8-2018

Methane Emissions from Rice Production on a Silt-loam Soil in Arkansas

Joshua Jeffry Humphreys
University of Arkansas, Fayetteville

Methane Emissions from Rice Production
on a Silt-loam Soil in Arkansas

A dissertation submitted in partial fulfillment
of the requirements for the degree of
Doctor of Philosophy in Crop, Soil, and Environmental Science

by

Joshua Jeffry Humphreys
University of Arkansas
Bachelor of Science in Agriculture, Food, and Life Sciences in Environmental, Soil, and Water Science, 2005
University of North Carolina – Wilmington
Master of Marine Science, 2013

August 2018
University of Arkansas

This dissertation is approved for recommendation to the Graduate Council.

Kristofor R. Brye Ph.D.
Dissertation Director

Richard J. Norman Ph.D.
Committee Member

Lanier Nalley Ph.D.
Committee Member

Jarrod Hardke Ph.D.
Committee Member

Benjamin Runkle Ph.D.
Committee Member

Abstract

Methane (CH_4) emissions from rice (*Oryza sativa* L.) production are a source of concern in the environmental and agricultural communities. New and/or revised agronomic methodologies will be needed to identify production practice combinations that reduce CH_4 emissions without decreasing yields. The objective of this multi-year study was to evaluate the effects of water management (i.e., full-season flood and mid-season drain) (2015), cultivar (i.e., pure-line cultivar 'LaKast' and the RiceTec hybrid "XP753") (2015), soil organic matter (SOM) concentration (2016), and tillage [conventional tillage (CT) and no-tillage (NT)] and urea-based fertilizers [N-(n-butyl) thiophosphoric triamide (NBPT)-coated urea and non-coated urea] (2017) on CH_4 fluxes over the growing season, season-long emissions, and emissions intensity from rice grown in the direct-seeded, delayed-flood production system on silt-loam soils in east-central Arkansas. Vented, non-flow-through, non-steady-state chambers were used to collect gas samples over a 60-min sampling interval for weekly measurements of CH_4 fluxes between flooding and harvest in each year of the study. During the 2015 sampling season, the full-season-flood (77.7 CH_4-C $ha^{-1}season^{-1}$) produced the greatest ($P < 0.01$), while the mid-season-drain (42.8 kg CH_4-C $ha^{-1}season^{-1}$) treatment produced the lowest season-long CH_4 emissions. The mid-season-drain/hybrid combination exhibited the lowest ($P < 0.05$) emissions intensity (2.5 kg CH_4-C Mg $grain^{-1}$). In the 2016 growing season, rice grown in the soil with the largest SOM content, a managed grassland, produced the second largest CH_4 emissions (1166 kg CH_4-C ha^{-1} $season^{-1}$). Methane emissions increased linearly ($P < 0.05$) with increasing SOM and total carbon concentrations (R^2 = 0.81 and 0.85, respectively). In the 2017 study, CH_4 fluxes differed ($P <$ 0.01) between tillage treatments over time and when averaged across tillage, mean season-long CH_4 emissions were 33.4 and 37.2 kg CH_4-C ha^{-1} $season^{-1}$ from NBPT-coated and non-coated

urea, respectively, but were unaffected (*P* > 0.05) by fertilizer treatment. Properly matching water management scheme with cultivar selection and other agronomic management options and soil properties can provide a means to reduce CH_4 emissions and reduce emissions intensity from rice production in the direct-seeded, delayed-flood production system on silt-loam soils.

Acknowledgements

I would like to give my immense gratitude and respect to my major advisor, Dr. Kristofor Brye, without his leadership, guidance, and patience I would have never achieved my goal of attaining my Ph.D. I must also thank he and Dr. Norman for the opportunity to study in the Crop, Soil, and Environmental Science department on a project that not only was interesting but also impactful. I would also like to thank my committee member Dr. Nalley for his advice and guidance in course work and genuinely expanding my skillsets to become a well-rounded scientist. I would also like to thank Dr. Hardke and Dr. Runkle for their professional insight into my project and feedback during my time at the University. Dr. Roberts and Dr. Slaton for allowing me to conduct my field trials in their plots, and the field crew at the Rice Research and Extension Center for my doing a wonderful job in the management of all the field plots. Finally, Casey Rector, who without I could not have completed my program and was instrumental in the design, execution, and data analysis of the treatment years, and who also became a great friend and confidant in our trips to Stuttgart. Thank you, Casey.

Dedication

I dedicate this my dissertation to my loving and supportive wife Ashley, throughout this incredible journey, you have pushed me and supported me in all ways. I would also like to dedicate this to my children Jadzia, Wolfram, and Aerza for always helping me keep perspective of my work-life balance and for the will power to succeed. Finally, to my family who have in many ways encouraged and guided me to become the person I am today and to attain a Ph.D.

Table of Contents

CHAPTER ONE
INTRODUCTION

CHAPTER TWO
LITERATURE REVIEW

CHAPTER THREE

Water management and cultivar effects on methane emissions from direct-seeded, delayed-flood rice production in Arkansas

CHAPTER FOUR

Methane emissions from rice production across a soil organic matter concentration gradient from direct-seeded, delayed-flood rice production in Arkansas

CHAPTER FIVE
Methane production as affected by tillage practice and urea fertilizer type from a silt-loam soil in Arkansas

List of Tables

CHAPTER FIVE

List of Figures

CHAPTER FIVE

CHAPTER ONE

INTRODUCTION

Introduction

Global climate change is the greatest challenge humans will collectively face in the next 100 years (IPCC, 2014). As rainfall patterns change, global temperatures increase, and human populations rise, increasing the efficiency of food production via soil health and water resource management will become paramount for continued survival. Crop breeding programs and natural resource management tools are needed in agricultural production to not only increase yield, but reduce climate-change drivers, such as greenhouse gas (GHG) emissions (IPCC, 2014). The challenges of population increase require that a clear understanding of current conditions and practices exists so that innovative techniques can developed and implemented to offset potential negative agronomic and ecological/environmental effects of climate change.

One area where these goals could have profound influence is in the arena of rice (*Oryza sativa L.*) production. Arkansas is the leading rice-producing state in the US, and, as such, is obligated to pursue a greater understanding of rice-production effects on the environment and how to make rice production more sustainable. Rice production systems differ from other row crops due to the practice of flood irrigation. Moreover, rice grown in the direct-seeded, delayed-flood rice production system common in Arkansas differs substantially from traditional rice systems, where rice is hand transplanted directly to a flooded field. These production differences create unique difficulties as well as opportunities for improving management of soil and water resources need to sustain rice production.

One opportunity for improvement on current rice production practices is by evaluating alternative water management practices to a delayed-flood system, which greatly promotes the production of methane (CH_4) and the subsequent release of CH_4 to the atmosphere. Methane is a potent greenhouse gas with a 100-yr global warming potential (GWP) 34 times greater on a molar basis than carbon dioxide (CO_2) (Forster et al., 2007). Methane is produced in flooded-soil conditions due to the absence of oxygen in the soil (i.e., anoxic or anaerobic conditions), as a byproduct of chemical C reduction. During C reduction, C in soil organic matter (SOM) is converted to CH_4 by a class of microorganisms known as methanogens. Methanogens use fermentation products, such as acetic acid, that are produced by other soil microbes as a food source and produce CH_4 as a waste product. Changing the physical and chemical environment of the topsoil by aeration, either through hybrid rice cultivars or other water management practices, has been shown to be instrumental in reducing CH_4 emissions.

Agronomic practices, such as cultivar selection and water management scheme, are two of the most important factors affecting CH_4 emissions from the saturated soil (Yagi et al., 1997; Wassman et al., 2000). Since agriculture is responsible for 10 to 12% of total global anthropogenic GHG emissions, accounting for nearly 50% of global CH_4 emissions (Smith et al., 2007), mitigation of CH_4 production and release in agricultural settings has profound importance. As of 2011, CH_4 emissions from rice cultivation represented 1.1% of total US CH_4 production (IPCC, 2014). Hybrid cultivars have shown decreased CH_4 emissions compared to pure-line cultivars and offer even greater yield potentials (Rogers et al., 2014). One reason is that the hybrid rice cultivars have more vigorous root growth, as well as increased transport of atmospheric oxygen to the root zone, or rhizosphere, to inhibit reduction of C in SOM and other C substrates to CH_4. Thus, the soil in the rhizosphere is kept from becoming anoxic longer and therefore minimizes CH_4 production by methanogens. Most CH_4 produced in rhizosphere is emitted to the atmosphere by passive transport through aerenchyma tissue. This tissue facilitates the removal of CH_4 from the rice rhizosphere to avoid having excess amounts of CH_4 near the roots.

Another way to reduce the CH_4 emissions from the soil is to alter the water management strategy used for rice production. Rice in the US is generally grown under a delayed-flood condition throughout the growing season. Utilizing a mid-season release of the flood (i.e., mid-season drain) aerates the topsoil again and reduces the time that the topsoil experiences anoxic conditions, which are required for CH_4 production. The mid-season drain water management alternative has historically been used only when controlling for straighthead, a disorder that causes sterility of the spikelets and reduces yield (IPCC, 2014). By using the mid-season drain strategy, the oxidation-reduction (redox) potential of the soil remains above the level needed for CH_4 production for a period of time (approximately 14 days) during the middle part of the growing season, thus reducing total CH_4 emissions from the field.

To reduce CH_4 emissions further from flooded rice, field management practices and cultivar combinations must be developed that will not only reduce CH_4 emissions, but also preserve yields (Lindau et al., 1993). One such field management option could be the use of no-tillage practices for rice production. No-tillage has been used to increase SOM, thus improving soil tilth and water and nutrient movement as well. However, little is known about the potential effects of tillage-practice alternatives on CH_4 production and emissions.

Consequently, research is still needed to characterize the magnitude of growing-season CH_4 fluxes and emissions in relation to common and alternative rice management practices. Rice grown in the direct-seeded, delayed-flood production system common to the Lower Mississippi River Delta region of eastern Arkansas offers the unique opportunity to further knowledge regarding the magnitude of GHG emissions, particularly CH_4, from rice production and potential mitigation strategies to reduce GHG emissions. The use of new hybrid rice cultivars and alternative water management schemes in large-scale rice production may be two ways to achieve the goals of producing enough food to feed the world's growing population, while mitigating GHG emissions to slow anthropogenic climate change.

Literature Cited

Forster, P., V. Ramaswamy, P. Artaxo, T. Berntsen, R. Betts, D.W. Fahey, J. Haywood, J. Lean, D.C. Lowe, G. Myhre, J. Nganga, R. Prinn, G. Raga, M. Schulz, and R. Van Dorland. 2007. Changes in atmospheric constituents and in radiative forcing. Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change [Solomon, S., D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M. Tignor and H.L. Miller (editors.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Intergovernmental Panel on Climate Change (IPCC). 2014. Intergovernmental Panel on Climate Change (IPCC), United States Greenhouse Gas Inventory Chapter 6. IPCC, Geneva, Switzerland. pp. 15-21.

Lindau, C.W., P.K. Bollich, R.D. DeLaune, A.R. Mosier, and K.E Bronson. 1993. Methane mitigation in flooded Louisiana rice fields. Biol. Fert. Soils: 15:174-178.

Rogers, C.W., K.R. Brye, A.D. Smartt, R.J. Norman, E.E. Gbur, and M.A. Evans-White. 2014. Cultivar and previous crop effects on methane emissions from drill-seeded, delayed-flood rice production on a silt-loam soil. Soil Sci. 179:28-36.

Smith, P., D. Martino, Z. Cai, D. Gwary, H. Janzen, P. Kumar, B. McCarl, S. Ogle, F. O'Mara, C. Rice, B. Scholes, and O. Sirotenko. 2007. Agriculture. In Climate Change 2007: Mitigation. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change [B. Metz, O.R. Davidson, P.R. Bosch, R. Dave, L.A. Meyer (editors)], Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Wassman, R., R.S. Lantin, H.U. Neue, L.V. Buendia, T.M. Corton, and Y. Lu. 2000. Characterization of methane emissions from rice fields in Asia. III. Mitigation options and future research needs. Nutr. Cycl. Agro. 58:23-36.

Yagi, K., H. Tsuruta, and K. Minami. 1997. Possible options for mitigating methane emission from rice cultivation. Nutr. Cycl. Agro. 49:213-220.

CHAPTER TWO

LITERATURE REVIEW

Literature Review

Global Atmospheric and Climate Changes

Anthropogenic climate change (ACC) due to increases in GHG emissions has become a concern in the scientific community and in the public health realm. Anthropogenic climate change is thought of as the influence of human activity over planetary systems regarding production of greenhouse gases. The main anthropogenically and naturally produced GHGs are carbon dioxide (CO_2), nitrous oxide (N_2O), and methane (CH_4), which have experienced concentrations increases to unprecedented levels not observed for 800,000 years (IPCC, 2014). Human activity over the last 30 years (i.e., 1983 to 2012) has caused the warmest climate of the last 1400 years in the Northern Hemisphere (IPCC, 2014). Sea ice reductions from 1978 to 2012, due to increased land and ocean temperatures, have occurred at a rate of 2.1 to 3.3% per decade, with a predicted ice-free Arctic ocean in the summer season by mid-century (2050) (IPCC, 2007). Accelerated ice melt has increased the global mean sea level over the last 100 years by 0.19 m, which is a larger mean increase than over the last 2000 years. Oceanic pH decreased 26% in the same 100 year timeframe due to increased oceanic absorbance of anthropogenically emitted CO_2 and the associated acidification is most likely leading to increased coral bleaching and reef destruction (IPCC, 2014). To better understand planetary temperature changes associated with increasing atmospheric GHG concentrations, land and ocean surface temperature data have been combined to calculate a globally averaged linear trend. This trend shows air temperatures increased by 0.85 [0.65 to 1.06] °C between 1880 and 2012 (IPCC, 2014).

Determining sources and magnitude of ACC are essential for predicting effects on environmental systems, most notably increased planetary temperature. A common base line for GHG concentrations is to use the pre-industrial revolution concentrations of GHGs in the atmosphere. Those GHG concentrations were 280 ppm for CO_2, 0.7 ppm for CH_4, and 0.18 to 0.26 ppm for N_2O, while 2005 concentrations of GHGs were 379 ppm for CO_2, 1.8 ppm for CH_4, and 0.32 ppm for N_2O (Forster et al., 2007). Determining a full inventory and understanding of GHGs and their increasing atmospheric concentrations are necessary to predict effects on environmental systems. Cumulative emissions of CO_2 from 1750 to 2012 were 2040 ± 310 Gt of CO_2, with 40% of those emissions remaining in the atmosphere, 30% being absorbed by the oceans, and the remaining 30% being sequestered in plants and soils (IPCC, 2014). Total GHG emissions peaked in the US during 2007 at 7263 Tg of CO_2 equivalents. Total US GHG emissions increased by 8.4% from 1990 to 2010, with a 1.6% decrease from 2010 to 2011 to 108 Tg of CO_2 equivalents. Overall CO_2 emissions from 1990 to 2011 increased by 504 Tg of CO_2 equivalents, while, during the same 21-yr span, CH_4 emissions decreased by 57.2 Tg of CO_2 equivalents (IPCC, 2014). Total CH_4 US emissions for 2011 were 587.2 Tg of CO_2 equivalents (IPCC, 2014) .

Global warming potential (GWP) is an expression of the relative radiative effect of a given substance compared to CO_2, integrated over a chosen time period, to determine CO_2 equivalents (IPCC, 2001). Global warming potentials are typically assigned based on CO_2 equivalents over a 100-yr time period, with CO_2 being the baseline with a value of 1. The GWP for CH_4 and N_2O are 23 and 296, respectively (IPCC, 2001). In other terms, 1 kg of CH_4 has the same GWP as does 23 kg of CO_2. The GWP expression helps determine the impact of any gas on the radiative forcing (RF) on the atmosphere. Over the last 250 years, GHGs have created a combined RF of +2.63 W m^{-2}, with CO_2 contributing +1.66 W m^{-2}, CH_4 contributing +0.48 W m^{-2}, halocarbons contributing +0.34 W m^{-2}, and N_2O contributing +0.16 W m^{-2} (Forster et al., 2007). Other human activities, which include increased stratospheric water vapor, tropospheric ozone, and contrails, collectively contribute a total of +0.35 W m^{-2} (Forster et al., 2007). Global warming potential also includes negative impacts on RF and include atmospheric aerosols contributing -0.5 W m^{-2} and indirect effects of aerosols on cloud albedo contributing -0.7 W m^{-2}. The net effects of RF are estimated to be +1.6 W m^{-2} from purely anthropogenic processes, which is approximately five times greater than from natural processes (Forster et al., 2007).

Global warming potential and RF can be combined to form the concept known as the greenhouse effect. The greenhouse effect is a collective mechanism that infers the ability of solar radiation to leave the Earth's atmosphere (IPCC, 2001). By measuring the absorption of long wave radiation and its atmospheric re-radiation and reflection as infrared radiation, the greenhouse effect on the planet can be determined. The greenhouse effect is positively correlated to atmospheric GHG concentration. Based on the direct correlation between atmospheric GHG concentration and the greenhouse effect, it is possible to project global surface temperature change for the latter part of the 21st century (i.e., 2081 to 2100), which is expected to likely exceed 2°C relative to 1850 to 1900 values and 0.3°C to 1.7°C relative to 1986 to 2005 values (IPCC, 2014). Weather events related to the increased global mean surface temperature include more frequent hot and fewer cold temperature extremes and heat waves with increased frequency (IPCC, 2014).

The increased atmospheric GHG concentrations that have occurred in the last 250 years have had a global effect on atmospheric chemistry and can have profound effects particularly on tropospheric chemistry. In the troposphere, the oxidation of CH_4 plays a key role as a source of carbon monoxide (CO) and dihydrogen gas (H_2) (Cicerone and Shetter, 1981). At greater altitudes (10 to 50 km) in the stratosphere, CH_4 oxidation is a vital chlorine acceptor in the ozone cycle and accounts for almost half of the water vapor and H_2 quantities in the atmosphere

(Cicerone et al., 1974). Increases in stratospheric water vapor act to cool the stratosphere, but act to warm the troposphere, whereas the reverse is true, as stratospheric water vapor decreases the troposphere cools (Solomon et al., 2010). Changes in stratospheric water vapor concentrations may point to a source of unforced decadal variability or even an environmental feedback loop that is influential in climate change and may be related to CH_4 oxidation (Solomon et al., 2010).

Methane Production

Unlike the majority of CO_2 production, CH_4 is produced under anoxic conditions when C-containing organic matter is converted to CH_4 by a class of microorganisms known as methanogens. Methanogenesis can occur in a variety of natural and anthropogenic systems.

As of 2005, agriculture contributes about 47% of total anthropogenic emissions, while the remaining non-agricultural sources of CH_4 production are natural gas systems, landfills, and coal mining, which make up over 50% of the total CH_4 emissions in the US (Smith et al., 2007). The main agricultural sources of CH_4 in the US are enteric fermentation and manure management, with over 95% of total agriculturally related CH_4 emissions as of 2012, with rice cultivation and field burning making up 3.7% of the total agricultural CH_4 releases (IPCC, 2014). As of 2012, CH_4 emissions from rice cultivation represented 1.1% of overall US CH_4 production (IPCC, 2014). As of 2013, atmospheric CH_4 inputs from enteric fermentation, manure management, rice production, and biomass burning contributed approximately 8.1% of total US anthropogenic GHG emissions to the environment (IPCC, 2014).

In the soil environment, whether natural and undisturbed or agricultural, the main source of CH_4 in the soil column is in the topsoil, where > 99% of the total soil-produced CH_4 is emitted (Mitra et al., 2002b). Under well-drained conditions, oxygen (O_2) is sufficiently available to sustain aerobic oxidation or decomposition of C-containing soil organic matter (SOM) that is concentrated in the topsoil. However, when the soil water content increases to saturation, and depending on soil temperature, soil texture, and SOM concentration, aerobic decomposition quickly depletes the available O_2 in the saturated soil zone as water displaces O_2-containing air and anaerobic respiration begins (IPCC, 2014). This change in O_2 concentration can be measured as the oxidation-reduction (redox) potential (Eh) in the soil. By using a platinum electrode embedded with a silver-chloride reference electrode, it is possible to observe a system's ability to donate or accept electrons. In well-aerated soils, the soil Eh may approach +700 mV and may decrease to as little as -300 mV in saturated soils with large SOM concentrations (Patrick et al., 1996).

When O_2 is no longer in sufficient concentration for aerobic processes to continue, the soil Eh begins to decrease. As a soil becomes anaerobic and O_2 becomes scarce as a reducing agent, acetic acid (CH_3COOH) and free hydroxyl radicals, which can be toxic to aerobic microorganisms, are produced. Many organisms in the soil would perish due to the accumulation of these fermentation products. Methanogens, however, sequentially use nitrate (NO_3^-; +280 to +220 mV), manganese (Mn^{4+}; +220 to +180 mV), iron (Fe^{3+}; +180 to +80 mV), sulfate (SO_4^{2-}; -140 to -170 mV), and eventually CO_2 (-200 to -280 mV) as electron acceptors for anaerobic respiration, which removes the fermentation products, but is a much slower process than aerobic respiration (van Breemen and Feijtel, 1990; Patrick et al., 1996). This sequential use of terminal electron acceptors plays a vital role in removing fermentation products that are produced in the environment (Mitra et al., 2002a).

Two main biochemical processes (i.e., hydrogenotrophic and acetoclastic) exist where CO_2 is reduced to CH_4, thus releasing energy for metabolic processes. These two biochemical processes contribute to three main pathways exist to produce CH_4 in an anoxic soil. One, H_2 reduction of CO_2 by a class of bacteria called chemoautotrophic methanogens (i.e., hydrogenotrophic): $CO_2 + 4H_2 \rightarrow CH_4 + 2H_2O$. Second, other strains of methanogens can also use HCOOH or CO as a C source for CH_4 production: $4HCOOH \rightarrow CH_4 + 3CO_2 + 2H_2O$ or $4CO + 2H_2O \rightarrow CH_4 + 3CO_2$. Third, CH_4 can also be produced by methylotrophic methanogens (i.e., acetoclastic) who use a methyl-group-containing C source, such as methanol, acetate, or trimethylamine: $4CH_3COOH \rightarrow 3CH_4 + CO_2$, $CH_3COOH \rightarrow CH_4 + CO_2$, $4(CH_3)^3\text{-}N + 6H_2O \rightarrow 9CH_4 + 3CO_2 + 4NH_3$ (Papen and Rennenberg, 1990; Sass et al., 1990; Ferry, 1992; Deppenmeier et al., 1996).

Acetoclastic methanogenesis accounts for almost 66% of the CH_4 produced in nature, while hydrogenotrophic methanogenesis accounts for the other 33% (Ferry, 1992). About 69% of the CH_4 sources are the result of these microbial processes with another 6% attributed to chemical production of CH_4 from plant matter. The remaining 25% of CH_4 sources are associated with mining, burning of biomass, and combustion of fossil fuels (Conrad, 2009). The natural sources of CH_4 are plants (6%), wetlands (23%), termites (3%), oceans (3%), and gas hydrates (2%)(Conrad, 2009). The anthropogenically influenced sources are rice fields (10%), ruminants (17%), landfills (7%), sewage treatment (4%), and biomass burning (7%), while the remaining 18% is attributed to fossil fuel burning (Conrad, 2009).

Rice Production and History

Historically, rice production dates back many thousands of years to as early as 4000 B.C. in the south Pacific region, 2800 B.C. in China, and 2500 B.C. in India (Chang et al., 2012).

Currently, rice is produced in 112 countries covering the latitudes of 53° north to 35° south, and 95% of rice produced is consumed in Asia (Chang et al., 2012).

Rice is the predominant staple food for 17 countries in Asia and the Pacific, nine countries in North and South America, and eight countries in Africa. Rice is produced differently from all other cultivated row crops in the world, as much of the global rice production, and most of the rice production in the US, occurs under flooded-soil conditions for most of the growing season. Other rice production strategies rely on different depths of field flooding, with some flooding more than a 1 m in depth in the south Pacific region. World rice production in 2012 was 738.1 million tons. In 2012, China and India produced 27.7% and 20.7%, respectively, while the US produced 1.2% of the world's rice (van Breemen and Feijtel, 1990; FAO, 2012).

Rice Production in the United States

Rice cultivation in the United States began around 1609, as an initial planting in Virginia which was believed to be brought in from Madagascar on a cargo ship (Chang et al., 2012). Other trial plots soon followed along the south Atlantic coast of the United States. Rice production in South Carolina was well-established by about 1690. Production then spread to Georgia and areas comprising Mississippi and southwest Louisiana in the Mississippi River Delta. Rice production in the Mississippi River Delta moved up to the Mississippi River flood plain in Arkansas and over to adjoining Texas. Rice was brought into the Hawaiian Kingdom by Chinese travelers between 1853 and 1862 (Chang et al., 2012). However, due to competition with sugarcane (*Saccharum officinarum*) and pineapple (*Ananas comosus*), rice production did not thrive as an agro-industry in Hawaii (Chang et al., 2012). California was the last state to begin producing rice, which occurred sometime between 1909 and 1912 (Chang et al., 2012).

Since 1973, Arkansas has been the nation's leading rice-producing state with rice grown in 40 of the state's 75 counties. Rice, as of 2013, continues to rank as one of the top three crop commodities in cash receipts for Arkansas farmers (Hardke, 2014).

As of 2015, approximately 1.3 million ha of rice was planted in the U.S. with an average yield of 8.4 Mg ha^{-1} for a total production of 11.93Tg of rough rice produced (USDA-NASS, 2015). The top rice producing states are Arkansas, California, Louisiana, Mississippi, Missouri, and Texas, with Arkansas leading production at 5.52Tg, equivalent to 46.3% of total U.S. production of rough rice in 2015 (USDA-NASS, 2015). California, Louisiana, Mississippi, Missouri, and Texas were 22.2, 13, 5.9, 6.7, 7.2%, respectively, of total U.S. rice production. However, California did average the largest 2015 per hectare production at 9977 Mg ha^{-1} in the U.S (USDA-NASS, 2015).

Rice Production in Arkansas

Rice production initially occurred in Arkansas in 1902 with 0.41 ha of rice grown in Lonoke County (Hardke and Wilson, 2012). Since then, Arkansas rice production has grown to producers planting 601,362 ha in 2014 and providing nearly 46% of the total rice production in the US. In Arkansas, the largest production area for rice is located in the eastern part of the state along the Mississippi River Alluvial Plain in Poinsett, Lawrence, and Jackson counties around the Stuttgart area (Hardke, 2014). The six largest rice-producing counties in Arkansas during 2015 included Arkansas, Cross, Jackson, Lawrence, Lonoke, and Poinsett representing 41.7% of the state's total rice acreage (Hardke, 2016). The average 2015 rice yield in Arkansas was 8047 kg ha^{-1}, with a total value of $1.1 billion (USDA, 2015). The majority, as of 2016, (53.6%) of rice is still produced on silt loam soils, while clay or clay loam soils (20.6% and 20.9%, respectively) has become static over recent years (Hardke, 2016).

Typical Agronomic Practices

Potential decision-making points for rice producers are cultivar selection, fertilizer form and application times, water management, herbicide/pesticide rate, and tillage practices. These decisions reflect the ability of the producer and the needs of the crop and the field. In Arkansas direct-seeded delayed-flood with multiple inlet irrigation using hybrid rice with Clearfield technology is the most abundant planting and cropping system in the Mississippi river alluvial plain. This system is heralded as the most profitable, environmentally friendly, and efficient rice system in Arkansas, and perhaps the world.

Rice Cropping Systems

Obtaining a level seedbed free of obstructions such as potholes and abundant trash or stubble is desired during field preparation for any production system. In Arkansas, over 60% of the rice produced was planted using conventional tillage methods in 2015 (Hardke, 2016). This historically involves fall tillage, followed by additional spring tillage to prepare the seedbed. The balance of rice acres were planted into a stale seedbed (30.1%) or using no-till (6.3%) systems (Hardke, 2016). No-till rice production is uncommon but is done in a few select regions around the state. Conventional tillage practices on a silt loam soil usually involve the use of a disk, followed by a field cultivator, then a land plane or roller to finish field preparations. However, tillage requirements may differ depending on soil texture, previous crop or other field conditions. In clay soils, aggressive tillage may produce clods which can impede planting efforts later in the season, and create field abnormalities. With a departure from convention, the use of reduced-tillage practices has increased from 2003 to 2013. Two no-tillage methods currently used in Arkansas: 1) stale seedbed, where the soil is tilled and floated in the fall or late winter, or 2) true no-tillage, where rice is directly seeded in the previous crop's stubble (Hardke, 2014). To speed emergence, the use of a roller behind the drill often increases seed-to-soil contact and by compacting the soil (Hardke, 2014). Stale seedbed or no till seeding has been show to increase seed-to-soil contact on clay soils. The type of tillage system helps dictate the cropping system, timing, cultivar, and weed management practice used for the producer.

Weed Management

Arkansas rice producers spend an estimated $100 million per year on weed control (Scott et al., 2014). The top five most costly weeds that afflict Arkansas rice producers are red rice (*Oryza S*.), barnyard grass (*Echinochloa crus-galli*), beaded sprangletop (*Leptochloa fascicularis*), Amazon sprangletop (*Leptochloa panicoides*), and broadleaf signal grass (*Urochloa platyphylla)*, with barnyard grass being the most common weed in rice (Scott et al., 2014). One of the most common and widely used herbicides to control grasses is propanil (N-(3,4-Dichlorophenyl) propanamide) which has been used for rice weed control for the last 40 years (Scott et al., 2014). Propanil is known as a contact herbicide with no residual activity and generally requires two applications before a permanent flood is established for complete grass control (Scott et al., 2014). Maximum application amounts used are 6.75 kg ha^{-1} active ingredient (a.i.) at the one to three leaf stage when temperatures are above 25°C (Scott et al., 2014). However, due to weed populations developing resistance to propanil and other herbicides new technology was needed to assist in rice production. To help combat weed pressure in Arkansas and give producers new options for weed control Clearfield rice was introduced into the market in 2002. Clearfield rice is a non-transgenic rice was developed to be tolerant to the imadazolinone family of herbicides such as Newpath and Beyond herbicides (Scott et al., 2014). Newpath is an herbicide that controls many grass and broad leaf weeds in rice, and is considered a long-residual herbicide that persists in the soil for more than one year. Red rice resistant management options dictate that Clearfield rice not be planted in consecutive years. With these considerations, soybeans are generally grown in rotation with Clearfield rice. Command, which is applied as a preemergent herbicide with a short-term residual effect (>14 days), provides excellent control of sprangletop, barnyard grass, and broadleaf signal grass. The Command rate determines the length of residual effectiveness. Residual grass control can be achieved using as little as 0.34 kg ha^{-1} of active ingedient on silt loam soils which has produced excellent results (Scott et al., 2014). Command is applied from 14 days before planting to as late as seven days after planting to ensure a clean weed free environment.

Planting

In Arkansas, rice planting typically begins during the last week of March and continues into early June. Planting dates have not changed appreciably over the last 30 years (Hardke, 2014). Approximately 50 and 95% of planting is completed by April 24 and June 1, respectively (Hardke, 2014). The majority (85%) of the rice in Arkansas is produced in a drill-seeded, delayed-flood production system with only 5.5% using a water-seeded system (Hardke, 2014). This system is also in majority use throughout the Mississippi River flood plain in southeastern Missouri and Louisiana and in Texas. The remainder of planted rice is either broadcast onto dry soil (i.e., dry-seeded) or into a field that is already flooded (i.e., water-seeded) (Hardke, 2014). For dry seeded beds, rice is broadcast on to a dry soil is then covered by flushing the levees or more commonly a final tillage operation (Hardke, 2014). However, in California the water-seeded rice production system is dominant. In a water-seeded production system the rice seed is first soaked in water for 48 hours and then flown on to a field by plane. The seed is dropped in to a flooded field and kept at a depth of 12 to 13 cm of water and is maintained at that depth throughout the entire growing season. In the majority of counties in Arkansas, rice is drill-seeded with 320 seeds m^{-2} for pure-line varieties or 110 to 160 seeds m^{-2} for hybrid varieties. These planting densities are used to obtain an optimum stand density (Hardke and Wilson, 2012). Seeding rates for both pure-line and hybrid varieties should be increased by 20% for broadcast seeding, poor seedbed condition, or clay soils and by 10% for no-tillage seedbeds (Hardke, 2014).

Cultivar Selection

Rice cultivars in the U.S. and Arkansas have seen a development boom in the last 15 years. The introduction of hybrid and Clearfield technologies in the U.S. as well as an expanded pure-line breeding program in Arkansas, Arkansas is currently the leader in pure-line and hybrid acres planted (Hardke and Wilson, 2012). The first hybrid rice cultivars were released in 2002 and 2003 and Clearfield rice was first planted on limited acreage in 2002. (Hardke and Wilson, 2012). Clearfield rice which has been bred through traditional techniques to be tolerant to imidazilanone and imazamox herbicides continues to play a significant role in rice production in Arkansas. This technology accounted for 44% of the total Arkansas rice acreage in 2015 of all cultivars combined. In Arkansas the most widely planted cultivar in 2015, a hybrid-Clearfield cultivar, was RiceTec CLXL745 which were planted to 19.9% of the acreage, followed by RiceTec XP753 (14.5%), Jupiter (14.4%), Roy J (13.1%), CL151 (12.4%), LaKast (5.0%), Mermentau (4.1%), CL111 (3.8%), RiceTec CLXL729 (3.2%), and Wells (1.6%) for the state of Arkansas (Hardke, 2016).

Water Management

As a semi-aquatic plant, rice requires between 1250 to 8500 m^3 ha^{-1} (4.9 to 33.5 in) globally of water per growing season, making water management and water conservation critical in the rice production system (de Avila et al., 2015; Henry et al., 2016). Groundwater is used to irrigate 76.4% of the rice acreage in Arkansas with 23.6% of remaining acres irrigated with surface water obtained from reservoirs, streams, or bayous. The primary irrigation practice in Arkansas is the use of the conventional levee and gate system. As of 2015, rice farmers utilize this practice on 40.6% of the rice acreage in Arkansas (Hardke, 2016). In Arkansas, the drill-seeded, delayed-flood rice production system is the predominate production system, accounting for 85% of total planted-rice area, for which annual irrigation-water use averaged 763 mm (30.0 in) over a 10-yr period between 2003 to 2012 (Henry et al., 2016). Two flood regimes that are currently used in Arkansas are the continuous flood and the mid-season drain. In the drill-seeded, delayed-flood production system that uses either flood system, flood establishment by irrigation typically occurs at the 4- to 5-leaf stage. The flood is maintained at a 5- to 10-cm flood depth until approximately two weeks prior to harvest when the flood is released for the soil to dry to facilitate combine harvesting (Hardke, 2014). To accomplish the mid-season drain regime, the initial flood is still established; however, a full drain of the field occurs approximately 20 days after initial flood establishment and reflooding occurs after the soil dries out to the point of surface cracking at roughly day 25 after initial flood. All other management practices are kept the same. In a drill-seeded, delayed-flood production system, fields are mostly flood-irrigated either by multiple-inlet irrigation systems or with a conventional levee and gate system. Rice production systems are mainly irrigated by pumping groundwater from the Alluvial Aquifer which amounts to 78% of the total acres flooded, with the remainder of the irrigation water split between surface water sources and precipitation (Hardke, 2014).

Fertilization

Many decisions need to be addressed before an effective nitrogen management program can be implemented. Understanding potential constraints, such as cultivar, equipment, or field management options, can have a tremendous impact on the choices made for nutrient fertilization. The most important nutrient for optimal/maximal rice production is nitrogen (N), but potassium (K) and phosphorus (P) are key nutrients as well (Norman et al., 2013).

Profitable rice grain yields are highly correlated with proper and effective N fertilizer management. Nitrogen is needed by rice in the largest quantities of any nutrient, and it is typically the largest input cost for rice producers. As such, the effective management of N fertilizer presents a greater challenge to the rice producer than does any other fertilizer nutrient. Nitrogen, in addition, can provide greater returns in increased rice yield for effective management. Common total nitrogen rates in Arkansas for hybrid varieties are 135 to 170 kg N ha^{-1} and for pure-line varieties 125 to 170 kg N ha^{-1} on a silt-loam soil following a soybean rotation (Norman et al., 2013). In Arkansas, the most common N fertilizers used are urea (46% N) and ammonium sulfate (21% N) (Hardke, 2014). However, other fertilizer choices can include organic fertilizers (i.e., chicken or swine manures), pelletized manures, liquid inorganic N-containing solutions, or pelletized inorganic N. On average, one metric ton of poultry litter contains 52 to 66 kg K_2O and 72 kg P_2O_5, making it equivalent to 86 to 110 kg of muriate of potash (0-0-60) and 162 kg of triple superphosphate (0-46-0) (Norman et al., 2013). When using organic sources of fertilizer, inorganic sources may also be needed in smaller quantities to complete the nutritional profile needed for the rice plant.

Depending on flood management capability, applying N in a single application early in the growing season or in multiple applications throughout the growing season is another critical decision for the producer (Norman et al., 2013). However, factors for producers considering the optimum pre-flood N application method are: can the field be flooded in two days or less for silt-loam soils and in five to seven days at most for clay soils, should the urease-inhibitor NBPT (N-(n-butyl) thiophosphoric triamide) be used or not with urea, and can the field be kept flooded for at least three weeks (Norman et al., 2013). Two effective application methods are viewed as the most practical N-fertilization methods: 1) an optimum pre-flood N application, and 2) a standard two-way split application (Norman et al., 2013). The two-way split application, which is most common in Arkansas, consists of a first application pre-flood (2 to 5 days before flooding) and a split application mid-season (Norman et al., 2013). The two-way split application can be used effectively on fields where large field size, limited irrigation capacity, or other factors can compromise the ability of the producer to establish and maintain the flood across the field (Norman et al., 2013). Mid-season N, typically 50 kg N ha^{-1}, should be applied for pure-line cultivars between internode elongation/panicle initiation and ½-inch internode elongation and for hybrid cultivars at the early boot stage (Norman et al., 2013). Optimum pre-flood N rates range from 100 to 118 kg ha^{-1} for pure-line varieties, and 100 to 135 kg ha^{-1} hybrid varieties (Hardke, 2014). In either N-application method, proper management of the pre-flood N application is essential to ensure high rice yields and reduced N losses.

Nitrogen loss to the atmosphere by volatilization or due to surface water runoff are of great concern to producers. These losses are usually related to the producer's ability to flood a field in a timely manner and to maintain the flood throughout the year (Norman et al., 2013). Urease inhibitors, such as NBPT, contribute to N retention in the soil in a flooded system by keeping the N in a stable form until the flood can be established and the N can adsorb to the soil particles (Norman et al., 2013). Urea is often treated with urease-inhibitor NBPT, which lowers potential ammonia volatilization losses from the fertilizer to the atmosphere (Norman et al., 2013).

Phosphorus fertilizer recommendations in Arkansas for rice are based on soil testing for soil pH and available P. Use of soil pH and available P accurately identifies soils that respond to P fertilization to produce optimal plant growth and yield in Arkansas. Optimum plant available phosphorus occurs when the pH is below 6.5 (Norman et al., 2013). For precision-graded soils which are routinely used as rice fields, 44 kg P_2O_5 ha^{-1} is the minimum recommended amount up to a high rate of 110 kg P_2O_5 ha^{-1} (Norman et al., 2013). These applications commonly use triple super phosphate (TSP, 0-46-0) as the pre-plant phosphorus fertilizer source in Arkansas. Higher rates of P (67-110 kg) are applied as a split application with one-half to two-thirds applied pre-plant and the remainder applied prior to flooding (Norman et al., 2013).

Potassium (K) fertilizer is recommended on soil test results lower than 131 ppm K (< 293 kg K ha^{-1} (Norman et al., 2013). Potassium fertilizer recommendations are 67 kg K_2O ha^{-1} that test less than 60 ppm K (≤135 kg K ha^{-1}) are considered to be very susceptible to K deficiency (Norman et al., 2013). Application of K fertilizer usually occurs in the fall or winter before seeding, due to the fact fertilization may help reduce the amount of salts in the root zone.

Harvesting, Milling, and Ratooning

Harvest of the primary rice crop typically occurs in the middle to end of August and finishes by the end of October to early November, which is somewhat earlier than during the past

20 years due to increased harvest efficiency and the development of shorter-season rice cultivars (Hardke and Wilson, 2012). Harvest conditions contribute greatly to rice milling quality. For instance if rice grain dries to 15% moisture in the field and is rewetted due to rain or heavy dew, fissuring of the kernel may occur, which in turn affects the rice grade and therefore the price paid per bushel (Hardke and Wilson, 2012).

Along with earlier planting dates and earlier maturing rice cultivars that allow for earlier harvest, there become opportunity to produce a second rice crop, known as a ratoon crop. The ratoon crop is a second rice crop in which the regrowth of tillers from the stubble that is harvested (IPCC, 2014). Ratooning was almost non-existent in Arkansas until 2012 when 10522 ha of ratoon rice were harvested, which was roughly 5% of Arkansas' total rice harvest (IPCC, 2014). The main ratooning states are Florida, Louisiana, and Texas with 44, 40, and 61%, respectively, of their total state rice harvest as a ratoon crop (IPCC, 2014). Ratoon crops produce a third of the harvest of the main crop. However, the input cost is significantly lower due to the fact that the producer only needs to fertilize, re-flood, and harvest a second time.

Methane Emissions from Rice

The first comprehensive measurements of CH_4 emissions from rice fields were reported in the early 1980s in California rice paddies on a Vertisol (Capay clay) (Cicerone and Shetter, 1981; Cicerone and Shetter, 1983). Results of these early field studies had a profound effect on the global estimations of CH_4 release from anthropogenically influenced sources.

Three CH_4 release mechanisms from rice fields have been identified: plant-mediated transport, molecular diffusion at soil-water interfaces, and ebullition of gas bubbles (Cicerone and Shetter, 1981). Investigations in Italy showed the transport of CH_4 through the rice plant and release from the culm as a main mode of CH_4 release from rice paddies rather than diffusion from the water surface (Holzapfel-Pschorn and Seiler, 1986). In Arkansas, Smartt et al. (2016) reported that CH_4 emissions on a Sharkey clay (very fine, smectitic, thermic Chromic Epiaquerts) were greater from N-fertilized rice 35.6 kg CH_4-C ha^{-1} $season^{-1}$ compared with 8.94 and 1.75 kg CH_4-C ha^{-1} $season^{-1}$ from non-N-fertilized rice and bare soil, respectively. These finding agree with previous studies examining plant-mediated transport as the main source of CH_4 from the soil profile. Smartt et al. (2016) also demonstrated the lack of molecular diffusion of CH_4 to the atmosphere by way of the soil surface based on very low CH_4 emissions from non-vegetated bare soil (1.8 kg CH_4-C ha^{-1} $season^{-1}$). When considering that between 58 and 80% of CH_4 produced in a rice paddy is oxidized by methanotrophic bacteria and not emitted to the atmosphere by diffusion or ebullition of gas bubbles, the crucial role of the rice plant in expelling CH_4 from the soil profile is apparent. Methane can also be removed from the soil profile as it is consumed as a carbon substrate for soil microbes (Holzapfel-Pschorn and Seiler, 1986; Sass et al., 1990).

As of 2013, the US Environmental Protection Agency (USEPA) uses a single CH_4 emission factor 178 kg CH_4-C ha^{-1} $season^{-1}$ to determine annual emissions from rice producing fields in the United States(USEPA, 2014). The USEPA CH_4 estimate is used for primary-crop rice production, however, ratoon crops have been shown to emit greater CH_4 than the primary rice crop. Methane emissions from a primary-crop rice have been reported to range from 61 to 500 kg CH_4 ha^{-1} $season^{-1}$, with ratoon-crop emissions ranging from 481 to 1490 kg CH_4 ha^{-1} $season^{-1}$ (IPCC, 2014). Greater CH_4 emissions occur from ratoon cropping because the stubble from the first crop has had no time to decompose aerobically because of extended flooded periods in the field. Keeping the field flooded for a ratoon crop results in a large amount of organic substrate that is decomposed anaerobically resulting in elevated CH_4 emissions (IPCC, 2014). Additionally, when the previous crop residue is abundant, such as in rice-rice rotation, there is greater CH_4 release from the field unlike in rice-soybean rotations where there is less substrate to decompose because of decreased soybean field residue from the previous growing season (Rogers et al., 2014).

The US Environmental Protection Agency has identified numerous factors that affect CH_4 emissions from rice. Rice cultivar, soil texture, crop rotation/previous crop, water management scheme, and the concentration of C-containing substrate to support methanogenesis are several of the major factors known to affected CH_4 emissions(USEPA, 2014).

Early field research in Arkansas documented multiple environmental and agronomic effects on CH_4 emissions from rice (Rogers et al., 2013; Brye et al., 2013; Rogers et al., 2014. Rogers et al. (2013) conducted the first study in Arkansas examining the influence of cultural practices associated with the drill-seeded, delayed flood production system on CH_4 emissions, for which the long-grain, pure-line rice cultivar 'Wells' and full-season flood regime were used. The field study was conducted at the Rice Research and Extension Center (RREC) near Stuttgart, AR on a Dewitt silt-loam soil (fine, smectitic, thermic Typic Albaqualfs) (Rogers et al., 2013). Rogers et al. (2013) reported CH_4 emissions averaged of 195 kg CH_4–C ha^{-1} $season^{-1}$ for the drill-seeded, delayed-flood rice production system using a no-N control and an optimal N rate of 168 kg N ha^{-1} as urea (46% N). Fertilizer N was applied in a split application, where 118 kg N ha^{-1} were applied pre-flood onto dry soil at the four- to five-leaf growth stage followed by an application of 50 kg N ha^{-1} at midseason into the floodwater after panicle differentiation. Methane emissions were nearly 20% greater than the USEPA 2011 emissions factor at the time of 160 kg CH_4-C ha^{-1} $season^{-1}$ (Rogers et al., 2013). Rogers et al. (2013) showed that N fertilization did not have a significant impact on weekly CH_4 fluxes over the growing season or on season-long emissions. Rogers et al. (2013) also observed a consistent and predictable pulse of CH_4 after release of the floodwater, which has been observed in other studies.

Brye et al. (2013) examined soil texture effects on CH_4 emissions and reported that N-fertilized rice grown on a clay soil at the Northeast Research and Extension Center (NEREC) at Keiser, AR exhibited increased CH_4 emissions to a maximum peak flux during heading and decreased thereafter until after the flood was released. The N-fertilized rice treatment emitted 75% less total CH_4 and had 70% lower CH_4 fluxes than that from the same field treatment combination on a silt-loam soil at RREC and CH_4 emissions were greater when rice plants were present than in the absence of plants (Brye et al., 2013). These findings support previous research that plant-mediated CH_4 release is the predominate mechanism of CH_4 release from the soil profile. In addition, soil texture has a considerable impact on the release of CH_4 from a drill-seeded, delayed-flood rice production system when comparing silt loam to a clay soil (Holzapfel-Pschorn and Seiler, 1986; Sass et al., 1990; Brye et al., 2013; Rogers et al., 2013). Brye et al. (2013) demonstrated that the CH_4 emissions reported for clay soils from N-fertilized rice were less than 23% (35.6 kg CH_4-C ha^{-1}) of silt-loam-soil emissions, which were lower than those used by governing bodies to make policies regarding GHG emissions. Discrepancies in CH_4 emissions between observed and estimated values used by policy makers, such as EPA's reported emissions factors, could contribute to negative consequences for rice producers and the rice-related economy in Arkansas and potentially other rice-producing regions (Brye et al., 2013).

Rice Cultivar Effects on CH_4 Emissions

Cultivar selection is vitally important when determining CH_4 emissions. The role of rice plants in regulating the CH_4 emissions to the atmosphere is influenced by the enormous genotypic and phenotypic variation (Aulakh et al., 2002). Early studies conducted in Louisiana (Crowley silt loam, Typic Albaqualf) and Texas (Verland silty clay loam, fine montmorillonitic, thermic Vertic Ochraqualf) reported CH_4 emissions ranged from 135 to 360 kg CH_4 ha^{-1} $season^{-1}$ (Lindau et al., 1993; Sass and Fisher Jr., 1997); however, the pure-line varieties used in these two studies are not widely used in current commercial production any more, thus their results are out of date. Nonetheless, these two early studies examined how different cultivars mediate CH_4 transport to the atmosphere (Lindau et al., 1993; Sass and Fisher Jr., 1997).

Cultivar effects on CH_4 were examined from 22 rice cultivars (18 pure-line varieties and 4 hybrids) from southeast Asia in a Maahas clay soil (Andaqueptic Haplaquoll) to assess the influence of cultivar on CH_4 emissions (Aulakh et al., 2002). Methane emissions ranged from 62 to 445 kg CH_4 ha^{-1}-season, indicating the wide variability and the control the rice plant has on transportation of CH_4 to the atmosphere (Aulakh et al., 2002). Differences in CH_4 release from multiple rice cultivars are a complicating factor in determining reasonable standards for an emissions factor to better predict rice agriculture's effect on CH_4 emissions.

Cultivar differences have been large between pure-line and hybrid cultivars with regard to CH_4 emissions. Averaged across previous crop, area-scaled seasonal emissions from hybrid cultivars, such as CLXL745 emitting 111 kg CH_4-C ha^{-1} per growing season (Rogers et al., 2014). Pure-line cultivars such as 'Cheniere', and 'Taggart' emitted 169 and 186 kg CH_4-C ha^{-1}, and 'Wells' another pure-line from the same production system averaged 195 kg CH_4-C ha^{-1} per growing season (Rogers et al., 2013 ; Rogers et al., 2014). Further research on a DeWitt silt-loam soil at RREC showed CH_4 emissions from a hybrid cultivar were nearly 38% lower than the current 2014 USEPA CH_4 emissions factor (178 kg CH_4-C ha^{-1} $season^{-1}$), and pure-line cultivars accounted for 55 to 70% more CH_4-C emissions than hybrid cultivars (Rogers et al., 2014; Brye et al., 2016). The difference in CH_4 emissions between hybrid and pure-line cultivars was also reported by Smartt et al. (2016), who measured CH_4 emissions from a hybrid cultivar (CLXL745) were 10.2 kg CH_4-C ha^{-1} less than that from two pure-line cultivars (Cheniere or Taggart) with mean emissions of 14.8 kg CH_4-C ha^{-1} (Smartt et al., 2016). This reduction in CH_4 emissions from hybrid rice compared pure-line cultivars is likely related to differences in CH_4 oxidation in the root zone due to the increased root mass in a hybrid providing greater oxygen to the soil microbial community thus delaying the reduction of organic matter to CH_4 (Rogers et al., 2014).

Hybrid cultivars displaying lower CH_4 emissions compared to pure-line varieties was demonstrated in Nalley et al. (2014) using results from Arkansas Rice Performance Trials (ARPT) during a review that was conducted for seven consecutive years between 2004 and 2010. Nalley et al. (2014) used yield data, emergence date, and the date of 50% heading from four silt-loam-soil locations throughout eastern Arkansas (RREC, near Stuttgart; Coring; Newport; and the Pine Tree Research Station, near Colt). Four cultivar categories were examined: conventional hybrids, Clearfield hybrids (RiceTec, Inc., Houston, TX), conventional pure-lines, and Clearfield pure-lines (Rogers et al., 2013; Nalley et al., 2014). Using a three-way, fixed-effects model, on average, for every 1 kg of hybrid rice grain yield, 0.001 Mg of CO_2 equivalents (CO_2e) were produced, whereas pure-line cultivars were estimated to release 0.00124 Mg CO_2e (kg grain yield)$^{-1}$ (Nalley et al., 2014). Hybrid cultivars were estimated to release more total GHGs per hectare (6037 CO_2e ha^{-1}) than either pure-line cultivar (5834 CO_2e ha^{-1}). However, hybrid cultivars have approximately 25% greater yield (10744 vs 8577 kg ha^{-1}) than pure-line cultivars, indicating that hybrid cultivars clearly have greater GHG efficiency than pure-line cultivars (Nalley et al., 2014). The use of high-yielding cultivars with a low CH_4 transport capacity could be economically and environmentally promising avenues for reducing CH_4 emissions from rice paddies (Aulakh et al., 2002). These reductions in CH_4 emissions using hybrid cultivars could be sold in the European Climate Exchange, which could be an economic boon for Arkansas by providing extra income for producers, particularly from increased yields with hybrids compared to pure-line cultivars (Nalley et al., 2014).

Soil Texture Effects on CH_4 Emissions

Soil texture plays a vital role in controlling CH_4 fluxes and total emissions. Methane fluxes were reported lower in fine-textured, clay soils than in more coarse-textured soils, such as silt loams (Sass et al., 1994; Smartt et al., 2016). Early studies on a Sacramento clay (Vertic Endoaquolls) in California, on bare soil and with low vegetation, reported CH_4 emissions of 8.85 and 10.5 kg CH_4-C ha^{-1} $season^{-1}$, respectively (Cicerone et al., 1992). Methane emissions from the treatments did not differ significantly, although emissions from both treatments were numerically less than the 21.6 kg CH_4-C ha^{-1} $season^{-1}$ released from a high-vegetation treatment under the same production system (Cicerone et al., 1992). Experimental data from a Capay silty clay (Typic Haploxererts) in California, where rice was seeded onto a flooded soil, showed maximum CH_4 fluxes of 0.9, 1.3, and 4.3 mg CH_4-C m^{-2} h^{-1} from unfertilized bare soil, unfertilized rice, and fertilized rice, respectively (Cicerone et al., 1992). Similarly, Rogers et al. (2013) measured maximum CH_4 fluxes of 11.6, 13.9, and 22.6 mg CH_4-C m^{-2} h^{-1} for unfertilized bare soil, unfertilized rice, and fertilized rice, on a DeWitt silt loam (fine, smectitic, thermic

Typic Albaqualf) under the drill-seeded, delayed-flood production system in Arkansas. However, CH_4 fluxes measured at Keiser, Arkansas (35°40′ N 90° 05′ W) from a Sharkey clay (very fine, smectitic, thermic Chromic Epiaquerts) were 35.6, 8.9, and 1.7 kg CH_4-C ha^{-1} $season^{-1}$ from N-fertilized rice, non-N-fertilized rice, and bare soil, respectively (Smartt et al., 2016). These differences in emissions from a Sharkey clay and DeWitt silt-loam soil in eastern Arkansas can be attributed to an inverse correlation between soil clay content and CH_4 emissions, which has been observed before on other clay and silt-loam soils (Mitra et al., 2002; Sass et al., 1994).

In continued efforts to better quantify CH_4 emissions in the drill-seeded, delayed-flood rice production system on a Sharkey clay soil (very-fine, smectitic, thermic Chromic Epiaquerts) in northeast Arkansas, sampling-chamber-size effects on growing-season CH_4 emissions were examined (Smartt et al., 2015). Chamber size (i.e., 15.2- or 30.4-cm inside diameter) did not result in differences in cumulative season-long CH_4 emissions (Smartt et al., 2015). Additionally, results from direct field measurements showed that CH_4 emissions from rice produced on a clay soil in the drill-seeded, delayed-flood rice production system in Arkansas may be greatly overestimated by the single USEPA emissions factor (178 kg CH_4-C ha^{-1} $season^{-1}$).

During a season-long emissions study on a Sharkey clay soil, it was reported that CH_4 emissions were 18 to 48% of the emissions reported from similar studies conducted on silt-loam soils in eastern Arkansas and almost 20% of the previous 2011 USEPA emissions factor of 160 CH_4-C ha^{-1} $season^{-1}$ (Smartt et al., 2016). The overestimation of CH_4 emissions from clay soils by the USEPA is additionally supported by results that showed silt-loam soils (Albaqualf) emitted 211% more CH_4-C than clay soils (Epiaquert) (Brye et al., 2016; Smartt et al., 2016); however, additional data are needed to better evaluate the numerous factors known to affect CH_4 emissions.

The inverse correlation between soil clay content and CH_4 emissions may also be related to increased tortuosity and decreased pore size in fine-texture clay compared to coarser-textured silt-loam soils, thus inhibiting gas movement in the soil column as the clay content increases. Therefore, decreased amounts of CH_4 are released to the atmosphere in clay soils because the CH_4 cannot reach the surface or come in to contact with root hairs of the rice plant to be transported to the atmosphere. This correlation indicates that rice production may be more environmentally friendly in clay than in silt-loam soils and that shifting the production areas of rice to areas of greater clay content may mitigate the atmospheric and environmental impact of CH_4 emissions from rice production (Brye et al., 2013).

Crop Rotation/Previous Crop Effects on CH_4 Emissions

The influence of previous crop was also examined with regards to CH_4 emissions from rice grown following soybean or rice (Rogers et al., 2014). There is substantially less soybean residue compared to rice residue, and soybean residue appears to be less recalcitrant and more readily decomposable than rice residue before flooding, thereby providing less substrate for soil microbial respiration (Rogers et al., 2014). When rice was grown following soybean in a crop rotation, CH_4 emissions were 21% lower than the previous 2011 USEPA emissions factor of 160 CH_4-C ha^{-1} $season^{-1}$ emissions factor estimate (Rogers et al., 2014; Brye et al., 2016). In addition, soybean-rice rotations produced 58% less CH_4-C emissions than rice-rice rotations (Rogers et al., 2014). In California, at the University of California, Davis on a Esquon-Neerdobe complex (Fine, smectitic, thermic Xeric Epiaquerts and Duraquerts), a four-yr fallow field study produced almost 92% less CH_4 emissions compared to previous 2011 USEPA emissions factor of

160 CH_4-C ha^{-1} $season^{-1}$ due to the reduced carbon substrate in the field that was limited to just weeds (Rogers et al., 2014; Simmonds et al., 2015; Brye et al., 2016).

Rogers et al. (2014) investigated both previous crop and cultivar effects on CH_4 emissions from a drill-seeded, delayed-flood rice production system on a DeWitt silt loam (fine, smectitic, thermic Typic Albaqualf) in eastern Arkansas. Methane emissions were shown to be significantly impacted by previous crop and cultivar. Averaged across cultivar, CH_4 emissions were greater when rice followed rice (184 kg CH_4-C ha^{-1} per growing season) than when rice followed soybean (127 kg CH_4-C ha^{-1} per growing season; (Rogers et al., 2014). Differences between pure-line cultivars, Cheniere and Taggart, and the hybrid cultivar CLXL745 were also significant. The hybrid CLXL745 emitted 56 to 111 kg CH_4-C ha^{-1} per growing season, while Cheniere and Taggart emitted approximately 34 and 40% more CH_4, respectively (Rogers et al., 2014). Other pure-line cultivars, Francis and Jupiter, emitted 77 to 72 kg CH_4-C ha^{-1}, respectively, when following soybean compared to following rice (Rogers et al., 2014; Simmonds et al., 2015). Compared to emissions from the pure-line cultivar Wells from an identical production system (195 kg CH_4-C ha^{-1} per growing season), CH_4 emissions from CLXL745, Cheniere, and Taggart were 43, 13, and 5% lower, respectively, overall (Rogers et al., 2013; Rogers et al., 2014).

Water Management Effects on CH_4 Emissions

Rice in the US is mostly grown under continuous, shallow-flood-water conditions (i.e., full-season-flood water management), which has the greatest documented CH_4 emissions (Sass et al., 1992). Upon flooding, there is a rapid decrease in the soil redox potential as the soil microbes consume the O_2 and C substrates, including root exudates, lysates, litter, and dead organic matter from incorporated vegetation (Sass et al., 1991; Cicerone et al., 1992). These conditions are prerequisites for CH_4 production by microbes in the soil.

The irrigation strategy that has been shown to dramatically decrease CH_4 emissions is a mid-season drain followed by re-flooding (Sass et al., 1990; Qin et al., 2010; IPCC, 2014). On a Dewitt silt loam in Arkansas, CH_4 emissions from a full-season-flood ranged from 76.4 to 195 kg CH_4–C ha^{-1} (Brye et al., 2013; Rogers et al., 2013; Rogers et al., 2014; Humphreys et al., 2016), while CH_4 emissions from a mid-season-drain strategy ranged from 28.9 to 56.6 CH_4–C ha^{-1} have been reported (Brye et al., 2013; Rogers et al., 2013; Humphreys et al., 2016). Draining floodwater has shown to decrease CH_4 emissions because soil aeration inhibits CH_4 production by methanogens, while at the same time depleting existing soil CH_4 build up through aerobic oxidation by methanotrophs (Sass et al., 1992; Humphreys et al., 2016). In most other production systems, mid-season drainage does not occur, except by accident or when controlling for straighthead, which is a disorder that causes sterility of the spikelets and reduces yield (IPCC, 2014).

As a consequence of the large amount of water used to produce a typical rice crop, water quantity and availability are quickly becoming major issues in many developed and developing countries, particularly in the Lower Mississippi River Delta region of eastern Arkansas. Therefore, developing irrigation strategies that reduce water use without decreasing yield or milling quality will also help to reduce CH_4 emission from flooded rice (Lindau et al., 1993).

Soil Organic Matter Concentration Effects on CH_4 Emissions

Though CH_4 production requires a C-containing substrate, the relationship between CH_4 emissions and SOM or soil organic C (SOC) concentration has not been well-investigated. It is expected that as SOM concentration increases, CH_4 production will also increase. Since soil microbes in an anaerobic setting require C as an electron acceptor to carry out metabolic processes, increasing the supply of SOM should increase microbial activity and therefore CH_4 production. However, in laboratory studies in Louisiana, using 16 soils ranging in texture from silt to clay, CH_4 emissions and SOM concentrations in the range of 0.7 to 2.4% (14 to 23.8 Mg ha^{-1}) were examined and it was determined that no correlation existed between CH_4 emissions and soil properties such as nitrogen, pH, or cation exchange capacity, but there was a significant increase in CH_4 soil entrapment in higher clay content soils < 0.001 to 0.005-mm suggesting soil texture plays a vital role in CH_4 emissions (Wang et al., 1993). Field trials are needed in Arkansas to assessCH_4 emissions across a range of SOM/SOC in silt-loam soils. This information can give researchers a better understanding on how to mitigate CH_4 release from silt-loam soils with large SOM concentrations.

Justification

Characterizing and understanding the magnitude and variability associated with CH_4 emissions are critically important to mitigating anthropogenic climate change. To reduce CH_4 emissions from flooded rice, field management practices must be first evaluated, then developed to reduce CH_4 emissions without decreasing yields (Lindau et al., 1993). Consequently, research is still needed to quantify the magnitude of growing-season CH_4 fluxes and emissions as a result of common and alternative management practices, such as cultivar selection, water management practices, and cultural practices, such as tillage, which has received little research attention thus far.

Due to the volume of water typically used to produce a rice crop, water quantity and availability are quickly becoming major issues in many developed and developing countries, as well as in the Lower Mississippi River Delta region of eastern Arkansas where aquifer stability and longevity is of utmost importance. Therefore, developing irrigation strategies that reduce water use without decreasing yield will also help to reduce CH_4 emission from flooded rice (Lindau et al., 1993). Since Arkansas is the leading rice-producing state in the US, rice grown in the direct-seeded, delayed-flood production system common to the Lower Mississippi River Delta region of eastern Arkansas offers the unique opportunity to further knowledge regarding GHG emissions, particularly CH_4 from rice production.

Research Goal and Objectives

The goal of this dissertation research is to further assess and quantify CH_4 released from silt-loam soils under a direct-seeded, delayed-flood rice production system in the Lower Mississippi River Delta region of eastern Arkansas. This goal will be achieved through field studies with the following three objectives: 1) evaluate the effects of water management strategy (i.e., delayed-permanent flood and mid-season drain) and rice cultivar (i.e., pure-line and hybrid) on CH_4 fluxes and growing-season emissions (conducted in 2015), 2) evaluate the effects of SOM concentration under full-season flood on CH_4 fluxes and growing-season emissions (conducted in 2016), and 3) evaluate the effects of tillage system (i.e., conventional and no-tillage) on CH_4 fluxes and emissions (conducted in 2017) from a direct-seeded, delayed-flood rice production system on a silt-loam soil. These field studies furthered our understanding of CH_4 production and release from rice agroecosystems and explore ways to reduce the environmental and C footprint of rice production.

Testable Hypotheses

For Objective 1, it was hypothesized that both rice cultivar and water management scheme will affect CH_4 emissions over the entire growing season with a reduction in total CH_4 emissions after 50% heading. Specifically, it was hypothesized that, based on previous field research results, the hybrid-cultivar/mid-season-drain will have the lowest and the pure-line-cultivar/full-season-flood treatment combination will have the largest growing-season-long CH_4 emissions.

For Objective 2, it was hypothesized that CH_4 fluxes and emissions from a transplanted, pure-line cultivar grown a silt-loam soil under full-season-flood management would increase with increasing SOM content. Specifically, it was hypothesized that CH_4 emissions would be directly related with SOM content due to an increase in labile organic C that could be readily reduced to CH_4, but that the relationship would be non-linear due to the passive transport the rice plant exhibits achieving a maximum, after which the emissions plateau despite increasing substrate availability in the soil.

For Objective 3, it is hypothesized that CH_4 fluxes and emissions from a pure-line cultivar grown on a silt-loam soil under full-season-flood management will be greater from long-term no-tillage than conventionally tilled management due to greater SOM in the long-term no-tillage system. Also, that N-(n-butyl) thiophosphoric triamide (NBPT)-coated urea would result in greater CH_4 fluxes and emissions due to the increased labile form of N compared to the non-coated urea.

Literature cited

Aulakh, M.S., R. Wassman, and H. Rennenberg. 2002. Methane transport capacity of twenty-two rice cultivars. Agric., Ecosyst. Environ. 91:5971.

Brye, K.R., L.L. Nalley, J.B. Tack, B.L. Dixon, A.P. Barkley, C.W. Rogers, A.D. Smartt, R.J. Norman, and K.S.V. Jagadish. 2016. Factors affecting methane emissions from rice production in the Lower Mississippi river valley, USA. Geod. Reg. 7:223-229.

Brye, K.R., C.W. Rogers, A.D. Smartt, and R.J. Norman. 2013. Soil texture effects on methane emissions from direct-seeded, delayed-flood rice production in Arkansas. Soil Sci. 178:519-529.

Chang, T., K.F. Kiple, and K.C. Ornelas. 2012. The Cambridge World History of Food, chapter II.A.7 - Rice, Cambridge University Press pp. 132-149.

Cicerone, R.J., C.C. Delwiche, S.C. Tyler, and P.R. Zimmerman. 1992. Methane emissions from California rice paddies with varied treatments. Global Biogeochem. Cycles 6:233-248.

Cicerone, R.J., and J.D. Shetter. 1981. Sources of atmospheric methane: measurements in rice paddies and a discussion. J. Geophys. Res. 86:7203-7209.

Cicerone, R.J., and J.D. Shetter. 1983. Seasonal variation of methane flux from a California rice paddy. J. Geophys. Res. 88:11022-11024.

Cicerone, R.J., R.S. Stolarski, and S. Walters. 1974. Stratospheric ozone destruction by man-made chlorofluoromethanes. Science 185:1165-1167.

Conrad, R. 2009. The global methane cycle: recent advances in understanding the microbial processes involved. Environ. Microbiol. Rep. 1:285-292.

de Avila, L.A., L.F.D. Martini, R.F. Mezzomo, J.P. Refatti, R. Campos, D.M. Cezimbra, S.L.O. Machado, J.H. Massey, R. Carlesso, and E. Marchesan. 2015. Rice water use efficiency and yield under continuous and intermittent irrigation. Agron. J. 107:442-448.

Deppenmeier, U., V. Müller, and G. Gottschalk. 1996. Pathways of energy conservation in methanogenic archaea. Arch. Microbiol. 165:149-163.

Ferry, J.G. 1992. Methane from acetate. J. Bacteriol. 174:5489-5495.

Food and Agriculture Organization (FAO) of the United Nations 2012. Food and Nutrition Division. Food and Agriculture Organization of the United Nations, http://faostat.fao.org/beta/en/#data/QC (accessed 11 November 2016).

Forster, P., V. Ramaswamy, P. Artaxo, T. Berntsen, R. Betts, D.W. Fahey, J. Haywood, J. Lean, D.C. Lowe, G. Myhre, J. Nganga, R. Prinn, G. Raga, M. Schulz, and R. Van Dorland. 2007. Changes in atmospheric constituents and in radiative forcing. Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change [Solomon, S., D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M.Tignor and H.L. Miller (editors.)].Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Hardke, J.T. 2014. Arkansas Rice Production Handbook. In: J. T. Hardke, editor Introduction. University of Arkansas Division of Agriculture, Cooperative Extension Service, Little Rock, AR.

Hardke, J.T. 2016. Trends in Arkansas rice production. In: R. J. Norman, and K. A. K. Moldenhauer, editors, B.R. Wells Arkansas Rice Research Studies, 2015. Univ. Ark. Agric. Exp. Sta. Res. Ser. 634., Univ. Ark. Agric. Exp. Sta., Fayetteville, AR. pp. 13-23.

Hardke, J.T., and C.E. Wilson. 2012. In: J. T. Hardke, editor Arkansas Rice Production Handbook. University of Arkansas Division of Agriculture, Cooperative Extension Service, Little Rock, AR.

Henry, C.G., S.L. Hirsh, M M. Anders, E.D. Vories, M.L. Reba, K.B. Watkins, and J.T. Hardke. 2016. Annual irrigation water use for Arkansas rice production. J. Irrig. Drain. Engin. 05016006:1-5.

Holzapfel-Pschorn, A., and W. Seiler. 1986. Methane emission during a cultivation period from an Italian rice paddy. J. Geophys. Res. 91:11803-11814.

Humphreys, J.J., K.R. Brye, A.D. Smartt, J.T. Hardke, D.L. Frizzell, E. Canstaneda-Gonzalez, G.J. Lee, and R.J. Norman. 2016. Methane emissions from direct-seeded, delayed-flood rice production as influenced by cultivar and water management. In: R.J. Norman, and K.A.K. Moldenhauer, editors, B.R. Wells Arkansas Rice Research Studies, 2015. Univ. Ark. Agric. Exp. Sta. Res. Ser. 634., Univ. Ark. Agric. Exp. Sta., Fayetteville, AR. pp. 275-281.

Intergovernmental Panel on Climate Change (IPCC). 2001. Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change [Houghton, J.T.,Y. Ding, D.J. Griggs, M. Noguer, P.J. van der Linden, X. Dai, K. Maskell, and C.A. Johnson (editors.)] Climate Change 2001: The Scientific Basis. pp. 881.

Intergovernmental Panel on Climate Change (IPCC). 2007. Intergovernmental Panel on Climate Change (IPCC), Climate Change 2007: Synthesis Report. Contribution of Working Groups I, II and III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change [Core Writing Team, Pachauri, R.K and Reisinger, A. (editors.)]. IPCC, Geneva, Switzerland. pp. 104.

Intergovernmental Panel on Climate Change (IPCC). 2014. Climate Change 2014: Synthesis Report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Core Writing Team, R.K. Pachauri and L.A. Meyer (editors.)]. IPCC, Geneva, Switzerland.

Intergovernmental Panel on Climate Change (IPCC). 2014. Intergovernmental Panel on Climate Change (IPCC), United States Greenhouse Gas Inventory Chapter 6. IPCC, Geneva, Switzerland. pp. 15-21.

Lindau, C.W., P.K. Bollich, R.D. DeLaune, A.R. Mosier, and K.E Bronson. 1993. Methane mitigation in flooded Louisiana rice fields. Biol. Fert. Soils 15:174-178.

Mitra, S., R. Wassman, M. Jain, and H. Patak. 2002b. Properties of rice soils affecting methane production protentials: 2. Differences in topsoil and subsoil. Nutr. Cycl. Agro. 64:183-191.

Mitra, S., R. Wassmann, M.C. Jain, and H. Pathak. 2002a. Properties of rice soils affecting methane production potentials: 1. Temporal patterns and diagnostic procedures. Nutr. Cycl. Agro. 64:169-182.

Nalley, L.L., B.L. Dixon, K.R. Brye, C.W. Rogers, H. Myrteza, and R.J. Norman. 2014. Estimating cultivar effects on water usage and greenhouse gas emissions in rice production. Agron. J. 106:1981.

Norman, R.J., N.A. Slaton, and T.L. Roberts. 2013. Soil Fertility. In: J. T. Harke, editor Arkansas Rice Production Handbook Misc. Publ. 192. University of Arkansas Division of Agriculture Cooperative Extension Service, Little Rock, AR, USA. pp. 69-102.

Papen, H., and H. Rennenberg. 1990. Microbial processes involved in emissions of radiatively important trace gases. Transactions 14th International Congress Soil Science, Kyoto, Japan, August 1990. Volume II. pp. 232-237.

Patrick, W.H., R.P. Gambrell, and S.P. Faulkner. 1996. Redox measurements of soils. In: D. L. Sparks, editor Methods of Soil Analysis. part 3 - Chemical Methods. 3rd ed. SSSA, Madison, WI, USA. pp. 1255-1273.

Qin, Y., S. Liu, Y. Guo, Q. Liu, and J. Zou. 2010. Methane and nitrous oxide emissions from organic and conventional rice cropping systems in Southeast China. Biol. and Fert. of Soils 46:825-834.

Rogers, C.W., K.R. Brye, R.J. Norman, E.E. Gbur, J.D. Mattice, T.B. Parkin, and T.L. Roberts. 2013. Methane emissions from drill-seeded, delayed-flood rice production on a silt-loam soil in Arkansas. J. Environ. Qual. 42:1059-1069.

Rogers, C.W., K.R. Brye, A.D. Smartt, R.J. Norman, E.E. Gbur, and M.A. Evans-White. 2014. Cultivar and previous crop effects on methane emissions from drill-seeded, delayed-flood rice production on a silt-loam soil. Soil Sci. 179:28-36.

Sass, R., and F. Fisher Jr. 1997. Methane emissions from rice paddies: a process study summary. Nutr. Cycl. Agro. 49:119-127.

Sass, R.L., F.M. Fisher, and P.A. Harcombe. 1990. Methane production and emission in a Texas rice field. Global Biogeochem. Cycles 4:47-68.

Sass, R.L., F.M. Fisher, S.T. Lewis, F.M. Jund, and F.T. Turner. 1994. Methane from rice fields: Effect of soil properties. Global Biogeochem. Cycles 8:135-140.

Sass, R.L., F.M. Fisher, F.T. Turner, and F.M. Jund. 1991. Methane emission from rice fields as influenced by solar radiation, temperature, and straw incorporation. Global Biogeochem. Cycles 5:335-350.

Sass, R.L., F.M. Fisher, F.T. Turner, and F.M. Jund. 1992. Methane emission from rice fields: The effect of flood water management. Global Biogeochem. Cycles 6:249-262.

Scott, B., J. Norsworthy, T. Barber, and J. T. Hardke. 2014. Arkansas Rice Production Handbook. In: J. T. Hardke, editor Rice weed control. University of Arkansas Division of Agriculture, Cooperative Extension Service, Little Rock, AR. pp. 53-62.

Simmonds, M.B., M. Anders, M.A. Adviento-Borbe, C. van Kessel, A. McClung, and B.A. Linquist. 2015. Seasonal methane and nitrous oxide emissions of several rice cultivars in direct-seeded systems. J. Environ. Qual. 44:103-114.

Smartt, A.D., K.R. Brye, C.W. Rogers, R.J. Norman, E.E. Gbur, J.T. Hardke, and T.L. Roberts. 2015. Chamber size effects on methane emissions from rice production. Open J. Soil Sci. 5:227-235.

Smartt, A.D., K.R. Brye, C.W. Rogers, R.J. Norman, E.E. Gbur, J.T. Hardke, and T.L. Roberts. 2016. Characterization of methane emissions from rice production on a clay soil in Arkansas. Soil Sci. 181:57-67.

Smartt, A.D., K.R. Brye, C.W. Rogers, R.J. Norman, E.E. Gbur, J.T. Hardke, and T.L. Roberts. 2016. Previous crop and cultivar effects on methane emissions from drill-seeded, delayed-flood rice grown on a clay soil. Appl. Environ. Soil Sci. 2016:1-13.

Smith, P., D. Martino, Z. Cai, D. Gwary, H. Janzen, P. Kumar, B. McCarl, S. Ogle, F. O'Mara, C. Rice, B. Scholes, and O. Sirotenko. 2007. Agriculture. In Climate Change 2007: Mitigation. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change [B. Metz, O.R. Davidson, P.R. Bosch, R. Dave, L.A. Meyer (editors)], Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Solomon, S., K.H. Rosenlof, R.W. Portmann, J.S. Daniel, S.M. Davis, T.J. Sanford, and G. Plattner. 2010. Contributions of stratospheric water vapor to decadal changes in the rate of global warming. Science 327:1219-1223.

United States Department of Agriculture(USDA-NASS), National Agricultural Statistics Service. 2015. Available at: https://www.nass.usda.gov/Quick_Stats/Ag_Overview/stateOverview.php?state=ARKANSAS (accessed 19 Jan. 2018).

United States Environmental Protection Agency (USEPA). 2014. United States Environmental Protection Agency (USEPA), Inventory of U.S. greenhouse gas emissions and sinks: 1990-2013. Available at: www.epa.gov/climatechange/emissions/usinventoryreport.html (accessed 11 Jan. 2016).

van Breemen, N., and T.C.J. Feijtel. 1990. Soil processes and properties involved in the production of greenhouse gases, with special relevance to soil taxonomic systems. In: A. F. Bouwman, editor Soils and the greenhouse effect. John Wiley and Sons Ltd, Chichester, West Sussex PO19 1UD; UK. pp. 195-223.

Wang, Z.P., C.W. Lindau, R.D. DeLaune, and W.H. Patrick. 1993. Methane emission and entrapment in flooded rice soils as affected by soil properties. Biology and Fert. of Soils 16:163-168.

Wassman, R., R.S. Lantin, H.U. Neue, L.V. Buendia, T.M. Corton, and Y. Lu. 2000. Characterization of methane emissions from rice fields in Asia. III. Mitigation options and future research needs. Nutr. Cycl. Agro. 58:23-36.

Yagi, K., H. Tsuruta, and K. Minami. 1997. Possible options for mitigating methane emission from rice cultivation. Nutr. Cycl. Agro. 49:213-220.

CHAPTER THREE

Water management and cultivar effects on methane emissions from direct-seeded, delayed-flood rice production in Arkansas

Abstract

Methane (CH_4) emissions from rice (*Oryza sativa* L.) *p*roduction are a source of concern in the environmental and agricultural communities. New and/or revised agronomic methodologies will be needed to identify production practice combinations that reduced CH_4 emissions without decreasing yields. The objective of this study was to evaluate the effects of water management (i.e., delayed-permanent flood and mid-season drain) and cultivar (i.e., pureline cultivar LaKast and the RiceTec hybrid XP753) on CH_4 fluxes and season-long emissions from rice grown in the direct-seeded, delayed-flood production system on a silt-loam soil in east-central Arkansas. Vented, non-flow-through, non-steady-state chambers were used to collect gas samples over a 60-min sampling interval for weekly measurements of CH_4 fluxes between flooding and harvest. Methane fluxes from all treatments started low then increased ($P < 0.01$) between 19 and 54 days after flooding (DAF), where the largest peak flux occurred from the full-season-flood/hybrid combination (229.3 mg CH_4-C m^{-2} d^{-1}) just after 50% of the panicles had emerged by 47 DAF. Methane fluxes from all four treatment combinations peaked between 47 and 54 DAF. After 54 DAF, CH_4 fluxes decreased ($P < 0.01$) in all treatment combinations leading up to flood release, with several treatment combinations exhibiting a temporary, at least numerically increased CH_4 flux just after flood release at 72 DAF. The full-season-flood (77.7 CH_4-C $ha^{-1}season^{-1}$) produced the greatest ($P < 0.01$), while the mid-season-drain (42.8 kg CH_4-C $ha^{-1}season^{-1}$) produced the lowest season-long CH_4 emissions. The mid-season-drain/hybrid combination exhibited the lowest ($P < 0.05$) emissions intensity (2.5 kg CH_4-C Mg $grain^{-1}$), while emissions intensity did not differ and averaged 6.4 kg CH_4-C Mg $grain^{-1}$ among the other three treatment combinations. Properly matching water management scheme with cultivar

selection can provide a means to reduce CH_4 emissions from rice production in the direct-seeded, delayed-flood production system on silt-loam soils.

Introduction

Total United States (US) greenhouse gas (GHG) emissions increased by 8.4% from 1990 to 2011, with a 1.6% decrease from 2010 to 2011, followed by a 2% increase in 2012 to a total 2015 US GHG emissions of 6568 million metric tons (MMT) of carbon dioxide (CO_2) equivalents (USEPA, 2017). The overall CO_2 equivalents from all sources from 1990 to 2011 increased by 504 Tg, while methane (CH_4) emissions specifically decreased by 57.2 Tg CO_2 equivalents over the same time period (IPCC, 2014). Despite the decline in CO_2 equivalents, CH_4 emissions from certain activities, namely agriculture, remain a concern.

As of 2005, agriculture was estimated to contribute about 47% of total anthropogenic CH_4 emissions, while the remaining non-agricultural sources of CH_4 production are from natural gas systems, landfills, and coal mining, which make up over 50% of the total CH_4 emissions in the US (Smith et al., 2007). The main agricultural sources of CH_4 emissions in the US are enteric fermentation and manure management, with over 95% of total agriculturally related CH_4 emissions as of 2012, with rice (*Oryza sativa* L.) cultivation and field burning making up 3.7% of the total agricultural CH_4 releases (IPCC, 2014). As of 2013, atmospheric CH_4 inputs from enteric fermentation, manure management, rice production, and biomass burning contributed approximately 8.1% of total US anthropogenic GHG emissions to the environment (IPCC, 2014). As of 2011, CH_4 emissions from rice cultivation represented 1.1% of the total US CH_4 emissions to the atmosphere (IPCC, 2014).

Between 1990 and 2014, annual CH_4 emissions from rice production fluctuated between 575 and 476 kT (kilotons), whereas CH_4 emissions in 2015 alone represented a 30% decrease compared to those in 1990 (USEPA, 2017). In 2015, estimated CH_4 emissions from rice cultivation were 11.2 MMT of CO_2 equivalents in the US (USEPA, 2017). However, CH_4

emissions from agricultural sources are closely tied to the regional geographic distribution of where rice production occurs, whereas Arkansas, California, Louisiana, and Missouri were the top four rice-producing states in the US in 2015 (NASS, 2016). Based on rice yields, Arkansas produced an estimated 3.8 MMT CO_2 equivalents in 2015 from rice cultivation alone (USEPA, 2017).

Rice production systems differ from other row crops due to the practice of flood irrigation. Moreover, rice grown in the direct-seeded, delayed-flood rice production system common in Arkansas differs substantially from traditional rice systems, where rice is hand-transplanted directly to a flooded field (Chang et al., 2012). These production differences create unique difficulties as well as opportunities for improving management of soil and water resources needed to sustain rice production and protect the environment (Henry, 2016).

As a potent GHG, CH_4 is produced under anoxic conditions commonly associated with lowland rice production when carbon (C) from organic matter is consumed and converted to CH_4 by methanogens (Ferry, 1992). Several biochemical processes exist where C is reduced to CH_4, thus releasing energy for metabolic processes (Ferry, 1992). Since soil organic matter (SOM) is generally concentrated near the soil surface in the A horizon, > 99% of the total soil-produced CH_4 is emitted from the topsoil (Mitra et al., 2002b). The main mechanism of CH_4 release to the atmosphere from below a column of water has been via passive transport through the aerenchyma tissue of the rice plants themselves (Cicerone and Shetter, 1981; Yu et al., 1997; Dannenburg and Conrad, 1999; Groot et al., 2005), while ebullition and diffusion are secondary and more minor emissions pathways (Cicerone and Shetter, 1981; Yu et al., 1997).

Along with soil texture (Brye et al., 2013), management practices associated with rice production are one of the most important factors affecting CH_4 emissions. Cultivar selection, or the choice to plant either a conventional pure-line or a hybrid cultivar, plays a major role in not only yield, but also potential CH_4 emissions (Simmonds et al., 2015; Smartt et al., 2016) . Hybrid cultivars have consistently shown decreased CH_4 emissions compared to pure-line cultivars grown on silt-loam (Adviento-Borbe et al., 2013; Rogers et al., 2014; Simmonds et al., 2015) and clayey soils (Adviento-Borbe et al., 2013; Brye et al., 2013; Smartt et al., 2016). Hybrid rice cultivars typically have more vigorous root growth, as well as increased transport of atmospheric oxygen to the rhizosphere (Dannenburg and Conrad, 1999; Aulakh et al., 2001; Conrad et al., 2006; Conrad et al., 2008) to inhibit reduction of C in SOM and other C substrates (i.e., organic soil amendments) to CH_4, which only occurs after the soil's oxidation-reduction (redox) potential has decreased to approximately -200 mV from prolonged saturated soil conditions. Consequently, when hybrid rice is grown, the soil in the rhizosphere is kept from becoming anoxic longer and therefore minimizes CH_4 production by methanogens. Since most CH_4 produced in the rhizosphere is transported to the atmosphere by passive transport through aerenchyma tissue, the typically greater biomass associated with hybrid compared to pure-line cultivars facilitates the removal of CH_4 from the rice rhizosphere to avoid having excess amounts of CH_4 trapped in the soil near the roots (Kludze et al., 1993; Aulakh et al., 2000; Wassman and Aulakh, 2000).

Along with cultivar selection, which is a relatively easily implemented management practice option for rice producers, water management scheme also is a main controlling factor for CH_4 emissions from rice (IPCC, 1996). However, water management alternatives are much less easily implemented compared to cultivar selection due to the potential constraints of water delivery to a field and fact that rice is a semi-aquatic plant that is adapted for optimal growth under flooded-soil conditions. As a semi-aquatic plant, globally rice requires between 1250 to

8500 m^3 ha^{-1} (4.9 to 33.5 in) of water per growing season, making water management and conservation critical in the rice production system worldwide (de Avila et al., 2015; Henry et al., 2016). Rice in the US is generally grown under continuously flooded conditions throughout the growing season. Groundwater is used to irrigate over 74.1% of the rice acreage in Arkansas with the remaining acres irrigated with surface water obtained from reservoirs, streams, or bayous (Hardke, 2016).

The primary irrigation practice in Arkansas is the use of a cascade levee system to establish and maintain a semi-permanent flood (Hardke, 2016). As of 2015, rice producers utilize this practice on 57% of the rice acreage in Arkansas (Hardke, 2016). In Arkansas, the drill-seeded, delayed-flood rice production system is the predominate production system, accounting for 85% of total planted-rice area, for which annual irrigation-water use averaged 763 mm (30.0 in) over a 10-yr period between 2003 to 2012 (Henry et al., 2016). Utilizing a mid-season release of the flood (i.e., mid-season drain) has historically been used in rice production to control for straighthead, a disorder that causes sterility of the spikelets and reduces yield, and decrease the bioavailability of arsenic to the plant by keeping the arsenic in a non-reduced state (IPCC, 2014). As an alternative water management practice, the mid-season drain aerates the topsoil and reduces the time that the topsoil experiences anoxic conditions, which are required for CH_4 production. Consequently, the mid-season drain may have positive implications for the sustainability of rice production if rice yields can be maintained, while reducing CH_4 emissions at the same time. However, this practice can be difficult to implement due to a narrow critical window in which to allow soil to dry and re-establish the flood before drought stress becomes yield-limiting. Rainfall during the desired mid-season drain period can also mitigate the success of this practice.

Since agriculture is responsible for 10 to 12% of total global anthropogenic GHG emissions, accounting for nearly 50% of global CH_4 emissions alone (Smith et al., 2007), mitigation of CH_4 production and release in agricultural settings, particularly in areas of concentrated rice production, have profound importance. Consequently, to reduce CH_4 emissions from rice production, field management practice combinations that promote reduced CH_4 emissions, without decreasing yields or milling quality, must be identified (Lindau et al., 1993). Therefore, the objective of this study was to evaluate the effects of water management (i.e., full-season flood and mid-season drain) and cultivar (i.e., a conventional pure-line and a hybrid cultivar) on CH_4 fluxes and season-long emissions from rice grown on a silt-loam soil in the direct-seeded, delayed-flood production system in eastern Arkansas. Based on previous field research results (Brye et al., 2013; Rogers et al., 2013; Simmonds et al., 2015; Smartt et al., 2016), it was hypothesized that the mid-season-drain/hybrid will have the lowest and the full-season-flood/pure-line treatment combination will have the largest season-long CH_4 emissions. It was also hypothesized that the mid-season-drain/hybrid will have the lowest CH_4 emissions per unit grain yield among the water management/cultivar treatment combinations.

Materials and Methods

Site Description

Field research, similar to that conducted recently by Rogers et al. (2014), was conducted in 2015 at the University of Arkansas System Division of Agriculture Rice Research and Extension Center (RREC) near Stuttgart, Arkansas (34°27’54.5” N, 91°25’8.6” W). The soil throughout the study area was a DeWitt silt loam (fine, smectitic, thermic Typic Albaqualf) (USDA, 2015). The RREC is located in Arkansas County within a region known as the Grand

Prairie, which is part of Major Land Resource Area 131D, the Southern Mississippi River Terraces (USDA, 2006). The study area has been managed in a rice-soybean (*Glycine max* L. [Merr.]) rotation, which is a common rotation for rice production in east-central Arkansas, for more than 25 years. The slope across the study area was approximately 0.15%. The regional climate throughout the study area is temperate with a mean annual air temperature of 17°C, which ranges from a mean minimum of 12.7°C to a mean maximum of 23.5°C (NOAA, 2015). The mean annual precipitation is 135 cm (NOAA, 2015).

Treatments, Experimental Design, and Agronomic Management

The study area consisted of 16 field plots, 1.6-m wide by 5-m long, with nine rice rows planted with an 18-cm row spacing, arranged in a randomized complete block (RCB) design with four replications of each treatment combination. Eight plots (i.e., four pure-line and four hybrid-planted plots) were established in a delayed, permanent flood bay, hereafter referred to full-season flood, and eight plots (i.e., four pure-line and four hybrid-planted plots) were established in a mid-season-drain bay. The pure-line rice cultivar ‘LaKast’ and the hybrid rice cultivar XP753 (RiceTec, Inc., Houston, TX) were drill-seeded on 6 May, 2015. The flood was established on 10 June, 2015 and was maintained at a depth of approximately 10 cm until maturity, at which time the flood was released on 24 October 2015 to prepare for harvest.

Recommended nitrogen (N) fertilization was used for optimal production of both cultivars (Norman et al., 2013). The pure-line received 117 kg N ha^{-1} that was broadcast manually as urea (46% N) 24 hr before the flood was established (10 June, 2015) and an additional split application of 45 kg N ha^{-1} was applied manually to the floodwater at beginning of internode elongation (1 July, 2015) approximately 20 days after flooding (DAF). The hybrid cultivar received 134 kg N ha^{-1} pre-flood (10 June, 2015) and a split application of 33 kg N ha^{-1} applied manually to the floodwater at the boot stage (14 July, 2015) approximately 34 DAF.

Initial Soil Sample Collection, Processing, and Analyses

Prior to flood establishment, two soil cores 4.8 cm in diameter were collected from the top 10 cm in each plot for a total of 32 cores collected from within the study area. Soil samples were dried at 70°C for 72 hr, crushed, and sieved through a 2-mm mesh screen for soil property determinations. One set of soil samples per plot was used for determining bulk density and particle-size analyses using a modified 12-hr hydrometer method (Gee and Or, 2002). The second set of soil samples was analyzed by inductively coupled, argon plasma, atomic emissions spectrometry (Spectro Arcos, Spectro Analytical Instruments, Kleve, Germany) using a 1:10 soil-mass-to-extractant-volume ratio (Tucker, 1992) for Mehlich-3 extractable nutrients (i.e., P, K, Ca, Mg, S, Na, Fe, Mn, Zn, and Cu). Total soil carbon (TC) and total nitrogen (TN) concentrations were measured by high-temperature combustion with a VarioMax CN analyzer (Elementar Americas, Inc., Mt. Laurel, NJ). Measured TC and TN concentrations were used to calculate C;N ratios on a plot-by-plot basis. Soil organic matter concentration was determined by weight-loss-on-ignition after 2 hr at 360°C. Soil pH and electrical conductivity (EC) were analyzed potentiometrically in a 1:2 (m/v) soil-water suspension. Based on measured bulk densities in each plot and the 10-cm sampling interval, all measured concentrations (mg kg^{-1}) were converted to contents (g, kg, or Mg ha^{-1}) for reporting purposes.

Soil Oxidation-Reduction Potential and Temperature Measurements

Immediately after flooding of the field plots began (15 June 2015), soil oxidation-reduction (redox) potential (Eh) sensors (Model S650KD-OR, Sensorex, Garden Grove, CA) with Ag/AgCl reference solution were installed vertically to a depth of approximately 7 cm. One Eh sensor was installed in the bulk soil and a second sensor was installed adjacent to a gas-sampling-chamber base collar, described below, in each plot. In addition to the Eh sensors, chromel-constantan thermocouples were installed horizontally in the bulk soil at a depth of approximately 7 cm in each plot. All sensors were connected to a datalogger (CR 1000, Campbell Scientific, Inc., Logan, UT), protected by an environmental enclosure, to record soil Eh and soil temperature at 15-minute intervals, while mean data were output every hour. Measured sensor data were collected weekly. Soil Eh values were corrected to the standard hydrogen electrode by adding 199 mV to each field-measured value (Patrick et al., 1996).

For the purposes of data reporting, both soil temperature and redox data from the hour during gas sample collection on each measurement data were extracted from the continuously recorded data for all replicate sensors. The individual hourly soil temperature and redox data from each weekly measurement date were subsequently used for statistical analyses.

Gas Sample Collection and Analyses

Similar to procedures used by Rogers et al. (2014), after planting and before flooding, a boardwalk system was constructed throughout the study area to reduce disturbances to the rice plants and allow easier access to the plots during the growing season for gas sample collection and other plot maintenance and access. The board walk was constructed of 5.1-cm x 30.5-cm x 3.6-m pressure-treated wooden boards laid upon 20- x 40-cm concrete blocks before chamber base collar installation in the plots. The base collars were then set into place to encompass the third and fourth rice rows in each plot for gas sampling.

Vented, non-flow-through, non-steady-state chambers (Livingston and Hutchinson, 1995) were used for the collection of gas samples for the determination of CH_4 fluxes. Schedule 40 polyvinyl chloride (PVC) was used in the construction of cylindrical base collars, 30 cm in diameter by 30-cm tall, that were inserted to a depth of approximately 10 cm. The collars were beveled on one end to a 45° angle to allow for easier insertion into the soil. Approximately 12 cm from the beveled end of each base collar, four 12.5-mm diameter holes were drilled to allow for flood water to enter and exit the collar. The collars were driven into the ground to a depth of 11 cm to allow for the drilled holes to be just above ground level. During sampling after flood release, the holes were plugged with gray butyl-rubber septa (Voigt Global, part# 73828A-RB, Lawrence, KS) to prevent convection currents inside the chambers that would dilute the ambient, headspace air.

Chamber extensions, 40 and 60 cm in length, were used to facilitate rice growth during the season. Chamber extensions were covered in reflective aluminum tape (CS Hyde, Mylar metallized tape, Lake Villa, IL) to reduce temperature variations inside the chamber during use. Tire inner tube cross sections, approximately 10-cm wide, were also taped to the bottom of all the extensions and functioned as a seal to the base collars and to the other extensions during chamber use.

Chamber caps were constructed with 10-cm tall cross sections of 30-cm diameter PVC, with a 5-mm thick sheet of PVC glued to the top and covered with reflective aluminum tape. Tire inner tube cross sections, approximately 10-cm wide, were also taped to the bottom of the caps to serve as a seal and attachment mechanism to the chamber base collar or extensions. A 15-cm long piece of 4.5-mm inside diameter (id) copper refrigerator tubing was installed on the side of each cap to maintain atmospheric pressure during use. On the top of the chamber caps, 12.5-mm diameter holes were created and plugged with gray butyl-rubber septa (Voigt Global, part# 73828A-RB, Lawrence, KS) for thermometer and syringe insertion. To ensure proper air mixing in the enclosed chamber, a 2.5-cm tall x 2.5-cm wide, battery-operated (9V), magnetic levitation fan (Sunon Inc., MagLev, Brea, CA) was installed that ran throughout the duration of gas sampling for headspace air mixing.

The collection of gas samples from the chambers was accomplished by using a 20-mL, B-D syringe with a detachable 0.5-mm diameter x 25-mm long needle (Beckton Dickson and Co., Franklin Lakes, NJ) that was inserted through the gray butyl-rubber septa installed in the chamber cap. After drawing a gas sample from the chamber, the collected sample was immediately injected into a pre-evacuated, 10-mL, crimp-top glass vial (Agilent Technologies, part# 5182-0838, Santa Clara, CA). Gas samples were collected at 20-minute intervals, beginning at 0 minutes when the chamber was capped and sealed, for 1 hr (i.e., the 0-, 20-, 40-, and 60-min marks). Gas sampling started 5 days after flood establishment in 2015 and continued weekly until flood release when sampling frequency changed to 1, 3, and 5 days after flood release. Similar to prior studies (Rogers et al., 2013, 2014), all gas sampling occurred in the morning between 0800 to 1000 hours to minimize potential temperature fluctuations in the chambers.

During each chamber sampling event, ambient air temperature, relative humidity, barometric pressure, 10-cm soil temperature, and the air temperature inside the chamber were recorded at every sampling interval (i.e., the 0-, 20-, 40-, and 60-min marks). At the end of gas sampling, the distance from the top of the chamber to the water level was recorded so that the interior chamber volume could be calculated. Samples of CH_4 gas standards (i.e., 2, 5, 10, 20, and 50 mg L^{-1}) were collected in the field using a 20-mL syringe with detachable needle that was immediately injected into pre-evacuated, 10-mL, crimp-top glass vials. Methane gas samples from the same four concentration standards were also collected in the laboratory immediately prior to gas sample analysis.

Using a flame ionization detector (250°C) equipped with a gas chromatograph (Model 6890-N; Agilent Technologies, Santa Clara, CA) with a 0.53-mm-diameter x 30-m HP-Plot-Q capillary column (Agilent Technologies, Santa Clara, CA), gas samples were analyzed for their CH_4 concentration within 48 hr of collection. Methane fluxes were calculated according to changes in concentrations in the chamber headspace over the 60-min sampling interval following procedures outlined by (Rogers et al., 2013). To determine the change in concentration over time, measured concentrations (mL L^{-1}; y axis) were regressed against time (min; x axis) of sample extraction (i.e., 0, 20, 40, and 60 min). The slope of the resulting best-fit line was then multiplied by the calculated chamber volume (L) and divided by the inner surface area of the chamber (m^2) resulting in flux units of µL CH_4 m^{-2} min^{-1} (Parkin and Venterea, 2010). The resulting units of the µL CH_4 were then converted using the Ideal Gas Law (PV = nRT) to µmol CH_4, where P was the pressure over the 60-min sampling interval in atmospheres (atm), V was the calculated volume of the interior of the chamber (L), n was the number of moles of the gas, R was the gas constant (0.8206 L atm Mol^{-1} K^{-1}), and T was the average temperature inside the chamber in Kelvin over the 60-min interval. To convert µmol CH_4 to the mass of CH_4, the molar mass of CH_4 was then used for a final flux unit of mg CH_4 m^{-2} d^{-1} (Parkin and Venterea, 2010).

Season-long emissions were calculated on a chamber-by-chamber basis by linear interpolation between sample dates. Emissions data were also divided into pre- and post-flood-release periods for data analyses due to differences in emissions mechanisms and to examine the impact of flood release and subsequent oxygenation on CH_4 emissions.

Plant Sampling and Processing

Seven days after the last gas sampling (i.e., 84 DAF), all aboveground biomass was collected from the interior of each base collar and dried at 55°C for 3 weeks then weighed to determine aboveground dry matter. Rice was harvested on 9 September 2015 (i.e. 86 DAF) with a research-grade plot combine, at which time a sub-sample of rice grain was collected to determine harvest grain moisture. The combine yield was corrected to 12% grain moisture for yield-reporting purposes. Total season-long CH_4 emissions (i.e., pre- plus post-flood-release emissions) were divided by total rice grain yield on a plot-by-plot basis to express emissions on a per-unit-grain-yield basis (i.e., an emissions intensity metric).

Statistical Analyses

Based on the RCB design with four replications of each treatment combination, a two-factor analysis of variance (ANOVA) was conducted using SAS (version 9.3, SAS Institute, Inc., Cary, NC) to determine pre-assigned treatment effects (i.e., cultivar, water management scheme, and their interaction) on initial soil properties (i.e., Mehlich-3 extractable P, K, Ca, Mg, S, Na, Fe, Mn, Zn, and Cu contents; soil pH and EC; SOM, TC, and TN contents; C:N ratio; bulk density; and sand, silt, and clay fractions) prior to flooding. A separate three-factor ANOVA was conducted to determine the effects of water management, cultivar, time (i.e., measurement date), and their interactions on CH_4 fluxes, soil temperature, and soil Eh. A separate two-factor ANOVA was conducted to determine the effects of water management, cultivar, and their interaction on rice grain yield; pre- and post-flood-release and total growing-season, area-based CH_4 emissions; and total growing-season, yield-based CH_4 emissions. All ANOVAs were conducted using the PROC MIXED procedure. When appropriate, means were separated by least significant difference (LSD) at the 0.05 level.

Results and Discussion

Initial Soil Properties

Initial soil properties in the top 10 cm prior to flooding were relatively uniform among pre-assigned treatment combinations throughout the study area (Table 1), where P, K, and Zn fertilizers were applied in March before study establishment. Most initial soil properties (i.e., EC, extractable K, Fe, Mn, Mg, S, Cu, Zn, TN, and SOM, bulk density, sand, silt, and clay) were unaffected ($P > 0.05$) by water management practice (i.e., full-season-flood and mid-season-drain), cultivar (i.e., the pure-line cultivar LaKast and the hybrid cultivar XL753), or their interaction (Table 1) and were all within recommended ranges for optimal rice production on a silt-loam soil (Hardke, 2014). Sand, silt, and clay averaged 0.21, 0.72, and 0.07 g g^{-1} in the top 10 cm, confirming a silt loam soil (Table 1). Soil organic matter, TC, and TN averaged 15.3, 7.6, and 0.92 Mg ha^{-1}, respectively for a mean C:N ratio of approximately 8:1. Extractable soil K and Zn (230 and 10.7 kg ha^{-1}) were within recommended optimum levels and extractable soil P (98.4 kg ha^{-1}) was above optimum for rice production on a silt loam soil (Norman et al., 2013). However, extractable soil Ca and Na, soil pH, and TC in the top 10 cm differed ($P \leq 0.01$) between water management schemes. Extractable soil Ca and Na were 120 and 36.5 kg ha^{-1}, respectively, and soil pH was 0.26 units greater in the full-season-flood than in the mid-season-drain water management scheme before flooding. Total carbon was also greater in the full-season-flood (7.77 Mg ha^{-1}) than in the mid-season-drain (7.49 Mg ha^{-1}). Despite the few differences in soil properties among pre-assigned treatments, all differences were small enough to cause no expected differences in rice growth or production. Consequently, any measured differences in CH_4 fluxes or emissions were assumed to be the result of actual treatment effects rather than due to inherent differences among plots.

Methane Fluxes

Similar to other reports in Arkansas (Brye et al., 2013; Rogers et al., 2014; Linquist et al., 2015; Smartt et al., 2016), CH_4 fluxes during the 2015 rice growing season followed a somewhat predictable temporal pattern throughout the rice growing season. Methane fluxes started low, increased to a numeric peak that ranged from 100 to 230 mg CH_4-C m^{-2} d^{-1} for the mid-season-drain/hybrid and full-season-flood/hybrid treatment combination, respectively, between 47 and 54 DAF, which was approximately 50% heading, and decreased thereafter until the flood was released at 68 DAF (Figure 1). The numeric peak flux from the full-season-flood/hybrid treatment combination was comparable to that of Brye et al. (2013) who reported a peak of 390 CH_4-C m^{-2} d^{-1} at 51 DAF from a pureline cultivar 'Taggart' grown on silt-loam soil under a full-season flood. In contrast, Simmonds et al. (2015) reported no relationship between the temporal pattern of weekly CH_4 emissions and the physiological growth stages of the rice crop. After flood release, CH_4 fluxes in all treatment combination at least slightly numerically increased within 5 days before decreasing to near zero by 81 DAF (Figure 1). This post-flood-release spike in CH_4 fluxes has been reported numerous times in both silt-loam and clay soils in Arkansas (Brye et al., 2013; Rogers et al., 2014; Linquist et al., 2015; Adviento-Borbe and Linquist, 2016; Smartt et al., 2016).

During the 2015 growing season, CH_4 fluxes differed ($P < 0.01$; Table 2) among water management/cultivar treatment combinations over time (Figure 1). Methane fluxes measured 5, 12, and 19 DAF in each treatment combination did not differ from a flux of zero. At 22 DAF, seven days after flood re-establishment following the mid-season drain at 15 DAF, CH_4 fluxes from both mid-season-drain treatments did not differ from a flux of zero, while fluxes from both full-season-flood combinations increased from that at 19 DAF, but did not differ from one another.

Between 33 and 72 DAF, CH_4 fluxes from the mid-season-drain/hybrid combination was lower than that from all other treatment combinations, with a peak average flux of 102.7 CH_4-C m^{-2} d^{-1} that occurred 47 DAF (Figure 1). The CH_4 fluxes at 40 DAF for the mid-season-drain/hybrid were different than zero and lower than the other three treatment combinations which were similar to one another on each date. By 47 DAF, both water management treatments with pure-line varieties did not differ from one another, while both were greater than that from the mid-season-drain/hybrid treatment combination. The CH_4 fluxes at 47 DAF did not differ from 40 DAF for the full-season-flood/pure-line or the mid-season-drain/hybrid, there was a difference for the mid-season-drain/pure-line and the full-season-flood/hybrid. The largest average peak flux occurred from the full-season-flood/hybrid combination (204.9 CH_4-C m^{-2} d^{-1}) between 40 and 47 DAF (Figure 1). The average peak fluxes for the mid-season-drain/pure-line and the full-season-flood/pure-line treatment combinations were 173.9 and 171.4 CH_4-C m^{-2} d^{-1}, respectively, which occurred at 54 to 61 and 40 to 47 DAF, respectively (Figure 1). The difference in peak fluxes between the mid-season-drain/hybrid and full-season-flood/hybrid combinations (i.e., 50%) highlights the impact of the alternative water management scheme at reducing CH_4 fluxes.

From 47 to 68 DAF, CH_4 fluxes decreased from all treatment combinations. At 54 and 61 DAF, CH_4 fluxes from the full-season-flood did not differ from one another and were greater than the fluxes from both mid-season-drain treatment combinations; however, CH_4 fluxes from the mid-season-drain/pure-line was greater than the fluxes from the mid-season-drain/hybrid treatment combination (Figure 1). At 67 DAF, four days prior to flood release, CH_4 fluxes from the full-season-flood/pure-line were greater than the fluxes from all other treatment combinations, while fluxes from the full-season-flood/hybrid and mid-season-drain/pure-line did not differ from one another and both were greater than the fluxes from the mid-season-drain/hybrid treatment combination.

After flood release (i.e., 72 DAF), CH_4 fluxes from the full-season-flood/hybrid treatment increased from 67 (95.2 CH_4-C m^{-2} d^{-1}) to 75 DAF (141.5 CH_4-C m^{-2} d^{-1}). However, the other treatment combinations only had small, numeric increases in CH_4 fluxes between 68 and 75 DAF, which peaked at 150.9, 154.9, and 56.8 CH_4-C m^{-2} d^{-1} from the full-season-flood/pure-line and the mid-season-drain/pure-line and hybrid treatment combinations, respectively (Figure 1). At 75 DAF, fluxes from the mid-season-drain/hybrid were lower than those from all other treatment combinations, which did not differ from one another. By 77 DAF, fluxes from both mid-season-drain treatment combinations did not differ from one another, but both were lower than fluxes from the full-season-flood treatment combinations. By 78 DAF, CH_4 fluxes for the mid-season-drain/pure-line and hybrid treatments did not differ from zero, while CH_4 fluxes from the full-season-flood/pureline and hybrid treatments were both greater than zero but did not differ from each other or the fluxes measured at 79 DAF. At 79 DAF, CH_4 fluxes for the full-season-flood/pure-line and hybrid and the mid-season-drain/hybrid treatment combinations did not differ among themselves and were all slightly greater than zero. Methane fluxes from the mid-season-drain/hybrid treatment did not differ from zero at 79 DAF, and, by 81 DAF, CH_4 fluxes from all treatment combinations did not differ from zero. The post-flood-release spike in CH_4 fluxes was consistent with similar previous reports (Brye et al., 2013; Rogers et al., 2014; Linquist et al., 2015). By 77 DAF, CH_4 fluxes from each treatment combination had decreased to similar to zero, indicating the cessation of CH_4 production and release.

Soil Temperature and Redox Potential Fluctuations

At the time of flood establishment, soil temperatures at the 7-cm depth averaged 29°C across both water management schemes, which then increased to the growing-season maximum of 33°C in the first few days after flooding (Figure 2). The soil temperature remained relatively constant and uniform between water management treatments, except for when the mid-season-drain occurred at 16 DAF when the average soil temperature for the mid-season-drain (25.5°C) was lower ($P < 0.01$) than that for the full-season-flood treatment (28°C; Figure 2). The 7-cm soil temperature did not differ between water management treatments on any other measurement date during the 2015 rice growing season and was unaffected by rice cultivar ($P > 0.05$). The results of this study were similar to those reported by Rogers et al. (2013), where a maximum 7-cm soil temperature of 32°C occurred at 19 DAF.

Similar soil temperature trends, but as expected, soil Eh started well-oxidized and decreased thereafter following flood establishment (Figure 2). The soil redox level of approximately -200 mV is necessary for CH_4 production (Reddy and DeLaune, 2008). Averaged across cultivar, soil Eh was greater (i.e., more oxidized; $P < 0.01$; Table 4) in the mid-season-drain than in the full-season-flood treatment at 19 and 26 DAF, whereas soil Eh was similar between water management treatments on each other weekly measurement date. Soil Eh at the 7-

cm depth in the full-season-flood treatment decreased to < -200 mV by 54 DAF and remained < -200 mV until the flood was released at 72 DAF, while that in the mid-season-drain did not reach < -200 mV until two weeks later at 68 DAF. However, after 36 DAF, which was three weeks after flood reestablishment in the mid-season-drain treatment, soil Eh in both water management treatments had similar magnitudes and followed the same pattern for the rest of the growing season.

In contrast to soil temperature, which was unaffected by cultivar, soil Eh differed among water treatment-cultivar combinations ($P < 0.02$; Table 4). Soil Eh at the 7-cm depth was greater in the mid-season-drain/hybrid (-14.0 mV) than in the other three treatment combinations, which did not differ and averaged -72.1 mV. The increase in soil Eh in the mid-season drain demonstrates the synergistic effect of the combination of the alternative water management practice and use of a hybrid cultivar on soil redox potential due to enhanced root zone oxygenation (Ma et al., 2009).

In a similar study on a silt-loam soil in east-central Arkansas, Rogers et al. (2013) reported soil Eh rapidly decreased to < -200 mV by 25 to 30 DAF in a full-season-flood treatment. Soil Eh in the current study, averaged over cultivar, also differed among water treatments over time ($P < 0.04$; Table 4). However, in contrast to the soil Eh trends under the full-season-flood treatment, after decreasing following flood establishment, soil Eh in the mid-season drain increased from +128 mV at 12 DAF to +226 mV at 19 DAF then decreased to +122 mV at 26 DAF, clearly indicating that the drained soil became more oxidized than the soil under the continuous flood. Directly after the mid-season-drain, CH_4 fluxes from the mid-season-drain/hybrid treatment decreased and did not increase again until after the flood was reestablished at 20 DAF. The increase in soil Eh measured in the mid-season-drain was a significant increase compared to soil Eh measured in the full-season-flood treatment, which, over the same time, continued to decrease from +115 to -66 mV by 26 DAF. Soil Eh did not differ between the two water management practices for the remained of the rice growing season.

In the current study, the soil Eh trends over time under the mid-season-drain treatment at least partially explain the low CH_4 fluxes from the mid-season-drain/hybrid treatment combination throughout most of the rice growing season and indicated at least two weeks less time available for CH_4 production under the mid-season-drain than under the full-season-flood treatment. It would be expected that less time available for CH_4 production due to more-oxidized soil conditions for some time under the mid-season-drain would result in lower CH_4 emissions than from the full-season-flood treatment that had a longer time available for CH_4 production due to more prolonged reducing conditions.

Area-scaled Methane Emissions

Since the presence or absence of the flood itself affects the mechanism by which CH_4 is released from the soil, emissions were analyzed separately for these two periods of the rice growing season. Between initial flooding and flood release, CH_4 emissions were unaffected by water management scheme and cultivar ($P > 0.05$; Table 2). Pre-flood-release CH_4 emissions averaged 50.8 kg CH_4-C ha^{-1} across all treatment combinations. In contrast, post-flood-release CH_4 emissions differed ($P = 0.02$; Table 2) between water management schemes, where emissions from the full-season-flood 14.0 kg CH_4-C ha^{-1}) were 1.7 times greater than emissions from the mid-season-drain (8.2 kg CH_4-C ha^{-1}) treatment.

During the complete 2015 growing season and in contrast to that hypothesized, total season-long, area-scaled emissions differed between water management treatments ($P < 0.01$), but were unaffected ($P > 0.05$) by cultivar (Table 2). Season-long, area-scaled CH_4 emissions were 1.8 times greater from the full-season-flood (77.7 kg CH_4-C ha^{-1} $season^{-1}$) than from the mid-season-drain (42.8 kg CH_4-C ha^{-1} $season^{-1}$) water management scheme (Table 3). These results support the expected emissions differences between the two water management schemes based on the soil Eh trends (Figure 2).

Post-flood-release CH_4 emissions represented 18.0 and 19.2% and averaged 18.6% of total season-long emissions for the full-season-flood and mid-season-drain treatments. This proportion of post-flood-release emissions is larger than that reported in recent studies under a full-season flood in east-central Arkansas (Brye et al., 2013; Rogers et al., 2013), which ranged from 3.4 to 13.2% from a silt loam soil under a continuous flooding, but from different pure-line cultivars (i.e., 'Taggart' and 'Wells').

A similar study conducted by Simmonds et al. (2015) investigated water management effects on CH_4 emissions in east-central Arkansas, but, to the best of the author's knowledge, this current study was the first to investigate the combination of mid-season-drain and full-season-flood water management schemes with pure-line and hybrid cultivars. Rogers et al. (2013) reported total season-long, area-scaled emissions from a full-season-flood on a silt-loam soil near Stuttgart, AR ranged from 54 kg CH_4–C ha^{-1} from N-fertilized bare soil to 220 kg CH_4–C ha^{-1} from an optimally N-fertilized pure-line cultivar 'Wells'. Simmonds et al. (2015) reported area-scaled CH_4 emissions from a silt-loam soil near Stuttgart, AR for a one-flush irrigation before continuous flooding and a continuous-flood regime ranged from 34 to 70 kg CH_4–C ha^{-1}, respectively, from the hybrid cultivar 'CLXP4534' and pure-line cultivars 'Francis', 'Jupiter', and 'Sabine'.

Rice Dry Matter and Yields

Neither aboveground dry matter nor rice yields differed ($P > 0.05$) between water management schemes or cultivars (Table 4). Rice dry matter ranged from 27.8 Mg ha^{-1} in the full-season flood/hybrid treatment to 37.8 Mg ha^{-1} in the full-season flood/pure-line treatment and averaged 33.1 Mg ha^{-1} across all treatment combinations. Similarly, rice yields ranged from 10.0 Mg ha^{-1} from the mid-season-drain/pure-line to 12.6 Mg ha^{-1} from the full-season-flood/hybrid (Table 3) and averaged 11.1 Mg ha^{-1} across all treatment combinations. For comparison, based on Arkansas Rice Performance Trials in 2015, the average yields for continuous-flood regime on a Dewitt silt-loam soil near Stuttgart, AR were 10.6 and 7.5 Mg ha^{-1} for the hybrid 'XP753' and the pure-line 'LaKast', respectively (Hardke et al., 2016).

Methane Emissions Intensity

Maintaining or increasing rice yields and improving C emissions intensity by reducing CH_4 emissions should be considered when developing new/alternative rice production management practice combinations, such as increasing the use of the mid-season drain for straighthead control. Methane emissions intensity differed ($P = 0.04$) between water management schemes across cultivars (Table 2). Methane emissions intensity for the mid-season-drain/hybrid combination (2.52 kg CH_4-C Mg grain^{-1}) was more than 50% greater, where a low CH_4 emissions per unit grain yield value represented lower intensity, than that for the other three treatment combinations, which did not differ and averaged 6.45 kg CH_4-C Mg grain^{-1} (Table 3). These results are similar to those of Simmonds et al. (2015), who reported an average CH_4 emissions intensity from a silt-loam soil near Stuttgart, AR for a one-flush irrigation before continuous flooding and continuous-flood water management regime of 5.57 and 9.72 kg CH_4-C

Mg grain^{-1}, respectively, across the hybrid cultivar 'CLXP4534' and pure-line cultivars 'Francis', 'Jupiter', and 'Sabine'. However, Rogers et al. (2013) reported an increased CH_4 intensity of 27.6 kg CH_4-C Mg grain^{-1} under a continuous flood with the pure-line cultivar 'Wells' compared to this study's full-season-flood/pure-line combination of 7.39 kg CH_4-C Mg grain^{-1}. The differences in emissions intensity could be attributed to yield differences between the various pure-line cultivars, coupled with the decreased season-long emissions for the particular study year compared to results of Rogers et al. (2013).

Potential Agronomic and Environmental Implications

Alternative water management practices, as well as cultivar selection, have been shown to decrease CH_4 emissions, thereby providing opportunities to potentially reduce excess loss of C. In a meta-analysis, Carrijo et. al (2017) reported an estimated 25% water-use reduction for water management practices that utilized some form of alternate wetting and drying compared to continuous, full-season-flood management practices and reduced global warming potentials associated with rice production. Alternate-wetting-and-drying and mid-season-drain water management strategies could alleviate the potential problems associated with arsenic bioavailability and straighthead by purposefully inducing a re-oxygenated soil environment part way through the rice growing season (Linquist et al., 2015). If other potentially negative agronomic ramifications, such as weed control and N-fertilizer uptake intensity, can be overcome such that rice yields are not compromised, use of the mid-season-drain water management practice may have significant positive effects on the future sustainability of rice production, soil health, and climate change in specific regions in the US, such as in the Lower Mississippi River Valley which is one region of concentrated rice production (de Avila, 2015).

Significant reductions in CH_4 emissions from the mid-season-drain compared to the full-season-flood water management scheme will decrease the C footprint of rice, which may increase the marketability of rice as a staple food crop relative to other staple foods, such as potato (*Solanum tuberosum*) and other small grains (National, 2018). Reducing the C footprint may make rice more desirable for those that wish to reduce their personal climate-change impact on the planet, which may translate into a tremendous economic opportunity for rice producers, suppliers, and retailers in markets sensitive to climate-change awareness.

Conclusions

Results of this study confirmed the potentially positive impacts of alternative water management schemes and specific cultivar selection by reducing CH_4 fluxes and season-long emissions. Similar to that hypothesized, this study showed that, regardless of cultivar selection, mid-season draining of flood water significantly reduced season-long, area-scaled CH_4 emissions compared to the full-season-flood water management practice from rice grown in 2015 in the direct-seeded, delayed-flood production system on a silt-loam soil in east-central Arkansas. Similarly, this study also clearly showed that the mid-season-drain/hybrid (XP753) combination had the lowest CH_4 emissions per unit grain yield (i.e., the least emissions intensity) among all water management/cultivar treatment combinations evaluated. The reduction in CH_4 emissions per unit grain yield from the mid-season-drain/hybrid combination was magnified due to the significantly lower emissions from the mid-season-drain treatment coupled with the numerically greater yield from the hybrid cultivar compared to the full-season-flood treatment and pure-line cultivar, respectively. Based on reduced season-long CH_4 emissions, the mid-season-drain water management scheme, regardless of cultivar selection, appears to be a more environmentally sustainable agronomic practice compared to the full-season-flood scheme.

Though the results of this study were based on one growing season of direct measurements, these results, coupled with the results of previous studies, indicate relatively consistent CH_4 emissions responses from year to year at least partially due to the presence of the flood water for most of the growing season attenuating climate variations and inter-annual differences in growing-season weather conditions. Therefore, it is reasonable to conclude that these results can be extrapolated to other years and over a longer time period, as minor differences in growing-season weather conditions from year to year likely have minimal effect on CH_4 emissions. Since climate change is at least partially driven by anthropogenic greenhouse gas emissions (IPCC, 2014), research efforts to identify logical and feasible alternative rice production practices that decrease CH_4 and other greenhouse gas emissions need to continue. Furthermore, continued investigation, over multiple years, particularly direct field measurements, will be critically necessary in the future to better understand the effects of various alternative water management practices, current rice cultivars, and their combinations on CH_4 emissions from silt-loam soils in Arkansas and other regions of concentrated rice production.

Literature Cited

Adviento-Borbe, M. A., and B. Linquist. 2016. Assessing fertilizer N placement on CH4 and N2O emissions in irrigated rice systems. Geoderma 266:40-45.

Adviento-Borbe, M. A., C. M. Pittelkow, M. Anders, C. van Kessel, J. E. Hill, A. M. McClung, J. Six, and B. A. Linquist. 2013. Optimal fertilizer nitrogen rates and yield-scaled global warming potential in drill seeded rice. J. Environ. Qual. 42:1623-1634.

Aulakh, M. S., R. Wassman, C. Bueno, and H. Rennenberg. 2001. Impact of root exudates of different cultivars and plant development stages of rice (Oryza sativa L.) on methane production in a paddy soil. Plant Soil 230:77-86.

Aulakh, M. S., R. Wassman, H. Rennenberg, and S. Fink. 2000. Pattern and amount of aerenchyma relate to variable methane transport capacity of different rice cultivars. Plant Biol. 2:182-194.

Brye, K.R., C.W. Rogers, A.D. Smartt, and R.J. Norman. 2013. Soil texture effects on methane emissions from direct-seeded, delayed-flood rice production in Arkansas. Soil Sci. 178:519-529.

Chang, T., K.F. Kiple, and K.C. Ornelas. 2012. The Cambridge World History of Food, chapter II.A.7 - Rice, Cambridge University Press pp. 132-149.

Cicerone, R.J., and J.D. Shetter. 1981. Sources of atmospheric methane: measurements in rice paddies and a discussion. J. Geophys. Res. 86:7203-7209.

Conrad, R., C. Erkel, and W. Liesack. 2006. Rice Cluster I methanogens, an important group of Archaea producing greenhouse gas in soil. Curr. Opin. Biotechnol. 17:262-267.

Conrad, R., M. Klose, M. Noll, D. Kemnitz, and P. L. E. Bodelier. 2008. Soil type links microbial colonization of rice roots to methane emission. Global Change Biol. 14:657-669.

Dannenburg, S., and R. Conrad. 1999. Effect of rice plants on methane production and rhizospheric metabolism in paddy soil. Biogeochem. 45:53-71.

de Avila, L.A., L.F.D. Martini, R.F. Mezzomo, J.P. Refatti, R. Campos, D.M. Cezimbra, S.L.O. Machado, J.H. Massey, R. Carlesso, and E. Marchesan. 2015. Rice water use efficiency and yield under continuous and intermittent irrigation. Agron. J. 107:442-448.

Ferry, J.G. 1992. Methane from acetate. J. Bacteriol. 174:5489-5495.

Gee, G.W., and D. Or. 2002. Particle size analysis. In: J. H. Dane, editor Methods of Soil Analysis: Part 4 Physical Methods. pp. 255-293.

Groot, T. T., P. M. van Bodegom, H. A. J. Meijer, and F. J. M. Harren. 2005. Gas transport through the root–shoot transition zone of rice tillers. Plant Soil 277:107-116.

Hardke, J. T., K.A.K Moldenhauer, S. Xueyan, Y. Wamishe, R. J. Norman, D.L. Frizzell, E. Castaneda, G. Lee, D. Wisdom, M. Blocker, J. Bulloch, T. Beaty, R. Mazzanti, R. Baker, S. Runsick, M. Duren, Y. Liyew, and J. Chlapecka. 2016. Arkansas rice performance trials: 2014-2016. Misc. Publ. 176.

Hardke, J.T. 2014. Arkansas Rice Production Handbook. In: J. T. Hardke, editor Introduction. University of Arkansas Division of Agriculture, Cooperative Extension Service, Little Rock, AR.

Hardke, J.T. 2016. Trends in Arkansas rice production. In: R. J. Norman, and K. A. K. Moldenhauer, editors, B.R. Wells Arkansas Rice Research Studies, 2015. Univ. Ark. Agric. Exp. Sta. Res. Ser. 634., Univ. Ark. Agric. Exp. Sta., Fayetteville, AR. pp. 13-23.

Henry, C.G., S.L. Hirsh, M M. Anders, E.D. Vories, M.L. Reba, K.B. Watkins, and J.T. Hardke. 2016. Annual irrigation water use for Arkansas rice production. J. Irrig. Drain. Engin. 05016006:1-5.

Intergovernmental Panel on Climate Change (IPCC). 1996. Intergovernmental Panel on Climate Change (IPCC), Guidelines for National Greehouse Gas Inventories Chapter 4 IPCC, Geneva, Switzerland. pp. 53-75.

Intergovernmental Panel on Climate Change (IPCC). 2014. Intergovernmental Panel on Climate Change (IPCC), United States Greenhouse Gas Inventory Chapter 6. IPCC, Geneva, Switzerland. pp. 15-21.

Kludze, H.K., R.D. DeLaune, and W.H. Patrick, Jr. 1993. Aerenchyma formation and methane and oxygen exchange in rice. Soil Sci. Soc. Am. J. 57:386-391.

Lindau, C.W., P.K. Bollich, R.D. DeLaune, A.R. Mosier, and K.E Bronson. 1993. Methane mitigation in flooded Louisiana rice fields. Biol. Fert. Soils 15:174-178.

Linquist, B., M. Anders, M. A. Adviento-Borbe, R. Chaney, L. Nalley, E. da Rosa, and C. van Kessel. 2015. Reducing greenhouse gas emissions, water use, and grain arsenic levels in rice systems. Global Change Biol. 21:407-417.

Livingston, G., and G. Hutchinson. 1995. Enclosure-based measurement of trace gas exchange: Applications and sources of error. In: P. A. Matso, and R. C. Harriss, editors, Biogenic trace gases measuring emissions from soil and water. Blackwell Science, Oxford [England]; Cambridge, MA, USA. pp. 14-51.

Mitra, S., R. Wassman, M. Jain, and H. Patak. 2002b. Properties of rice soils affecting methane production protentials: 2. Differences in topsoil and subsoil. Nutr. Cycl. Agro. 64:183-191.

National Agricultural Statistics Service, (NASS). 2016. Agricultural Statistics 2016. In: United States Department of Agriculture, editor United States Government, United States Government Printing Office Washington, D.C.

National Geographic Society, (National). 2018 Staple crops of the world. Available at: https://www.nationalgeographic.org/maps/wbt-staple-food-crops-world/ (Accessed on 6 Sept. 2018)

National Oceanic & Atmospheric Administration (NOAA). 2015. National Environmental Satellite, Data, and Information Service. Annual Climatological Summary (2015). National Centers for Environmental Information, U.S. Department of Commerce, 151 Patton Avenue, Asheville, NC 2880.

Norman, R.J., N.A. Slaton, and T.L. Roberts. 2013. Soil Fertility. In: J. T. Harke, editor Arkansas Rice Production Handbook Misc. Publ. 192. University of Arkansas Division of Agriculture Cooperative Extension Service, Little Rock, AR, USA. pp. 69-102.

Parkin, T.B., and R.T. Venterea. 2010. Sampling protocols. In: R.F. Follet USDA-ARS GRACEnet Project Protocols, editor Chapter 3. Chamber-Based Trace Gas Flux Measurements. pp. 3-1 to 3-39.

Patrick, W.H., R.P. Gambrell, and S.P. Faulkner. 1996. Redox measurements of soils. In: D. L. Sparks, editor Methods of Soil Analysis. part 3 - Chemical Methods. 3rd ed. SSSA, Madison, WI, USA. pp. 1255-1273.

Reddy, K.R., and R.D. DeLaune. 2008. Biogeochemistry of wetlands: Science and applications. 1st ed. CRC Press, Boca Raton, FL.

Rogers, C.W., K.R. Brye, R.J. Norman, E.E. Gbur, J.D. Mattice, T.B. Parkin, and T.L. Roberts. 2013. Methane emissions from drill-seeded, delayed-flood rice production on a silt-loam soil in Arkansas. J. Environ. Qual. 42:1059-1069.

Rogers, C.W., K.R. Brye, A.D. Smartt, R.J. Norman, E.E. Gbur, and M.A. Evans-White. 2014. Cultivar and previous crop effects on methane emissions from drill-seeded, delayed-flood rice production on a silt-loam soil. Soil Sci. 179:28-36.

Simmonds, M.B., M. Anders, M.A. Adviento-Borbe, C. van Kessel, A. McClung, and B.A. Linquist. 2015. Seasonal methane and nitrous oxide emissions of several rice cultivars in direct-seeded systems. J. Environ. Qual. 44:103-114.

Smartt, A.D., K.R. Brye, C.W. Rogers, R.J. Norman, E.E. Gbur, J.T. Hardke, and T.L. Roberts. 2016. Characterization of methane emissions from rice production on a clay soil in Arkansas. Soil Sci. 181:57-67.

Smartt, A.D., K.R. Brye, C.W. Rogers, R.J. Norman, E.E. Gbur, J.T. Hardke, and T.L. Roberts. 2016. Previous crop and cultivar effects on methane emissions from drill-seeded, delayed-flood rice grown on a clay soil. Appl. Environ. Soil Sci. 2016:1-13.

Smith, P., D. Martino, Z. Cai, D. Gwary, H. Janzen, P. Kumar, B. McCarl, S. Ogle, F. O'Mara, C. Rice, B. Scholes, and O. Sirotenko. 2007. Agriculture. In Climate Change 2007: Mitigation. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change [B. Metz, O.R. Davidson, P.R. Bosch, R. Dave, L.A. Meyer (editors)], Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Tucker, M.R. 1992. Determination of phosphorous by Mehlich 3 extraction. In: S. J. Donohue, editor, Soil and media diagnostic procedures for the Southern Region of the United States. Virginia Agric. Exp. Stn. Ser. Bull. 374., Blacksburg, VA, USA. pp. 6-8.

United States Department of Agriculture (USDA) 2006. Major Land Resource Regions Handbook 296. Available at: http://www.cei.psu.edu/mlra, (accessed 10 Jan. 2018).

United States Department of Agriculture (USDA). 2015. Soil Survey. National Resources Conservation Service Soils, Available at: http://www.nrcs.usda.gov (accessed 19 Jan. 2018).

United States Environmental Protection Agency (USEPA). 2017. United States Environmental Protection Agency (USEPA), Inventory of U.S. greenhouse gas emissions and sinks: 1990-2015. Available at: https://19january2017snapshot.epa.gov/ghgemissions_.html (accessed 10 Jan. 2018).

Wassman, R., and M. S. Aulakh. 2000. The role of rice plants in regulating mechanisms of methane missions. Biol. Fert. Soils 31:20-29.

Yu, K.W., Z. P. Wang, and G. X. Chen. 1997. Nitrous oxide and methane transport though rice plants. Biol. Fert. Soils 24:341-343.

Appendices

Table 1. Analysis of variance summary of the effects of rice cultivar, pre-assigned water management (WM) practice, and their interaction on soil physical and chemical properties from the top 10 cm of a Dewitt silt loam prior to flood establishment at the Rice Research and Extension Center near Stuttgart, Arkansas during the 2015 growing season. Overall means (n = 16) and standard errors (SE) are also reported.

Soil Property	Cultivar	WM	Cultivar x WM	Overall Mean (± SE)
		P		
Sand ($g\ g^{-1}$)	0.22	0.34	0.31	0.21 (< 0.01)
Silt ($g\ g^{-1}$)	0.24	0.68	0.30	0.72 (< 0.01)
Clay ($g\ g^{-1}$)	0.45	0.15	0.65	0.07 (< 0.01)
pH	0.43	**< 0.01**	0.27	6.7 (0.05)
Bulk density ($g\ cm^{-3}$)	0.70	0.33	0.70	1.38 (0.01)
Electrical conductivity ($dS\ m^{-1}$)	0.62	0.24	0.69	315 (53)
Extractable nutrients ($kg\ ha^{-1}$)				
P	0.87	0.19	0.59	98.4 (3.5)
K	0.92	0.25	0.86	230 (7.9)
Ca	0.88	**< 0.01**	0.28	1599 (21)
Mg	0.48	0.06	0.84	159 (2.0)
S	0.45	0.24	0.83	14.6 (0.54)
Na	0.44	**< 0.01**	0.87	148 (5.7)
Fe	0.51	0.27	0.89	646 (11)
Mn	0.25	0.68	0.98	293 (6.9)
Zn	0.54	0.25	0.28	10.7 (1.3)
Cu	0.73	0.31	0.43	1.3 (0.05)
Soil organic matter ($Mg\ ha^{-1}$)	0.48	0.79	0.78	15.3 (0.01)
Total N ($Mg\ ha^{-1}$)	0.28	0.64	0.42	0.92 (< 0.01)
Total C ($Mg\ ha^{-1}$)	0.26	**0.01**	0.18	7.6 (0.01)
C:N ratio	0.06	0.66	0.88	8.3(0.17)

Table 2. Analysis of variance summary of the effects of cultivar, water management, time, and their interaction on methane fluxes and the effects of cultivar, water management, and their interaction on pre- and post-flood-release and season-long, area-scaled and yield-scaled methane emissions during the 2015 growing season at the Rice Research and Extension Center near Stuttgart, Arkansas.

Variable/Source of Variation	*P*
Methane fluxes	
Cultivar	0.34
Water management	0.07
Time	< 0.01
Cultivar x water management	< 0.01
Cultivar x time	0.54
Water management x time	< 0.01
Cultivar x water management x time	**< 0.01**
Pre-flood-release, area-scaled emissions	
Cultivar	0.45
Water management	0.16
Cultivar x water management	0.48
Post-flood-release, area-scaled emissions	
Cultivar	0.11
Water management	**0.02**
Cultivar x water management	0.42
Season-long, area-scaled emissions	
Cultivar	0.40
Water management	**0.01**
Cultivar x water management	0.16
Season-long yield-scaled emissions	
Cultivar	0.43
Water management	0.01
Cultivar x water management	**0.04**

Table 3. Summary of mean season-long, area-scaled methane (CH_4) emissions, rice yield, and methane emissions intensity for the various water management/cultivar treatment combinations and water management practices averaged across cultivars during the 2015 growing season at the Rice Research and Extension Center near Stuttgart, Arkansas.

Water Management/ Cultivar Combination	Methane Emissions (kg CH_4-C ha^{-1} $season^{-1}$)	Rice Yield (Mg ha^{-1})	Emissions Intensity (kg CH_4-C Mg $grain^{-1}$)
Mid-season-drain/LaKast	56.6	10.0	5.67a†
Mid-season-drain/XL753	28.9	11.5	2.52b
Mid-season-drain Mean	42.8b†	10.7	3.99
Full-season-flood/LaKast	76.4	10.3	7.39a
Full-season-flood/XL753	79.1	12.6	6.29a
Full-season-flood Mean	77.7a	11.4	6.79

† Values in same column followed by different letters are significantly different ($P < 0.05$)

Table 4. Analysis of variance summary of the effects of water management, time, and their interactions on soil temperature and soil oxidation-reduction (redox) potential and the effects of cultivar, water management, and their interaction on rice dry matter and yield during the 2015 growing season at the Rice Research and Extension Center near Stuttgart, Arkansas.

Variable/Source of Variation	*P*
Soil temperature	
Cultivar	0.74
Water management	< 0.01
Time	< 0.01
Cultivar x time	0.18
Cultivar x water management	0.38
Water management x time	**< 0.01**
Cultivar x water management x time	0.99
Soil Redox	
Cultivar	< 0.63
Water management	< 0.16
Time	< 0.01
Cultivar x time	0.90
Cultivar x water management	**0.02**
Water management x time	**0.04**
Cultivar x water management x time	0.26
Rice dry matter	
Cultivar	0.39
Water management	0.94
Cultivar x water management	0.38
Rice yield	
Cultivar	0.87
Water management	0.61
Cultivar x water management	0.06

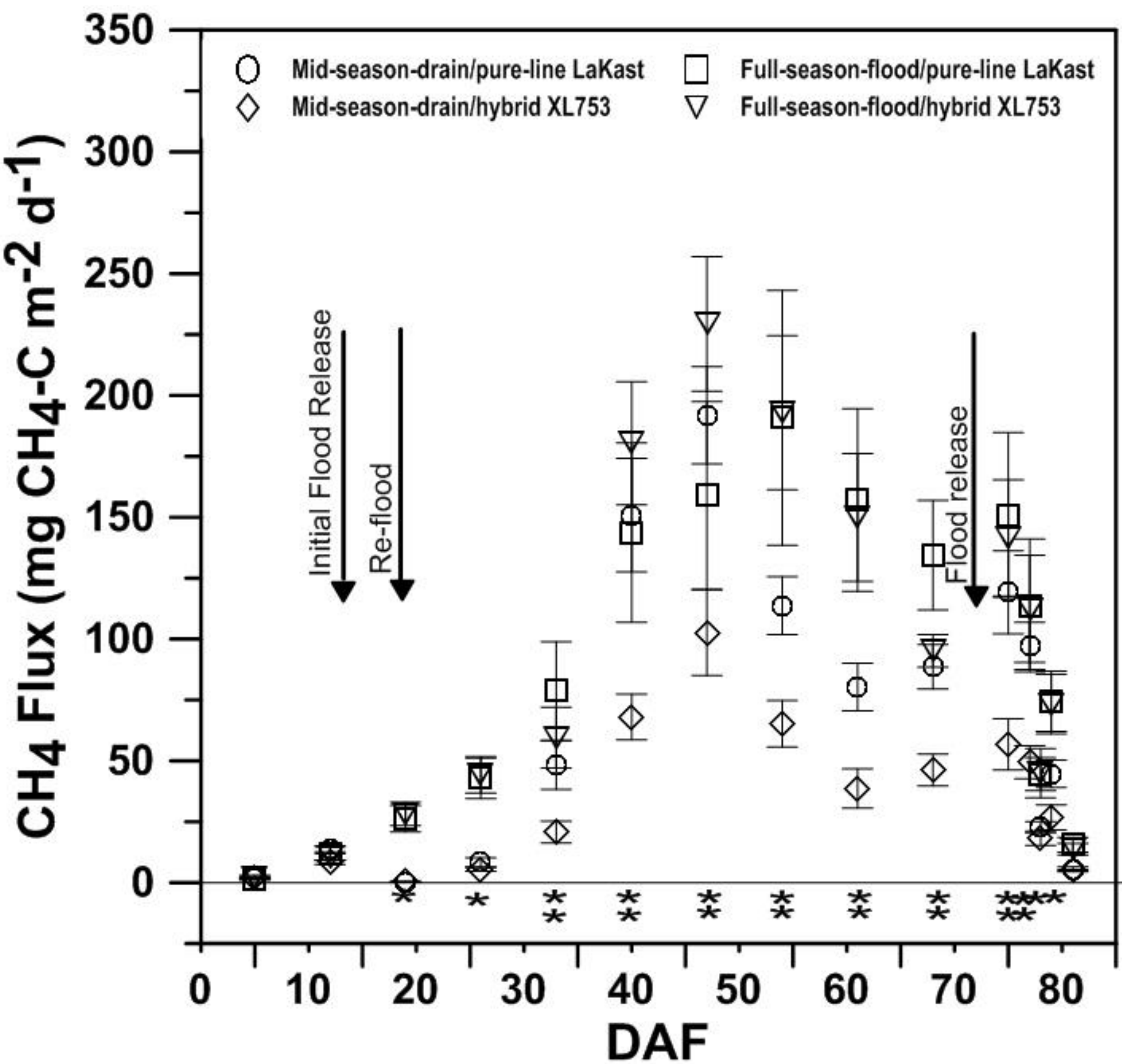


Figure 1. Season-long profile of methane (CH_4) flux trends over time for four water management scheme (mid-season-drain and full-season-flood) and cultivar (pure-line LaKast and hybrid XL753) treatment combinations from a DeWitt silt-loam soil during 2015 at the Rice Research and Extension Center near Stuttgart, Arkansas. The thick vertical lines indicate the timing of (1) flood release at 15 days after flooding (DAF) for the mid-season drain, (2) flood re-establishment at 20 DAF 5 days after the mid-season drain, and (3) flood release at 72 DAF from all plots prior to harvest. Standard error bars accompany treatment means (n = 4). A single asterisk on a given measurement date indicates a significant ($P < 0.05$) difference exists among treatment combinations, while a double asterisk indicates the mid-season-drain/XL753 combination is significantly ($P < 0.05$) lower than all other treatment combinations.

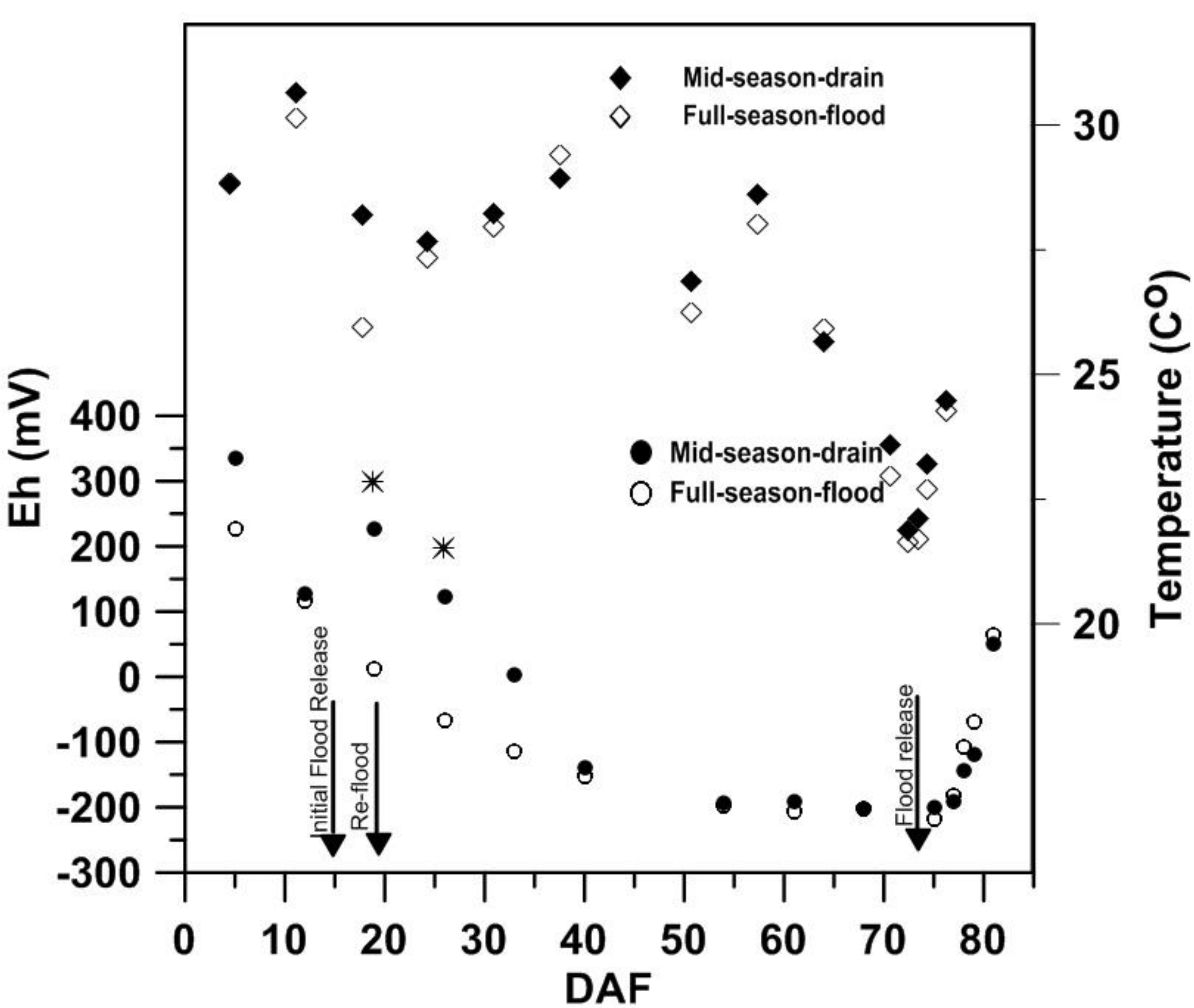
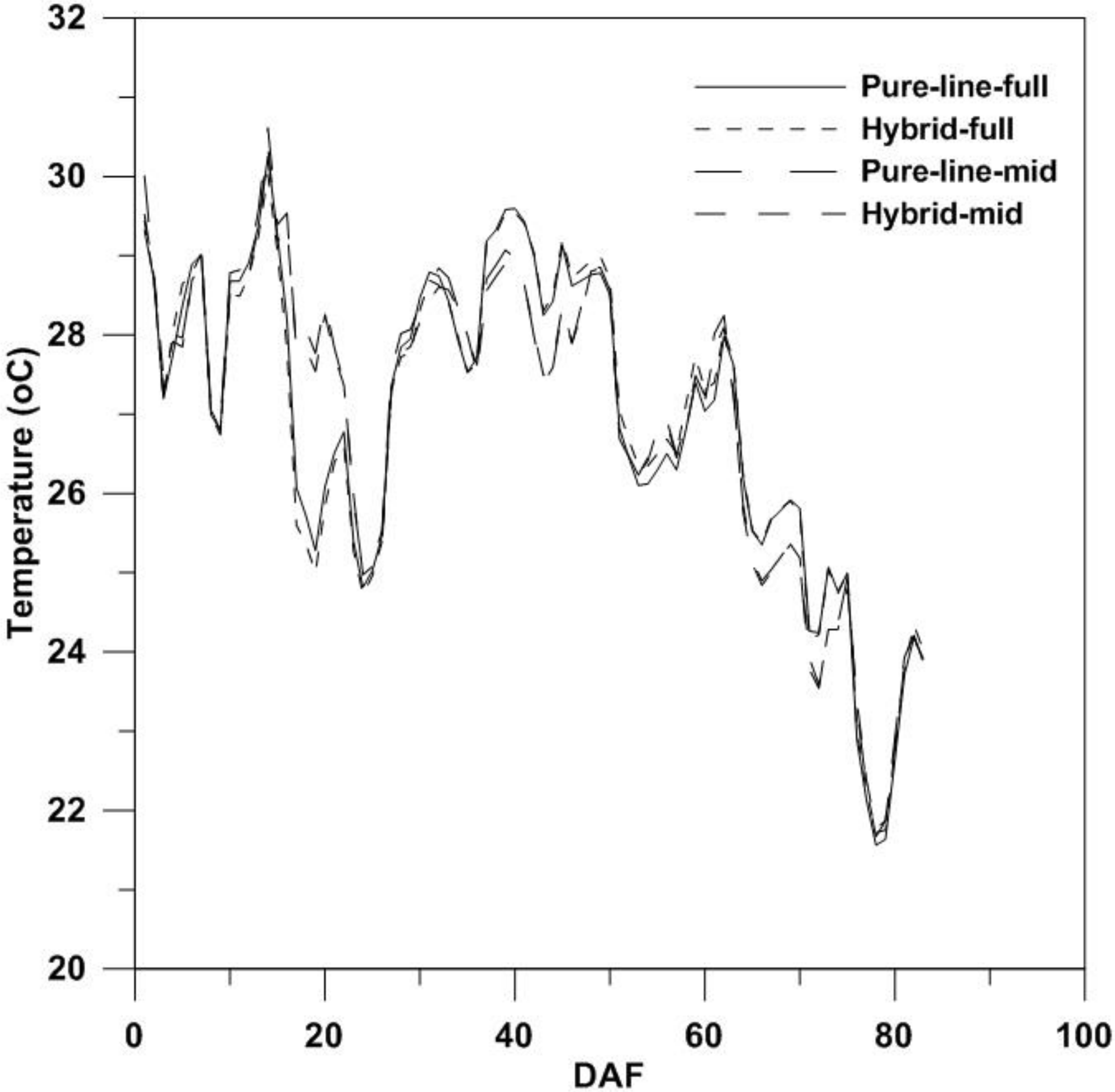
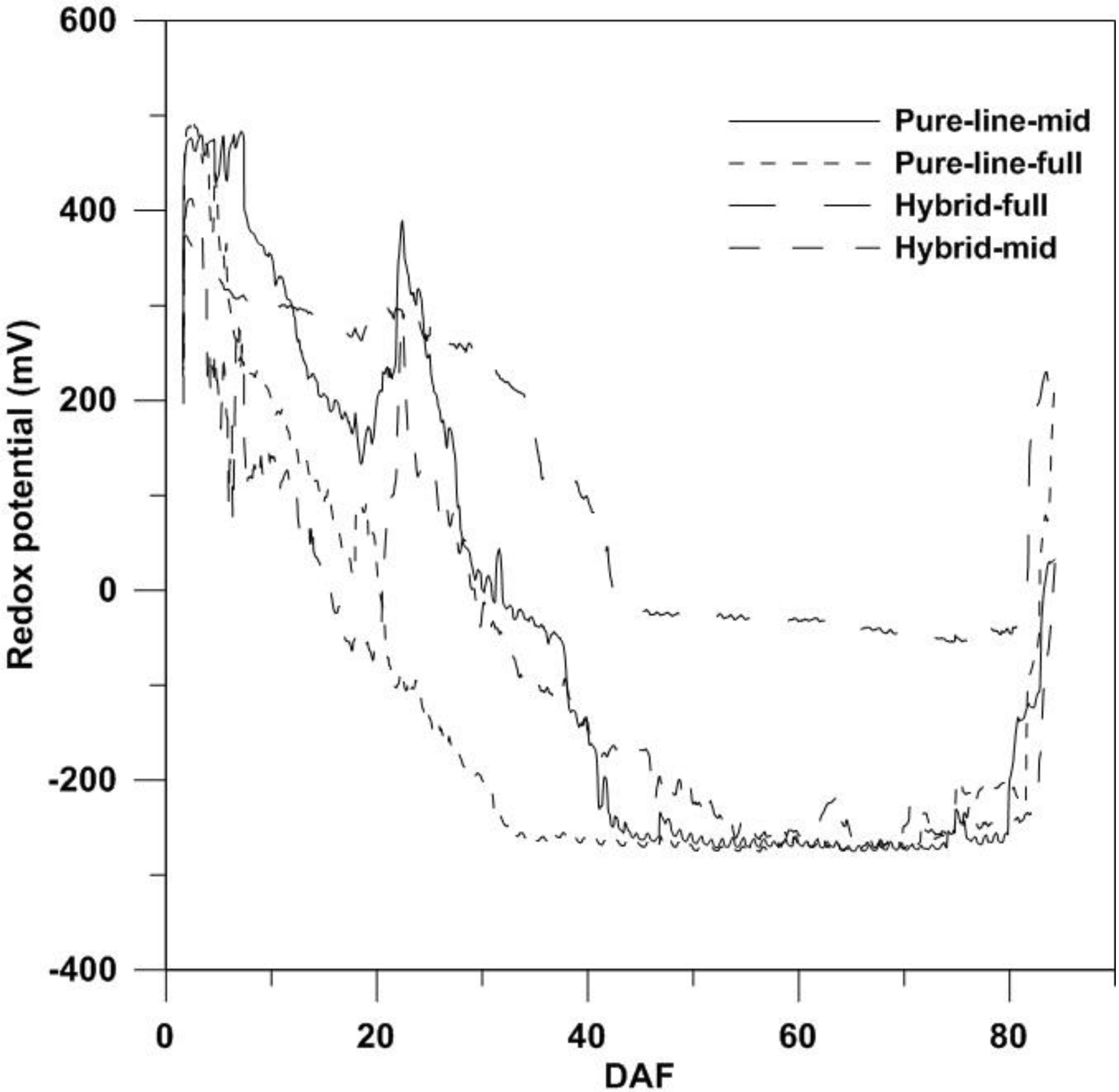


Figure 2. Season-long profile of soil temperature and oxidation-reduction potential (Eh) trends over time for the mid-season-drain and full-season-flood water management practices measured at a depth of 7 cm in a DeWitt silt-loam soil during 2015 at the Rice Research and Extension Center near Stuttgart, Arkansas. The thick vertical lines indicate the timing of (1) flood release at 15 days after flooding (DAF) for the mid-season drain, (2) flood re-establishment at 20 DAF 5 days after the mid-season drain, and (3) flood release at 72 DAF from all plots prior to harvest. Asterisks indicates a significant difference in soil temperature or soil Eh between water management schemes ($P < 0.05$).

Appendix A



Season-long profile of soil temperature trends over time [days after flooding (DAF)] for four water management schemes [mid-season-drain (mid) and full-season-flood (full)) and cultivar (pure-line LaKast (pure-line) and hybrid XL753 (hybrid)] treatment combinations at a depth of 7 cm in a DeWitt silt-loam soil during 2015 at the Rice Research and Extension Center near Stuttgart, Arkansas.



Season-long profile of soil oxidation-reduction potential (Eh) trends over time [days after flooding (DAF)] for four water management schemes [mid-season-drain (mid) and full-season-flood (full)] and cultivar (pure-line LaKast (pure-line) and hybrid XL753 (hybrid)] treatment combinations at a depth of 7 cm in a DeWitt silt-loam soil during 2015 at the Rice Research and Extension Center near Stuttgart, Arkansas.

Appendix B

Example of a SAS program evaluating CH_4 fluxes over time among water management treatment and cultivar for the 2015 growing season.

```
title 'Methane Field Study 2015 - Joshua Humphreys';
title2 'Methane Fluxes 2015 ANOVA';
data methane2015;
  infile 'CH4Flux2015.prn' firstobs=2;
  input ID DAF block treatment $ cultivar $ flux;
 run;

proc sort data=methane2015; by DAF;
quit;

proc print data=methane2015 noobs;by DAF;
id DAF;
var treatment cultivar flux;
run;

proc mixed data=methane2015 method=type3;
class cultivar treatment DAF block;
model flux = treatment cultivar treatment*cultivar DAF DAF*cultivar DAF*treatment DAF*
treatment*cultivar / ddfm=kr ;
random block block*treatment block*cultivar ;
quit;
```

Example SAS program evaluating CH_4, season-long emissions, post-season long emissions, yield, and emissions intensity for the 2015 growing season.

```
title 'Methane Field Study 2015 - Joshua Humphreys';
title2 'Seasonal Methane 2015 ANOVA';
data methane2015;
  infile 'Post2015.prn' firstobs=2;
  input id $ cultivar $ treatment $ full post yield intensity ;
 run;

proc sort data=methane2015; by plot block cultivar;
quit;

proc print data=methane2015 noobs; by plot;
  id plot;
  var full post yield intensity ;
  run;

proc mixed data=methane2015 method=type3;
class cultivar treatment ;
model full = cultivar treatment cultivar*treatment ;
random cultivar;
quit;
```

Example SAS program for evaluating biomass for the 2015 growing season.

```
Title 'Methane Field Study - Biomass 2015 - Joshua Humphreys';
data Bio2015;
  infile 'Biomass2015.prn' firstobs=2;
  input block chamber $ cultivar $ treatment $ bio ;
 run;

proc sort data=bio; by cultivar treatment block;
quit;

proc print data=bio noobs; by cultivar ;
  id ;
  var  chamber treatment block bio ;
  run;
quit;

proc mixed data=bio method=type3;
class    block cultivar treatment ;
model bio = cultivar treatment cultivar*treatment  ;
random  block block*cultivar ;
quit;
```

Example of a SAS program evaluating redox potential among water management treatment and cultivar for the 2015 growing season.

```
title 'Methane Field Study - ORP 2015 - Joshua Humphreys';
title2 'ORP 2015 ANOVA';
data ORP;
  infile 'ORP2015.prn' firstobs=2 ;
  input Block DAF cultivar $ treatment $ MV ;
 run;

proc sort data=ORP; by Treatment Block;
quit;

proc print data=ORP noobs; by Treatment;
    var  Block DAF cultivar Treatment MV ;
  run;
quit;

proc mixed data=ORP ;
class      DAF cultivar treatment ;
model MV =   DAF treatment cultivar cultivar*treatment cultivar* DAF treatment* DAF
cultivar*treatment* DAF   ;
random Block Block*treatment ;
quit;
```

CHAPTER FOUR

Methane emissions from rice production across a soil organic matter concentration gradient from direct-seeded, delayed-flood rice production in Arkansas

Abstract

Quantifying greenhouse gas (GHG) emissions in agricultural settings has become critically important in determining the magnitude of agricultural impacts on global climate change. Methane (CH_4) is a leading GHG emitted from rice (*Oryza sativa* L.) production and, since soil organic matter (SOM) serves as a substrate for methanogenesis, understanding the relationship between SOM and CH_4 emissions will be essential for attenuating the excessive release of CH_4 to the atmosphere from rice production. The objective of this field study was to evaluate the effects of SOM on CH_4 emissions from rice grown under a full-season flood across several silt-loam soils in eastern Arkansas. Eight soils were collected from various locations around east-central Arkansas to represent a SOM gradient (22.1 to 51.0 Mg ha^{-1} in the top 10 cm) for this field study. Approximately 0.08 m^3 of soil were placed in plastic tubs that were buried in a single bay with the pure-line rice cultivar ('LaKast') transplanted into each tub and grown to harvest maturity under a full-season flood. Season-long, area-scaled CH_4 emissions ranged from 63 to 1521 kg CH_4-C ha^{-1} $season^{-1}$ for in-situ, field-plot soil and native prairie soil, respectively, and differed ($P < 0.01$) among soil treatments. Rice grown in soil from under a managed grassland, which had the largest SOM content in the top 10 cm, produced the second largest CH_4 emissions (1166 kg CH_4-C ha^{-1} $season^{-1}$). Methane emissions increased linearly ($P < 0.05$) with increasing SOM and total carbon concentration (R^2 = 0.81 and 0.85, respectively). Greater understanding of the influence of SOM on CH_4 emissions is essential for assessing GHG impacts from rice production and for refining predictions of CH_4 and total GHGs emissions.

Introduction

Anthropogenically induced climate change may be the greatest environmental challenge humans will collectively face in the next 100 years (IPCC, 2014). As rainfall patterns change, global temperatures increase, and human populations rise, increasing food production via improved soil health and water resource management will become paramount for sustainable resource use and continued survival (IPCC, 2014). Natural resource management tools are needed in agricultural production to not only increase yield, but reduce climate-change drivers, such as greenhouse gas (GHG) emissions.

The main anthropogenically and naturally produced GHGs are carbon dioxide (CO_2), nitrous oxide (N_2O), and methane (CH_4), all of which have experienced increased atmospheric concentrations to unprecedented levels not observed for 800,000 years (IPCC, 2014). A common baseline for GHG concentrations is to use the pre-Industrial Revolution atmospheric GHG concentrations, which were 280 mg L^{-1} for CO_2, 0.7 mg L^{-1} for CH_4, and 0.18 to 0.26 mg L^{-1} for N_2O (Forster et al., 2007). However, by 2005, these same GHGs had mean atmospheric concentrations of 379 mg L^{-1} for CO_2, 1.8 mg L^{-1} for CH_4, and 0.32 mg L^{-1} for N_2O (Forster et al., 2007). Cumulative CO_2 emissions from 1750 to 2012 were 2040 ± 310 Gt CO_2, with ~ 40% of those emissions remaining in the atmosphere, ~ 30% being absorbed by the oceans, and the remaining ~ 30% being sequestered in plants and soils (IPCC, 2014). Total US GHG emissions increased by 8.4% from 1990 to 2010, with a 1.6% decrease from 2010 to 2011 to 108 Tg of CO_2 equivalents (IPCC, 2014). Total GHG emissions peaked in the US during 2007 at 7263 Tg of CO_2 equivalents.

Methane is a potent GHG that is produced in saturated- and/or flooded-soil conditions, due to the absence of oxygen (i.e., anoxic or anaerobic conditions), as a byproduct of chemical carbon (C) reduction (Ferry, 1992). During C reduction, C in soil organic matter (SOM) is converted to CH_4 by methanogens. Methanogens use fermentation products, such as acetic acid and CO_2, which are produced by other soil microbes, as food and an energy source, where CH_4 is produced as a by-product of the reactions. Since agriculture is responsible for 10 to 12% of total global anthropogenic GHG emissions, accounting for nearly 50% of global CH_4 emissions alone (Smith et al., 2007), mitigation of CH_4 production and release in agricultural settings, particularly in areas of concentrated rice (*Oryza sativa* L.) production, will have profound importance for future resource sustainability.

The main agricultural sources of CH_4 emissions in the US are enteric fermentation and manure management, with over 95% of total agriculturally related CH_4 emissions as of 2012 (IPCC, 2014). The natural sources of CH_4 emissions are wetlands (23%), plants (6%), termites (3%), oceans (3%), and gas hydrates (2%) (Conrad, 2009). The anthropogenically influenced sources of CH_4 emissions are ruminants (17%), rice fields (10%), landfills (7%), biomass burning (7%), and sewage treatment (4%), while the remaining 18% is attributed to fossil fuel burning (Conrad, 2009). As of 2011, total CH_4 emissions from rice production represented 1.1% of the total US budget of CH_4 emissions to the atmosphere (IPCC, 2014). However, rice cultivation and residue burning make up 3.7% of the total agricultural CH_4 releases (IPCC, 2014). Between 1990 and 2014, annual CH_4 emissions from rice cultivation varied between 575 and 476 kT (kilotons), whereas CH_4 emissions in 2015 alone represented a 30% decrease compared to 1990 emissions (USEPA, 2017). In 2015, total estimated CH_4 emissions from rice production were 11.2 MMT (million megatons) of CO_2 equivalents in the US (USEPA, 2017). This substantial amount of GHG production substantiates further examination into mitigation of GHG emissions, particularly for CH_4, from rice production must be examined.

The importance of quantifying the impact of CH_4 emissions from rice production cannot be overstated. Rice is the predominant staple food for 17 countries in Asia and the Pacific, nine countries in North and South America, and eight countries in Africa (FAO, 2004). Compared to other cultivated grain crops, rice is unique in that the majority of global rice production, and most of the rice production in the US, occurs under flooded-soil conditions for most of the growing season. In the US, CH_4 emissions from agricultural sources are closely tied to the regional geographic distribution of where rice production occurs, whereas Arkansas, California, Louisiana, and Missouri were the top four rice-producing states in the US in 2015 (NASS, 2016). Based on rice yields, Arkansas produced an estimated 3.8 MMT CO_2 equivalents in 2015 from rice cultivation alone (USEPA, 2017).

Rice grown in the direct-seeded, delayed-flood rice production system common in Arkansas also differs substantially from more traditional rice production systems, where rice is hand-transplanted directly to a flooded field. The direct-seeded, delayed-flood rice production system initiates the flood on a rice field four to six weeks after planting, thus limiting the time the flood is present over the entire growing season, which is unlike the water-seeded or hand-transplanted rice systems where flooded soil conditions persist nearly year-round. In Arkansas, the direct-seeded, delayed-flood rice production system is the dominate system, accounting for ~ 85% of the total planted-rice area, for which annual irrigation water use averaged 763 mm over a 10-yr period between 2003 to 2012 (Henry et al., 2016). In Arkansas, a unique and main irrigation standard is the use of the multiple-inlet irrigation, which uses poly-tubing as a means of irrigating rice to conserve water and labor (Henry et al., 2016). As of 2015, rice producers utilized multiple-inlet irrigation on ~ 41% of the rice area in Arkansas (Hardke, 2016).

Groundwater is used to irrigate over 74% of the rice area in Arkansas, with the remaining area irrigated with surface water obtained from reservoirs, streams, or bayous (Hardke, 2016).

In the soil environment, whether undisturbed or agricultural, the main source of CH_4 in the soil column is in the topsoil, where > 99% of the total soil-produced CH_4 is emitted (Mitra et al., 2002b). Due to the flooded-soil nature associated with rice production, CH_4 diffusion through the water column is generally slow. However, the main release mechanism of CH_4 from rice cultivation to the atmosphere from below a column of water has been via passive transport through the aerenchyma tissue of the rice plants themselves (Cicerone and Shetter, 1981; Yu et al., 1997; Dannenburg and Conrad, 1999; Groot et al., 2005). Since soil microbes in an anaerobic setting eventually require C as an electron acceptor to carry out metabolic processes, increasing the supply of SOM would likely increase microbial activity and therefore CH_4 production. However, single or multiple soil property, particularly SOM, correlations with CH_4 emissions have been inconclusive (Wang et al., 1993; Watanabe and Kimura, 1999).

The challenges of population growth require a clear understanding of soil conditions and management practices so that innovative techniques can be developed and implemented to off-set potential negative agronomic and ecological/environmental effects of climate change. The pressure to expand production into previously uncultivated land is tremendous. Previously uncultivated land has the temptations of increased soil fertility leading to greater yields, but also potentially negative environmental drawbacks. One of these potential drawbacks is increased CH_4 production from rice production due to large initial SOM.

Since CH_4 can only be produced if there is a source of reducible C in the soil, it stands to reason that soils with a greater initial SOM concentration would produce greater amounts of CH_4 in the flooded-soil condition associated with rice production (Ferry, 1992). However, to date, this relationship has not been well demonstrated. Therefore, the objective of this field study was to evaluate the effect of SOM on season-long CH_4 emissions from a pure-line cultivar planted in numerous silt-loam soils and grown under a full-season flood in eastern Arkansas. It was hypothesized that CH_4 emissions would vary among soils with differing initial SOM concentrations and, specifically, CH_4 emissions would increase linearly as SOM concentration increased. It was also hypothesized that the emissions intensity (kg CH_4-C (Mg grain)$^{-1}$) would be inversely related to SOM concentration.

Materials and Methods

Site Description

Field research was conducted in 2016 at the University of Arkansas System Division of Agriculture's Rice Research and Extension Center (RREC) near Stuttgart, AR (34°27'54.5" N, 91°25'8.6" W) and closely followed procedures outlined in Rogers et al. (2014). The RREC is located in a region in east-central AR known as the Grand Prairie, which is part of the Major Land Resource Area 131D, Southern Mississippi River Terraces, within Arkansas County (USDA, 2006). The study area has been managed in a rice-soybean (*Glycine max* L. [Merr.]) rotation, which is a commonly used rotation for rice production in Arkansas, for more than 25 years. The slope across the study area was approximately 0.2% to facilitate irrigation water application and removal. The regional climate throughout the study area is temperate, with a mean annual air temperature of 17°C, which ranges from a mean minimum of 12.7°C in January to a mean maximum of 23.5°C in July (NOAA, 2015). The mean annual precipitation for the study area is 135 cm (NOAA, 2015).

Field Treatments and Establishment

Field treatments for this study consisted of eight soils collected from various locations from the agricultural region of east-central Arkansas that established a SOM concentration gradient. Two of the eight soils were collected from the University of Arkansas System Division of Agriculture's Pine Tree Research Station (PTRS; 35° 22.1" N, 90° 55' 45.2" W) in St. Francis County near Colt, AR. One soil was from a Calhoun silt loam (fine-silty, mixed, active, thermic Typic Glossaqualfs) under cultivated agriculture (CA) in a rice-soybean rotation (CA-PT). The second soil was collected from a Henry silt loam (coarse-silty, mixed, active, thermic Typic Fragiaqualfs) under Conservation Reserve Program (CRP) managed grassland landuse that had not been used for cultivated agriculture for at least 15 years. The dominant vegetation in the CRP field was big bluestem (*Andropogon gerardi*) and little bluestem (*Schizachyrium scoparium*). Four of the eight soils were collected from a private farmstead (i.e., the Seidenstricker Farm) (34° 43' 40.26" N, 91° 33' 10.76" W) north of Stuttgart, AR, where one soil was a DeWitt silt loam (fine, smectitic, thermic Typic Albaqualfs; USDA, 2015) under native tallgrass prairie (NP) landuse, which had been subject to periodic annual burning, while the other three soils were collected from agricultural landuse immediately adjacent to the native prairie that had been under continuous annual cultivation in a rice-wheat (*Triticum aestivum*)-soybean rotation for 30 (CA-30; DeWitt silt loam), 41 [CA-41; Stuttgart silt loam (fine, smectitic, Albaquultic Hapludalfs)], and 59 (CA-59; Stuttgart silt loam) years. The remaining two of eight soils were collected from the RREC, where one soil had been under cultivated agriculture in a rice-soybean rotation for at least 25 years (CA-25; DeWitt silt loam), while the other soil was from a managed grassland (MG; DeWitt silt loam) mix of fescue (*Festuca* spp.) and Bermudagrass (*Cynodon dactylon*) (i.e., a manicured lawn). Table 1 summarizes additional characteristics of the eight soils and the sites from which the soils were collected.

Between 18 March and 6 May 2016, soils were collected from each site. At each site, soil was manually excavated to a depth of ~ 50 cm. First, the upper ~ 20 cm of soil were removed and temporarily set aside on a tarp, while the remaining sub-soil, ~ 20- to 45-cm depth interval, was manually excavated and placed into a 33-cm wide × 60.7-cm long × 42.6-cm deep, high-density, commercially available plastic bin. Once the sub-soil was in place in the plastic bin, which occupied the bottom ~ 20 cm of the bin, the upper 20 cm of topsoil was placed in the bin on top of the sub-soil to recreate the original soil profile horizon sequence as best as possible. Each of the eight soils collected from the various sites were collected in triplicate for a total of 24 bins.

All soil-containing bins were transported to the RREC and, on 7 May 2016, the bins were randomly placed within two, 5-m wide × 3-m long areas adjacent to one another that were manually excavated to a depth of ~ 40 cm. Once all 24 bins had been placed in one of the two excavated areas, soil was manually back-filled around the bins to bury them such that the soil level inside the bins was at the approximate level of the surrounding natural soil. After back-filling soil around the bins, the top ~ 10 cm of the soil surface in each bin was manually disturbed to simulate tillage by breaking up large clods to create a semi-smooth, uniformly appearing, level seed bed into which rice seedlings would be transplanted.

On 20 May, 2016, ~ 10-cm-tall rice seedlings, which had 4 to 5 leaves, from a nearby area, which had been drill-seeded with the pure-line rice cultivar ‘LaKast’ on 23 April, 2016, were manually transplanted 2- to 4-cm deep into two rows 18-cm apart in each bin to match the planting density in the surrounding drill-seeded area, which was approximately 320 plants m^{-2}.

On 8 June 2016, the transplanted rice plants in the bins were fertilized with a manually broadcast, pre-flood, optimum recommended rate of 117 kg N ha^{-1} as urea (46% N) [i.e., 5.77 g surface-applied urea per bin]. Since the rice seedlings had already been pre-flood fertilized once before transplanting, the pre-flood N application was used to offset the transplant shock to the rice plants.

A levee that had been previously established around the buried-bin area contained the permanent full-season flood that was established immediately after N fertilization on 9 June 2016 and was maintained at a depth of ~ 10 cm until harvest maturity. On 27 June 2016, 18 days after flood establishment, the mid-season, split N application of 117 kg N ha^{-1} was manually broadcast-applied [i.e., 5.77 g surface-applied urea per bin] to the floodwater at the beginning of internode elongation.

In addition to the 24 buried bins containing transplanted rice, four field plots were established adjacent to the buried-bin area in the same full-season-flood bay to evaluate the effect of growing transplanted rice in the bins compared to direct, drilled-seeded rice into native soil in typical field plots. Field plots were 1.6-m wide by 5-m long, with nine drill-seeded rice rows with 18-cm row spacing. Field plots were planted with the pure-line rice cultivar 'LaKast' on 23 April 2016. Similar to the transplanted bins, on 8 June 2016, field plots were manually broadcast-fertilized pre-flood at a rate of 117 kg N ha^{-1} as urea. On 27 June 2016, 18 days after flood establishment, 45 kg N ha^{-1} were manually broadcast-applied to the floodwater for the mid-season, split N application. The field plots were not provided with any extra nitrogen since rice in the field plots was not transplanted. On 23 August 2016, the flood was released from the bay containing the 24 buried bins with transplanted rice and the four field plots to prepare for harvest.

Soil Sample Collection, Processing, and Analyses

Prior to flood establishment, on 28 May, 2016, two soil cores, 4.8 cm in diameter, were collected with a core chamber and slidehammer from the top 10 cm in each bin and field plot for soil property analyses. All soil samples were dried at 70°C for 72 h, crushed, and sieved through a 2-mm mesh screen. One set of soil samples per bin/plot was used for and particle-size analyses using a modified 12-hr hydrometer method (Gee and Or, 2002).

The second set of soil samples was used for soil chemical property determinations. Electrical conductivity (EC) and soil pH were analyzed potentiometrically in a 1:2 (m/v) soil-water suspension. Soil organic matter concentration was determined by weight-loss-on-ignition after 2 h at 360 °C. Inductively coupled, argon-plasma, atomic emissions spectrometry (Spectro Arcos, Spectro Analytical Instruments, Kleve, Germany) was used to determine Mehlich-3 extractable nutrient (i.e., P, K, Ca, Mg, S, Na, Fe, Mn, Zn, and Cu) concentrations using a 1:10 soil-mass-to-extractant-volume ratio (Tucker, 1992). Total nitrogen (TN) and total carbon (TC) concentrations were measured by high-temperature combustion with a VarioMax C:N analyzer (Elementar Americas, Inc., Mt. Laurel, NJ). Since all soils did not effervesce upon treatment with dilute hydrochloric acid, all measured TC was assumed to be organic C. Measured TN and TC concentrations were used to calculate soil C:N ratios on a bin-by-bin or plot-by-plot basis. Based on measured sand and clay fractions and SOM concentrations from the top 10 cm of each bin/plot, soil bulk densities on a bin-by-bin and plot-by-plot basis were estimated from generalized multiple regression equations (Saxton et al., 1986). All measured soil concentrations (mg kg^{-1}) were converted to contents (kg or Mg ha^{-1}) using the estimated bulk densities and 10-cm sampling interval.

Soil Oxidation-Reduction Potential and Temperature Measurements

Immediately after flooding of the bay containing the 24 buried bins and four field plots began (9 June, 2016), soil oxidation-reduction (redox) potential (Eh) sensors (Model S650KD-OR, Sensorex, Garden Grove, CA), with Ag/AgCl reference solution, were installed vertically to a depth of ~ 7 cm. One Eh sensor was installed adjacent to a gas-sampling-chamber base collar, described below, in each plot and randomly in two of the three bin replications per soil treatment. In addition to the Eh sensors, chromel-constantan thermocouples were installed horizontally in the bulk soil at a depth of ~ 7 cm in each plot and in the one remaining bin replication per soil treatment. All sensors were connected to a datalogger (CR 1000, Campbell Scientific, Inc., Logan, UT), which was housed in an environmental enclosure, to record soil temperature and soil Eh at 15-minute intervals, while mean data were output every hour. Measured sensor data were collected weekly. Soil Eh values were corrected to the standard hydrogen electrode by adding 199 mV to each field-measured value (Patrick et al., 1996).

For the purposes of data reporting, both soil Eh and temperature data from the hour during gas sample collection on each measurement date were extracted from the continuously recorded data series for all replicate sensors. The individual hourly soil Eh and temperature data from each weekly measurement date were subsequently used for statistical analyses.

Gas Sample Collection and Analyses

Similar to procedures used by Rogers et al. (2014), after planting/transplanting and before flooding, a wooden boardwalk system was erected throughout the study area to reduce disturbances to the soil and rice plants and allow easier access to the plots/bins during the growing season for gas sample collection and other plot/bin maintenance and access. The boardwalk was constructed out of 5.1-cm thick x 30.5-cm wide x 3.6-m long pressure-treated, wooden planks set upon 20- x 40-cm concrete blocks before chamber base collar placement in the plots/bins. One chamber base collar, 30-cm in diameter × 30-cm tall, was installed to encompass the third and fourth rice rows in each field plot for gas sampling. One base collar for gas sampling was then set into place in the center of each bin encompassing the majority of both manually transplanted rice rows. Base collars were constructed out of 0.6-cm thick, Schedule 40 polyvinyl chloride (PVC) material and beveled to a 45° angle to facilitate installation. Base collars were inserted ~ 10 cm into the soil so that four 1.25-cm diameter holes 12 cm from the bottom of the base collar were ~ 1 cm above the soil when properly inserted to facilitate flood-water movement into and out of the base collar.

Vented, non-steady-state, non-flow-through chambers (Livingston and Hutchinson, 1995) made out of 30-cm diameter Schedule 40 PVC were used for gas sample acquisition for the purpose of CH_4 flux determinations (Rogers et al., 2014). To prevent convection currents inside the chambers that would dilute the ambient, headspace air during sampling, the holes in the base collars were plugged with gray butyl-rubber septa (Voigt Global, part# 73828A-RB, Lawrence, KS) during sampling after flood release.

Chamber extensions, 40 and 60 cm in length depending on the height of the rice plants at the time of sampling, were used to accommodate rice growth during the season. Reflective aluminum tape (CS Hyde, Mylar metallized tape, Lake Villa, IL) was used to cover chamber extensions to reduce temperature variations inside the chamber during use. Tire inner tube cross sections were cut to an ~ 10-cm width and taped to the bottom of all the extensions to function as a seal between the base collar and the chamber extensions during gas sampling.

Chamber caps were constructed with 10-cm-tall sections of 30-cm-diameter PVC, with a 5-mm-thick sheet of PVC glued to the top and covered with reflective aluminum tape. Tire inner tube cross sections, ~ 10-cm wide, were also taped to the bottom of the caps to serve as a seal between the chamber base collar early in the growing season or upper-most extension later in the season. A 15-cm-long piece of 4.5-mm-inside-diameter (id) copper refrigerator tubing was installed into the side of each cap to maintain atmospheric pressure during gas sampling. On the top of the gas-chamber caps, two 12.5-mm-diameter holes were drilled and plugged with gray butyl-rubber septa for syringe and thermometer insertion. To ensure adequate air mixing in the enclosed gas chamber, a 2.5-cm tall × 2.5-cm wide, 9V-battery-operated, magnetic levitation fan (Sunon Inc., MagLev, Brea, CA) was installed on the underside of the chamber cap and operated for the duration of gas sampling.

The collection of gas samples from the enclosed chambers was achieved using a 20-mL B-D syringe with a removable 0.5-mm diameter × 25-mm long needle (Beckton Dickson and Co., Franklin Lakes, NJ) that was inserted through the gray butyl-rubber septa installed in the chamber cap. After drawing a gas sample from the chamber, the collected sample was immediately injected into a pre-evacuated, 10-mL, crimp-top glass vial (Agilent Technologies, part# 5182-0838, Santa Clara, CA). Gas sampling occurred weekly between flooding and flood release starting 5 d after flooding. On each sample date, gas samples were collected at 20-min intervals for 1 h, after the chamber was capped and sealed (i.e., the 0-, 20-, 40-, and 60-min marks). At the end of the growing season, prior to harvest, gas sampling occurred 1, 5, and 6 d after flood release. Similar to prior studies (Rogers et al., 2013, 2014), all gas sampling started in the morning between 0800 to 0830 hours to minimize temperature fluctuations in the chambers and to maintain continuity with previous research.

During each chamber sampling event, 10-cm soil temperature, relative humidity, ambient air temperature, barometric pressure, and the air temperature inside the chamber were measured. At the end of each gas sampling event, the chamber height to the current water level was recorded so that the interior chamber volume could be accurately calculated. Samples of CH_4 gas standards (i.e., 2, 5, 10, 20, and 50 mg L^{-1}) were collected in the field using a 20-mL B-D syringe with a detachable 0.5-mm-diameter × 25-mm-long needle that was immediately injected into a pre-evacuated, 10-mL, crimp-top glass vial. Immediately prior to field sample analyses, CH_4 gas samples from the same five gas standards were also collected in the laboratory.

Using a flame ionization detector (250°C) equipped with a gas chromatograph (Model 6890-N; Agilent Technologies, Santa Clara, CA), with a 0.53-mm diameter × 30-m HP-Plot-Q capillary column (Agilent Technologies), gas samples were analyzed for CH_4 concentrations within 48 h of collection. Based on procedures described by Rogers et al. (2014), CH_4 fluxes were calculated by linear regression according to changes in concentrations in the chamber headspace over the 60-min sampling interval. To determine the change in concentration over time, measured concentrations (mL L^{-1}; y axis) were regressed against time (min; x axis) of sample extraction (i.e., 0, 20, 40, and 60 min). The slope of the resulting best-fit line was then multiplied by the calculated chamber volume (L) and divided by the inner surface area of the chamber (m^2) resulting in flux units of μL CH_4 m^{-2} min^{-1} (Parkin and Venterea, 2010). The resulting units of the μL CH_4 were then converted using the Ideal Gas Law (PV = nRT) to μmol CH_4, where P was the measured pressure over the 60-min sampling interval in atmospheres (atm), V was the calculated volume of the interior of the chamber (L), n was the number of moles of the gas, R was the gas constant (0.8206 L atm Mol^{-1} K^{-1}), and T was the average measured temperature inside the chamber in Kelvin over the 60-min interval. To convert μmol

CH_4 to the mass of CH_4, the molar mass of CH_4 was then used for a final flux unit of mg CH_4 m^{-2} d^{-1} (Parkin and Venterea, 2010). Season-long emissions were calculated on a chamber-by-chamber basis by linear interpolation among measured fluxes between sample dates.

Plant Sampling and Processing

Eight days after the last gas sampling (6 September 2016), all aboveground biomass was collected from the interior of each base collar. Plants were cut ~2 cm above the soil surface and dried at 55°C for 3 weeks then weighed to determine aboveground dry matter. Yield from the field plot was determined using a research-grade plot combine, at which time a sub-sample of rice grain was obtained to determine harvest grain moisture. The combine yield was corrected to 12% grain moisture for yield-reporting purposes. To obtain grain yields from the bins, the panicles were removed from the aboveground dry matter samples from the bins, manually threshed to separate the grain from the panicles, and weighed. Yield was calculated based on grain mass per collar area. Rice grain yields from the bins were corrected to 12% grain moisture. Total season-long CH_4 emissions were divided by total rice grain yield on a bin-by-bin basis to express emissions on a per-unit-grain-yield basis, which has been used as an emissions intensity metric.

Statistical Analyses

Based on a completely random design with three replications of each treatment combination, a single-factor analysis of variance (ANOVA) was conducted using SAS (version 9.4, SAS Institute, Inc., Cary, NC) to determine the pre-assigned treatment (i.e., soil) effects on initial soil properties (i.e., bulk density; sand, silt, and clay fractions; Mehlich-3 extractable P, K, Ca, Mg, S, Na, Fe, Mn, Zn, and Cu contents; soil pH and EC; SOM, TC, and TN contents; and

C:N ratio) prior to flooding. A separate two-factor ANOVA was conducted using SAS to determine the effects of soil treatment and time (i.e., measurement date), and their interactions on CH_4 fluxes, soil temperature, and soil Eh. A separate single-factor ANOVA was conducted using SAS to determine the effect of soil treatment on rice yield and season-long area- and yield-scaled CH_4 emissions. All ANOVAs were conducted using the PROC MIXED procedure. When appropriate, means were separated by least significant difference (LSD) at the 0.05 level. Correlation and regression analyses were performed among TC, SOM, sand, silt, and clay and area- and yield-scaled CH_4 emissions using Minitab (ver. 13.31, Minitab Inc., State College, PA).

Results and Discussion

Initial Soil Properties

With the exception of soil EC, all other initial soil properties in the top 10 cm prior to flooding differed ($P < 0.02$) among soil treatments (Table 2). Soil particle-size distributions in the top 10 cm ranged from 0.13 to 0.24 g g^{-1} for sand, from 0.67 to 0.73 g g^{-1} for silt, and from 0.07 to 0.13 g g^{-1} for clay, where each differed ($P \leq 0.01$) somewhat among soil treatments (Table 2). However, particle-size analyses confirmed all soil treatments had a silt-loam texture. Soil pH was mostly alkaline (pH = 7.3) in the 41- and 59-yr-old conventionally tillage agricultural soils, which did not differ, presumably due to the longest history of periodic liming and was mostly acidic (pH = 4.8) in the managed grassland and native prairie soils, which did not differ, presumably due the longest period of undisturbed weathering (Table 2). The recommended soil pH for rice production is between 6.0 and 6.5 (Norman et al. 2013). However, no pH adjustments were made to any soil treatment. Bulk density was similar and largest across all five cultivated agricultural soils put in bins, which averaged 1.46 g cm^{-3}, and was smallest (1.21 g cm^{-3}) in the managed grassland soil (Table 2). Soil organic matter content was greatest in the managed grassland soil (51.0 Mg ha^{-1}) and lowest in the 30-, 41-, and 59-r-old cultivated agricultural soils placed in bins and the field-plot soil, which did not differ and averaged 24.5 Mg ha^{-1} (Table 2). Total C, which was considered all organic C, was greatest in the managed grassland soil (24.8 Mg ha^{-1}) and lowest in the 30-yr-old cultivated agricultural soil placed in bins (7.2 Mg ha^{-1}; Table 2). Mehlich-3 extractable soil nutrients ranged from a low of 2.1 times different for K to 5.1 times different for Mg across all soil treatments (Table 2). Soil K tested low for the 25-yr-old cultivated agriculture and field plot (140 and 136 kg ha^{-1}, respectively) with all other soil treatments in the very low category, with the lowest soil-test K from the native prairie at 67.5 kg ha^{-1}. Soil P was in the optimum range for rice production for the 41-yr-old cultivated agriculture (78.5 kg ha^{-1}), in the medium range for cultivated agriculture -25 and -59 and in the low soil P range for all other soil treatments, with the lowest of 19.9 kg ha^{-1} for the CRP soil (Norman et al., 2013). No soil amendments were added to correct for any deficiencies. Soil EC averaged 292 dS m^{-1} across all soil treatments. The measured differences among soil treatments were expected, as soils were specifically chosen from various locations and under various landuses to establish a SOM and/or TC gradient for evaluation of SOM/TC concentration effects on CH_4 fluxes and emissions.

Methane Fluxes

During the 2016 rice growing season, CH_4 fluxes followed a predictable temporal pattern, which was similar to previous observations from silt-loam and clay soils in Arkansas (Brye et al., 2013; Rogers et al., 2014; Linquist et al., 2015; Smartt et al., 2016b). Methane fluxes started low, increased to numeric peaks that ranged from 232 to 3815 mg CH_4-C m^{-2} d^{-1} between 39 and 53 DAF, which was approximately 50% heading, for the field-plot and native prairie soil, respectively, and decreased thereafter until the flood was released at 75 DAF (Figure 1). After flood release, CH_4 fluxes in all treatment combinations at least slightly numerically increased within 6 days before decreasing to near zero by 81 DAF (Figure 1). The occurrence of a post-flood-release increase in CH_4 fluxes has been measured numerous times in both clay and silt-loam soils in Arkansas (Brye et al., 2013; Rogers et al., 2014; Linquist et al., 2015; Adviento-Borbe and Linquist, 2016; Smartt et al., 2016b).

During the 2016 growing season, CH_4 fluxes differed ($P < 0.01$; Table 3) among soil treatments over time (Figure 1). Due to large overall measured variability among all soil treatment, CH_4 fluxes from the CA-30 soil treatment did not differ from a flux of zero on any measurement date throughout the entire growing season. Methane fluxes measured at 5 DAF from all soil treatments did not differ from a flux of zero. At 12 DAF, five soil treatments did not different from a flux of zero (CA-PT, CA-25, CA-41, CA-59, and CA-30), while CH_4 fluxes from the CRP, NP, and MG soil were all greater than a flux of zero. The mean CH_4 flux was larger from the MG than from the NP and CRP soils, which did not differ, at 12 DAF (Figure 1), presumably due to their large concentration of readily reducible C. At 19 DAF, CH_4 fluxes from four soil treatments (CA-PT, CA-41, CA-59, and CA-30) did not differ from a flux of zero; however, the NP soil had a larger CH_4 flux than the MG soil, while CH_4 fluxes from both the NP and MG soils were greater than fluxes from the CRP and CA-25 soil treatments, which did not differ. At 27 and 32 DAF, CH_4 fluxes from three treatments (CA-41, CA-59, and CA-30) did not differ from a flux of zero, while CH_4 fluxes from the remaining soil treatments ranked, from greatest to smallest, as follows: NP > MG > CRP > CA-25 = CA-PT. At 39 DAF, CH_4 fluxes from two treatments (CA-59 and CA-30) did not differ from a flux of zero, while CH_4 fluxes from the remaining soil treatments ranked, from greatest to smallest, as follows: NP > MG > CRP > CA-PT = CA-25 > CA-41. At 47 DAF, CH_4 fluxes from one treatment (CA-30) did not differ from a flux of zero, while CH_4 fluxes from the remaining soil treatments ranked, from greatest to smallest, as follows: NP > MG > CRP = CA-PT = CA-25 > CA-41 = CA-59.

Between 39 and 53 DAF, CH_4 fluxes numerically peaked for all soil treatments, with the largest numeric peak flux from the NP (3815 mg CH_4-C m^{-2} d^{-1}) and the smallest numeric peak flux from the CA-59 (352 mg CH_4-C m^{-2} d^{-1}) soil at 47 DAF. The CH_4 flux from the CA-59 soil treatment was similar to that reported by Rogers et al. (2014), where CH_4 fluxed from the hybrid CLXL745 ranged from 199 to 448 mg CH_4-C m^{-2} d^{-1}. The peak fluxes reported by Rogers et al. (2014) were substantially lower than the peak fluxes measured from the NP or the MG (2730 mg CH_4-C m^{-2} d^{-1}) treatments, which were five to seven times more than peak fluxes reported by Rogers et al. (2013) (542 mg CH_4-C m^{-2} d^{-1}) on a similar silt-loam soil under Arkansas rice production practices. This dramatic difference in peak CH_4 fluxes is likely due to the native prairie and managed grassland never being under cultivation and their greater concentration of readily reducible C substrate. At 53 DAF, CH_4 fluxes from three treatments (CA-59, CA-41, and CA-30) did not differ from a flux of zero, while CH_4 fluxes from the remaining soil treatments ranked, from greatest to smallest, as follows: NP > MG > CRP = CA-PT = CA-25. At 61 and 74 DAF, CH_4 fluxes from three treatments (CA-59, CA-41, and CA-30) did not differ from a flux of zero, while CH_4 fluxes from the remaining soil treatments ranked, from greatest to smallest, as follows: NP = MG > CRP = CA-PT = CA-25. At 76 DAF, which was one day after flood release and similar to 51 and 74 DAF, CH_4 fluxes from three treatments (CA-59, CA-41, and CA-30) did not differ from a flux of zero, while CH_4 fluxes from the remaining soil treatments ranked, from greatest to smallest, as follows: NP = MG > CRP = CA-PT = CA-25. At 80 DAF, five days after flood release, CH_4 fluxes from the same three treatments (CA-59, CA-41, and CA-30) did not differ from a flux of zero, while CH_4 fluxes from the remaining soil treatments ranked, from greatest to smallest, as follows: MG = NP and NP = CRP > CA-PT = CA-25. At 81 DAF, six days after flood release, CH_4 fluxes from four treatments (CA-59, CA-41, CA-30, and CA-25) did not differ from a flux of zero, while CH_4 fluxes from the remaining soil treatments ranked, from greatest to smallest, as follows: NP > MG > CRP > CA-PT. The numeric, post-flood-release increases in CH_4 fluxes were consistent with previous observations (Brye et al., 2013; Rogers et al., 2014; Linquist et al., 2015), however not all fluxes returned to zero by the end of the sampling six days after flood release on 81 DAF due to the bins not being fully drained of water.

Soil Temperature and Redox Potential Fluctuations

At the time of flood establishment, soil temperatures at the 7-cm depth averaged 25°C across all treatments, then increased to the growing-season maximum of 28.6°C by 6 weeks after flooding (39 DAF), and remained relatively uniform thereafter (Figure 2). The soil temperature variations measured in this study were similar to those reported by Rogers et al. (2013), where a maximum, 7-cm soil temperature of 32°C occurred at 19 DAF.

As expected, soil Eh started well-oxidized and decreased ($P < 0.05$) thereafter following flood establishment (Figure 2). Averaged over soil treatments, soil Eh steadily declined to 39 DAF, then stabilized at around -200 mV. In a similar study on a silt-loam soil in east-central Arkansas, Rogers et al. (2013) reported soil Eh rapidly decreased to < -200 mV by 25 to 30 DAF in a full-season-flood treatment, where a soil Eh of approximately -200 mV is necessary for maximum CH_4 production (Reddy and DeLaune, 2008). Averaged across time, soil Eh was greater (i.e., more oxidized; $P < 0.01$) in the CA-25 soil (131 mV) than that in the CA-41, CA-30, CRP, and NP soils, which did not differ and averaged -169.5 mV (Table 4). The soil treatment differences in Eh correspond well to the measures differences in CH_4 flux trends and peak-flux rankings (Figure 1).

Area-scaled Methane Emissions

During the 2016 growing season, season-long, area-scaled emissions ranged from 63 kg CH_4-C ha^{-1} from the in-situ field-plot soil to 1521 kg CH_4-C ha^{-1} from the native prairie soil contained in the bins (Table 4). This 24-fold difference in season-long, area-scaled emission is likely the result of a larger SOM pool in the native prairie soil and/or from the disturbance of the soil from transporting. Similar to that hypothesized, season-long CH_4 emissions differed among soil treatments ($P < 0.01$; Table 3). Season-long CH_4 emissions were greatest from the non-agricultural soils, which also had the greatest SOM and total C contents in the top 10 cm, compared to the current agricultural soils. Season-long CH_4 emissions were 1.3 times greater from the native prairie (1521 kg CH_4-C ha^{-1}) than that from the managed grassland soil (1166 kg CH_4-C ha^{-1}), while both of which were greater than all other agricultural soils (Table 4). The large CH_4 emissions from these two soils (NP and MG) was likely the result of larger SOM contents (45.8 and 51.0 Mg ha^{-1}, respectively) in both soils compared to the other soils for which the lowest average SOM content was 22.1 Mg ha^{-1} for the in-situ field plot soil treatment. Season-long emissions from the managed grassland were 1.9 times greater than that from the CRP soil. Season-long CH_4 emissions from the CRP were 1.8 times greater than that from the cultivated agricultural soil from PTRS (CA-PT) and the 30-yr-old cultivated agricultural soil

(CA-30), which did not differ and averaged 352 kg CH_4-C ha^{-1}. Season-long CH_4 emissions from the CA-PT and the CA-30 soils were 2.4 times greater than that from the 25- (CA-25), 41- (CA-41), and 59-yr-old cultivated agricultural (CA-59) soils, which did not differ and averaged 148 kg CH_4-C ha^{-1}. Season-long emissions were lowest among all treatments from the in-situ field-plot soil (Table 4), which also had low SOM and total C contents in the top 10 cm (Table 2). Season-long CH_4 emissions results generally support the expected variations in CH_4 emissions from the differences in SOM contents, where the larger the SOM content in the top 10 cm, the greater the season-long CH_4 emissions.

Similar to the in-situ field-plot and the 30-yr-old cultivated agricultural soil (CA-30) evaluated in this study, Simmonds et al. (2015) reported season-long, area-scaled CH_4 emissions from a silt-loam soil near Stuttgart, AR 56, 77, 72, and 75 kg CH_4-C ha^{-1} $season^{-1}$ from the hybrid cultivar 'CLXL745' and pure-line cultivars 'Francis', 'Jupiter', and 'Sabine', respectively, grown under a continuous, full-season flood. In addition, Rogers et al. (2013) reported total season-long, area-scaled CH_4 emissions from a full-season-flood on a silt-loam similar near Stuttgart, AR ranged from 54 kg CH_4-C ha^{-1} $season^{-1}$ from N-fertilized bare soil to 220 kg CH_4-C ha^{-1} $season^{-1}$ from the optimally N-fertilized, pure-line cultivar 'Wells'. The substantially larger CH_4 emissions measured in the current study from the native prairie and the managed grassland were likely due to the increased amount of readily reducible C substrate compared to the other soil treatments. With a greater amount of readily reducible amount of C substrate there is a greater potential for increased CH_4 emissions given that a soil C electron acceptor is needed for CH_4 production (Ferry, 1992). There are no other studies that report large CH_4 emissions, such as those measured in the current study from the managed grassland and native prairie. Multiple CH_4 emissions studies in the US (Brye et al., 2013; Rogers et al., 2013;

Brye et al., 2016; Smartt et al., 2016a; Smartt et al., 2016b) do not report CH_4 emissions within the same order of magnitude as what was measured in this study with regards to the native prairie (1521 kg CH_4-C ha^{-1}) or the managed grassland (1166 kg CH_4-C ha^{-1}). In addition, when taking into account international studies, CH_4 fluxes from deep-water rice in Thailand were lower and averaged ~ 99 kg CH_4 ha^{-1} $season^{-1}$ and rain-fed systems averaged 52 to 91 kg CH_4 ha^{-1} $season^{-1}$ for wet and dry seasons, respectively (Wassman et al., 2000).

Season-long emissions from the 30-yr-old cultivated agricultural soil placed in the bins was two-fold greater than that from the same in-situ soil left in field plots. The difference in emissions between these two treatments was likely due to the soil disturbance that occurred while preparing the bins, where the additional disturbance was apparently enough of a perturbation to result in more readily reducible soil C in the bins. Consequently, it is likely that the season-long emissions measured from all soil treatments placed in bins were artificially elevated, potentially by a factor of two, compared to what might be expected from the same soil that was left in-situ, cultivated, and cropped to rice under an optimally N-fertilized, full-season flood management system. Regardless of the potential over-estimation of season-long CH_4 emissions, results of this study clearly demonstrate a relationship exists between season-long CH_4 emissions and initial SOM and/or soil C that supplies reducible C substrate for methanogenesis.

Rice Dry Matter and Yields

Rice dry matter ranged from 23.2 to 38.5 Mg ha^{-1} from the in-situ field-plot and managed grassland soil, respectively, while rice yields ranged from 9.7 to 17.5 Mg ha^{-1} from the 25-yr-old cultivated agricultural soil, which was the disturbed-soil counterpart to the in-situ field-plot soil, and managed grassland soil, respectively (Table 4). The large yield produced from the managed grassland soil was out of the range of typical plot-scale yields (Hardke et al., 2016). However, the large rice yield also demonstrates the potential substantial influence that an undisturbed soil can have on plant productivity due to the inherent natural soil fertility associated with non-cultivated grassland soils. Despite the same rice variety being planted and grown in the field plots and in all prepared soil bins, rice dry matter ($P < 0.01$) and rice yields differed ($P < 0.01$) among soil treatments (Table 3). Rice dry matter was more than 25% greater from the managed grassland than from all other currently cultivated agricultural soils. Rice dry matter was the lowest from the in-situ field-plot soil. Similar to dry matter, rice yield was more than 35% greater from the managed grassland than from the 25-, 41-, and 59-yr-old and PT cultivated agricultural and the in-situ field-plot soil. For comparison, based on Arkansas Rice Performance Trials in 2016, yields for rice cultivar 'LaKast' grown under a continuous, full-season-flood regime on a Dewitt silt-loam soil near Stuttgart, AR averaged 9.5 Mg ha^{-1} (Hardke et al., 2016). Consequently, rice growth and productivity from field plots and bins in this study performed reasonably similar to production-scale rice productivity. In contrast to season-long CH_4 emissions, neither rice dry matter nor yield differed between the in-situ field-plot soil and the same soil placed in the prepared bins (Table 4), suggesting that plant growth was similar when rice was grown in the prepared soil bins compared to rice grown under typical conditions in in-situ field-plot soil. Though season-long CH_4 emissions were two times greater from the soil placed in the bins compared to the in-situ field-plot soil, plant-response results demonstrated that preparing small-scale bins to evaluate CH_4 emissions from widely differing soils from various sites at a single location to impose uniform management was a reasonable approach.

Methane Emissions Intensity

Improving CH_4 emissions intensity by reducing CH_4 emissions per unit grain yield produced should be a management goal for rice producers to maintain sustainable rice production and resource use into the future. For the 2016 rice growing season, CH_4 emissions intensity ranged from 5.6 kg CH_4-C (Mg grain)$^{-1}$ in the in-situ field-plot soil to 93.7 kg CH_4-C (Mg grain)$^{-1}$ in the native prairie soil (Table 4). Consequently, as suspected, CH_4 emissions intensity differed ($P < 0.01$; Table 4) among soil treatments. Emissions intensity, where, based on how the calculation was conducted, the larger the value, the larger the intensity, from the native prairie was greater from than that from the managed grassland, which was greater than from the CRP soil (Table 4). Emission intensity was lowest from the 25-, 41-, and 59-yr-old cultivated agricultural soils placed in bins and the in-situ field-plot soil, which did not differ and averaged 12.2 kg CH_4-C (Mg grain)$^{-1}$.

Emissions intensity results from this study for the cultivated agricultural soils placed in bins and for the in-situ field-plot soil were comparable to those from Rogers et al. (2013) and Simmonds et al. (2015), who both measured CH_4 emissions from rice grown under a continuous, full-season flood on a silt-loam soil near Stuttgart, AR. Rogers et al. (2013) reported a CH_4 emissions intensity of 27.6 kg CH_4-C (Mg grain)$^{-1}$ with the pure-line cultivar ‘Wells’, whereas Simmonds et al. (2015) reported an average CH_4 emissions intensity of 6.8 kg CH_4-C Mg grain^{-1} for the hybrid cultivar ‘CLXP4534’ and the pure-line cultivars ‘Francis’, ‘Jupiter’, and ‘Sabine’ averaged 10.9 kg CH_4-C Mg grain^{-1}. The differences in emissions intensity can be attributed to yield differences in the Simmonds et al. (2015) field study, whereas yields were greater for the current field study for the lowest-emitting soil treatments, thus improving/reducing the emissions intensity. The largest-yielding soil treatments in the current study also had CH_4 emissions that were orders of magnitude larger than what has been documented in any study thus far on a silt-loam soil, which greatly increased/worsened the emissions intensity.

Relationship between Soil Properties and CH_4 Emissions

Several measured soil properties in the top 10 cm correlated with CH_4 emissions (Table 5 and 6). Both SOM and TC concentrations were strongly, positively correlated ($r > 0.86$; $P < 0.01$) with season-long CH_4 emissions and emissions intensity (Table 5). However, both SOM and TC contents were unrelated to season-long CH_4 emissions or emissions intensity likely due to the added variation in estimated bulk density as part of the content calculation, but potassium (r = -0.41, $P = 0.05$) and zinc (r = -0.46, $P = 0.01$) contents were both moderately negatively correlated season-long CH_4 emissions and emissions intensity (Table 6). Similar to that hypothesized, the result of the correlations indicates that both CH_4 emissions and emissions intensity increase as SOM or TC concentration increase, which further validates the initial goal of the selected soils representing a gradient of SOM/TC concentration for CH_4 emissions evaluation in this study. However, in laboratory studies in Louisiana, using 16 soils ranging in texture from silt to clay, CH_4 emissions and SOM concentrations in the range of 14 to 23.8 g kg^{-1} were examined (Wang et al., 1993). Wang et al. (1993) reported that no correlation existed between CH_4 emissions and soil properties such as nitrogen, pH, or cation exchange capacity, but there was a significant increase in CH_4 entrapment in soils with large clay contents, which suggested that soil texture plays a vital role in CH_4 emissions (Wang et al., 1993). In a study based in Japan, no correlation was reported between total CH_4 emissions and any single measured soil property, which included amorphous Fe(III), free iron (Fe)(III), easily reducible manganese (Mn), nitrate (NO_3^-), and sulfate (SO_4^{2-}), and reducing agents including total carbon (C), total nitrogen (N), and easily decomposable C (Watanabe and Kimura, 1999).

Soil particle-size fractions were unrelated (P > 0.05) to season-long CH_4 emissions and emissions intensity, likely due to narrow ranges since only silt-loam soils were targeted for evaluation in this study. Other studies have indicated that there are greater CH_4 emissions from silt-loam than from more clayey soils (Brye et al., 2013; Smartt et al., 2016b). With the large and significant linear correlations, the possibility exists to estimate season-long CH_4 emissions and emissions intensity with some confidence from only a few basic measured soil properties from the top 10 cm. Both SOM ($R^2 = 0.81$; $P < 0.01$) and TC ($R^2 = 0.85$; $P < 0.01$) concentrations from the top 10 cm produced strong, positive linear relationships with season-long, area-scaled CH_4 emissions (Figure 3). Similar studies from Brye et al. (2013) and Rogers et al. (2014) have produced season-long CH_4 emissions of 159.6 and 190 kg CH_4-C ha^{-1}, respectively, with corresponding mean TC contents of 8.7 and 11.4 Mg ha^{-1}, respectively.

Potential Agronomic and Environmental Implications

This field experiment provided an opportunity to examine the relationship between CH_4 production and initial SOM and/or TC contents. Results clearly demonstrated that initial SOM/TC content affects CH_4 fluxes and season-long emissions, specifically in that both CH_4 fluxes and season-long emissions tended to be greater when the initial SOM/TC contents were large. The demonstrated relationship has ramifications for SOM conservation, soil C sequestration, climate change, and soil health with regards to rice production in the Lower Mississippi River Valley of the US.

The general decrease in SOM under rice production from extensive tillage also decreases soil's native C and decreases soil tilth and structure, and overall health, which in turn can result in decreased soil fertility (Mosier, 2004). Native, or undisturbed, soils often act as a C sink, sequestering large quantities of C through natural processes (Mosier, 2004). When those soils are disturbed, that sink can transforms into a tremendous C source by way of anaerobic and aerobic microbial activity, adding to the C load to the atmosphere and increasing GHG concentrations. Utilizing soils that have increased SOM/TC for rice production will likely increase CH_4 emissions, as shown with the current research, which may contribute to potential negative effects of global climate change. If rice production is expanded into previously uncultivated land areas, the environmental impact with respect to GHG emissions may be more severe than previously thought. Though it is still unclear due to the lack of research, the full range of potential/expected GHG emissions following the conversion of previously minimally managed, non-agricultural land to intensively managed, row-crop production, the results of the current study provide an initial baseline for what could potentially occur with regards to CH_4 emissions in the short-term following land-use change. As humans become more interested in long-term sustainability and seek ways to mitigate the sources and impacts of climate change, this research can serve as piece of the framework to help determine the C budget needed to reduce the effects of agriculture on climate change.

Conclusions

To the author's knowledge, this was the first field experiment to select, transport, and combine multiple soil treatments from various locations into a single study at one location so that production practices (i.e., planted rice cultivar, N fertilization, water management) and environmental conditions (i.e., precipitation, air temperature variations) could be uniform among treatments, where the main variable was SOM/TC content. Verifying the hypothesis, results of this study showed that CH_4 emissions and emissions intensity were greatly affected by initial SOM/TC content and confirmed a strong, positive relationship between season-long, area-scaled CH_4 emissions and TC and SOM contents in the top 10 cm. Though season-long CH_4 emissions were greater from soil placed in bins than from in-situ field-plot soil, rice dry matter and yields were unaffected, indicating that the bin approach used in this field study was a reasonable methodology for evaluating the effects of initial SOM/TC contents on CH_4 fluxes and emissions across a wide range of initial SOM/TC contents from geographically diverse areas.

Based on increased season-long, area-scaled CH_4 emissions from previously non-agriculturally managed soils, conversion to intensive agricultural practices and rice production may have a large climate-change impact, at least in the short-term. Though the results of this study were based on one growing season of measurements, these results, indicate relatively consistent CH_4 emissions responses from year to year at least partially due to the presence of the flood water for most of the growing season attenuating climate variations and inter-annual differences in growing-season weather conditions. Therefore, it is reasonable to conclude that these results can be extrapolated to future years and similar field conditions, as minor differences in growing-season weather conditions from year to year likely have minimal effect on CH_4 emissions. As efforts increase to mitigate climate change globally and public discourse shifts to align itself with environmental stewardship, one way to forward the goal of reduced GHG emissions from rice production is to be acutely aware of all soil and management practice factors that affect GHG emissions and plan agricultural practices accordingly to minimize the C footprint and maximize future resource sustainability. Rice production must attain a level of sustainability that will aid the goal of feeding an ever-growing human population, and GHG emissions are a key part of this modern puzzle. There is a responsibility to maximize production of staple grains, while bearing in mind that humans must equally protect future generations from the devastating effects of global climate change.

Literature Cited

Adviento-Borbe, M. A., and B. Linquist. 2016. Assessing fertilizer N placement on CH4 and N2O emissions in irrigated rice systems. Geoderma 266:40-45.

Brye, K.R., L.L. Nalley, J.B. Tack, B.L. Dixon, A.P. Barkley, C.W. Rogers, A.D. Smartt, R.J. Norman, and K.S.V. Jagadish. 2016. Factors affecting methane emissions from rice production in the Lower Mississippi river valley, USA. Geod. Reg. 7:223-229.

Brye, K.R., C.W. Rogers, A.D. Smartt, and R.J. Norman. 2013. Soil texture effects on methane emissions from direct-seeded, delayed-flood rice production in Arkansas. Soil Sci. 178:519-529.

Cicerone, R.J., and J.D. Shetter. 1981. Sources of atmospheric methane: measurements in rice paddies and a discussion. J. Geophys. Res. 86:7203-7209.

Conrad, R. 2009. The global methane cycle: recent advances in understanding the microbial processes involved. Environ. Microbiol. Rep. 1:285-292.

Dannenburg, S., and R. Conrad. 1999. Effect of rice plants on methane production and rhizospheric metabolism in paddy soil. Biogeochem. 45:53-71.

Ferry, J.G. 1992. Methane from acetate. J. Bacteriol. 174:5489-5495.

Food and Agricultural Organization of the United Nations (FAO). 2004. International year of rice, rice is life. Available at: http://www.rice2004.org (accessed 5 Sept. 2018)

Forster, P., V. Ramaswamy, P. Artaxo, T. Berntsen, R. Betts, D.W. Fahey, J. Haywood, J. Lean, D.C. Lowe, G. Myhre, J. Nganga, R. Prinn, G. Raga, M. Schulz, and R. Van Dorland. 2007. Changes in atmospheric constituents and in radiative forcing. Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change [Solomon, S., D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M.Tignor and H.L. Miller (editors.)].Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Gee, G.W., and D. Or. 2002. Particle size analysis. In: J. H. Dane, editor Methods of Soil Analysis: Part 4 Physical Methods. pp. 255-293.

Groot, T. T., P. M. van Bodegom, H. A. J. Meijer, and F. J. M. Harren. 2005. Gas transport through the root–shoot transition zone of rice tillers. Plant Soil 277:107-116.

Hardke, J. T., K.A.K Moldenhauer, S. Xueyan, Y. Wamishe, R. J. Norman, D.L. Frizzell, E. Castaneda, G. Lee, D. Wisdom, M. Blocker, J. Bulloch, T. Beaty, R. Mazzanti, R. Baker, S. Runsick, M. Duren, Y. Liyew, and J. Chlapecka. 2016. Arkansas rice performance trials: 2014-2016. Misc. Publ. 176.

Hardke, J.T. 2016. Trends in Arkansas rice production. In: R. J. Norman, and K. A. K. Moldenhauer, editors, B.R. Wells Arkansas Rice Research Studies, 2015. Univ. Ark. Agric. Exp. Sta. Res. Ser. 634., Univ. Ark. Agric. Exp. Sta., Fayetteville, AR. pp. 13-23.

Henry, C.G., S.L. Hirsh, M M. Anders, E.D. Vories, M.L. Reba, K.B. Watkins, and J.T. Hardke. 2016. Annual irrigation water use for Arkansas rice production. J. Irrig. Drain. Engin. 05016006:1-5.

Intergovernmental Panel on Climate Change (IPCC). 2014. Climate Change 2014: Synthesis Report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Core Writing Team, R.K. Pachauri and L.A. Meyer (editors.)]. IPCC, Geneva, Switzerland.

Intergovernmental Panel on Climate Change (IPCC). 2014. Intergovernmental Panel on Climate Change (IPCC), United States Greenhouse Gas Inventory Chapter 6. IPCC, Geneva, Switzerland. pp. 15-21.

Linquist, B., M. Anders, M. A. Adviento-Borbe, R. Chaney, L. Nalley, E. da Rosa, and C. van Kessel. 2015. Reducing greenhouse gas emissions, water use, and grain arsenic levels in rice systems. Global Change Biol. 21:407-417.

Livingston, G., and G. Hutchinson. 1995. Enclosure-based measurement of trace gas exchange: Applications and sources of error. In: P. A. Matso, and R. C. Harriss, editors, Biogenic trace gases measuring emissions from soil and water. Blackwell Science, Oxford [England]; Cambridge, MA, USA. pp. 14-51.

Mitra, S., R. Wassman, M. Jain, and H. Patak. 2002b. Properties of rice soils affecting methane production protentials: 2. Differences in topsoil and subsoil. Nutr. Cycl. Agro. 64:183-191.

Mosier, A., R. Wassman, L. Verchot, J. King, and C. Palm. 2004. Methane and nitrigen oxide fluxes in tropical agricultural soils: Sources, sinks, and mechanisms. Environ. Dev. Sustainability 6:11-49.

National Agricultural Statistics Service, (NASS). 2016. Agricultural Statistics 2016. In: United States Department of Agriculture, editor United States Government, United States Government Printing Office Washington, D.C.

National Oceanic & Atmospheric Administration (NOAA). 2015. National Environmental Satellite, Data, and Information Service. Annual Climatological Summary (2015). National Centers for Environmental Information, U.S. Department of Commerce, 151 Patton Avenue, Asheville, NC 2880.

Norman, R.J., N.A. Slaton, and T.L. Roberts. 2013. Soil Fertility. In: J. T. Harke, editor Arkansas Rice Production Handbook Misc. Publ. 192. University of Arkansas Division of Agriculture Cooperative Extension Service, Little Rock, AR, USA. pp. 69-102.

Parkin, T.B., and R.T. Venterea. 2010. Sampling protocols. In: R.F. Follet USDA-ARS GRACEnet Project Protocols, editor Chapter 3. Chamber-Based Trace Gas Flux Measurements. pp. 3-1 to 3-39.

Patrick, W.H., R.P. Gambrell, and S.P. Faulkner. 1996. Redox measurements of soils. In: D. L. Sparks, editor Methods of Soil Analysis. part 3 - Chemical Methods. 3rd ed. SSSA, Madison, WI, USA. pp. 1255-1273.

Reddy, K.R., and R.D. DeLaune. 2008. Biogeochemistry of wetlands: Science and applications. 1st ed. CRC Press, Boca Raton, FL.

Rogers, C.W., K.R. Brye, R.J. Norman, E.E. Gbur, J.D. Mattice, T.B. Parkin, and T.L. Roberts. 2013. Methane emissions from drill-seeded, delayed-flood rice production on a silt-loam soil in Arkansas. J. Environ. Qual. 42:1059-1069.

Rogers, C.W., K.R. Brye, A.D. Smartt, R.J. Norman, E.E. Gbur, and M.A. Evans-White. 2014. Cultivar and previous crop effects on methane emissions from drill-seeded, delayed-flood rice production on a silt-loam soil. Soil Sci. 179:28-36.

Saxton, K.E., W.J. Rawls, J.S. Romberger, and R.I. Papendick. 1986. Estimating generalized soil-water characteristics from texture. Soil Sci. Soc. Am. J. 50:1031-1036.

Simmonds, M.B., M. Anders, M.A. Adviento-Borbe, C. van Kessel, A. McClung, and B.A. Linquist. 2015. Seasonal methane and nitrous oxide emissions of several rice cultivars in direct-seeded systems. J. Environ. Qual. 44:103-114.

Smartt, A.D., K.R. Brye, C.W. Rogers, R.J. Norman, E.E. Gbur, J.T. Hardke, and T.L. Roberts. 2016a. Characterization of methane emissions from rice production on a clay soil in Arkansas. Soil Sci. 181:57-67.

Smartt, A.D., K.R. Brye, C.W. Rogers, R.J. Norman, E.E. Gbur, J.T. Hardke, and T.L. Roberts. 2016b. Previous crop and cultivar effects on methane emissions from drill-seeded, delayed-flood rice grown on a clay soil. Appl. Environ. Soil Sci. 2016:1-13.

Smith, P., D. Martino, Z. Cai, D. Gwary, H. Janzen, P. Kumar, B. McCarl, S. Ogle, F. O'Mara, C. Rice, B. Scholes, and O. Sirotenko. 2007. Agriculture. In Climate Change 2007: Mitigation. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change [B. Metz, O.R. Davidson, P.R. Bosch, R.

Dave, L.A. Meyer (editors)], Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Tucker, M.R. 1992. Determination of phosphorous by Mehlich 3 extraction. In: S. J. Donohue, editor, Soil and media diagnostic procedures for the Southern Region of the United States. Virginia Agric. Exp. Stn. Ser. Bull. 374., Blacksburg, VA, USA. pp. 6-8.

United States Department of Agriculture (USDA) 2006. Major Land Resource Regions Handbook 296. Available at: http://www.cei.psu.edu/mlra, (accessed 10 Jan. 2018).

United States Environmental Protection Agency (USEPA). 2017. United States Environmental Protection Agency (USEPA), Inventory of U.S. greenhouse gas emissions and sinks: 1990-2015. Available at: https://19january2017snapshot.epa.gov/ghgemissions_.html (accessed 10 Jan. 2018).

Wang, Z.P., C.W. Lindau, R.D. DeLaune, and W.H. Patrick. 1993. Methane emission and entrapment in flooded rice soils as affected by soil properties. Biology and Fert. of Soils 16:163-168.

Wassman, R., H. U. Neue, R. S. Lantin, K. Makarim, N. Chareonsilp, L.V. Buendia, and H. Rennenberg. 2000. Characterization of methane emissions from rice fields in Asia. II. Differences among irrigated, rainfed, and deepwater rice. Nutr. Cycl. Agro. 58:13-22.

Watanabe, A., and M. Kimura. 1999. Influence of chemical properties of soils on methane emission from rice paddies. Commun. Soil Sci. Plant Anal. 30:2449-2463. doi:10.1080/00103629909370386.

Yu, K.W., Z. P. Wang, and G. X. Chen. 1997. Nitrous oxide and methane transport though rice plants. Biol. Fert. Soils 24:341-343.

Appendices

Table 1. Summary of landuse, Major Land Resource Area (MLRA) classification, soil series (sub-group taxonomic classification), and unique landuse feature associated with the various soil treatments selected for evaluation in this study.

Landuse (Abbreviation)	MLRA	Soil Series (Soil Sub-group)	Unique Feature
Native prairie (NP)	131D	Dewitt silt loam (Typic Albaqualfs)	Native tall grass prairie
Managed grassland (MG)	131D	Dewitt silt loam (Typic Albaqualfs)	Manicured lawn
Conservation Resource Program (CRP)	134	Henry silt loam (Typic Fragiaqualfs)	> 15 years in CRP
Cultivated agriculture (CA-PT)	134	Calhoun silt loam (Typic Glossaqualfs)	> 10 years in rice/soybean rotation
Cultivated agriculture (CA-25)	131D	Dewitt silt loam (Typic Albaqualfs)	> 25 years in rice/soybean rotation
Cultivated agriculture (CA-30)	131D	Dewitt silt loam (Typic Albaqualfs)	> 30 years in rice/soybean rotation
Cultivated agriculture (CA-41)	131D	Stuttgart silt loam (Albaquultic Hapludalfs)	41 years in rice-wheat-soybean rotation
Cultivated agriculture (CA-59)	131D	Stuttgart silt loam (Albaquultic Hapludalfs)	59 years in rice-wheat-soybean rotation
Cultivated agriculture (FP)	131D	Dewitt silt loam (Typic Albaqualfs)	> 25 years in rice/soybean rotation

Table 2. Analysis of variance summary of the effect of soil treatments on initial soil physical and chemical properties from the top 10 cm prior to flood establishment at the Rice Research and Extension Center near Stuttgart, AR during the 2016 growing season. Means are based on three replications, with the exception of the in-situ field-plot (FP) soil that had four replications.

Soil Property	*P*	NP	MG	CRP	CA-PT	CA-25	CA-30	CA-41	CA-59	FP
Sand (g g^{-1})	< 0.001	0.24a	0.22abc	0.15d	0.13d	0.23ab	0.20bc	0.23a	0.23a	0.20c
Silt (g g^{-1})	< 0.001	0.69c	0.67c	0.73ab	0.73a	0.68c	0.68c	0.70bc	0.67c	0.73a
Clay (g g^{-1})	0.01	0.07c	0.11ab	0.12ab	0.13a	0.09bc	0.12ab	0.07c	0.10bc	0.08c
pH	< 0.001	4.8e	4.8e	5.6d	6.0c	6.1c	6.8b	7.3a	7.3a	6.9b
Bulk density (g cm^{-3})	< 0.001	1.28d	1.21e	1.40bc	1.44ab	1.44ab	1.48a	1.48a	1.47a	1.37c
Extractable nutrients (kg ha^{-1})										
P	< 0 .001	33.2def	35.1def	19.9f	27.6ef	39.1cde	45.8c	78.5a	41.7cd	68.9b
K	< 0 .001	67.5g	87.3fg	98.3de	84.1ef	140.9a	117.8bc	104.8cd	86.0ef	136.0ab
Ca	< 0.001	417.8f	1,063e	1,002d	1,361b	1,163c	1,343b	1,614a	1,370b	1,450b
Mg	< 0.001	67.3g	168.7e	214.2c	341.4a	183.3d	139.7e	233.1b	196.0c	121.0f
S	< 0.001	47.3b	52.8ab	21.4d	19.9d	30.2c	15.9d	51.0a	30.6c	15.1d
Na	< 0.001	21.2e	30.2de	25.4de	41.3c	27.5de	103.5a	51.7b	36.0cd	67.4b
Fe	< 0.001	210.5d	241.8d	230.8d	416.7b	341.5c	312.5c	501.5a	338.8c	489.8ab
Mn	< 0.001	234.7d	296.7bc	289.4a	265.7ab	127.2e	233.8bc	138.1e	203.3d	231.5cd
Zn	< 0.001	1.1d	4.9c	1.4d	2.6cd	4.4c	6.6ab	4.5bc	4.6bc	8.2a
Cu	0.008	1.2d	1.8bc	1.4cd	2.1a	1.7abc	1.7abc	1.5bc	1.8ab	1.7bc
Total N (Mg ha^{-1})	< 0.001	1.57b	2.3a	1.0c	0.9cd	1.1c	0.6d	1.0c	0.83cd	0.66d
Total C (Mg ha^{-1})	< 0 .001	23.7b	24.8a	14.4c	12.5d	13.8cd	7.2g	13.2d	10.91e	8.7f
C:N ratio	< 0.001	15.1a	10.7d	14.9a	14.4ab	13.2bc	12.0cd	13.3bc	13.2bc	13.2bc
Soil organic matter (Mg ha^{-1})	< 0.001	45.8b	51.0a	33.0c	27.9de	29.6cd	22.9f	25.5def	25.1def	22.1f

Means in same row followed by different letters are significantly different ($P < 0.05$) Native prairie (NP), managed grassland (MG), conservation resource program (CRP), cultivated agriculture (CA-25), cultivated agriculture (CA-30), cultivated agriculture (CA-41), cultivated agriculture (CA-59), field plot (FP).

Table 3. Analysis of variance summary of the effects of soil treatment and time and their interactions on methane fluxes, area-scaled emissions, yield scaled emissions, soil oxidation-reduction (redox) potential, rice dry matter, and rice yield during the 2016 growing season at the Rice Research and Extension Center near Stuttgart, Arkansas.

Variable/Source of Variation	— *P*—
Methane Fluxes	
Treatment	**< 0.01**
Time	< **0.01**
Treatment x time	**< 0.01**
Season-long, area-scaled emissions	
Treatment	**< 0.01**
Season-long, yield-scaled emissions	
Treatment	**< 0.01**
Soil redox	
Treatment	**< 0.01**
Time	**< 0.01**
Treatment x time	0.99
Rice dry matter	
Treatment	**0.0005**
Rice yield	
Treatment	**< 0.01**

Table 4. Summary of soil treatment effects on soil oxidation-reduction (redox) potential of dates sampled, rice dry matter and yield, season-long, area-scaled methane (CH_4) emissions, and CH_4 emissions intensity during the 2016 rice growing season at the Rice research and Extension Center near Stuttgart, AR. Means in a column followed by the same letter do not differ (*P* > 0.05)

Treatment	Redox Potential (mV)	Methane Emission (kg CH_4-C ha^{-1} season	Dry matter (Mg ha^{-1})	Rice Yield (kg ha^{-1})	Emissions Intensity [kg CH_4-C (Mg grain) $^{-1}$]
Native prairie (NP)	-137bcd	1521a	35.3ab	16319a	93.7a
Managed grassland (MG)	-121bc	1166b	38.5a	17489a	66.9b
Conservation Reserve Program (CRP)	-149cd	623c	34.3ab	15447ab	41c
Cultivated agriculture (CA-PT)	-77b	336d	28.9cd	12099bc	28d
Cultivated agriculture (CA-25)	131a	126e	23.5fe	9374c	13.9ef
Cultivated agriculture (CA-30)	-197d	368d	30.5bc	14578ab	24.9de
Cultivated agriculture (CA-41)	-195d	153e	28.6cde	12786bc	12.1f
Cultivated agriculture (CA-59)	-122bc	166e	24.2def	9698c	17.4def
Cultivated agriculture (FP)	--	63f	23.2f	11289c	5.6f

Table 5. Linear correlation summary among measured soil properties and concentrations in the top 10 cm and season-long, area-scaled methane (CH4) emissions and emissions intensity [kg CH_4-C (Mg grain)$^{-1}$] for the 2016 growing season at the Rice Research and Extension Center near Stuttgart, AR. Single asterisk (*) indicates $P < 0.001$.

Soil Property	Season-long CH_4 emissions	Emissions intensity
	r	
Sand (g g^{-1})	0.159	0.109
Silt (g g^{-1})	-0.019	0.018
Clay (g g^{-1})	-0.214	-0.188
Soil organic matter (mg kg^{-1})	0.899*	0.860*
Total carbon (mg kg^{-1})	0.924*	0.884*

Table 6. Linear correlations summary among measured soil property contents in the top 10 cm and season-long, area-scaled methane (CH4) emissions and emissions intensity [kg CH_4-C (Mg grain)$^{-1}$] for the 2016 growing season at the Rice Research and Extension Center near Stuttgart, AR. Single asterisk (*) indicates $P < 0.05$.

Soil Property	Season-long CH_4 emissions	Emissions intensity
	———— *r* ————	
Zinc (kg ha^{-1})	-0.495*	-0.504*
Potassium (kg ha^{-1})	-0.407*	-0.439*
Soil organic matter (Mg ha^{-1})	0.355	0.35
Total carbon (Mg ha^{-1})	0.338	0.344

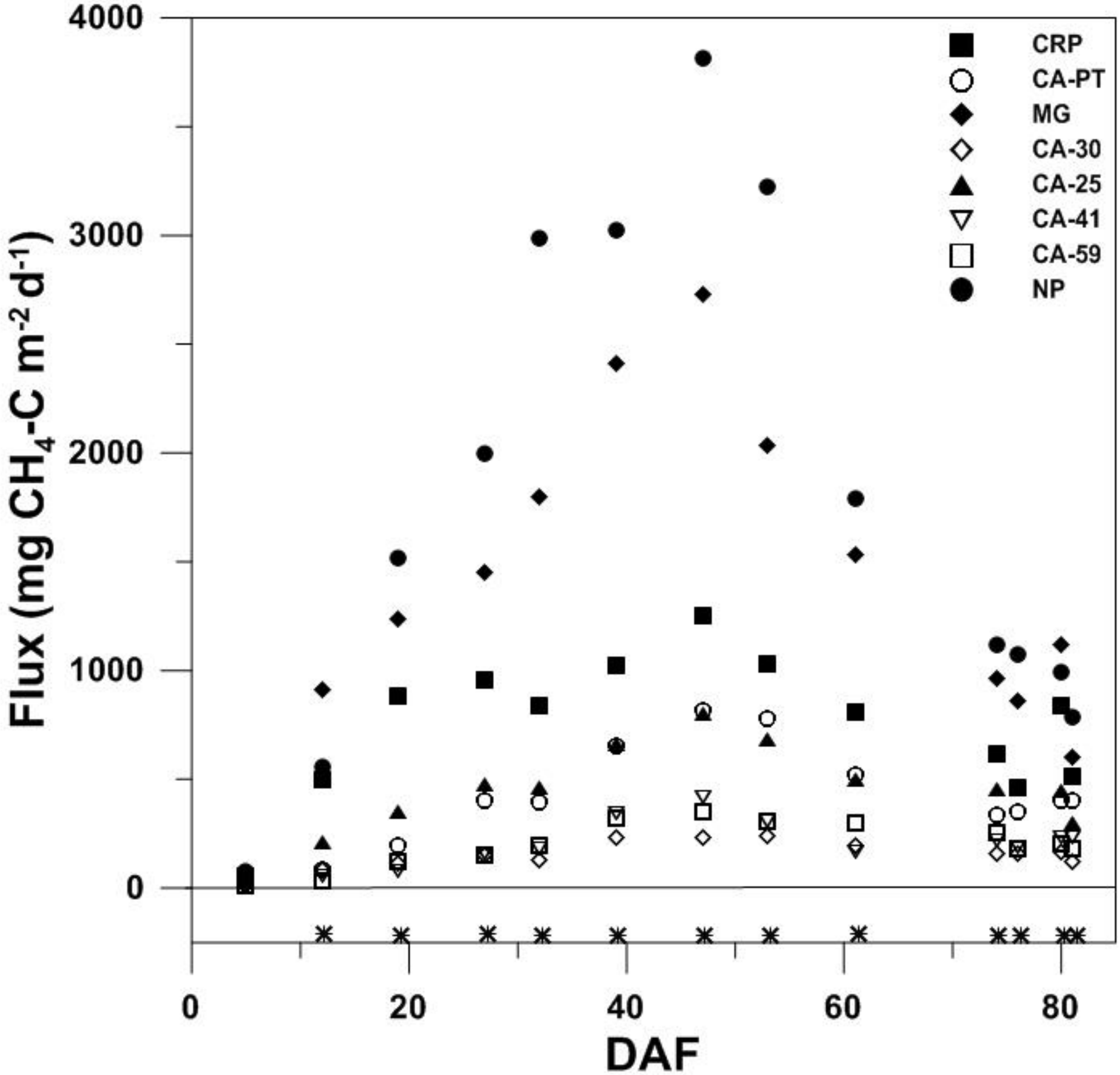


Figure 1. Season-long profile of methane (CH_4) flux trends over time [i.e., days after flooding (DAF)] for eight soil treatments : native prairie (NP), managed grassland (MG), conservation resource program (CRP), cultivated agriculture (CA-PT) cultivated agriculture (CA-25), cultivated agriculture (CA-30), cultivated agriculture (CA-41), and cultivated agriculture (CA-59), during the 2016 rice growing season at the Rice Research and Extension Center near Stuttgart, AR. A single asterisk (*) on a given measurement date indicates a significant (*P* < 0.05) difference exists from a flux of zero and among soil treatments.

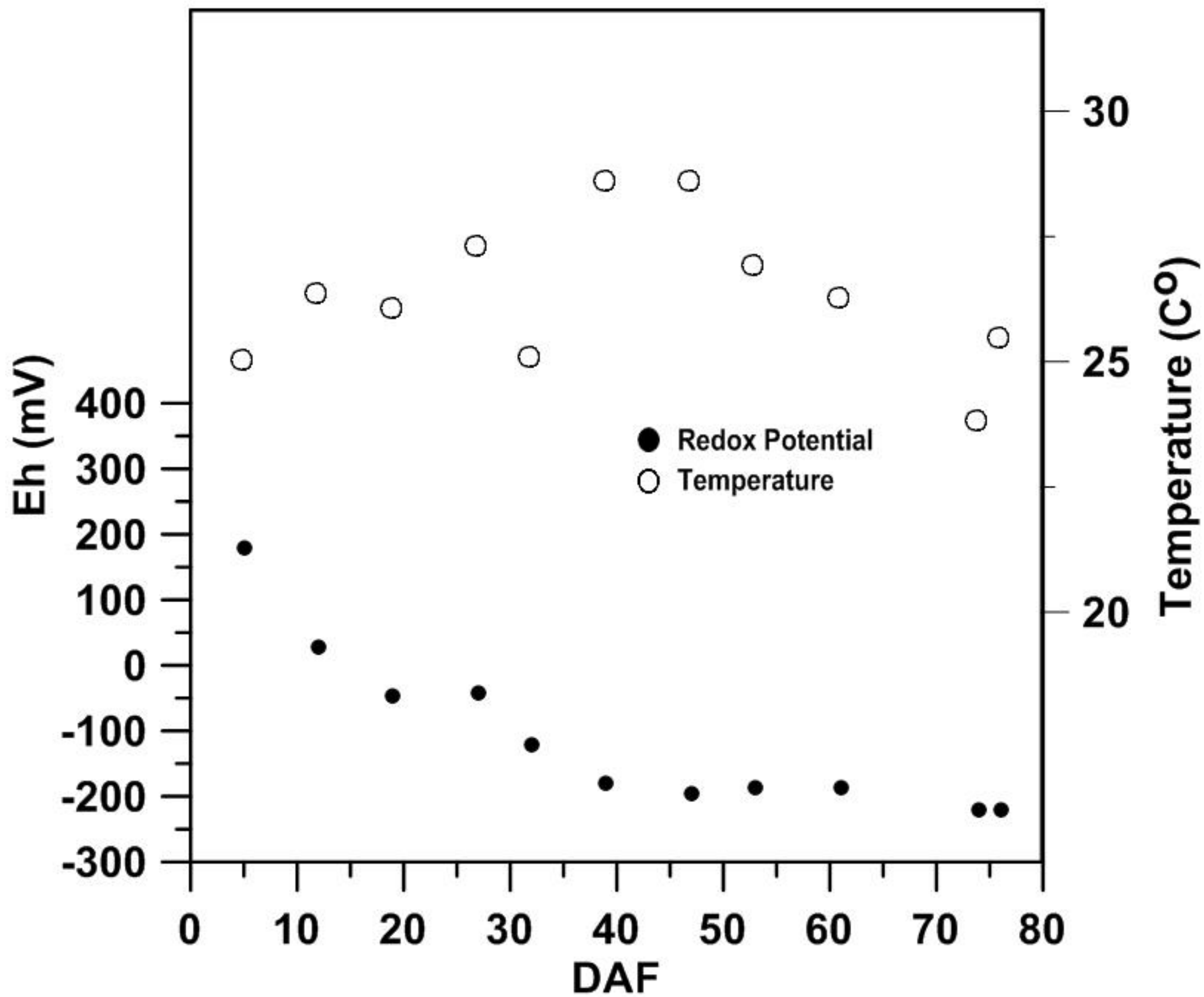


Figure 2. Season-long profile of soil temperature and oxidation-reduction potential (Eh), measured at the 7-cm soil depth, averaged over time [i.e., days after flooding (DAF)] during the 2016 season at the Rice Research and Extension Center near Stuttgart, AR.

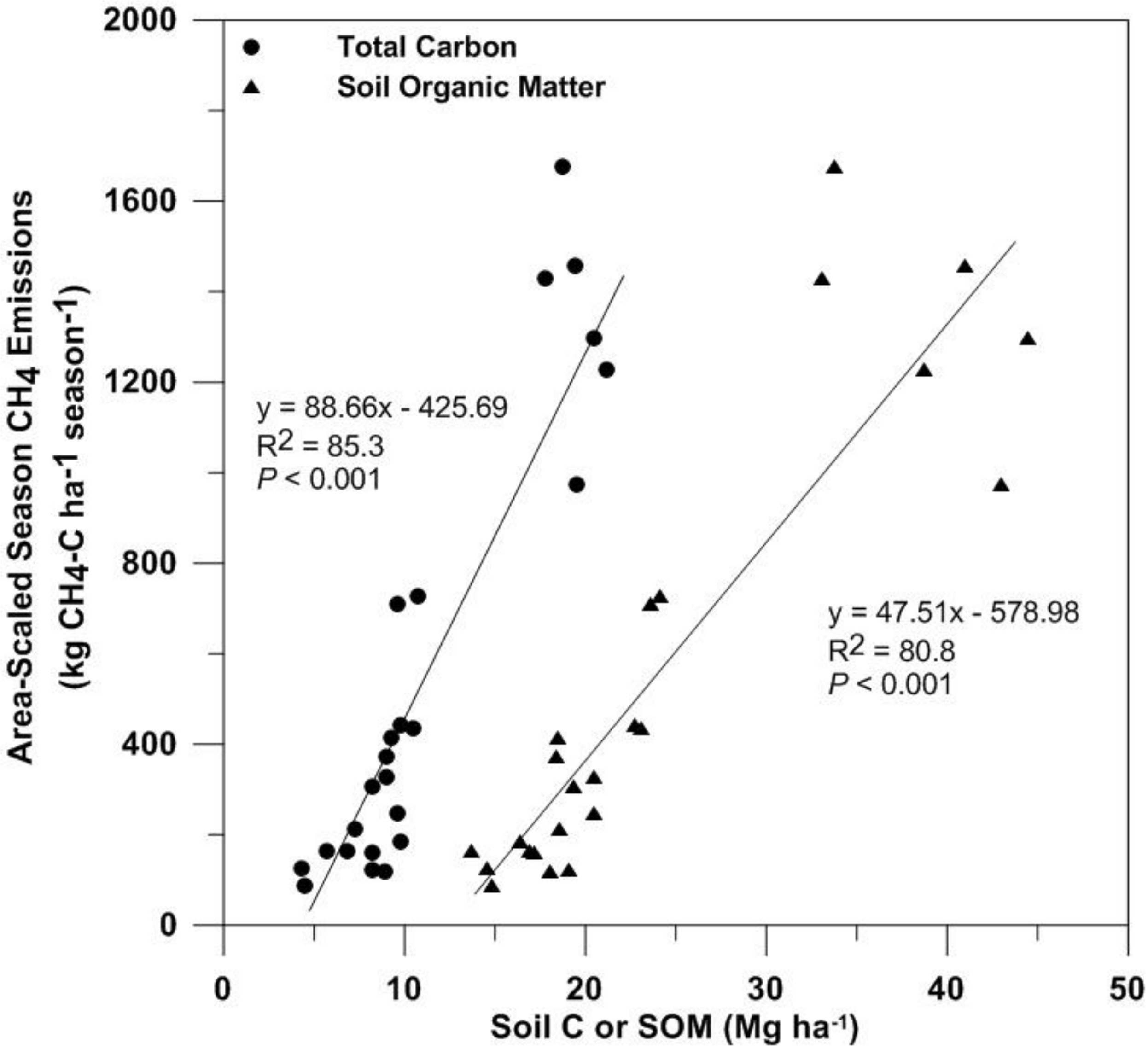
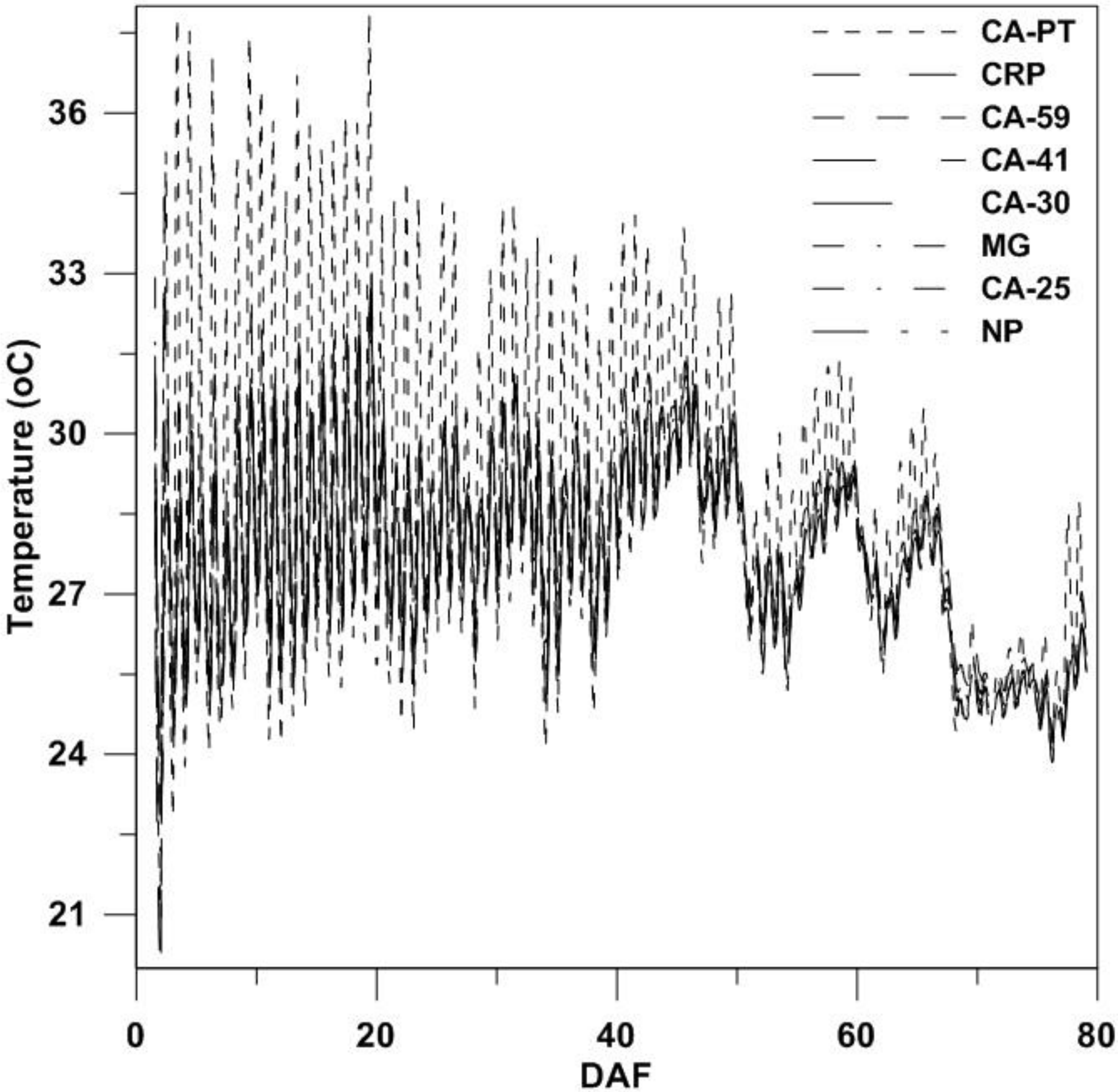
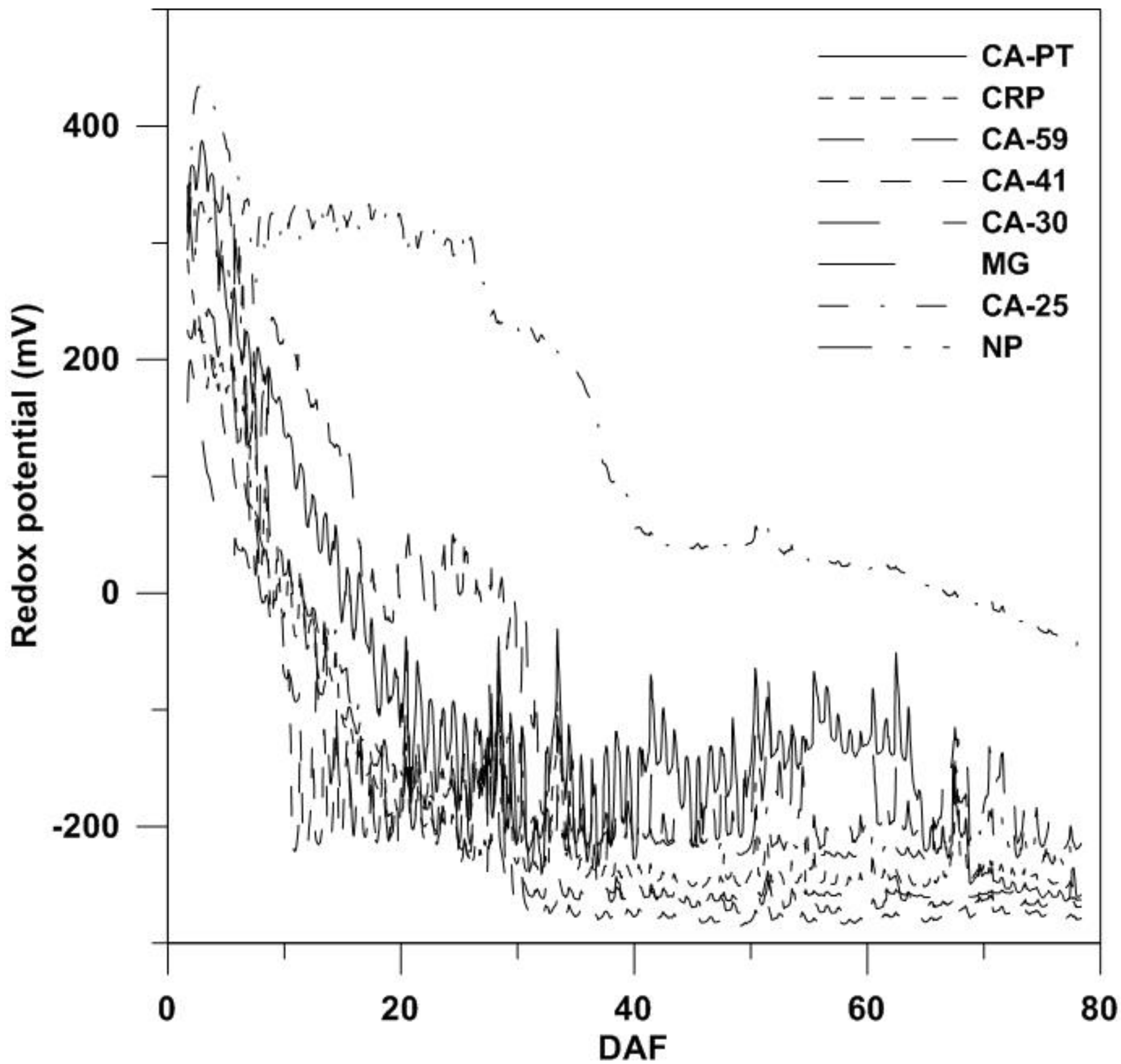


Figure 3. Relationship between soil organic matter (SOM) and total carbon (TC) contents in the top 10 cm and season-long, area-scaled methane (CH_4) emissions from the 2016 rice growing season at the Rice Research and Extension Center near Stuttgart, AR.

Appendix A



Season-long profile of soil temperature measured at the 7-cm soil depth [i.e., days after flooding (DAF)] for eight soil treatments, including native prairie (NP), managed grassland (MG), Conservation Reserve Program (CRP), cultivated agriculture at the Pine Tree Research Station (CA-PT), 25-year-old cultivated agriculture (CA-25), 30-year-old cultivated agriculture (CA-30), 41-year-old cultivated agriculture (CA-41), and 59-year-old cultivated agriculture (CA-59), during the 2016 season at the Rice Research and Extension Center near Stuttgart, AR.



Season-long profile of oxidation-reduction potential (Eh), measured at the 7-cm soil depth [i.e., days after flooding (DAF)] for eight soil, including native prairie (NP), managed grassland (MG), Conservation Reserve Program (CRP), cultivated agriculture at the Pine Tree Research Station (CA-PT), 25-year-old cultivated agriculture (CA-25), 30-year-old cultivated agriculture (CA-30), 41-year-old cultivated agriculture (CA-41), and 59-year-old cultivated agriculture (CA-59), during the 2016 season at the Rice Research and Extension Center near Stuttgart, AR.

Appendix B

Example of SAS program for evaluating CH_4 fluxes over time among pre-assigned soil treatments for the 2016 season.

```
title 'Methane Field Study 2016 - Joshua Humphreys';
title2 'Methane Fluxes 2016 ANOVA';
data methane2016;
  infile 'CH4Flux2016.prn' firstobs=2;
  input ID DAF treatment $ flux;
 run;

proc sort data=methane2016; by DAF;
quit;

Proc print data=methane2016 noobs;by DAF;
id DAF;
var treatment flux;
run;

proc mixed data=methane2016 method=type3;
class   treatment DAF;
model flux = treatment DAF DAF*treatment / ddfm=kr ;
ods exclude FitStatistics Tests3 IterHistory ;
quit;
```

Example of SAS program for evaluating season-long, area-and yield-scaled flood-release CH_4 for pre-assigned soil treatments for the 2016 season.

```
title 'Methane Field Study 2016 - Joshua Humphreys';
title2 'Emission Methane 2016 ANOVA';
data methane2016;
  infile 'CH4Emissions2016.prn' firstobs=2;
  input treatment ID $ Emission;
 run;

proc sort data=methane2016; by treatment;
quit;

proc print data=methane2016 noobs; by treatment;
  id ;
  var Emission;
  run;

proc mixed data=methane2016 method=type3;
class  treatment;
model emission = treatment / ddfm=kr  ;
ods exclude FitStatistics Tests3 IterHistory   ;
quit;
```

Example of SAS program for evaluating yield, aboveground biomass, soil redox potential, and soil temperature between pre-assigned soil treatments for the 2016 season.

```
title 'Methane Field Study 2016 - Joshua Humphreys';
title2 'Emission Methane 2016 ANOVA';
data methane2016;
  infile 'CH42016.prn' firstobs=2;
  input ID treatment bio;
 run;

proc sort data=methane2016; by treatment;
quit;

proc mixed data=methane2016 method=type3;
class  treatment;
model bio = treatment / ddfm=kr  ;
ods exclude FitStatistics Tests3 IterHistory   ;
quit;
```

Example of SAS program data for evaluating initial soil properties between pre-assigned soil treatments for the 2016 season.

```
title 'Methane Field Study Soil Properties 2016- Joshua Humphreys';
title2 'Soil properties CH4 2016 ANOVA';
data soildata2016;
  infile 'soil properties2016.prn' firstobs=2;
  input id treatment $ ph ec p k ca mg su na fe mn zn cu N C LOI CN ;
 run;

proc sort data=soildata2016; by treatment;
quit;

proc print data=soildata2016 noobs; by id;
    var   ph ec p k ca mg su na fe mn zn cu N C LOI CN ;
  run;
quit;

proc mixed data=soildata2016 method=type3 ;
class treatment;
model ph = treatment / ddfm=kr ;
random treatment;
ods exclude FitStatistics Tests3 IterHistory;

quit;
```

CHAPTER FIVE

Methane production as affected by tillage practice and urea fertilizer type from a silt-loam soil in Arkansas

Abstract

Greenhouse gas (GHG) emissions from agricultural settings have come under great scrutiny in the past 20 years and the impact of GHGs in the environment regarding global climate change is alarming. Understanding the conditions and mechanisms that produce GHGs, specifically methane (CH_4), are needed to better attenuate the release of CH_4 from various agronomic practices in agricultural settings, particularly from rice (*Oryza sativa* L.) production. The objective of the study was to evaluate the effects of tillage [conventional tillage (CT) and no-tillage (NT)] and urea-based fertilizers [N-(n-butyl) thiophosphoric triamide (NBPT)-coated urea and non-coated urea] on CH_4 fluxes and emissions from rice grown on a Dewitt silt-loam soil (fine, smectitic, thermic Typic Albaqualfs) in the direct-seeded, delayed-flood rice production system in Arkansas. Gas samples were a collected in 2017 from vented, non-flow through chambers at 20-minute intervals (0, 20, 40, and 60 minutes) every week from flood establishment to four days after end-of-season flood release. Methane fluxes differed ($P < 0.01$) between tillage treatments over time during the 2017 growing season. Methane fluxes ranged from 452.8 g CH_4-C ha^{-1} day^{-1} by 41 days after flood (DAF) establishment to 611.2 g CH_4-C ha^{-1} day^{-1} by 70 DAF under CT and ranged from 405.2 g CH_4-C ha^{-1} day^{-1} by 13 DAF to 784.6 g CH_4-C ha^{-1} day^{-1} by 41 DAF under NT. Averaged across tillage, mean season-long CH_4 emissions were 33.4 and 37.2 kg CH_4-C ha^{-1} $season^{-1}$ from NBPT-coated and non-coated urea, respectively, but were unaffected ($P > 0.05$) by fertilizer treatment. Greater understanding of the effects of tillage and urea fertilizer type on CH_4 and other GHG emissions is essential for ascertaining GHG impacts from rice production and for determining GHG loads to the atmosphere.

Introduction

Global climate change will be one of the foremost challenges for humankind over the next 50 years (IPCC, 2014). As air temperatures increase globally and the human population rises, developing new techniques to improve or sustain soil health and water resources will become necessary for continued survival (IPCC,2014). Developing alternative agronomic techniques will be paramount for increasing agricultural production, as well as reducing climate-change drivers, such as greenhouse gas (GHG) emissions. Rising levels of the main naturally and anthropogenically produced GHGs [i.e., carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O)] are clear when contrasting a baseline of pre-Industrial Revolution concentrations with recently recorded concentrations, which set record levels unseen for the last 800,000 years (IPCC, 2014). Pre-Industrial Revolution GHG concentrations were 280 mg L^{-1} for CO_2, 0.7 mg L^{-1} for CH_4, and 0.18 to 0.26 mg L^{-1} for N_2O, while 2005-reported GHG concentrations were 379 mg L^{-1} for CO_2, 1.8 mg L^{-1} for CH_4, and 0.32 mg L^{-1} for N_2O (Forster et al., 2007). More recently, total US GHG emissions increased by 8.4% from 1990 to 2010 (IPCC, 2014).

Agriculture alone is responsible for nearly 50% of global CH_4 emissions and for 10 to 12% of total anthropogenic GHG emissions worldwide (Smith et al., 2007). Of all widely grown row crops, particularly in the United States (US), rice (*Oryza sativa* L.) production specifically has been under scrutiny for its atmospheric-CH_4 contributions due to the unique water management system used for rice production, which entails maintaining a continuous flood for most to all of the rice growing season, as rice is semi-aquatic plant (IPCC, 2014). The flood-irrigation system differs from all other cultivated row crops in the world, as most crops are irrigated or watered when needed. In the flooded-soil environment, anaerobic and reducing conditions develop gradually to facilitate CH_4 production by methanogens, if a reducible form of carbon (C) is present (IPCC, 2014). Since C, and soil organic matter (SOM) in general, is concentrated near the soil surface, the main source of CH_4 production in the soil column, regardless of landuse type, is in the topsoil, where > 99% of the total soil-produced CH_4 is typically emitted (Mitra et al., 2002). Methane diffusion through the water column is slow, consequently passive transport of CH_4 through the aerenchyma tissue of the rice plants themselves provides the main mechanism of CH_4 release to the atmosphere from rice cultivation (Cicerone and Shetter, 1981; Yu et al., 1997; Dannenburg and Conrad, 1999; Groot et al., 2005).

As of 2011, estimates of total CH_4 emissions from rice production represented 1.1% of the total US CH_4 emissions to the atmosphere (IPCC, 2014); however, residue burning and rice cultivation combined make up 3.7% of the total agricultural CH_4 releases (IPCC, 2014). In 2015, the total estimated CH_4 emissions from rice production in the US were 11.2 MMT (million megatons) of CO_2 equivalents (USEPA, 2017). In 2016, 47% of all US rice was grown in Arkansas (Hardke et al., 2017). Consequently, Arkansas produced an estimated 3.8 MMT CO_2 equivalents in 2015 from rice cultivation alone (USEPA, 2017). This large magnitude of GHG production from the soil and its effects on global climate change justify why characterization of GHG emissions, in particular CH_4, from common rice production practices, specifically in Arkansas, is crucial (Rector et al., 2018).

Along with conventional tillage (CT), no-tillage (NT) agriculture is a relatively widely adopted, alternative management practice being used with many upland crops, where the goal is to reduce soil erosion, decrease input costs, and sustain long-term crop productivity (Pittelkow et al., 2015). No-tillage also generally increases SOM, which not only enhances essential nutrients in the soil, but may potentially supply an increased amount of C substrate to methanogens, which could have a significant effect on CH_4 emissions (Liu et al., 2006; Ahmad et al., 2009). For rice production in Arkansas, NT methods account for approximately 4% of the total planted area, where CT makes up approximately 60%, while the remaining 36% uses a stale-seedbed approach (Hardke et al., 2016). One reason for the rather low NT adoption rate is that rice produced under NT has exhibited up to a 7.5% reduction in yield compared to under traditional CT (Pittelkow et al., 2015), which is a barrier for many producers to overcome when contemplating switching tillage systems to reap the environmental benefits of conservation production practices, such as increased SOM, that can be realized from conversion to NT. It is anticipated that more producers will consider conversion to NT rice production in the future for a variety of reasons, including agronomic, environmental, and economic reasons. Consequently, evaluation of CH_4 emissions from rice production under CT and NT practices, which has not been done in Arkansas, is not only timely, but is also critical to document potential impacts of tillage practice on CH_4 emissions to help guide future agronomic decisions, such as whether to convert to NT or not.

In addition to tillage practice as a major agronomic decision point for rice production, optimal rice production requires careful nitrogen (N) management to maximize yields. Conventional production practices often expose N-fertilized crops to potentially increased N-loss mechanisms, such as volatilization, denitrification, and/or leaching. For rice production, urea is the common fertilizer-N source due to urea's large N concentration (46% N; Norman et al., 2013). Urea has two amine groups, which help reduce N loss through nitrification after application, compared to other potential fertilizer-N sources like ammonium nitrate, which adds readily mobile nitrate directly to the soil that is also prone to denitrification (Rector et al., 2018). To further reduce potentially substantial N losses via ammonia volatilization and denitrification after application, the urease inhibitor N-(n-butyl) thiophosphoric triamide (NBPT) is commonly used as a coating on urea prills (Norman et al., 2013). Although significant loss of N can occur through ammonia volatilization, particularly to wet soil, establishing the flood quickly after N application as NBPT-coated urea slows down the activity of the urease enzyme that resides in the soil (Norman et al., 2013). More specifically, NBPT-coated urea is the common urea treatment used in Arkansas to inhibit urease activity after application and slow the release of plant available N in the soil (Norman et al., 2003, 2013). Examining the relationship between non-coated-urea fertilization and an unfertilized control, Rogers et al. (2013) demonstrated no difference with regards to season-long CH_4 emissions from rice grown on a silt-loam soil in east-central Arkansas. Furthermore, Rector et al. (2018) reported no effect of urea fertilizer type (i.e., NBPT-coated or non-coated) on season-long N_2O emissions from rice grown on a silt-loam soil in east-central Arkansas. Minimizing N volatilization losses and prolonging N release in the soil from NBPT-coated urea compared to non-coated urea have the potential to increase aboveground biomass production. However, it has not been clearly shown whether CH_4 emissions increase with greater aboveground biomass (Ahmad et al., 2009, Rogers et al., 2013). Furthermore, the potential effects of NBPT-coated compared to non-coated urea on CH_4 emissions have not been examined in Arkansas.

The lack of field studies directly assessing the potential effects of tillage options and urea fertilizer types on CH_4 emissions is a severe limitation for evaluating the present and potential future sustainability of rice production in Arkansas and elsewhere in areas of concentrated rice cultivation. Therefore, the objective of this field study was to evaluate the effects of tillage practice (CT and NT) and urea fertilizer type (NBPT-coated urea and non-coated urea) on CH_4 fluxes and season-long emissions from a pure-line cultivar grown under a full-season flood in the direct-seeded, delayed-flood production system on a silt-loam soil in Arkansas. It was hypothesized that CH_4 fluxes and emissions would be greater from NT than from CT because of the increased labile organic matter on the soil surface under NT to provide more C substrate for CH_4 production compared to CT. It was also hypothesized that NBPT-coated urea would result in greater CH_4 fluxes and emissions due to the increased labile form of N compared to the non-coated urea. Specifically, the NBPT-coated urea will keep N in the soil longer and more plant available, giving the plant a greater opportunity to establish greater aboveground biomass, which will result in greater CH_4 fluxes and season-long emissions than from the non-coated urea, which may result in greater N volatilization losses

.

Materials and Methods

Site Description

Research was performed in 2017 at the University of Arkansas Division of Agriculture's Rice Research and Extension Center (RREC) east of Stuttgart in Arkansas County, in east-central AR (34°27'54.5" N, 91°25'8.6" W), closely following procedures outlined in Rogers et al. (2014), on a Dewitt silt-loam (fine, smectitic, thermic Typic Albaqualfs) soil with < 1% slope throughout the research site. The study area had been managed in a rice-soybean (*Glycine max* L. [Merr.]) rotation, which is a commonly used rotation for rice production in Arkansas, for more than 25 years. Replicate research plots for this study have been managed under long-term NT for at least 10 years (Slaton et al., 2013, 2017; Parvej et al., 2016) and an adjacent area that had been under continuous CT for over 75 years. The NT treatment used in this study was border area of larger NT plots that were part of an on-going long-term NT potassium (K) fertilization study (Slaton et al., 2013, 2017; Parvej et al., 2016).

The regional climate throughout the study area is temperate, with a mean annual air temperature of 16.5°C, which ranges from a mean minimum of 12.7°C in January to a mean maximum of 23.5°C in July (NOAA, 2015). The mean annual precipitation for the study area is 135 cm (NOAA, 2015). The 2017 growing season (i.e., May through September) had an average daily air temperature of 25.0°C, which was similar to the 30-year (i.e., 1981 to 2010) average of 25.1°C for the same months (NOAA, 2015). The precipitation for the entire growing season was 55.0 cm while the 30-year average is 43.0 cm of rainfall.

Treatments and Experimental Design

A randomized complete block (RCB) design with a factorial arrangement of each tillage (CT and NT)-fertilizer type [NBPT-coated urea and non-coated urea] treatment combination replicated four times was used to address the objective of this study. Two long-term NT plots (4.6-m wide by 7.6-m long) were used with an 18-cm row spacing. Each large plot had two areas fertilized with NBPT-coated urea and two areas fertilized with non-coated urea. Each plot had four base collars (described below) installed: two for the NBPT-coated urea treatment and two for the non-coated urea treatment. Conventional tillage plots (1.6-m wide by 4.6-m long) with 18-cm row spacing were established adjacent to the long-term NT plots and had one base collar placed per plot, for a total of four base collars per plot receiving NBPT-coated urea and four base collars per plot receiving non-coated urea. The CT and NT areas, situated adjacent to one another, were separated by a levee, but were each treated with a full-season-flood water management scheme. There was a total of 16 gas-sampling base collars for the tillage-fertilizer-type treatment combinations (i.e., CT/NBPT-coated urea, CT/non-coated urea, NT/NBPT-coated urea, and NT/non-coated). Tillage and fertilizer-type treatments represented a split-plot design, where tillage was the whole-plot and fertilizer type was the split-plot factor, while time (i.e., gas flux measurement date) was a split-split-plot factor for CH_4 flux analyses.

Plot Management

On 22 March, 2016, the year prior to this field study, pre-plant fertilizer, 83.8 kg K ha^{-1} as muriate of potash 29.4 kg P ha^{-1} as triple superphosphate, and 11.2 kg Zn ha^{-1} as $ZnSO_4$, were applied to all CT plots. On 22 March, 2016, the NT plots were pre-plant fertilized with only 83.8 kg K ha^{-1} as muriate of potash and rice seeds were pre-treated with Zn. The CT plot area was left fallow, while the NT plots were cropped to soybean during the 2016 growing season. On 20 November, 2016, the CT plots were disked with one pass, then on 25 April, 2017 the CT plots were manipulated with two passes of a land plane to smooth the soil surface to prepare for planting.

The pure-line cultivar 'CL172', which is a long-grain, semi-dwarf cultivar that was created by the University of Arkansas, was planted on 9 May and 11 May, 2017 in the NT and CT plots, respectively. A single, pre-emergence mixture of Obey (FMC Corp., Philadelphia, PA), which is a mixture of clomazone (2-[(2-chlorophenyl)methyl]-4,4-dimethyl-3-isoxazolidinone and quinclorac (3,7-dichloro-8-quinolinecarboxylic acid), and Permit Plus [halosulfuron-methyl, methyl 3-chloro-5-(4,6-dimethoxypyrimidin-2-ylcarbamoylsulfamoyl)-1-methylpyrazole-4-carboxylate; Gowan Co., Yuma, AZ] herbicide was applied on 9 May, 2017, with no additional herbicide application throughout the growing season.

A recommended, single, pre-flood N application (118 kg N ha^{-1} as either coated or non-coated urea) was broadcast manually to dry soil within each collar in both CT and NT plots on 12 June, 2017. The N recommendation was determined according to the N-Soil Test for Rice (N-STaR; Norman et al., 2013) in the NT portion of the study area. The N-STaR fertilizer-N recommendation is based on soil samples to a depth of 46 cm and is further refined based on soil textural class and cultivar selection (Norman et al., 2013). On 13 June, 2017, the full-season flood was established at the 4- to 5-leaf stage of the rice, after which the flood was maintained at a 6-cm to 10-cm depth until two weeks prior to harvest when the flood was released.

Soil Redox Potential and Temperature

Soil oxidation-reduction (redox, Eh) potential sensors (Model S650KD-ORP, Sensorex, Garden Grove, CA) with Ag/AgCl reference solution and chromel-constantan thermocouples (Type E) were installed adjacent to two NT/NBPT-coated-urea and two NT/non-coated-urea base collars to a depth of 7 cm on the day of flood establishment (13 June, 2017). Thermocouples and redox sensors were also installed adjacent to two CT/NBPT-coated-urea and two CT/non-coated-urea base collars to a depth of 7 cm the day prior to flood establishment (12 June, 2017). All redox sensors were installed vertically, while all thermocouples were installed horizontally. All sensors were connected to a datalogger (CR 1000, Campbell Scientific, Inc., Logan, UT), protected by an environmental enclosure, to record soil Eh and soil temperature at 15-minute intervals, while mean data were output every hour until after the flood was released to prepare for harvest. Soil Eh values were corrected to the standard hydrogen electrode by adding 199 mV to each field-measured value (Patrick et al., 1996). Recorded sensor data were collected weekly and soil Eh and soil temperature data were summarized based on the values recorded at 0900 hours on each gas sampling date. Sensors were removed from the field on 9 September, 2017.

Soil Sampling and Analyses

On 30 May, 2017, two weeks before flood establishment, soil samples were collected from the top 10 cm near each base collar prior to fertilizer-N application and flooding. Soil samples were collected for bulk density determinations using a stainless-steel, 4.8-cm-diameter core chamber and slide hammer. Eight additional soil samples per plot were collected from the top 10 cm using a 2-cm-diameter push probe that were used for particle-size and chemical analyses. Soil samples were dried at 70°C for at least 48 hr and weighed. Dried soil samples were crushed and sieved to pass through a 2-mm mesh screen. A modified 12-hr hydrometer method was used to determine particle-size distribution (Gee and Or, 2002). Soil pH and electrical conductivity (EC) were analyzed potentiometrically in a 1:2 (m/v) soil-water suspension. Mehlich-3 extractable nutrients (i.e., P, K, Ca, Mg, Fe, Mn, Na, S, Zn, and Cu) were analyzed by inductively coupled argon plasma atomic emissions spectrometry (Spectro Arcos, Spectro Analytical Instruments, Kleve, Germany) using a 1:10 soil-mass-to-extractant-volume ratio (Tucker, 1992). Total soil C (TC) and total N (TN) concentrations were determined by high-temperature combustion with a VarioMax CN analyzer (Elementar Americas Inc., Mt. Laurel, NJ; Nelson and Sommers (1996). Measured TC and TN concentrations were used to calculate C:N ratios on a plot-by-plot basis. Soil organic matter (SOM) was determined by weight-loss-on-ignition after 2 hours at 360°C. Based on measured bulk densities in each plot and the 10-cm sampling depth, all measured concentrations (mg kg^{-1}) were converted to contents (kg or Mg ha^{-1}) for reporting purposes.

Gas Sampling and Analyses

Similar to procedures used by Rogers et al. (2014) and Humphreys et al. (2018), after planting and before flooding, a boardwalk system was constructed throughout the study area to reduce stresses and disturbances to the rice plants and facilitate easier access to the plots during the growing season for gas sample collection. The boardwalk was constructed of 5.1-cm x 30.5-cm x 3.6-m pressure-treated wooden planks laid upon 20- x 40-cm concrete blocks before base-collar installation in the plots. The base collars were then set into place to contain portions of the second and third rice rows in each plot for gas sampling.

For the determination of CH_4 fluxes, vented, non-flow-through, non-steady-state chambers (Livingston and Hutchinson, 1995; Rogers et al., 2014; Humphreys et al., 2018) were used for the collection of gas samples. In the construction of cylindrical base collars (30 cm in diameter by 30-cm tall), schedule 40 polyvinyl chloride (PVC) was used and beveled at the bottom to facilitate insertion to a depth of approximately 10 cm. Four, 12.5-mm diameter holes were drilled approximately 12 cm from the beveled end of each base collar to allow for flood water to enter and exit the base collars. The collars were driven into the ground such that the drilled holes were just above or level with the soil surface. The holes were plugged during sampling and after flood release with gray butyl-rubber septa (Voigt Global, part# 73828A-RB, Lawrence, KS) to prevent convection currents inside the chambers that would dilute the ambient headspace air.

To facilitate rice growth during the season, 40- and/or 60-cm-long chamber extensions were used to increase the height of the chamber. Extensions were covered in reflective aluminum tape (CS Hyde, Mylar metallized tape, Lake Villa, IL) to reduce temperature variations due to reflecting solar energy inside the chamber during use. Tire inner tube cross sections, approximately 10 cm wide, were taped to the bottom of all the extensions to function as a seal to the base collars and to the other extensions during chamber use.

Chamber caps (30-cm-diameter PVC by 10 cm tall) with a 5-mm thick sheet of PVC glued to the top were also covered with reflective aluminum tape. Approximately 10-cm-wide tire inner tube cross sections were also taped to the bottom of the caps to serve as a seal and attachment mechanism to the base collar or extensions. A 4.5-mm inside diameter (id), 15-cm-long piece of copper refrigerator tubing was installed on the side of each cap to maintain atmospheric pressure during sampling. On the top of each chamber cap, a single, 12.5-mm diameter hole was drilled and plugged with gray butyl-rubber septa (Voigt Global, part# 73828A-RB, Lawrence, KS) for syringe and thermometer insertion. To ensure proper air mixing in the enclosed chamber, a 2.5-cm^2, battery-operated (9V), magnetic levitation fan (Sunon Inc., MagLev, Brea, CA) ran throughout the duration of gas sampling for headspace air mixing.

The acquisition of gas samples from the chambers was completed by using a 20-mL, B-D syringe with a detachable 0.5-mm diameter x 25-mm long needle (Beckton Dickson and Co., Franklin Lakes, NJ) that was inserted through the gray butyl-rubber septa installed in the chamber cap. After drawing a gas sample from the chamber into the syringe, the collected sample was immediately injected into a pre-evacuated, 10-mL, crimp-top glass vial (Agilent Technologies, part# 5182-0838, Santa Clara, CA). Gas samples were acquired at 20-minute intervals, beginning at 0 minutes when the chamber was capped and sealed, for 1 hr (i.e., the 0-, 20-, 40-, and 60-min marks). Gas sampling started 1 day after flood establishment in 2017 and continued weekly until flood release when sampling frequency changed to 1, 2, 3, 4 and 5 days after flood release. Similar to prior studies (Rogers et al., 2013, 2014; Humphreys et al., 2018), all gas sampling occurred in the morning between 0800 to 1000 hours CST to minimize temperature fluctuations in the chambers.

Relative humidity, ambient air temperature, 10-cm soil temperature, barometric pressure, and the air temperature inside the chamber were recorded during each chamber sampling event and at every sampling interval (i.e., the 0-, 20-, 40-, and 60-min marks). During gas sampling, the distance from the top of the chamber to the water level, if any water was present, was measured to properly calculate the interior chamber volume. Methane gas standards (i.e., 2, 5, 10, 20, and 50 mg L^{-1}) were collected in the field using a 20-mL syringe with a detachable needle that was immediately injected into pre-evacuated, 10-mL, crimp-top glass vials. Methane gas standards from the same five concentration standards were also collected in the laboratory immediately prior to gas sample analysis to evaluate potential leakage from sample transport from the field.

Utilizing a flame ionization detector (250°C), a Shimadzu GC-2014 gas chromatograph (Shimadzu North America/Shimadzu Scientific Instruments Inc., Columbia, MD) was used to analyze gas samples for their CH_4 concentration within 48 hr of collection in the field. According to procedures described by Rogers et al. (2013), CH_4 fluxes were calculated using changes in concentrations in the chamber headspace over the 60-min sampling interval. To assess the change in concentration over time, measured concentrations (mL L^{-1}; y axis) were regressed against time (in minutes; x axis) of sample extraction (i.e., 0, 20, 40, and 60 minutes). The slope of the resulting best-fit line was then multiplied by the calculated chamber volume (L) and divided by the inner surface area of the chamber (m^2) resulting in flux units of µL CH_4 m^{-2} min^{-1} (Parkin and Venterea, 2010). The units of the µL CH_4 were then converted using the Ideal Gas Law (PV = nRT) to µmol CH_4, where P was the measured pressure over the 60-min sampling interval in atmospheres (atm), V was the calculated volume of the interior of the chamber (L), n was the number of moles of the gas, R was the gas constant (0.8206 L atm Mol^{-1} K^{-1}), and T was the average measured temperature inside the chamber in Kelvin over the 60-min interval. To convert µmol CH_4 to the mass of CH_4, the molar mass of CH_4 was then used for a final flux unit of mg CH_4 m^{-2} d^{-1} (Parkin and Venterea, 2010).

On a chamber-by-chamber basis, season-long emissions were calculated by linear interpolation between sample dates. Emissions data were also divided into pre- and post-flood-release periods for data analyses due to differences in emissions mechanisms.

Plant Sampling

Aboveground biomass in each base collar was collected on 10 September 2017, four days after flood release, by cutting rice plants 2 cm above the soil surface. To determine aboveground dry matter, samples were dried at 55°C for 3 weeks and weighed. A yield estimate was determined on a chamber-by-chamber basis by clipping panicles, which were then weighed and adjusted to 20% moisture. Methane emissions on a per-unit-yield-basis for each treatment combination (i.e., NT/NBPT-coated urea, NT/non-coated urea, CT/NBPT-coated urea, and CT/non-coated urea) were determined by dividing season-long emissions by rice yields on a chamber-by-chamber basis to evaluate emissions intensity.

Statistical Analyses

A three-factor analysis of variance (ANOVA) was performed using SAS 9.4 (SAS Institute, Inc., Cary, NC) to evaluate that effects of tillage, N-fertilizer type, time, and their interactions on CH_4 fluxes. A two-factor ANOVA was performed to determine the effects of pre-assigned treatments (i.e., tillage practice, N-fertilization type, and their interaction) on initial soil properties in the top 10 cm. A two-factor ANOVA was performed to evaluate the effects of tillage practice, N-fertilizer type, and their interaction on grain yield, pre- and post-flood-release, CH_4 emissions, area- and yield-scaled, season-long CH_4 emissions. When appropriate, means were separated by least significant difference (LSD) at the $\alpha = 0.05$ level.

Results and Discussion

Soil Physical and Chemical Properties

Early season soil properties were evaluated to determine potential differences among plots associated with the tillage (NT and CT) and pre-assigned fertilizer treatments. Sand, silt, and clay contents, 0.14, 0.71, and 0.15 g g^{-1}, respectively, in the top 10 cm were unaffected ($P > 0.05$) by tillage or fertilizer treatment, thus confirming a silt-loam soil surface texture throughout the study area (Table 1). In addition, soil EC, extractable soil Ca, S, and Cu and TN, TC, and SOM content, and C:N ratio in the top 10 cm were also unaffected ($P > 0.05$) by tillage or fertilizer treatment (Table 1). However, several minor differences existed among tillage and pre-assigned fertilizer treatments.

Soil bulk density and extractable soil K differed ($P < 0.05$) by tillage between pre-assigned fertilizer treatments. However, bulk density did not differ between pre-assigned fertilizer treatments under CT, which averaged 1.38 g cm^{-3}, but was 19 and 11% greater than that in the NT/non-coated-urea (1.15 g cm^{-3}) and NT/NBPT-coated-urea (1.23 g cm^{-3}) treatment combinations, which also differed between one another. Similar to soil bulk density, pre-flood extractable soil K content did not differ between pre-assigned fertilizer treatments under CT but was greater in the NT/NBPT-coated urea (156 kg ha^{-1}) than in the NT/non-coated urea (135 kg ha^{-1}) treatment combination. However, all treatment combinations had extractable soil K concentrations within the "Medium" (i.e., 91 to 130 mg K kg^{-1}) soil-test category for fertilizer recommendations for rice grown in Arkansas, with any additional K fertilizer having a little to no expected effect on rice growth or productivity (Norman et al., 2013).

In contrast to soil bulk density and extractable soil K, soil pH, and extractable soil P, Mg, Na, Fe, Mn, and Zn differed ($P < 0.05$) slightly between tillage treatments and was unaffected (P

> 0.05) by pre-assigned fertilizer treatment (Table 1). Soil pH under both CT and NT fell within the optimal ~ 5.0 to 6.75 pH range for rice production (Norman et al., 2003; Havlin et al., 2014), but, averaged across pre-assigned fertilizer treatments, pre-flood soil pH was 13% greater in the top 10 cm under CT (pH = 6.1) than under NT (pH = 5.4) (Table 1). Averaged across pre-assigned fertilizer treatments, pre-flood extractable soil P, Mg, Na, and Mn contents were also 12, 60, 45, and 24%, respectively, greater under CT than under NT, while extractable soil Fe and Zn contents were 1.2 and 2.1 times, respectively, greater under NT than under CT (Table 1). However, soil P concentrations in both tillage treatments were in the "Low" (i.e., 16-25 mg kg^{-1}) soil-test category, which would have suggested additional P fertilizer be applied, but additional P was not applied due to maintaining research continuity with the long-term NT study, which could have potentially impacted plant health and productivity (Norman et al., 2013). Mean extractable soil Zn concentrations were 5.1 and 2.1 mg kg^{-1} for NT and CT, respectively, where the soil-test Zn category was "Low" for CT and "Optimum" for NT. However, according to Norman et al. (2013), neither Zn levels required additional Zn fertilizer for rice grown on a silt-loam soil in Arkansas.

Considering only a few pre-flood differences in soil properties existed among treatments early in the rice growing season, with the exception of extractable soil P, the differences were relatively minor and were generally expected to have little agronomic impact on rice growth and productivity. Consequently, it was reasonably assumed that any subsequently measured differences in CH_4 fluxes and/or emissions among treatments were actually due to imposition of those treatments rather than to large and numerous inherent differences among plots representing the imposed treatments.

Methane Fluxes

Over the 2017 rice growing season, as expected, CH_4 fluxes followed a similar pattern as reported in previous studies (Brye et al., 2013; Rogers et al., 2013; Smartt et al., 2016), with fluxes starting low, increasing to a mid-season peak, then decreasing towards the end-of-season drain, with a small flux increase after flood release before declining within one week after flood release. Methane fluxes differed between tillage treatments over time ($P < 0.01$) but were unaffected ($P > 0.05$) by urea fertilizer type (Table 2; Figure 1). At 1, 2, and 6 DAF, CH_4 fluxes from both tillage treatments did not differ from a flux of zero. By 13 DAF, CH_4 fluxes from CT still did not differ from a flux of zero, while CH_4 fluxes from NT were both greater than zero and greater than that from CT (405 mg CH_4-C m^{-2} d^{-1}). Between 13 and 41 DAF, analytical equipment error prevented analysis of collected gas samples, therefore no data could be presented. By 41 DAF, CH_4 fluxes from CT (452 mg CH_4-C m^{-2} d^{-1}) were lower than the seasonal peak from NT (784 mg CH_4-C m^{-2} d^{-1}) but did not differ from CT fluxes measured 48 DAF. Between 41 and 55 DAF, CH_4 fluxes at least numerically decreased over time, where CH_4 fluxes remained greater from NT than from CT at both 48 and 55 DAF. Between 55 and 89 DAF, which represented the end of gas sampling in the field, CH_4 fluxes did not differ between tillage treatments on any measurement date (Figure 1). However, CH_4 fluxes from CT numerically peaked at 70 DAF (611.2 mg CH_4-C m^{-2} d^{-1}) then decreased until a post-flood-release spike occurred at 87 DAF (501.6 mg CH_4-C m^{-2} d^{-1}). After peaking at 41 DAF, CH_4 fluxes from NT generally decreased until a post-flood-release spike also occurred at 87 DAF (686.1 mg CH_4-C m^{-2} d^{-1}). The general pattern of a post-flood-release spike in CH_4 flux has been observed previously from silt-loam soils (Brye et al., 2013; Rogers et al., 2013, Humphreys et al., 2018). The post-flood-release spike in CH_4 flux is thought to be caused by the degassing of entrapped CH_4 in the soil profile (Smith et al., 2003) after the water column has been released from the field to prepare for harvest. Despite measured CH_4 fluxes still being greater than a flux of zero, gas sampling in the field ceased at 89 DAF because of the need to harvest the rice crop.

Aboveground Dry Matter and Yield

Aboveground dry matter produced by CL172 was unaffected by urea fertilizer type (P = 0.61) but differed between tillage practices (P < 0.01; Table 3). Aboveground dry matter was 17.85 and 18.07 Mg ha^{-1} for the NBPT-coated and non-coated urea, respectively, and averaged 17.96 Mg ha^{-1}. Aboveground dry matter was 15% lower from NT (16.5 Mg ha^{-1}) than from CT (19.4 Mg ha^{-1}).

Similar to aboveground dry matter, rice yield produced by CL172 was unaffected by urea fertilizer type (P = 0.54) but differed between tillage treatments (P < 0.01; Table 3). Rice yields were 8.3 and 8.5 Mg ha^{-1} for the NBPT-coated and non-coated urea, respectively, and averaged 8.4 Mg ha^{-1}. The lack of a urea-fertilizer effect on aboveground dry matter and yield support the similar lack of a urea-fertilizer effect on CH_4 fluxes, where both urea-fertilizer treatments resulted in similar dry matter production and resulting yields (Rector et al., 2018). These results indicate that greater N-volatilization loss from the non-coated compared to the NBPT-coated urea likely did not occur, which contradicted the original hypothesis that greater fluxes would occur from the NBPT-coated urea because more N would be retained in the soil to stimulate greater aboveground biomass production. From the same Dewitt silt-loam soil and N-fertilization treatments as used in the current study, Rector et al. (2018) also reported that season-long N_2O emissions did not differ between NBPT-coated and non-coated urea. Consequently, the lack of a urea-fertilizer-type effect on dry matter production, yield, and season-long CH_4 and N_2O demonstrates that substantial rice-plant morphological differences, specifically with aerenchyma tissue, do not arise from using either NBPT-coated or non-coated urea to potentially differentially facilitate GHG emissions.

Rice yield was 12% lower from NT (7.8 Mg ha^{-1}) than from CT (8.9 Mg ha^{-1}). Though both tillage treatments had mean soil-test P levels in the low category before planting, the slightly, though significantly, greater P content in CT compared to NT (Table 1) may have contributed to the yield difference between the two tillage treatments. Rice yield measured in this study from CT practices were also slightly lower than expected yield for CL172 (9.2 Mg grain ha^{-1}) grown in Arkansas based on a summary of recent yield trials (Hardke et al., 2014), where site-specific yields measured in this study could have been impacted by the fungal disease false smut (*Ustilaginoidea virens*), which was visually observed to a small degree in 2017 associated with rice grown in both tillage treatments. In contrast to the results of this study, through a global meta-analysis, Pittelkow et al. (2015) reported that NT had no significant effect on rice yield compared to CT. Both NT and CT plots received the same quantity of fertilizer N, but NT was not fertilized with P, whereas CT plots were fertilized with P due to the nature of the P-fertilization treatments the NT plots were a part of that were used in this study.

Soil Redox and Temperature

Methane production is optimal in the soil redox potentials (Eh) range of approximately -200 to -250 mV (Patrick et al., 1996). Based on measured values from the hour during CH_4 flux measurements, soil Eh at the 7-cm depth started near 200 mV but decreased to near 0 mV by 6 DAF under NT and by 24 DAF under CT (Figure 2). Once reached, soil Eh remained near or below -200 mV for the remainder of the season (Figure 2).

Similar to CH_4 fluxes, soil Eh differed ($P < 0.01$) between tillage practices over time during the growing season, but also differed ($P < 0.01$) among tillage-urea-fertilizer-type treatment combinations (Table 2). Soil Eh was greater under CT than NT at 2 and 6 DAF (Figure 2). Averaged across measurement dates, mean soil Eh was greater in the NT/non-coated-urea (-55.6mV) than in the other three treatment combinations, which did not differ and averaged -241 mV. An explanation for the apparent inconsistent differences in soil Eh is not immediately obvious, but may relate to the degree of rhizosphere oxygenation, which would tend to maintain greater soil Eh when well-oxygenated and a lower soil Eh when poorly oxygenated.

Soil temperatures at the 7-cm depth started around 26°C, increased to around 28°C mid-season by 41 DAF, then decreased to below 20°C and continued to decrease after the end-season-drain (86 to 88 DAF; Figure 2). The numerically largest soil temperature was achieved in CT at 41 DAF, with the numerically lowest soil temperature occurring in NT at 87 DAF (Figure 2).

Soil temperature differed between tillage practices over time during the growing season ($P < 0.01$) and differed among tillage-urea fertilizer type treatment combinations ($P = 0.03$) and (Table 3). Averaged over urea fertilizer type, the soil temperature was significantly cooler under NT than CT during the middle of the flooded portion of the rice growing season (i.e., 34, 41, 48, 55, 62, and 70 DAF), but did not differ by more than 2°C on any given date (Figure 2). The cooling effect under NT management likely occurred because of unincorporated residue left by the NT treatment on the soil surface, which attenuated soil profile heating during the middle of the sampling season more than under CT. Averaged over measurement dates, mean soil temperatures were lower and did not differ between urea fertilizer types, averaging 23.5ºC, under NT compared to under CT, where soil mean temperatures were slightly warmer and differed between urea fertilizer types (24.5 and 23.8°C for NBPT-coated and non-coated urea, respectively) under CT. The soil warming was likely due to the lack of crop residue and greater subsequent heating of the soil profile by radiative solar energy under CT than under NT. Brye et al. (2016) reported that diurnal fluctuations of air temperature significantly impacted CH_4 emissions from silt-loam soils in Arkansas. However, the presence of the flood water likely attenuates and minimizes the diurnal fluctuations of air temperature.

Methane Emissions

In contrast to that hypothesized, pre- and post-flood-release and season-long, area- and yield-scaled CH_4 emissions were unaffected ($P > 0.05$) by tillage treatment and urea fertilizer type (Table 3). Though not significant, pre-flood-release CH_4 emissions ranged from 19.1 to 37.2 kg CH_4-C ha^{-1} $period^{-1}$ and averaged 27.8 kg CH_4-C ha^{-1} $period^{-1}$ from CT and ranged from 27.2 to 51.4 kg CH_4-C ha^{-1} $period^{-1}$ and averaged 40.6 kg CH_4-C ha^{-1} $period^{-1}$ from NT (Table 4). Similarly, though not significant, pre-flood-release CH_4 emissions ranged from 27.2 to 51.3 kg CH_4-C ha^{-1} $period^{-1}$ and averaged 36.4 kg CH_4-C ha^{-1} $period^{-1}$ from non-coated urea and ranged from 19.1 to 51.4 kg CH_4-C ha^{-1} $period^{-1}$ and averaged 32.0 kg CH_4-C ha^{-1} $period^{-1}$ from NBPT-coated urea (Table 4). Post-flood-release CH_4 emissions were numerically smaller than those before the flood was released (Table 4) and represented only 4.3 and 3.7% of the measured season-long CH_4 emissions from CT and NT, respectfully. The relatively small proportion of post-flood-release CH_4 emissions was similar what has been reported in recent studies (3.4 to 13.2%), but from different pure-line cultivars (i.e., ‘Taggart’ and ‘Wells’) grown on silt-loam soils under CT and a full-season flood in east-central Arkansas (Brye et al., 2013; Rogers et al., 2013).

Though not significant, season-long, area-scaled CH_4 emissions ranged from 20.3 to 39.2 kg CH_4-C ha^{-1} $season^{-1}$ and averaged 29.0 kg CH_4-C ha^{-1} $season^{-1}$ from CT, whereas season-long, area-scaled CH_4 emissions ranged from 28.3 to 53.8 kg CH_4-C ha^{-1} $season^{-1}$ and averaged 42.2

kg CH_4-C ha^{-1} season^{-1} from NT (Table 4). Though SOM and TC contents in the top 10 cm did not differ between tillage treatments early in the growing season (Table 1), it was likely that both SOM and C were concentrated more towards the soil surface (i.e., upper-most few millimeters), due to the lack of incorporation, which limited the availability of reducible substrate to methanogens, hence limited the production and release of CH_4 from under NT management. Mitra et al. (2002) suggested that the main source of CH_4 in the soil column is in the topsoil, where > 99% of the total soil-produced CH_4 is emitted regardless of the landuse being agriculturally disturbed or natural and relatively undisturbed. Furthermore, since the aerenchyma tissue of the rice plants themselves provides the main mechanism of CH_4 release to the atmosphere via passive transport from below a column of water (Cicerone and Shetter, 1981; Yu et al., 1997; Dannenburg and Conrad, 1999; Groot et al., 2005) and the SOM/C substrate was likely stratified and concentrated right at the soil surface, there was likely little to no opportunity for produced CH_4 molecules to enter the aerenchyma tissue of the rice plant and therefore no mechanism for release to the atmosphere, except for ebullition which is slower than the passive aerenchyma transport (Butterbach-Bahl et al., 1997; Smith et al., 2003). Though not measured directly in this study, it was also possible that the soil redox status right at the soil surface was not reduced enough for substantial CH_4 production, despite the presence of ample SOM/C substrate. In a recent study using the same plots as were used in the current study, Rector et al. (2018) also reported no difference in N_2O emissions between CT and NT practices.

Similar to the lack of a tillage effect, though not significant, season-long, area-scaled CH_4 emissions ranged from 24.8 to 53.3 kg CH_4-C ha^{-1} season^{-1} and averaged 37.8 kg CH_4-C ha^{-1} season^{-1} from non-coated urea, whereas season-long, area-scaled CH_4 emissions ranged from 20.3 to 53.8 kg CH_4-C ha^{-1} season^{-1} and averaged 33.4 kg CH_4-C ha^{-1} season^{-1} from NBPT-coated urea (Table 4). Though it was expected that the N from NBPT-coated urea would create greater biomass due to slower release and greater N retention in the soil, consequently resulting in greater CH_4 emissions, than from non-coated urea, this was not observed as hypothesized, as aboveground dry matter production and yield were similar between urea fertilizer types. Thus, it was concluded that the same amount of aerenchyma tissue was produced between the two urea-fertilizer-type treatments that facilitated the same magnitude of season-long CH_4 emissions. Regardless of urea fertilizer type, the magnitude of season-long CH_4 emissions from optimally N-fertilized rice measured in this study were lower than that reported from recent studies conducted on silt-loam soils in east-central Arkansas (Rogers et al., 2013; Humphreys et al., 2018).

Similar to area-scaled emissions, season-long, yield-scaled CH_4 emissions, which represented an emissions intensity metric, ranged from 4.1 to 4.4 kg CH_4-C (Mg grain)$^{-1}$ and averaged 4.25 kg CH_4-C (Mg grain)$^{-1}$ across urea fertilizer types, whereas season-long, yield-scaled CH_4 emissions ranged from 3.2 to 5.4 kg CH_4-C (Mg grain)$^{-1}$ and averaged 4.3 kg CH_4-C ha^{-1} season^{-1} across tillage treatments (Table 4). The emissions intensities measured in this study are similar to and within the range [2.52 to 7.39 kg CH_4-C (Mg grain)$^{-1}$] reported by Humphreys et al. (2018) for rice grown in 2015 in a Dewitt a silt loam in Arkansas.

Agronomic and Environmental Implications

Reducing the GHG load to the atmosphere will be necessary to mitigate global climate change and its potentially disastrous long-term effects on the environment (IPCC, 2014). However, before the GHG load can be reduced, it will be necessary to increase understanding of the agronomic practices that affect GHG emissions, which necessitates careful characterization of rice production practices that affect CH_4 emissions. Though measurements were made over the course of only one growing season, results of this field study, the first of which conducted in Arkansas, the leading rice-producing state in the United States, to evaluate the effects of tillage practice and urea fertilizer type, clearly showed that season-long CH_4 emissions did not differ between CT and NT or between NBPT-coated and non-coated urea.

Rice producers considering the adoption of alternatives practices for increased sustainability may not achieve substantial benefits from NT, in terms of reduced CH_4 emissions, as might be expected for other soil properties and processes. However, implementing NT compared to continuing with CT, coupled with similar, rather than greater, CH_4 emissions from NT compared to CT, may provide an impetus for changing tillage practices.

Though designed to inhibit urea breakdown, fertilizing rice with NBPT-coated urea is also more costly than using non-coated urea. However, results of this study showed that non-coated urea could potentially be used in place of NBPT-coated urea without increasing CH_4 emissions, which was also shown recently to be the case for N_2O emissions (Rector et al., 2018). In addition, season-long N_2O emissions were also low from a full-season flood treatment, which minimized the fluctuations in soil Eh that would have promoted N_2O production and release (Rector et al., 2018).

Since numerous other factors have been shown to significantly influence CH_4 emissions from rice production, such as cultivar selection (Rogers et al., 2014; Humphreys et al., 2018), soil texture (Brye et al., 2013; Smartt et al., 2016), and water management scheme (Humphreys et al., 2018), results of this study suggest that climate-change modelers may not need to account for tillage practice or urea fertilizer type when attempting to estimate large-scale, regional CH_4 emissions from rice produced from a silt-loam soil in a direct-seeded, delayed flood production system. Consequently, the results of this study have provided evidence to narrow the pool of significant soil and agronomic factors needed to consider for model estimation purposes. It is studies like the present study that will continue to be necessary to conduct under field conditions to further refine current knowledge regarding factor affecting CH_4 emissions in regions of concentrated rice production, such as is eastern Arkansas.

Conclusions

This field study was the first to examine the effects of tillage (CT and NT), urea fertilizer type (NBPT-coated and non-coated urea), and their interaction on CH_4 fluxes and emissions from a pure-line rice cultivar grown in a silt-loam soil in the direct-seeded, delayed-flood production system in east-central Arkansas. Similar to that hypothesized, CH_4 fluxes were greater from NT than CT at times over the 2017 rice growing season. However, in contrast to that hypothesized, CH_4 fluxes were unaffected by urea fertilizer type and CH_4 emissions were unaffected both tillage treatment and urea fertilizer type. Results of this study will be valuable information when contemplating new policies and recommendations for future rice production practices and sustainability in the mid-southern United States, particularly eastern Arkansas.

Though the results of this study were based on one growing-season of measurements, these results, indicate consistent CH_4 emissions and flux trend responses from year to year at least partially due to the presence of the flood water for most of the growing season attenuating inter-annual differences in growing-season weather conditions. Therefore, it is reasonable to conclude that these results can be extrapolated to similar field conditions, as minor differences in growing-season weather conditions from year to year likely to continue to have minimal effect on CH_4 emissions.

The importance of rice production to the state of Arkansas makes continued quantification of GHG emissions, specifically CH_4, from traditionally common and alternative rice production practices vital to mitigating global climate change. With rice a staple food for a substantial portion of the current human population, continued research into the effects of rice production practices on CH_4 emission is warranted as the global population continues to rise, which will require increased, yet sustainable, production, while simultaneously protecting the environment.

Literature Cited

Ahmad, S., C. Li, G. Dai, M. Zhan, J. Wang, S. Pan, and C. Cao. 2009. Greenhouse gas emission from direct seeding paddy field under different rice tillage systems in central China. Soil Till. Res. 106:54-61.

Brye, K.R., C.W. Rogers, A.D. Smartt, and R.J. Norman. 2013. Soil texture effects on methane emissions from direct-seeded, delayed-flood rice production in Arkansas. Soil Sci. 178:519-529.

Brye, K.R., L.L. Nalley, J.B. Tack, B.L. Dixon, A.P. Barkley, C.W. Rogers, A.D. Smartt, R.J. Norman, and K.S.V. Jagadish. 2016. Factors affecting methane emissions from rice production in the Lower Mississippi river valley, USA. Geod. Reg. 7:223-229.

Butterbach-Bahl, K., H. Papen, and H. Rennenberg. 1997. Impact of gas transport through rice cultivars on methane emission from rice paddy fields. Plant Cell Environ. 997:1175-1183.

Cicerone, R.J., and J.D. Shetter. 1981. Sources of atmospheric methane: measurements in rice paddies and a discussion. J. Geophys. Res. 86:7203-7209.

Dannenburg, S., and R. Conrad. 1999. Effect of rice plants on methane production and rhizospheric metabolism in paddy soil. Biogeochem. 45:53-71.

Forster, P., V. Ramaswamy, P. Artaxo, T. Berntsen, R. Betts, D.W. Fahey, J. Haywood, J. Lean, D.C. Lowe, G. Myhre, J. Nganga, R. Prinn, G. Raga, M. Schulz, and R. Van Dorland. 2007. Changes in atmospheric constituents and in radiative forcing. Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change [Solomon, S., D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M.Tignor and H.L. Miller (editors.)].Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Gee, G.W., and D. Or. 2002. Particle size analysis. In: J.H. Dane, editor Methods of Soil Analysis: Part 4 Physical Methods. pp. 255-293.

Groot, T.T., P.M. van Bodegom, H.A.J. Meijer, and F.J.M. Harren. 2005. Gas transport through the root–shoot transition zone of rice tillers. Plant Soil 277:107-116.

Hardke, J.T. 2014. Arkansas Rice Production Handbook. In: J. T. Hardke, editor Introduction. University of Arkansas Division of Agriculture, Cooperative Extension Service, Little Rock, AR.

Hardke, J.T. 2016. Trends in Arkansas rice production. In: R.J. Norman, and K.A.K. Moldenhauer, editors, B.R. Wells Arkansas Rice Research Studies, 2015. Univ. Ark. Agric. Exp. Sta. Res. Ser. 634., Univ. Ark. Agric. Exp. Sta., Fayetteville, AR. pp. 13-23.

Hardke, J.T. 2017. Trends in Arkansas rice production. In: R. J. Norman, and K. A. K. Moldenhauer, editors, B.R. Wells Arkansas Rice Research Studies, 2016. Univ. Ark. Agric. Exp. Sta. Res. Ser. 634., Univ. Ark. Agric. Exp. Sta., Fayetteville, AR. pp. 11-14.

Havlin, J.L., S.L. Tisdale, and W.L. Nelson. 2014. Soil Fertility and Fertilizers: An introduction to nutrient management. 8th ed. Pearson Education Inc., Upper Saddle River, NJ. p. 516.

Humphreys, J., K.R. Brye, C. Rector, E.E. Gbur, and J.T. Hardke. 2018. Water management and cultivar effects on methane emissions from direct-seeded, delayed-flood rice production in Arkansas. J. Rice Res. Dev. 1:14-24.

Intergovernmental Panel on Climate Change (IPCC). 2014. Climate Change 2014: Synthesis Report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Core Writing Team, R.K. Pachauri and L.A. Meyer (editors.)]. IPCC, Geneva, Switzerland.

Intergovernmental Panel on Climate Change (IPCC). 2014. Intergovernmental Panel on Climate Change (IPCC), United States Greenhouse Gas Inventory Chapter 6. IPCC, Geneva, Switzerland. pp. 15-21.

Liu, X.J., A.R. Mosier, A.D. Halvorson, and F.S. Zhang. 2006. The impact of nitrogen placement and tillage on NO, N2O, CH4, and CO2 fluxes from a clay loam soil. Plant Soil 280:177-188.

Livingston, G., and G. Hutchinson. 1995. Enclosure-based measurement of trace gas exchange: Applications and sources of error. In: P. A. Matso, and R. C. Harriss, editors, Biogenic trace gases measuring emissions from soil and water. Blackwell Science, Oxford [England]; Cambridge, MA, USA. pp. 14-51.

Mitra, S., R. Wassman, M. Jain, and H. Patak. 2002. Properties of rice soils affecting methane production protentials: 2. Differences in topsoil and subsoil. Nutr. Cycl. Agro. 64:183-191.

National Oceanic & Atmospheric Administration (NOAA). 2015. National Environmental Satellite, Data, and Information Service. Annual Climatological Summary (2015). National Centers for Environmental Information, U.S. Department of Commerce, Asheville, NC.

Nelson, D.W., and L.E. Sommers. 1996. Total carbon, organic carbon, and organic matter. Methods of soil analysis. Part 3: Chemical analysis 3rd ed. Soil Science Society of America, Madison, WI.

Norman, R.J., C.E. Wilson, and N.A. Slaton. 2003. Chapter 3.4: Soil fertilization and mineral nutrition in U.S. mechanized rice culture. In: C. Smith, and R. Dilday, editors, Rice: Origin, history, technology, and production. John Wiley & Sons, Inc., Hoboken, NJ.

Norman, R.J., N.A. Slaton, and T.L. Roberts. 2013. Soil Fertility. In: J.T. Harke, editor Arkansas Rice Production Handbook Misc. Publ. 192. University of Arkansas Division of Agriculture Cooperative Extension Service, Little Rock, AR, USA. pp. 69-102.

Parkin, T.B., and R.T. Venterea. 2010. Sampling protocols. In: R.F. Follet USDA-ARS GRACEnet Project Protocols, editor Chapter 3. Chamber-Based Trace Gas Flux Measurements. pp. 3-1 to 3-39.

Parvej, M., N.A. Slaton, L. Purcell, and T. Roberts. 2016. Critical trifoliate leaf and petiole potassium concentrations during the reproductive stages of soybeans. Agron. J. 108:2502-2518.

Patrick, W.H., R.P. Gambrell, and S.P. Faulkner. 1996. Redox measurements of soils. In: D.L. Sparks, editor Methods of Soil Analysis. part 3 - Chemical Methods. 3rd ed. SSSA, Madison, WI, USA. pp. 1255-1273.

Pittelkow, C.M., B.A. Linquist, M.E. Lundy, X. Liang, K.J. Groenigen, J. Lee, N. Gestel, J. Six, R.T. Venterea, and C. Kessel. 2015. When does no-tillage yield more? A global meta-analysis. Field Crops Res. 183:156-168.

Rector, C., K.R. Brye, J. Humphreys, R.J. Norman, N.A. Slaton, E.E. Gbur, C. Willett, and M.A. Evans-White. 2018. Tillage and coated-urea effects on nitrous oxide emissions from direct-seeded, delayed-flood rice production in Arkansas. J. Rice Res. Dev. 1:25-37.

Rogers, C.W., K.R. Brye, R.J. Norman, E.E. Gbur, J.D. Mattice, T.B. Parkin, and T.L. Roberts. 2013. Methane emissions from drill-seeded, delayed-flood rice production on a silt-loam soil in Arkansas. J. Environ. Qual. 42:1059-1069.

Rogers, C.W., K.R. Brye, A.D. Smartt, R.J. Norman, E.E. Gbur, and M.A. Evans-White. 2014. Cultivar and previous crop effects on methane emissions from drill-seeded, delayed-flood rice production on a silt-loam soil. Soil Sci. 179:28-36.

Slaton, N.A., T.L. Roberts, R.J. Norman, J. Hardke, R.E. DeLong, J.B. Shafer, G. Massey, and S.D. Clark. 2013. Rice and soybean response to short- and long-term phosphorus and potassium fertilization rate. In: R. J. Norman, and K. A. K. Moldenhauer, editors, B.R. Wells Arkansas Rice Research Studies, 2013. Univ. Ark. Agric. Exp. Sta. Res. Ser. 617., Univ. Ark. Agric. Exp. Sta., Fayetteville, AR. pp. 332-338.

Slaton, N.A., J.T. Hardke, and T.L. Roberts. 2017. Evaluation of Mehlich-3 extraction of field-moist and oven-dried soil from long-term fertilization trials. In: N.A. Slaton, editor Wayne E. Sabbe Arkansas Soil Fertility Studies, 2016. Res. Series 642. pp. 29-33.

Smartt, A.D., K.R. Brye, C.W. Rogers, R.J. Norman, E.E. Gbur, J.T. Hardke, and T.L. Roberts. 2016. Characterization of methane emissions from rice production on a clay soil in Arkansas. Soil Sci. 181:57-67.

Smith, K.A., T. Ball, F. Conen, K.E. Dobbie, J. Massheder, and A. Rey. 2003. Exchange of greenhouse gases between soil and atmosphere: interactions of soil physical factors and biological processes. Eur. J. Soil Sci. 54:779-791.

Smith, P., D. Martino, Z. Cai, D. Gwary, H. Janzen, P. Kumar, B. McCarl, S. Ogle, F. O'Mara, C. Rice, B. Scholes, and O. Sirotenko. 2007. Agriculture. In Climate Change 2007: Mitigation. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change [B. Metz, O.R. Davidson, P.R. Bosch, R. Dave, L.A. Meyer (editors)], Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Tucker, M.R. 1992. Determination of phosphorous by Mehlich 3 extraction. In: S. J. Donohue, editor, Soil and media diagnostic procedures for the Southern Region of the United States. Virginia Agric. Exp. Stn. Ser. Bull. 374., Blacksburg, VA, USA. pp. 6-8.

United States Environmental Protection Agency (USEPA). 2017. United States Environmental Protection Agency (USEPA), Inventory of U.S. greenhouse gas emissions and sinks: 1990-2015. Available at: https://19january2017snapshot.epa.gov/ghgemissions_.html (accessed 10 Jan. 2018).

Yu, K.W., Z.P. Wang, and G.X. Chen. 1997. Nitrous oxide and methane transport though rice plants. Biol. Fert. Soils 24:341-343.

Appendices

Table 1. Analysis of variance (ANOVA) summary of the effects of tillage practice [conventional tillage (CT) and no-tillage (NT)], urea fertilizer type [i.e., N-(n-butyl) thiophosphoric triamide (NBPT)-coated and non-coated urea], and their interaction on sand, silt, clay, bulk density, pH, electrical conductivity (EC), extractable soil nutrient (P, K, Ca, Mg, Fe, Mn, Na, S, Zn, and Cu) contents, total nitrogen (TN), total carbon (TC), and soil organic matter (SOM) contents, and the C:N ratio in the top 10 cm from rice grown on a silt-loam soil in the direct-seeded, delayed-flood production system at the Rice Research and Extension Center near Stuttgart, AR in 2017. Overall mean values by tillage treatment are also reported for each soil property. Bolded values represent significant effects ($P < 0.05$).

Soil property	Tillage	Fertilizer	Tillage x fertilizer	Overall mean (NT)	Overall mean (CT)
		P			
Sand (g g^{-1})	0.38	0.24	0.24	0.15a	0.13a
Silt (g g^{-1})	0.76	0.18	0.30	0.71a	0.71a
Clay (g g^{-1})	0.24	0.99	0.45	0.14a	0.16a
Bulk density (g cm^{-3})	< 0.01	0.02	**0.04**	1.19	1.38
pH	**0.03**	0.08	0.38	5.43b	6.09a
EC (dS m^{-1})	0.38	0.93	0.25	0.19a	0.21a
P (kg ha^{-1})	**0.04**	0.48	0.70	15.9b	18a
K (kg ha^{-1})	0.80	0.02	**0.03**	146	143
Ca (Mg ha^{-1})	0.10	0.38	0.22	1.16a	1.49a
Mg (kg ha^{-1})	**0.04**	0.91	0.30	162	260a
S (kg ha^{-1})	0.69	0.76	0.78	15.1a	14.6a
Na (kg ha^{-1})	**< 0.01**	0.40	0.28	52b	97.4a
Fe (kg ha^{-1})	**0.02**	0.54	0.66	507a	424b
Mn (kg ha^{-1})	**< 0.01**	0.67	0.33	219b	289a
Zn (kg ha^{-1})	**< 0.01**	0.79	0.64	6.09a	2.91b
Cu (kg ha^{-1})	0.16	0.91	0.98	1.41a	1.62a
TN (kg ha^{-1})	0.66	0.22	0.35	903a	853a
TC (Mg ha^{-1})	0.53	0.20	0.21	9.23a	8.49a
SOM (Mg ha^{-1})	0.70	0.27	0.17	23.1a	23.6a
C:N ratio	0.23	0.68	0.34	10.20a	9.97a

Table 2. Analysis of variance summary of the effects of tillage practice (conventional tillage and no-tillage), urea fertilizer type [i.e., N-(n-butyl) thiophosphoric triamide (NBPT)-coated and non-coated urea], time as days after flooding (DAF), and their interactions on methane fluxes, soil oxidation-reduction (redox) potential, and soil temperature from rice grown on a silt-loam soil in the direct-seeded, delayed-flood production system at the Rice Research and Extension Center near Stuttgart, AR in 2017. Bolded values represent significant effects ($P < 0.05$).

Property/Treatment effect	*P*
Methane flux	
Tillage	0.17
Fertilizer	0.22
DAF	< 0.01
Tillage x fertilizer	0.60
Tillage x DAF	**< 0.01**
Fertilizer x DAF	0.81
Tillage x fertilizer x DAF	0.35
Soil redox potential	
Tillage	0.96
Fertilizer	0.48
DAF	< 0.01
Tillage x fertilizer	**< 0.01**
Tillage x DAF	**< 0.01**
Fertilizer x DAF	0.95
Tillage x fertilizer x DAF	0.94
Soil temperature	
Tillage	0.53
Fertilizer	0.22
DAF	< 0.01
Tillage x fertilizer	**0.03**
Tillage x DAF	**< 0.01**
Fertilizer x DAF	0.65
Tillage x fertilizer x DAF	0.67

Table 3. Analysis of variance (ANOVA) summary of the effects of tillage practice (conventional tillage and no-tillage, urea fertilizer type [i.e., N-(n-butyl) thiophosphoric triamide (NBPT)-coated and non-coated urea], and their interactions on aboveground dry matter, grain yield, pre- and post-flood-release and season-long, area- and yield-scaled methane emissions from rice grown on a silt-loam soil in the direct-seeded, delayed-flood production system at the Rice Research and Extension Center near Stuttgart, AR in 2017. Bolded values represent significant effects ($P < 0.05$).

Property/Treatment effect	*P*
Aboveground dry matter	
Tillage	< **0.01**
Fertilizer	0.61
Tillage x fertilizer	0.48
Grain yield	
Tillage	< **0.01**
Fertilizer	0.54
Tillage x fertilizer	0.41
Pre-flood-release emissions	
Tillage	0.11
Fertilizer	0.15
Tillage x fertilizer	0.55
Post-flood-release emissions	
Tillage	0.32
Fertilizer	0.99
Tillage x fertilizer	0.94
Season-long, area-scaled emissions	
Tillage	0.11
Fertilizer	0.21
Tillage x fertilizer	0.71
Season-long, yield-scaled emissions	
Tillage	0.06
Fertilizer	0.21
Tillage x fertilizer	0.14

Table 4. Mean pre- (i.e., establishment of the flood to end-of-season flood release) and post-flood-release (i.e., after end-of-season flood release) methane (CH_4) emissions and emissions intensity among tillage practices (conventional tillage and no-tillage) and urea fertilizer types [i.e., N-(n-butyl) thiophosphoric triamide (NBPT)-coated and non-coated urea] from rice grown on a silt-loam soil in the direct-seeded, delayed-flood production system at the Rice Research and Extension Center near Stuttgart, AR in 2017.

Treatment	Pre-flood-release CH_4 emissions (kg CH_4-C ha^{-1} $season^{-1}$)	Post-flood-release CH_4 emissions (kg CH_4-C ha^{-1} $season^{-1}$)	Emissions intensity [kg CH_4-C (Mg grain) $^{-1}$]
Conventional tillage	27.8 (2.0)	1.24 (0.16)	3.2 (0.22)
No-tillage	40.6 (3.2)	1.57 (0.17)	5.4 (0.34)
NBPT-coated urea	32 (3.6)	1.40 (0.17)	4.1 (0.53)
Non-coated urea	36.4 (3.4)	1.40 (0.19)	4.4 (0.46)

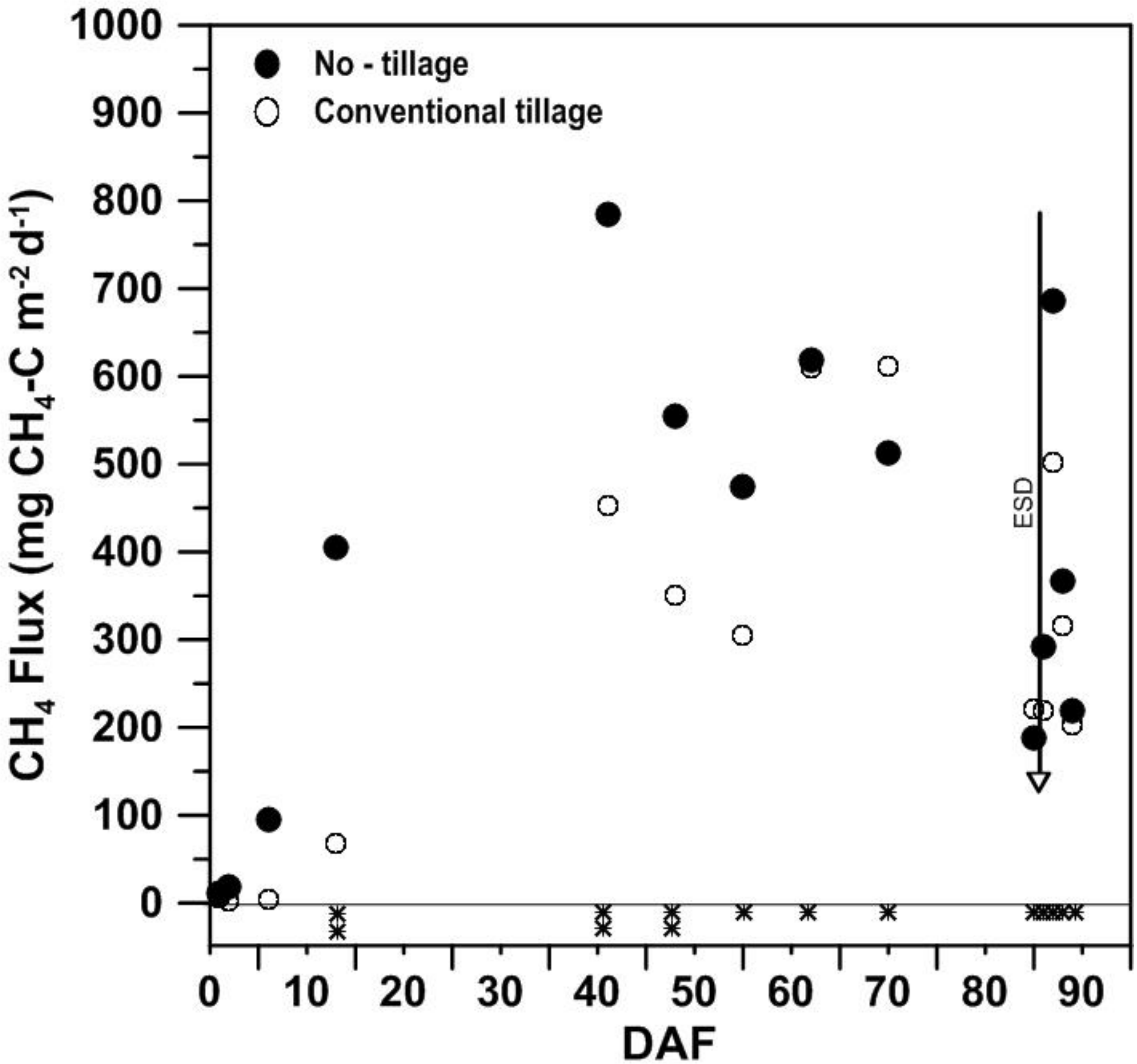


Figure 1. Tillage differences in methane (CH_4) fluxes over time [days after flooding (DAF)] during the 2017 rice growing season at the Rice Research and Extension Center near Stuttgart, AR. The arrow (↓) indicates the date of the end-of-season (ESD) of the flood from the field (85 DAF). A single asterisk (*) on a given measurement date indicates a significant ($P < 0.05$) difference exists from a flux of zero. A double asterisk (*) on a given measurement date indicates a significant ($P < 0.05$) difference exists from a flux of zero and between tillage treatments.

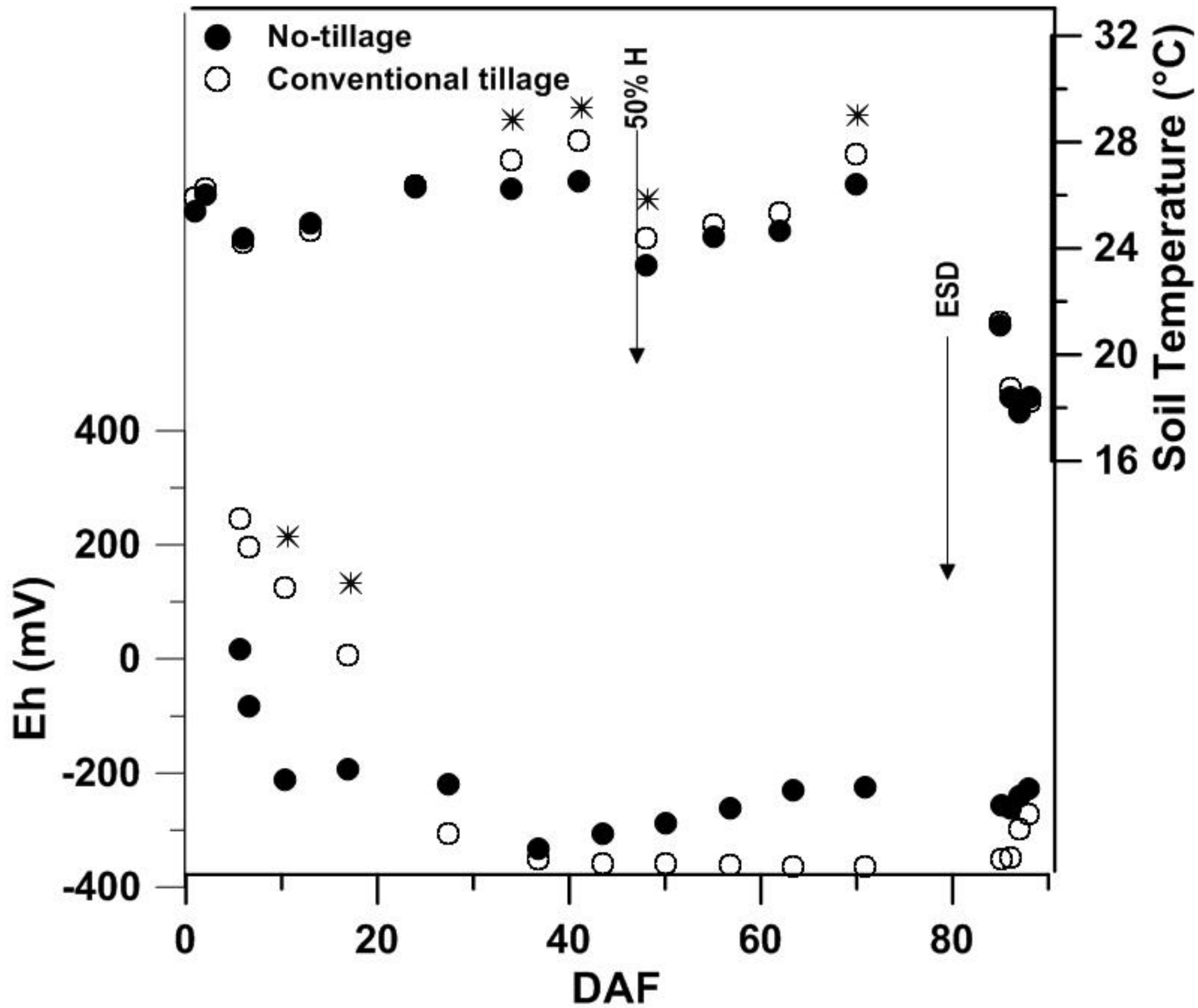
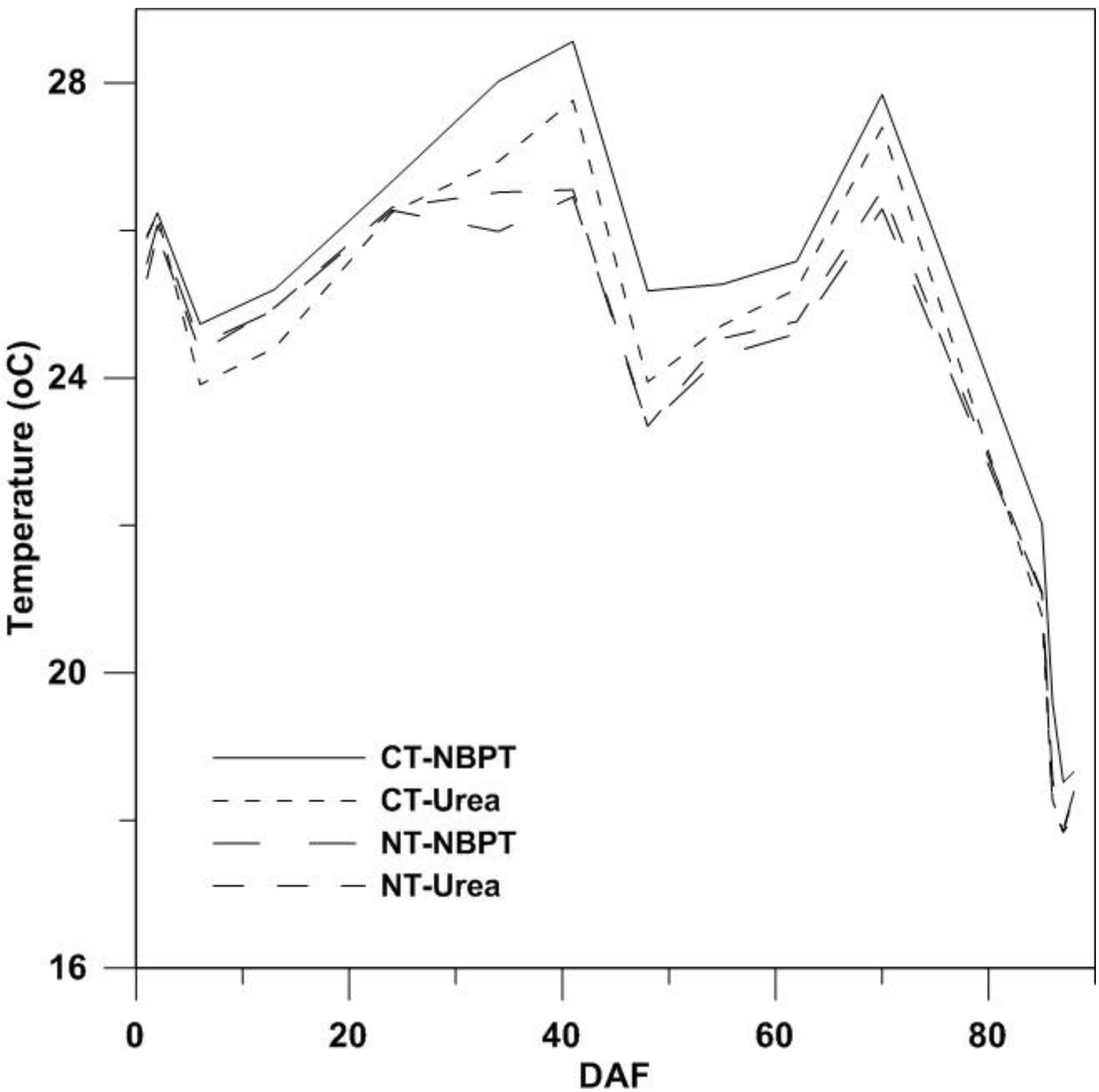
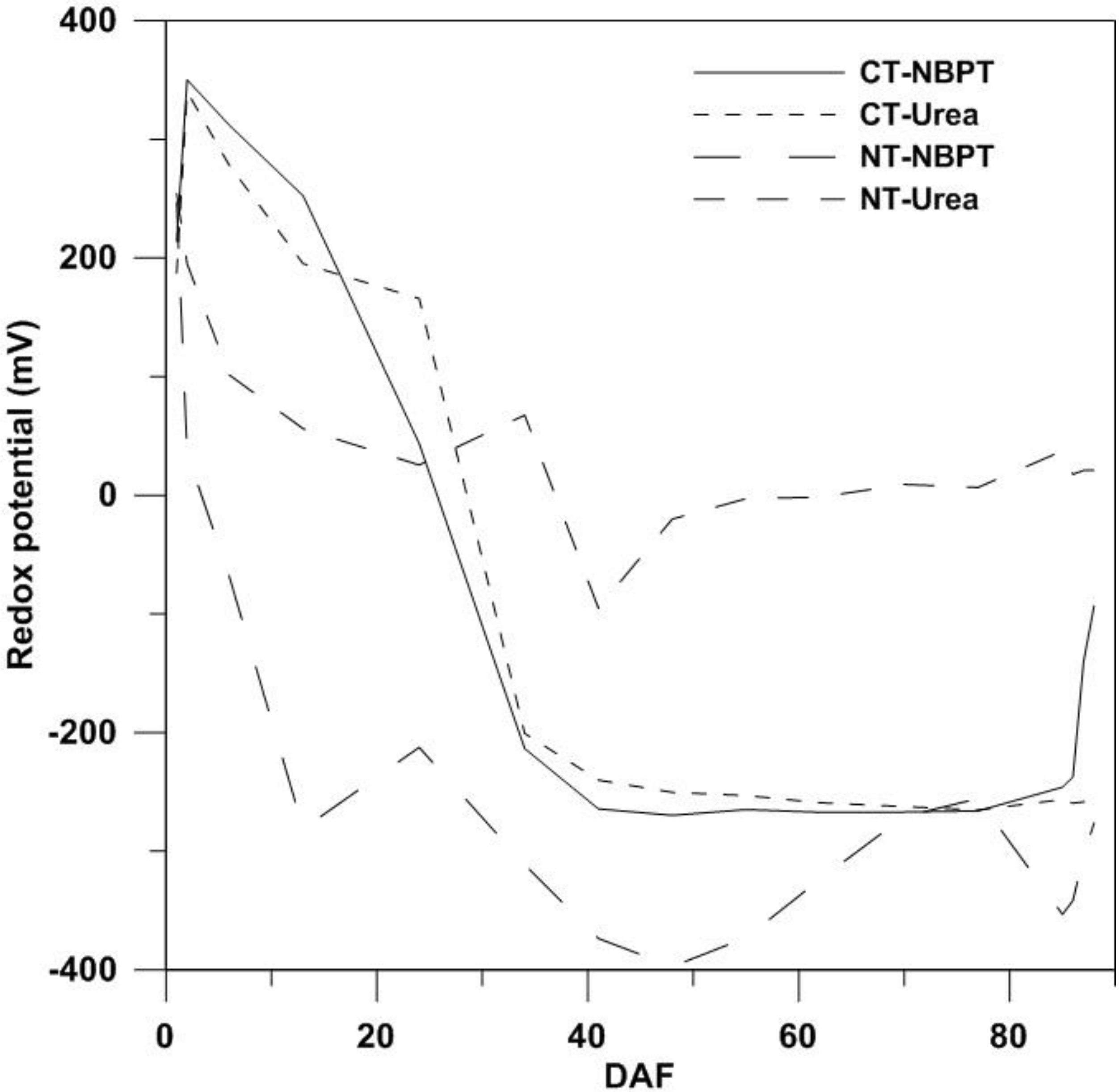


Figure 2. Tillage differences, averaged across urea fertilizer types, in soil redox potential (Eh) and soil temperature at the 7.5-cm depth over time [days after flooding (DAF)] during the 2017 rice growing season at the Rice Research and Extension Center near Stuttgart, AR. Arrows (↓) indicate the occurrence of 50% heading (50% H; 53 DAF)] and the end-of-season (ESD) drain of the flood (85 DAF). An asterisks (*) represents a significant difference ($P < 0.05$) between tillage treatment on that date.

Appendix A



Season-long profile of soil temperature measured at the 7-cm soil depth over time [days after flooding (DAF)] for no-tillage (NT) and conventional tillage (CT) and urea treatment [coated (NBPT) and non-coated urea (Urea)] during the 2017 season at the Rice Research and Extension Center near Stuttgart, AR.



Season-long profile of oxidation-reduction potential (Eh), measured at the 7-cm soil depth, over time [days after flooding (DAF)] for no-tillage (NT) and conventional tillage (CT) and urea treatment [coated (NBPT) and non-coated urea (Urea)] during the 2017 season at the Rice Research and Extension Center near Stuttgart, AR

Appendix B

Example of SAS program for evaluating CH_4 fluxes over time between tillage practices and pre-assigned type of urea fertilizer for the 2017 season.

```
title 'Methane Field Study 2017 - Joshua Humphreys';
title2 'Methane Fluxes 2017 ANOVA';
data methane2017;
  infile 'CH4Flux2017.prn' firstobs=2;
  input ID DAF Block tillage $ fert $ flux;
 run;

proc sort data=methane2017; by  DAF;
quit;

Proc print data=methane2017 noobs;by DAF;
id DAF;
var tillage fert flux;
run;

proc mixed data=methane2017 method=type3;
class  cultivar tillage fert block;
model flux = tillage fert tillage*fert DAF DAF*fert DAF*tillage DAF* tillage*fert / ddfm=kr ;
random  Block block*tillage block*fert ;
ods exclude FitStatistics Tests3 IterHistory ;
lsmeans DAF
```

Example of SAS program for evaluating season-long, area-and yield-scaled, and pre- and post-flood-release CH_4 emissions between tillage practices and type of urea fertilizer for the 2017 season.

```
title 'Methane Field Study 2017 - Joshua Humphreys';
title2 'Emission Methane 2017 ANOVA';
data methane2017;
  infile 'CH4Emissions2017.prn' firstobs=2;
  input ID Block tillage $ fert $ Emission;
 run;

proc sort data=methane2017; by tillage fert;
quit;

proc print data=methane2017 noobs; by fert;
  id ;
  var tillage Emission;
  run;

proc mixed data=methane2017 method=type3;
class  fert tillage block;
model emission = tillage fert tillage*fert  / ddfm=kr  ;
random   Block  block*tillage ;
ods exclude FitStatistics Tests3 IterHistory   ;
*lsmeans tillage  ;
quit
```

Example of SAS program for evaluating yield, aboveground biomass, soil redox potential, and soil temperature between tillage practices and type of urea fertilizer for the 2017 season.

```
title 'Methane Field Study 2017 - Joshua Humphreys';
title2 'Emission Methane 2017 ANOVA';
data methane2017;
  infile 'CH4 Bio 2017.prn' firstobs=2;
  input ID Block tillage $ fert $ bio;
 run;

proc sort data=methane2017; by tillage fert;
quit;

proc print data=methane2017 noobs; by fert;
  id ;
  var tillage;
  run;

proc mixed data=methane2017 method=type3;
class  fert tillage block;
model bio = tillage fert tillage*fert  / ddfm=kr  ;
random   Block  block*tillage ;
ods exclude FitStatistics Tests3 IterHistory   ;
*lsmeans tillage  ;
quit;
```

Example of SAS program data for evaluating soil properties between tillage practices and pre-assigned type of urea fertilizer for the 2017 season.

```
title 'Methane Field Study - Initial Soil Sample Analysis 2017 - Joshua Humphreys';
title2 'Soil Data CH4 2017 ANOVA';
data soildata2017;
  infile 'soil properties2017.prn' firstobs=2;
  input id block tillage $ fert $ ph ec p k ca mg su na fe mn zn cu N C LOI CN ;
 run;

proc sort data=soildata2017; by tillage fert;
quit;

proc print data=soildata2017 noobs; by fert;
  id ;
  var fert block tillage ph ec p k ca mg su na fe mn zn cu N C LOI CN ;
  run;
quit;

title3 ANALYSIS OF VARIANCE FOR SOIL PROPERTIES';
proc mixed data=soildata2017 method=type3 ;
class block fert tillage;
model ph =  fert tillage fert*tillage / ddfm=kr ;
random block block*tillage;
ods exclude FitStatistics Tests3 IterHistory;
lsmeans tillage  / diff ;
quit;
```

Conclusions

Results of this dissertation indicate potentially positive impacts of alternative growing techniques and their impacts on trace gas emissions in southeastern Arkansas rice culture. The first study initially focused on water management schemes (mid-season drain and full-season flood) combined with specific cultivar selection ('LaKast' and 'XL753') to reduce methane (CH_4) fluxes and season-long emissions in Arkansas rice production. Similar to that hypothesized, the 2015 growing season demonstrated that, regardless of cultivar selection, mid-season draining of flood water significantly reduced season-long, area-scaled CH_4 emissions compared to the full-season-flood water management practice in the direct-seeded, delayed-flood production system on a silt-loam soil in east-central Arkansas. This study also clearly showed that the mid-season-drain/hybrid (XL753) combination had the lowest CH_4 emissions per unit grain yield (i.e., the lowest emissions intensity) among all water management/cultivar treatment combinations evaluated.

The 2016 study was, to the author's knowledge, the first field experiment to select, transport, and combine multiple soil treatments from various locations around Arkansas into a single study at one location so that environmental factors (i.e., precipitation, air temperature variations) and production treatments (i.e., planted rice cultivar 'LaKast', N fertilization, water management) could be uniform among soil treatments, with the main variables being soil organic matter (SOM) and/or total carbon (TC) content. Verifying the hypothesis, results of this study showed that CH_4 emissions and emissions intensity were greatly affected by initial SOM/TC contents and confirmed a strong, positive relationship between season-long, area-scaled CH_4 emissions and TC and SOM contents in the top 10 cm. This information can be useful in determining potential greenhouse (GHG) impacts when deciding to bring previously undisturbed land into rice production.

The 2017 study was the first to examine the effects of tillage [conventional tillage (CT) and no-tillage (NT)], urea fertilizer type (N-(n-butyl) thiophosphoric triamide (NBPT)-coated and non-coated urea), and their interaction on CH_4 fluxes and emissions from a pure-line rice cultivar grown in a silt-loam soil in the direct-seeded, delayed-flood production system in east-central Arkansas. Similar to that hypothesized, CH_4 fluxes were greater from NT than CT at times over the 2017 rice growing season. However, in contrast to that hypothesized, CH_4 fluxes were unaffected by urea fertilizer type and CH_4 emissions were unaffected by tillage treatment (CT and NT) and urea fertilizer type (NBPT-coated and non-coated urea).

Climate change is at least partially driven by anthropogenic GHG emissions. Consequently, research efforts to identify logical and feasible alternative rice production practices, such as the mid-season drain/hybrid combination, that decrease CH_4 and other GHG emissions need to continue. Furthermore, continued investigation, particularly direct field measurements, is critically necessary to better understand the effects of various alternative water management practices, current rice cultivars, SOM/TC, and their combinations on CH_4 emissions from silt-loam soils in Arkansas and other regions of concentrated rice production. Rice production must attain a level of sustainability that will aid the goal of feeding an ever-growing human population, and GHG emissions are a key part of this modern puzzle. There is a responsibility to maximize production of staple grains while bearing in mind that humans must equally protect future generations from the devastating effects of global climate change. This dissertation will provide valuable information when contemplating new policies and recommendations for future rice production practices and sustainability in the mid-southern

United States, particularly eastern Arkansas. The importance of rice production to the state of Arkansas makes continued quantification of GHG emissions, specifically CH_4, from traditionally common and alternative rice production practices vital in the future.