kg a.i. ha^{-1}	0.717 kg a.i. ha^{-1} + 0.439 kg a.i. ha^{-1}	0.717 kg a.i. ha^{-1} + 0.439 kg a.i. ha^{-1}
Atrazine 4L (Atrazine) ‡	0.235 kg a.i. ha^{-1}		
Glystar plus (Glyphosate) §	0.919 kg a.i. ha^{-1}	0.316 kg a.i. ha^{-1}	0.316 kg a.i. ha^{-1}
Post-emergent			
Glystar plus (Glyphosate) §	0.460 kg a.i. ha^{-1}	0.316 kg a.i. ha^{-1}	
Insecticide date	None	None	None
Fungicide date	None	None	None
Irrigation management	None	None	None
Harvest date	8-Oct	Not harvested	20-Oct

¶ (46.8%) 2,4-Dichlorophenoxyacetic acid

ƛ Atrazine (33.0%) 2-chloro-4-ethylamino-6-isopropylamino-striazine; S-Metolachlor (26.1%) Acetamide, 2-chloro-N-(2-ethyl-6-methylphenyl)-N-(2-methoxy-1-methylethyl],(S)

‡ Atrazine (41.9%) 2-chloro-4-ethylamino-6-isopropylamino-striazine

§ Glyphosate (41.0%) (N-(phosphonomethyl)glycine)

Table 4. Values for selected soil routine analysis on baseline samples (0-15 cm depth) taken previously to residue treatment allocation at the Blacksburg and New Kent locations, VA.

Location	pH †	P	K	Ca	Mg	Zn	Mn	Cu	Fe	B	CEC	Acidity	Base Sat	Ca Sat	Mg Sat	K Sat	P	K	Ca	Mg
			mg / kg soil ‡								Cmol(+)/kg ⁋	%					Rating €			
BB	6.0	43.0	66.5	492.5	97.0	1.3	26.6	0.9	14.2	0.2	4.9	28.8	71.2	51.2	16.6	3.4	H	M	M	H+
NK1	6.2	78.7	113.3	548.0	92.7	1.6	6.7	0.1	16.3	0.3	4.1	9.1	90.9	65.3	18.5	7.2	VH	H	M	H
NK2	5.5	16.3	64.7	368.0	62.3	1.0	11.1	0.6	18.4	0.1	3.6	30.0	70.0	51.0	14.4	4.6	M+	M	M-	M+

† Water pH was determined in a 1:1 (vol/vol) ratio of deionized water:soil (Thomas, 1996).
‡ Nutrients available for plant uptake were extracted with Mehlich 1 solution (0.05N HCl in 0.025N H_2SO_4) (Mehlich, 1953) using a 1:5 vol:vol soil to extractant ratio and analyzed using ICP-AES analysis.
⁋ Cation Exchange Capacity (CEC) is estimated by summation of the Mehlich 1 extractable non-acid generating cations (Ca, Mg and K), plus the acidity estimated from the Mehlich soil-buffer pH after conversion of all analytical results to meq/100 cm3 or cmol(+)/kg.
€ Rating levels based on soil test recommendations for Virginia; Virginia Cooperative Extension (2017). Letters M, H, and VH stand for medium, high and very high extractable levels of either P, K, Ca or Mg. Each category is subdivided in 3 subcategories to account for differences within a soil nutrient level availability (i.e., M-, M, and M+ stand for medium low, medium average and medium high extractable levels).

Table 5. Residue sources and application rates for residue management field-based experiments NK1 and NK2 in a corn-wheat/soybean rotation in New Kent, VA.

Treatment	Corn stover†	Wheat straw‡	Total residue⁋
		——— $Mg\ ha^{-1}$ ———	
1	0.00	0.00	0.00
2	0.00	1.00	1.00
3	0.00	2.00	2.00
4	0.00	3.00	3.00
5	3.33	0.00	3.33
6	3.33	1.00	4.33
7	3.33	2.00	5.33
8	3.33	3.00	6.33
9	6.66	0.00	6.66
10	6.66	1.00	7.66
11	6.66	2.00	8.66
12	6.66	3.00	9.66
13	10.00	0.00	10.00
14	10.00	1.00	11.00
15	10.00	2.00	12.00
16	10.00	3.00	13.00

† Corn stover retained rates either applied in NK2 in September 2015 or in NK1 in September 2016.
‡Wheat straw retained rates either applied in NK1 in June 2015 or in NK2 in June 2016.
⁋ Total residue resulting from the summation of corn stover and wheat straw residues applied in a 2 yr period. Soybean residues, even when part of the rotation, were not considered here.

Table 6. Detail of the management practices performed in NK1 in the period 2014-2017 at Lanexa, New Kent, VA*

	2014	2015		2016		2017
	Wheat	Wheat (cont.)	Soybean	Corn	Wheat	Wheat (cont.)
Fertilizer application date	5-Nov	19-Mar	-		2-Nov	15-Mar
		2-Apr				5-Apr
N–P_2O_5–K_2O, kg ha^{-1}	40-60-80	67-0-0	-		40-60-80	67-0-0
		67-0-0	-			67-0-0
Seeding date	28-Oct	-	23-Jun	15-Apr	25-Oct	-
Tillage	No-till	No-till	No-till	No-till	No-till	No-till
Genotype	USG 3404	-	Asgrow 5332	Channel 197-31	USG 3404	USG 3404
Seed rate, seeds ha^{-1}	-	-	346,000	59,300		
Seed rate, kg ha^{-1}	0	-	-	-	135	-
Row width, cm	19	19	19	76	19	19
Herbicides			-			
Pre-emergent				15-Apr		
Atrazine 4L (Atrazine) ꝸ	-	-	-	0.235 kg a.i. ha^{-1}	-	-
Glystar plus (Glyphosate) ∑	-	-	-	0.919 kg a.i. ha^{-1}	-	-
Post-emergent	23-Nov	1-Apr	-	15-May	21-Nov	3-Apr
Metribuzin 75 (Metribuzin) ¥	0.039 kg a.i. ha^{-1}	-	-	-	0.039 kg a.i. ha^{-1}	-
Prowl H2O (Pendimethalin) ₤	0.309 kg a.i. ha^{-1}	-	-	-	0.309 kg a.i. ha^{-1}	-
Glystar plus (Glyphosate) ∑	-	-	-	0.460 kg a.i. ha^{-1}	-	-
Harmony SG (Thifensulfuron-methyl) ¶	-	0.092 kg a.i. ha^{-1}	-	-	-	0.092 kg a.i. ha^{-1}
Insecticide date	23-Nov	1-Apr	-	-	23-Nov	1-Apr
Baythroid XL (ß-cyfluthrin) Ω	0.002 kg a.i. ha^{-1}	-	-	-	0.002 kg a.i. ha^{-1}	-
Warrior T (Lambda-cyhalothrin) †	-	0.001 kg a.i. ha^{-1}	-	-	-	0.001 kg a.i. ha^{-1}
Fungicide date	-	1-Apr; 14-May	-		-	2-Apr; 11-May
QuiltXcel (Azoxystrobin + Propiconazole) ƛ	-	0.022 + 0.017 kg a.i. ha^{-1}		-	-	0.022 + 0.017 kg a.i. ha^{-1}
Prosaro (Prothioconazole + Tebuconazole) §	-	0.020 + 0.020 kg a.i. ha^{-1}		-	-	0.020 + 0.020 kg a.i. ha^{-1}
Growth regulator date	None	1-Apr	-	-	-	3-Apr
Palisade EC ‡	-	0.002 kg a.i. ha^{-1}	-	-	-	0.002 kg a.i. ha^{-1}

Irrigation management	None	None		None		None
Harvest date	-	22-Jun		6-Sep	-	22-Jun

*NK2, initiated after corn harvest in September 2015, had similar treatments than NK1.

§ Atrazine (33.0%) 2-chloro-4-ethylamino-6-isopropylamino-striazine; S-Metolachlor (26.1%) Acetamide, 2-chloro-N-(2-ethyl-6-methylphenyl)-N-(2-methoxy-1-methylethyl],(S)

∑ Glyphosate (41.0%) (N-(phosphonomethyl)glycine)

¥ Metribuzin (75%): 4-Amino-6-(1,1-dimethylethyl)-3-(methylthio)-1,2,4-triazin-5 (4H)-one

₤ Pendimethalin (38.%): N-(1 -ethylpropyl)-3,4-dimethyl-2,6-dinitrobenzenamine

¶ Thifensulfuron-methyl (50%): Methyl 3-[[[[(4-methoxy-6-methyl-1,3,5-triazin-2-yl) amino]carbonyl]amino]sulfonyl]-2-thiophenecarboxylate

Ω ß-cyfluthrin (12.7%): Cyano(4-fluoro-3-phenoxyphenyl)methyl-3-(2,2-dichloro-ethenyl)-2,2-dimethyl-cyclopropanecarboxylate

† Lambda-cyhalothrin (11.4%) : [1α(S*),3α(Z)]-(±)-cyano-(3-phenoxyphenyl)methyl-3-(2-chloro-3,3,3-trifluoro-1-propenyl)-2,2-dimethylcyclopropanecarboxylate

ƛ Azoxystrobin (13.5%): Methyl (E)-2-{2-[6-(2-cyanophenoxy)pyrimidin-4-yloxy]phenyl}-3-methoxyacrylate ; Propiconazole (11.7%): 1-[[2-(2,4-dichlorophenyl)-4-propyl-1,3-dioxolan-2-yl]methyl]-1H-1,2,4-triazole

§ Prothioconazole (19%): 2-[2-(1-Chlorocyclopropyl)-3-(2-chlorophenyl)-2-hydroxypropyl]-1, 2-dihydro-3H-1, 2,4-triazole-3-thione ; Tebuconazole (19%): alpha-[2-(4-chlorophenyl)ethyl]-alpha-(1, 1-dimethylethyl)-1H-1, 2,4-triazole-1-ethanol

‡ Trinexapac-ethyl (12.0%): Cyclohexanecarboxylic acid, 4-(cyclopropylhydroxymethylene)- 3,5-dioxo-, ethyl ester(95266-40-3)

Table 7. Quadratic regression parameters, R square coefficient of determination, p-value, minimum (min), maximum (max) and mean response values for corn and wheat grain yield, and corn stover and wheat straw biomass production (in Mg ha^{-1}) in 2016 and 2017 in Blacksburg and New Kent, Virginia.

Blacksburg

Year		Parameter a	Parameter b	Parameter c	R-sq	Pr>t	Min	Max	Mean
2017	Corn Grain Yield, Mg ha^{-1}	0.0005	0.0239	10.300	0.100	0.794	9.39	12.11	10.55
	Corn Stover, 0-30 cm, Mg ha^{-1}	0.0029	-0.0547	2.890	0.166	0.101	2.23	3.54	2.78
	Corn Stover, >30 cm, Mg ha^{-1}	0.0107	-0.1917	11.170	0.162	0.137	9.00	13.67	10.85
	Corn Stover, total, Mg ha^{-1}	0.0136	-0.2463	14.060	0.185	0.103	11.62	17.22	13.63

New Kent, experiment 1

Year		Parameter a	Parameter b	Parameter c	R-sq	Pr>t	Min	Max	Mean
2016	Corn Grain Yield, Mg ha^{-1}	0.0264	-0.3552	11.790	0.188	0.892	9.28	13.54	11.60
	Corn Stover, Mg ha^{-1}	-1.1472	2.6896	12.130	0.200	0.190	5.84	17.61	12.00
2017	Wheat Grain Yield, Mg ha^{-1}	0.0010	-0.0254	3.600	0.007	0.763	1.89	4.66	3.49
	Wheat Straw, 0-10 cm, Mg ha^{-1}	0.0026	-0.0238	0.880	0.070	0.419	0.49	1.41	0.88
	Wheat Straw, >10 cm, Mg ha^{-1}	0.0037	-0.0258	3.930	0.019	0.636	2.39	5.48	3.98
	Wheat Straw, total, Mg ha^{-1}	0.0080	-0.0810	4.930	0.026	0.396	3.00	6.88	4.86

New Kent, experiment 2

Year		Parameter			R-sq	Pr>t	Min	Max	Mean
		a	b	c					
2016	Wheat Grain Yield, Mg ha^{-1}	0.0097	-0.0404	2.690	0.484	0.182	2.03	3.67	2.90
2017	Corn Grain Yield, Mg ha^{-1}	-0.0070	0.0409	11.140	0.014	0.893	6.22	14.29	10.99
	Corn Stover, 0-30 cm, Mg ha^{-1}	-0.0012	0.0117	1.440	0.006	0.828	1.02	2.11	1.45
	Corn Stover, >30 cm, Mg ha^{-1}	0.0029	-0.0291	8.560	0.003	0.821	6.78	10.31	8.53
	Corn Stover, total, Mg ha^{-1}	0.0017	-0.0176	10.000	0.001	0.905	7.98	11.91	9.99

Table 8. Quadratic regression parameters, R square coefficient of determination, p-value, minimum (min), maximum (max) and mean response values for Nitrogen (N), Phosphorus (P), Potassium (K) and Sulfur (S) concentration (%) and uptake (kg ha^{-1}), theoretical ethanol potential (L Mg^{-1}) and theoretical ethanol yield (L ha^{-1}) calculated in 2017 in Blacksburg and New Kent, Virginia.

Blacksburg

Fraction		Parameter			R-sq	Pr>t	Min	Max	Mean
		a	b	c					
Corn Stover, <30 cm	N, %	0.0001	-0.0020	0.217	0.116	0.240	0.19	0.23	0.21
	P, %	0.0004	-0.0066	0.239	0.131	0.228	0.14	0.29	0.23
	K, %	-0.0004	0.0284	1.033	0.438	0.745	0.83	1.55	1.22
Corn Stover, >30cm	N, %	0.0000	-0.0001	0.233	0.072	0.900	0.22	0.24	0.23
	P, %	0.0000	-0.0017	0.104	0.035	0.976	0.00	0.24	0.08
	K, %	0.0010	-0.0083	1.197	0.427	0.223	0.98	1.51	1.25
	N uptake, kg ha^{-1}	0.0000	-0.0002	0.070	0.041	0.848	21.3	26.2	23.6
	P uptake, kg ha^{-1}	0.0012	-0.1591	23.410	0.023	0.317	0.1	24.7	8.3
	K uptake, kg ha^{-1}	0.0118	-0.4147	10.650	0.031	0.834	94.4	167.9	129.4
	TEP, L Mg^{-1}	0.2109	-2.5178	123.590	0.520	0.010	223	358	297
	TEY, L Ha^{-1}	0.0083	0.1815	7.310	0.093	0.336	2163	3698	3032

New Kent, experiment 1

Fraction		Parameter a	Parameter b	Parameter c	R-sq	Pr>t	Min	Max	Mean
Wheat Straw, <10 cm	N, %	0.0000	0.0015	0.232	0.070	0.988	0.19	0.30	0.24
	P, %	-0.0001	0.0033	0.112	0.093	0.446	0.07	0.16	0.12
	K, %	-0.0010	0.0159	0.993	0.004	0.702	0.60	1.84	1.04
	S, %	-0.0002	0.0037	0.119	0.131	0.045	0.10	0.16	0.13
Wheat Straw, >10cm	N, %	-0.0003	0.0015	0.250	0.082	0.107	0.23	0.37	0.27
	P, %	-0.0003	0.0053	0.113	0.070	0.149	0.07	0.21	0.13
	K, %	-0.0004	0.0062	0.542	0.002	0.834	0.21	0.96	0.56
	S, %	0.0001	-0.0009	0.116	0.005	0.651	0.09	0.16	0.11
	N uptake, kg ha^{-1}	-0.0038	0.1504	9.901	0.029	0.869	6.9	14.9	10.7
	P uptake, kg ha^{-1}	-0.0046	0.1612	4.361	0.064	0.765	2.1	8.9	5.2
	K uptake, kg ha^{-1}	-0.0008	0.1071	21.801	0.002	0.993	8.0	42.8	22.5
	S uptake, kg ha^{-1}	-0.0520	0.0476	4.504	0.012	0.606	2.8	6.7	4.5
	TEP, L Mg^{-1}	-0.1002	1.4614	295.440	0.010	0.495	264	322	299
	TEY, L Ha^{-1}	1.5693	-20.6180	933.570	0.094	0.310	705	1709	1196

New Kent, experiment 2

Fraction		Parameter			R-sq	Pr>t	Min	Max	Mean
		a	b	c					
Corn Stover, <30 cm	N, %	0.0000	-0.0007	0.197	0.017	0.826	0.18	0.23	0.19
	P, %	-0.0001	0.0016	0.199	0.010	0.742	0.13	0.25	0.20
	K, %	-0.0034	0.0456	1.080	0.075	0.138	0.79	1.53	1.17
	S, %	0.0000	0.0001	0.055	0.001	0.938	0.04	0.07	0.06
Corn Stover, >30cm	N, %	0.0001	-0.0009	0.207	0.033	0.416	0.19	0.22	0.21
	P, %	0.0001	-0.0035	0.086	0.090	0.855	0.01	0.12	0.07
	K, %	0.0000	0.0024	0.981	0.003	0.991	0.66	1.25	0.99
	S, %	0.0000	0.0001	0.054	0.003	0.880	0.04	0.08	0.05
	N uptake, kg ha^{-1}	0.0123	-0.1386	17.000	0.009	0.650	14.1	20.9	17.6
	P uptake, kg ha^{-1}	0.0084	-0.3111	7.300	0.087	0.808	1.2	10.5	6.0
	K uptake, kg ha^{-1}	0.0377	-0.1428	83.630	0.009	0.854	47.4	118.7	84.9
	S uptake, kg ha^{-1}	-0.0022	0.0178	4.610	0.005	0.823	3.2	6.0	4.6
	TEP, L Mg^{-1}	-0.3577	4.9277	244.800	0.074	0.149	218	291	256
	TEY, L Ha^{-1}	-2.1489	33.0870	2096.000	0.015	0.590	1510	2752	2188

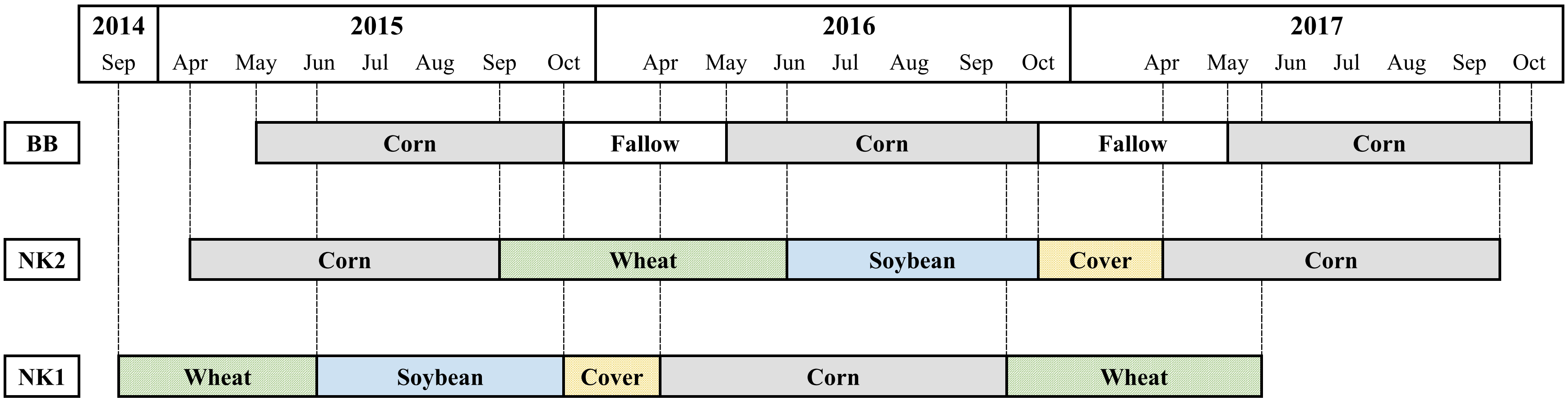


Figure 1. Chronological chart with the detail of the full crop rotations in Blacksburg (BB) and New Kent (NK1 and NK2), VA.

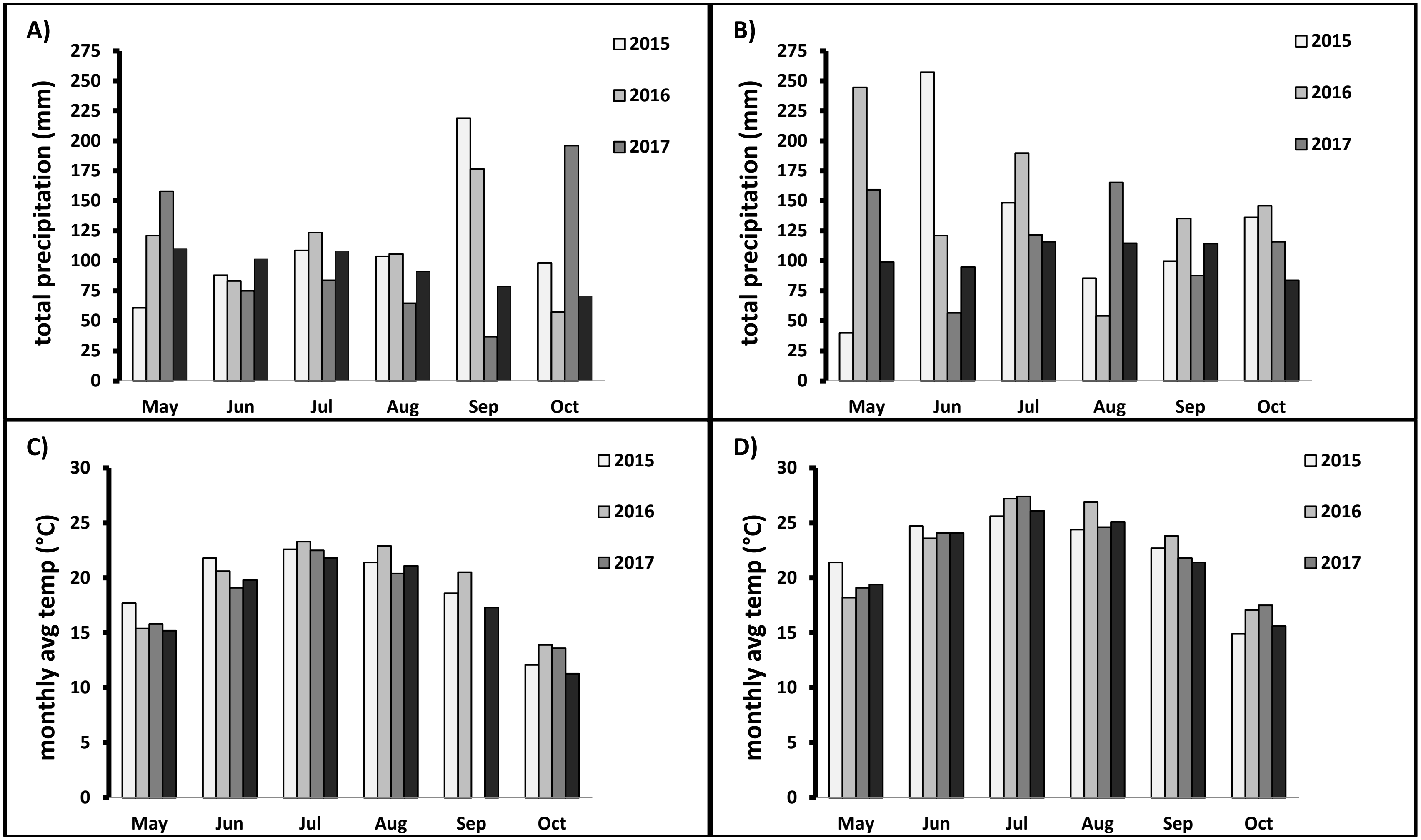


Fig. 2. Total monthly and 30-yr monthly averages precipitation (mm), and air temperatures (°C) from May through October during the period 2015-2017 in Blacksburg (A and C) and New Kent (B and D), VA, respectively. Source: National Oceanic and Atmospheric Administration, National Centers for Environmental Information (https://www.ncdc.noaa.gov/cdo-web/search).

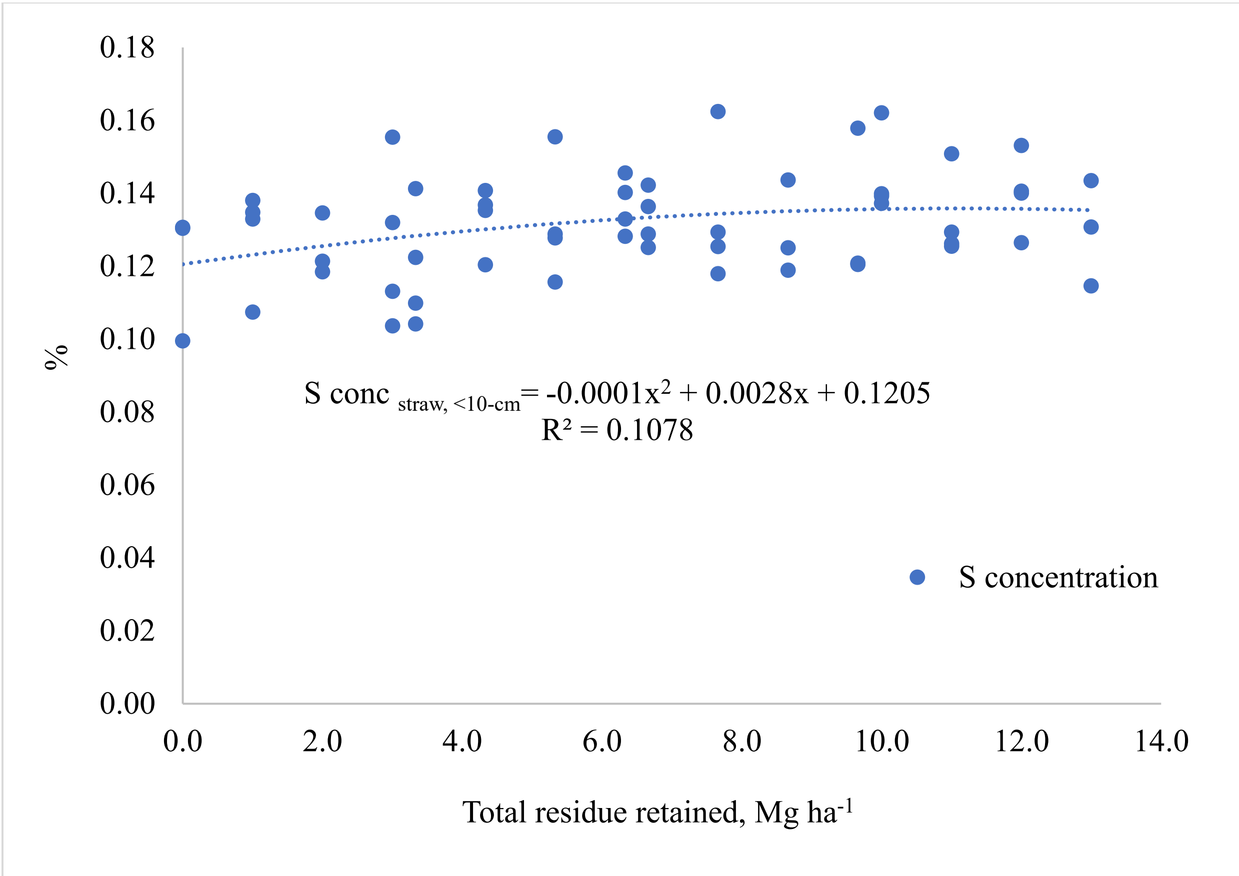


Figure 3. Sulfur concentration (%) for the wheat straw portion between 0 and 10-cm in a factorial design of sixteen total residue retained rates (in Mg dry matter ha^{-1}) from corn and wheat in a corn - wheat/soybean rotation in 2017 in New Kent, experiment 1 (NK1), Virginia. Polynomial best fit line, quadratic equation and R-square value are presented in the figure.

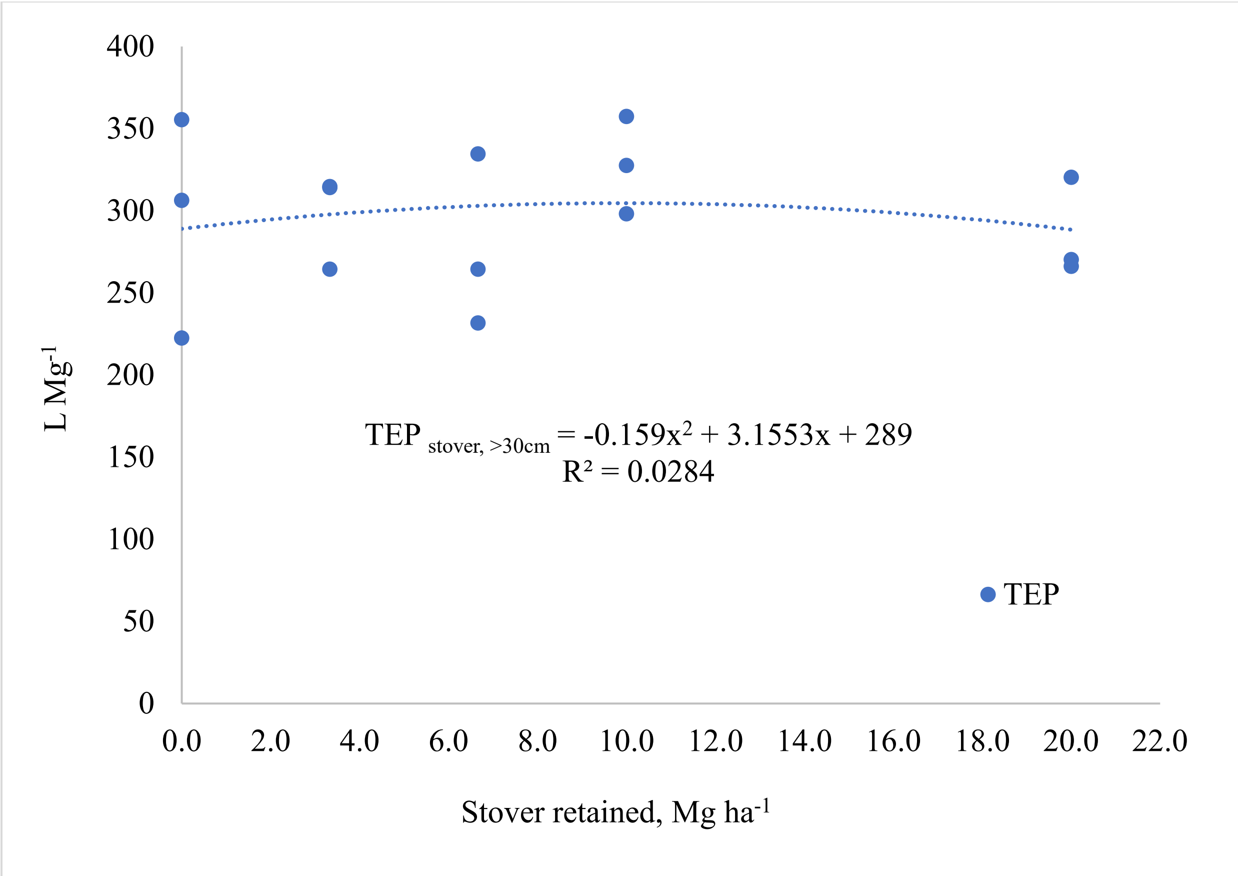


Figure 4. Theoretical ethanol production (TEP, in L Mg^{-1}) for the corn stover portion above 30-cm under five corn stover retained rates (in Mg dry matter ha^{-1}) in a continuous corn rotation in 2017 in Blacksburg (BB), Virginia. Polynomial best fit line, quadratic equation and R-square value are presented in the figure.