LAND USE EFFECTS ON GREENHOUSE GAS PRODUCTION IN LOWER MISSOURI RIVER FLOODPLAIN SOILS

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by

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The undersigned, appointed by the dean of the Graduate School, have examined the thesis entitled

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ABSTRACT

Agricultural sources account for significant proportions of global anthropogenic production of some major greenhouse gases (GHG), such as nitrous oxide and methane. The effects of these gases have been strongly linked to past and present climate change, and are predicted to correlate with future climate change based on the predictions of model simulations. Soil-based fluxes of GHG are produced primarily through plant and microbial processes and are affected by soil physical, chemical, and biological properties. The lower Missouri River Floodplain (MRF) region encompasses many different land use systems including agriculture and riparian forest. The effects of these different land use systems in the MRF on soil GHG (i.e., carbon dioxide, nitrous oxide and methane) emissions has been little studied possibly partly because of the challenges of measuring GHG fluxes due to the occurrence of periodic flooding and the high spatial variability in soil properties in this region. The goal of this study was to investigate the influences of lower MRF land use systems on the spatial and temporal variations of soil GHG emissions. The specific objectives were to evaluate soil GHG emissions (CO_2, CH_4, N_2O) in floodplain soils under agroforestry, row-crop agriculture, and forested systems in

response to differences in soil water content, temperature, land use, and N inputs. The project contains two distinct portions, a laboratory and in situ field study.

For the laboratory incubation study intact soil surface cores (0 – 10 cm depth) were obtained from row crop agricultural, agroforestry and riparian forest sites within the MRF near New Franklin, Missouri and incubated for a period of 94 days. Cores were subjected to three water regime treatments: Flooded (FLD), Optimal for CO_2 efflux (OPT), and Fluctuating (FLX). For each of 20 sampling dates, the cores were placed in sealed 2 L plastic jars and the headspace flushed with He gas. The soil CO_2, CH_4, and N_2O fluxes were determined by sampling the head space at three time points (0, 20, 40 min. after sealing).

Soil CO_2 emissions were associated over time by both the MRF land use system and the soil moisture regime. Riparian forest (FOR) soils had higher cumulative CO_2 emission rates than those in fertilized agriculture (AG-N) despite no significant differences ($p \leq 0.05$) in initial soil particulate organic matter (POM) or $KMnO_4$-oxidizable C. OPT sample cumulative emissions were significantly higher ($p \leq 0.05$) than those of FLD and FLX. Soil CH_4 emissions showed no influence of land use, but were influenced by the water regime and length of time under flooding conditions. FLD samples showed a significant difference in cumulative soil CH_4 emissions from other samples at Day 51 onward when E_h had significantly decreased. FLD CH_4 efflux rates were significantly higher than under the OPT and FLX soil moisture regimes. Time, land use, and water regime all showed independent and interactive influences on the surface cumulative N_2O emissions. AG-N and AF-N emitted the most N_2O among the land use and N treatments. Loss of N through soil N_2O emissions decreased as the incubation

progressed. These results suggest that land use management, the soil moisture regime, and soil N content can influence the amounts of GHG emitted in the MRF.

The in situ study component was comprised of three spatially intensive samplings through the 2011 growing season. Soil GHG (CO_2, CH_4, N_2O) flux and soil properties were measured from agriculture, agroforestry, and riparian forested areas. Within each land use type, a representative 0.10 ha area was delineated and 45 sample chambers were installed for assessment of spatial variation in soil GHG flux. Samplings occurred in May, June, and August with the resulting range of coefficient of variation values for CH_4, 157-496, CO_2, 47-132.6, and N_2O, 66-351%. Spatial interpolation displayed the GHG efflux rate variations across the seasonal land use sampling, and the relative importance of individual hot spot efflux rates upon total emissions. No significant relationships were found between soil properties and gas emissions. Limited significant relationships at $p \leq 0.05$ were found between soil properties and log transformed GHG emissions. Considerations of the closed chamber methodologies for GHG assessment need to occur to expand field studies within the MRF region. Specifically, the dimensions and deployment strategy of the chambers must be adapted to individual site characteristics due to the widespread temporal and spatial variation among soil conditions.

CHAPTER 1: INTRODUCTION AND OBJECTIVES

1.1 Literature Review

1.1.1 Problem of Greenhouse Gas Emissions

Concentrations of greenhouse gases (GHG) from anthropogenic and natural sources have increased in recent centuries with numerous potential changes on both natural and managed systems including regional and global climates, ecosystems, water resources, and agricultural production. The three naturally produced GHG's with the greatest global impact are carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) accounting for 63%, 18% and 6% of total long lived GHG radiative forcing, respectively (Intergovernmental Panel on Climate Change, 2007c). Atmospheric CO_2 is a major concern because it has increased 100 ppm over pre industrial period values, circa 1750, to a mean global concentration of 379 ppm in 2005, and its now rapid rise during the last 30 years (Intergovernmental Panel on Climate Change, 2007c). Besides water vapor, CO_2 has a higher atmospheric concentration than any other GHG. Atmospheric CH_4 and N_2O concentrations have also risen in recent years with the average atmospheric CH_4 concentration at 1774 ppb and atmospheric N_2O at 319 ppb, much greater than the pre-industrial concentration levels of 700 and 270 ppb, respectively (Intergovernmental Panel on Climate Change, 2001). Although, atmospheric CH_4 and N_2O concentrations are magnitudes lower than atmospheric CO_2, the two trace gases have 25 and 298 times more 100-year global warming potential (GWP) compared to that of CO_2, respectively (Intergovernmental Panel on Climate Change, 2007c).

The increases of GHG correspond with the average rise of global temperatures by 0.74°C +/- 0.18°C between 1906-2005 (Intergovernmental Panel on Climate Change,

2007c). The spatial distribution of this change is uneven with differences among continents, bodies of water, and latitude. The ongoing warming trend is associated with changes to local and regional hydrologic cycles. Higher temperatures increase evaporation rates and the amount of atmospheric moisture (Olsen, 2006).

The effects of rising GHG concentration on precipitation rates are estimated to differ by region with some areas becoming much drier while others, such as the United States Midwest, are to become much wetter (Intergovernmental Panel on Climate Change, 2007a). Increases in seasonal influences in the Midwest are predicted with much of the rainfall coming during the spring season, while droughts may occur during the already typically dry late summer months (Intergovernmental Panel on Climate Change, 2007b). In an evaluation by Goudie (2006), precipitation events on average will occur in higher intensities and consequently less water will infiltrate as more becomes surface runoff. The combination of these precipitation trends and land use changes have led to a doubling of flood catastrophic events in the span between 1996-2005 as compared the amount per decade between 1950-1980 (Kron and Bertz, 2007).

From 2000-2011 portions of the state of Missouri were declared flooding disaster areas by the federal government 18 times (Federal Emergency Management Agency, 2011). The seasonal balance of precipitation, strength of individual weather events, timing of snowmelt, and vegetation changes are all predicted to lead to increased spring flooding and summer drought across the Midwest (U. S. Global Change Research Program, 2009). During spring 2011, flooding along the Missouri and Mississippi Rivers in the state of Missouri damaged thousands of hectares to the extent where conventional agricultural production was prevented throughout the growing season. As of winter

2012, a comprehensive review has not been completed, but initial estimates place damages to infrastructure, cleanup and restoration, and losses in agricultural production in the magnitude of billions of dollars (Grigg et al., 2012).

The predicted increase of Midwestern flooding frequency necessitates the importance to further the understanding of the influences of flooding upon the associated floodplain regions. Floodwaters can alter floodplain soil physical, chemical, and biological processes and have subsequent effects on crop and vegetation growth. Floodwaters can physically alter soils, especially in fallow fields, scouring and depositing sediments on field and regional scales. Plant roots may be dislodged as the surrounding material is lost, or buried under fresh deposition. The volatility of these forces limits the formation of well developed soils, which can require centuries or more to form. While some MRF plants are flood tolerant, the germination rates of many row crop and riparian forest species seedling are lowered (Kozlowski, 2002). Species may also suffer due to the loss of plant available nutrients in soluble forms. Soil NO_3^- is readily lost through leaching, but can also denitrify in the forms of N_2, NO, and N_2O gases (Baldwin and Mitchell, 2000). As a plant essential nutrient, agricultural managers apply considerable amounts of N fertilizers to increase biomass production. When used in the MRF, the risk of N losses due to flooding should be considered in management decisions.

Mitigation of current flood-induced soil GHG production may prevent additional future climate change and flooding. The basis of this relationship is focused around the theory that increasing atmospheric GHG will only further the effects of climate changes, such as the frequency of flooding within the Midwest US, thereby creating optimal conditions for additional GHG production. Improving the GHG impact of agriculture

will require managers to have knowledge of the interactions between land use, soils, and hydrology and have some research-based quantitative determinations of cumulative GHG emissions under current and alternative land use systems in the MRF. Decisions must also be aligned with socio-economic interests as areas, such as the MRF, rely heavily upon an agricultural economy. Further understanding of these floodplain systems could result in the selection and/or development of flood tolerant production systems that reduce, or even act as a sink for GHG gases.

1.1.2 Factors Affecting Soil GHG Emissions

Soils contain the second largest global pool of C after ocean sediments, more than twice the amount contained in atmospheric and living biomass pools (Schimel, 1995). Globally a total of approximately 1500 Pg C exists in the first meter of soil depth alone (Sparks, 2003). This soil C pool is the net accumulation of total organic carbon (TOC) that occurs when the rates of biomass growth and deposition are greater than the rate of decomposition. Soil microorganisms comprise 1-5% of the TOC and produce numerous compounds including GHG. In a temperate forest community, 60-90% of the total ecosystem respiration can be attributed to the soil, including both microbial and root respiration components (Goulden et al., 1996). Since microbial activity dominates soil CO_2 production, factors affecting microbial communities heavily influence gas production (Liikanen et al., 2003). These factors include changes to the microclimate and soil properties, such as temperature, moisture, and C and N transformations.

Carbon dioxide is a relatively abundant gas evolved during cellular respiration. Each time organic residues are decomposed a portion of the C is transformed to CO_2 with the remaining fraction retained in organic compounds. (Sylvia et al., 2005). The most

biologically labile C fractions in soil are consumed within days leaving the more recalcitrant C materials. Eventually, only the most stable organic C compounds are left in the soil, taking decades or centuries to decompose. Overall, aerobic microbes retain approximately 35% of the total C and the remaining 65% is liberated as CO_2.

The hydrology of floodplains and poorly drained soils often results in anaerobic conditions which impacts the prevalence of differing soil microbial consortiums. Sub-oxic and anoxic environments do not have enough O_2 for many soil microbial species to survive. However, certain microbial species have adapted strategies to utilize other compounds for energy sources. Methanogens are a category of anaerobic archaebacteria that live in O_2-depleted environments utilizing CO_2 as an electron source; the result of their respiration is CH_4. At this redox level CO_2 is among the highest energy yielding materials remaining as higher yielding substrates, including NO_3^-, MnO_2, $Fe(OH)_3$, and SO_4^{2-}, have already been reduced. Methane production begins when the soil redox potential (E_h) value is equal to -150 mV at a neutral pH, and grows exponentially with further E_h decreases (Sumner, 2000). In acidic soils anoxic conditions are reached at higher E_h values, and consequently the production of CH_4 develops sooner under saturated conditions. On average, this E_h value is reached in soils subject to submergence for 1-2 weeks (Kirk, 2004).

Due to the heterogeneous nature of soil, gases including CH_4 can simultaneously be produced and consumed through oxidation as locations millimeters apart can possess widely differing O_2 levels. During flooding the soil surface can be in an anaerobic state while the underlying soil remains aerobic, or vice-versa depending on the situation. Often in an anaerobic subsurface soil zone, CH_4 will be produced and rise until it reaches

the aerobic surface, where it's re-oxidized (Frenzel et al., 1992). Vegetation also influences CH_4 flux by altering pathways to the soil surface. For example, many wetland and flood tolerant plants feature aerenchyma cell structures allowing direct atmospheric pathways, bypassing the aerobic surface soil (Sumner, 2000). Through these pathways, diffusion can occur 10,000 times faster than through the adjacent saturated soil pore space. Papen (1997) considers non-vegetative greenhouse gas emissions from rice fields to be 'insignificant.' As much as 80-90% of an agricultural field's CH_4 emissions have been attributed to plant pathways (Yu et al., 1997)

Soil N_2O emissions are caused by soil microbial activity during the nitrification ($NH_4^+ \rightarrow NO_3^-$) and denitrification ($NO_3^- \rightarrow N_2$) processes that are part of the soil N cycle. These transformations do not directly produce N_2O, but 1-2% of the total N evolved is transformed into N_2O depending on the physical, chemical, and biological soil conditions. These reactions cumulate into 9.5 Tg N_2O-N yr^{-1} accounting for 65% of annual global N_2O emissions, 3.5 Tg N_2O-N yr^{-1} of which are emitted from arable soils (Stohl et al., 1996).

The largest N_2O impacting factor is the soil water content (Sylvia et al., 2005). During denitrification at lesser water filled pore space (WFPS) conditions, N is converted into NO gas while near saturated WFPS conditions yield N_2 gas. Emissions of N_2O are a result of median soil moisture conditions with its production peaking at 70% WFPS (Sylvia et al., 2005). Acidity is another factor controlling N transformations, as N_2O fluxes are greaterin acidic soils. Gilliam and Weier (1986) tested coastal soils finding higher percentages of N_2O emissions in NO_3^- production in conditions where the pH $\leq$ 5.8.

The use of manufactured N fertilizers allows for increased GHG loss by dramatically increasing the amount of biologically reactive N in soil. At the turn of the century, 15 Tg fertilizer-N was utilized from manure, legumes, and other natural N sources. Currently, 100 Tg of synthetic fertilizer-N is manufactured and 40 Tg N is fixed by legumes (Scott, 2004). Globally, 36% of N_2O emissions can be attributed to the application of synthetic N fertilizers (Mosier et al. 1998). To meet the rising demand for food and energy production, N fertilizers are increasingly being applied to degraded arable soils and marginal lands which may require higher rates due to low N use efficiency and have higher potential for environmental N losses.

1.1.3 Spatial Variation in Soil GHG Emissions

The spatial variation of soil GHG emissions have been the focus of studies in recent years and have shown a wide range in coefficients of variation (CV). In a North American temperate deciduous forest Davidson et al. (2002) found CV's of soil surface CO_2 emissions from 25-500%. Laville et al. (1999) reported a range of CV's to a maximum of 200% for soil N_2O emissions. These differences can happen over short distances in space as GHG emissions one meter apart can show as much variability as locations much farther away (Ball et al., 1997; Nakayama, 1990). The variation is linked to the interactions between the numerous production and transportation factors of the individual gases (Fang and Moncrieff, 1998). Knowledge of the spatial variation can help researchers collect the necessary amount of samples to accurately characterize GHG data. The improvement of global C pool and GHG emission assessments, such as those by the IPCC, will require precise and accurate data from all regions and land uses to limit the need for estimation by extrapolation, a current large source of error (IPCC, 2007).

Soil water content affects microbial populations and at low levels can limit these gas producing organisms, but can also limit soil gas exchange at high levels. Linn and Doran (1984) measured CO_2 emissions between conventional and no till practices. The results supported the hypothesis that CO_2 emissions and microbial activity was maximized at 60% WFPS. The effects of soil water content have often been studied in relation to soil temperature. McKenzie et al. (1998) incubated intact soil surface cores from a boreal forest wetland in four temperature treatment groups: 4, 15, 20, and 25C°. The emissions of CO_2 among all soil types tripled as the temperature increased by 10°C. Mielnick and Dugas (2000) investigated the effects of both soil moisture and soil temperature on CO_2 emissions within the Texas Blackland Prairies ecoregion. The researchers used multiple regression to assess the influence of the variables and cited them as the source of 52% of the flux variance. They evaluated their findings on independent data from the Konza Prairie in central Kansas, in which soil moisture and soil temperature accounted for 76% of flux variance.

Soil physical properties also influence the exchange of gases between the soil surface and atmosphere. The rate of diffusion through porous media such as soil has been estimated in several papers in the 20th century, including two of particular note by Penman (1940) and Millington (1959). Xia et al. (1992) measured the gas diffusion from four different soil types. Increased gas diffusion was related to larger percentages of soil air-filled porosity as well as higher ratios of macropores/micropores.

A study by Medeiros et al. (2011) focused on CO_2 emissions from subtropical soils under no till and convention tillage. Crops planted for the study were a rotation of corn and oats. Through time the no till plots had less soil disturbance preserving

macropore volume and soil structure, and accumulated higher amounts of SOC. Medeiros cited these factors as the causation of the significantly higher CO_2 emissions as compared to those under conventional tillage. The results from a 2008 study by Cabaneiro support SOC as a factor in soil CO_2 emissions. The study centered on fallow agricultural plots in the Northern Andean Páramo Ecosystem. Emission measurements were taken from a mosaic of plots under different stages of cropping and aspect ratios. Soil emissions were correlated with SOC levels with the highest CO_2 coming from NE facing soils with elevated SOC. The lowest emissions were emitted from low SOC content SW facing slopes.

In addition to soil physical influences, soil chemical properties have also been shown to affect the levels of soil GHG. The results of Van den Heuvel et al. (2011) demonstrate a strong negative relationship between soil pH and N_2O emissions due to the decrease in N_2O:(N_2O+N_2) ratio under acidic conditions. Efflux rates were highest at pH 5. The researchers hypothesize isolated acidic conditions as the cause of N_2O emission hotspots.

Land management practices, such as tillage, compaction, removal of biomass, fertilizer applications, and vegetation changes alter GHG transportation and production factors, and consequently affect their emissions. These practices influence not only the quantity of GHG emitted, but also their spatial heterogeneity (Smith et al., 2008). For example, Rayment (2000) found the most influential spatial factor to be differences in soil water content due to micro-topographical features, which are often removed by conventional tillage. Determination of the variability of an area is important to an accurate assessment of its emissions. Turner et al. (2008) researched spatial variation in a

2.33 ha pasture measuring soil GHG emissions twice after rotational grazing, urea fertilization at a rate of 50 kg N ha^{-1}, and 50 mm of irrigation. Cattle were spread at a density of 3.95 cows ha^{-1} and estimated to cover 4-6% of the pasture with feces and urine. The influences of soil, plant, and animal factors led to large measurement variations and the estimation that 181 samples would be necessary from the same area for the sample mean to be within 10% of the true mean. This finding possibly questions the results of the large body of GHG research using relatively few measurements per area often due to research and financial constraints. This practice of using relatively few replicates for measurement of soil GHG emissions may result in the obscuring of results from treatments showing insignificance due to very widespread confidence intervals (Maljanen et al., 2003; McHale et al., 1998; Tomoaki et al., 2011; Ussiri et al., 2009). Improvements in sampling will alleviate this issue allowing for a clearer understanding of land use and management effects on soil GHG emissions.

1.1.4 History of the Lower Missouri River and Floodplain

The Missouri River is the longest river in the United States now measuring 3,768 km which is hundreds of kilometers shorter than in its pre-European state. Its watershed covers 1,360,000 km^2 accounting for approximately 1/6 of the continental US (National Research Council and Committee on Missouri River Ecosystem Science, 2002). The river's natural flow spreads across the floodplain in a complex series of shifting braided channels, sloughs, and side channels, but this flow pattern has changed after intensive management began in the first half of the 20^{th} century. The sediment the Missouri River carried in its natural state caused it to be the second muddiest river in the world, second only to the silt-laden Colorado River (Ferrell, 1996). Galat et al. (1998) reports that

during non-flood conditions the width ranged between 610-1,830 m, whereas Schneiders (1999) asserts a width of 300 to 3,100 m wide during normal conditions, and between 7,600 to 10,700 m during floods. Traditional flow surges in April and June caused flooding and scouring, redepositing sediment in backwater areas with lower flow velocities (National Research Council and Committee on Missouri River Ecosystem Science, 2002).

In 1819 Congress recognized the river's potential to serve as a navigation course and approved money to survey it for that purpose (Ferrell, 1996). In 1832 additional legislation was passed and channelization efforts began and intensified over the following decades. Through the 20th century the river was increasingly controlled by a series of dams, levees, dikes, and other control structures to regulate flow and to promote the desired river and land uses. Today, the river remains managed under a suite of government agencies led by the Army Corps of Engineers.

Similar to the river itself, the lower MRF, the region below Sioux City, IA, has been heavily used and managed after the arrival of eastern settlers. Rich, alluvial sediments allowed for diverse biological communities, and when drained and cleared supported high yielding agricultural production across the 76,000 ha of MRF within Missouri as well as in the surrounding states of Kansas, Nebraska, and Iowa. While often fertile, repetitive flooding acts to prevent the formation of highly developed soil structure (Faust, 2006) Explorer Henry Brackenridge (1814) noted the floodplain soils on an expedition as a "…light soil of a texture extremely loose…". The soil textural distribution of MRF alluvial deposits follows a general pattern of finer textured materials being deposited upwards and away from the river (Heisner, 1997). While this pattern

holds true for the deposition of single events, the uniqueness of each deposition leads to MRF soils being highly variable over small distances in lateral and vertical dimensions (Gregory et al., 1991).

John Bradbury (1817) travelled the Missouri River in the early 19th century and detailed the species he observed. In Mid-Missouri he reported rushes (*Equisetum hyemale*) growing thick along the floodplain and banks and listed the common tree species as cottonwoods (*Populus deltoides*), elm (*Ulmus* sp.), and mulberry (*Morus* sp.). These plant communities form successionally depending on the frequency of flooding, due to species disturbance tolerance and soil specificity. Cottonwoods (*Populus* sp.) and willows (*Salix* sp.) are very successful on fresh alluvial materials, while on the higher benches, where flooding was less frequent and soils more developed, species of ash (*Fraxinus* sp.) and box elder (*Acer negundo*) were more numerous (Johnson et al., 1976). Species richness increases towards the river's mouth with the addition of elm, oak, maple, hickory, and sycamore (National Research Council and Committee on Missouri River Ecosystem Science, 2002).

Currently 58% of Midwestern riparian areas have been converted to agriculture, 4% developed for urban use, and 38% are in natural vegetation (Palik et al., 2004). The majority of the current natural areas have previously been managed for intensive uses, such as timber harvesting or grazing, during their history and as such the communities are primarily secondary growth or restored lands (Faust, 2006). Alterations to the region's soils, hydrology, and microtopography hinder restoration of native species in the floodplain (Dey et al., 2000; Jacobson et al., 2011).

Currently Missouri riparian forests vary, but conform to several general characteristics. Soils are generally deep (>150cm), moderately acidic to neutral ($5.6 \leq$ pH ≤ 7.3) and slopes are less than 8%. Soils are saturated 10-20% of the year often early in the growing season (Nelson, 1987). Flooding in forests adjacent to smaller streams often is flashy, lasting several days or less, while communities on larger rivers can be inundated with floodwater for weeks or months. The ample moisture leads to low fire frequency. Nelson (1987) describes the species in Missouri riverfront forests as the following:

> **Dominant Plants**
>
> **Canopy**: silver maple (*Acer saccharinum*), cottonwood (*Populus deltoides*), green ash (*Fraxinus pennsylvanica*), black willow (*Salix nigra*), sycamore (*Planus occidentalis*), american elm (*Ulmus Americana*), river birch (*Betula nigra*), box elder (*A. negundo*), hackberry (*Celtis occidentalis*), sugarberry (*C. laevigata*)
>
> **Understory:** gray dogwood (*Cornus foemina*), swamp dogwood (*C. amomum ssp. Oblique*)
>
> **Shrubs/Vines**: poison ivy (*Toxicodendron radicans*), Virginia creeper (*Parthenocissus quinquefolia*), raccoon grape (*Ampelopsis cordata*), grape (*Vitis cinerea, V. riparia, V. vulpine*)
>
> **Herbaceous layer:** White woodland aster (*Aster lateriflorus*), wild ryes (*Elmus villosus, E. virginicus*).

The woody species found in northern Missouri riparian forests are associated with the types in northern mesic forest ecosystems present in other Midwestern states, such as Illinois, Michigan, Iowa, Minnesota, and Indiana (Dollar et al., 1992). Faust (2006) found very similar species composition in an extensive survey of northern Missouri riparian areas.

Riparian forests provide numerous ecosystem services for landowners and downstream residents. Such services include stabilizing stream banks, protecting water quality, and providing food and habitat for aquatic and terrestrial species (Westphal and Ostry, 2006). Vellidis and Lawrance (2004) go as far as to call riparian forests a "silver bullet" for maintaining water quality.

1.1.5 Historical and current land use in Howard County, Missouri

Howard County has experienced land use changes common to the lower MRF region. The majority of the county is classified in the EPA Level IV scheme as the River Hills Ecoregion (Schroeder et al., 2002). The often wide and flat river floodplain is comprised of moderate to poor draining alluvial soils with sandy or silty textures. Deep loess covers the upland hills and thins away from the river adjoining glacial till covered soils to the north. Elevations range from 120-300 m above sea level (Grogger and Landtiser, 1978). Historically between the cities of Booneville and Glasgow the bordering river channel often reached widths of two miles (Brackenridge, 1814), but is now channeled and provides space in the drained floodplain for agricultural use. As Schroeder (1968) described, "virtually all of Howard County was wooded at the beginning of the nineteenth century." The largest of the trees lined the bottomlands and creek valleys. Brackenridge (1814) wrote in his journal of the local vegetation including rushes "which grow as high as a mans head, and are matted with vines and briars" and the "clean and open woods" of hickory (Carya sp.) and oak (Quercus sp.).

In the years surrounding the Lewis and Clark Expedition, immigrants began streaming into the county. A Frenchman Joseph Marie settled in (what would later become) Franklin township becoming the first European settler in 1800. The sons of

famed pioneer Daniel Boone, Nathan and Daniel W. Boone, formed a settlement around the widely publicized salt lick in 1807 (Grogger and Landtiser, 1978). Three years later in 1810 Benjamin Cooper led 150 Kentucky families to settle in the floodplain around Franklin that would later bear his name and the adjacent uplands (Schroeder, 1968). Timber was cut and sold downriver while the land was cleared for crops and expanding settlements. Franklin saw great growth due to nearby productive land, its role in river commerce, and its placement as the Howard County seat. The city was the largest and considered the most important Missouri town west of St. Louis (Barns et al., 1877; CAFNR, 2008). The soils there were known to be "generally friable and fertile drawing in farmers and land speculators" (Schroeder, 1968). Expansion continued for a number of years in Howard County until 1829 when the town of Franklin was washed away in a flood. The prominence never returned as businesses and services moved more stable locations.

The county remains home to over 10,000 people and a viable agriculture based economy. The NRCS soil survey (1978) estimated county land use to be 50% row-crop agriculture, 25% open pasture, 21% forest/woodland, and 4% urban/other . Current row crop estimates are lower at 36%, possibly due to the inclusion of the Crop Reserve Program (Table 1.1) (National Agriculture Statistics Service, 2011). A USFS survey (2011) projected forest cover at only 12% (Table 1.2).

In 1953 the University of Missouri opened a research facility in Howard County concentrating on horticulture. In the early 1990's the center expanded to 269 ha including agroforestry in its research focus, and was renamed the University of Missouri Horticulture and Agroforestry Research Center, otherwise known as HARC (CAFNR,

2008). The property encompasses the river hills as well as Missouri River Floodplain providing opportunities to test agroforestry practices in a variety of settings. Integrated land use systems, such as agroforestry, produces food, fiber, and fuels and while reducing environmental impact in comparison to conventional agricultural production. HARC operates the basic tenets of agroforestry management through practices including silvopasture, alley cropping, riparian buffers, and forest farming (United States Dept. of Agriculture, 1997). These practices encourage product diversification, emphasizing plant and animal interaction to increase productivity and produce in a sustainable manner. Agroforestry is performed globally and relied on by 1.2 billion people, but is most prevalent in the tropics (Sanchez et al., 1997). Previous research has shown the advantages of agroforestry on soil and water conservation, but claims of lowering GHG impacts are still largely untested (Johnson, 1995). Despite the potential for adoption within the fertile MRF region, agroforestry practices are not widespread. The increased understanding of agroforestry benefits over recent decades has provided land owners solutions to land degradation issues associated with monoculture production systems (Nair, 2007). The inclusion of agroforestry in national land use inventories, such as those completed by US Forest service and NRCS, could provide updated statistics on adoption of individual practices (Udawatta and Jose, 2011).

1.2 Potential Significance of the Research

The knowledge of agricultural management practices has greatly improved over the 20th century helping land owners, policy makers, and researchers to understand the agronomic and environmental impacts of those practices and to improve future management decisions. Agroforestry practices have been demonstrated to provide region

wide ecosystem services while granting economic benefits to landowners. The economics of responsible agricultural production can be aided by policy decisions as seen currently by soil and water conservation programs at state and federal levels. Land owners may be persuaded to voluntarily change management to reduce GHG if yields can be maintained or improved. The profitability of carbon sequestration would increase with the development of a carbon 'cap and trade' system, allowing individuals to seek primary incomes from the externality benefits from sustainable management, including numerous agroforestry systems.

Scientists working with current GHG models lack site-specific information and must make assumptions about GHG production. The GHG impacts of both the temperate floodplain and agroforestry practices are not well understood. Errors in the measurement of soil properties and emissions are extrapolated to larger areas having great effects on overall estimates (Robertson, 1993). More accurate inputs concerning region specific factors and management techniques will lead to higher quality models. Knowing the spatial variation of emissions from these areas and practices will improve field measurement techniques as well as the subsequent data extrapolation simulation purposes.

1.3 Objectives and Hypotheses

Primary Research Objective

To investigate the influences of MRF land use systems on the spatial and temporal variations of soil GHG emissions

Specific Research Objectives

1. To evaluate soil GHG emissions (i.e., CO_2, CH_4, N_2O) in floodplain soils under agroforestry, row-crop agriculture, and forested systems in response to differences in soil moisture, temperature, land use, and N inputs.
2. To characterize the spatial variation of GHG emissions (CO_2, CH_4, N_2O) in these land-use systems in the Missouri River floodplain.

Specific Research Hypotheses

1. Floodplain soils historically under row-crop agriculture and N inputs will have higher soil GHG flux than agroforestry and forested systems.
2. The total and spatial variation of soil GHG emissions will differ due to land management and subsequent changes in microclimate and substrate availability.

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1.5 Tables

Table 1.1. Total area of crops harvested in Howard County, MO during 2009 and 2010 as projected by the National Agriculture Statistics Service (2011).

Crop Type	Area (ha)	
	2009	2010
Corn	13,436	13,638
Soybeans	17,199	16,835
Wheat	2,954	1,133
Hay	10,117	9,712
Total Crop Production	43,706	41,318
Howard County	--	121,900

Table 1.2. Total area of forest within Howard County, MO estimated between 2005-2009 (USFS, 2011).

Forest Type	Area (ha)
White oak/red oak/hickory	7,135
White oak	1,178
Mixed upland hardwoods	2,543
River birch/sycamore	1,930
Cottonwood	774
Sugarberry/hackberry/elm/green ash	882
Total Forest Area	14,442
Howard County	121,900

CHAPTER 2: EFFECT OF LAND USE AND WATER CONTENT ON SOIL GREENHOUSE GAS EMISSIONS

2.1 Abstract

The lower Missouri River Floodplain (MRF) encompasses over 76,000 ha within the state of Missouri and has experienced large-scale land use changes primarily to agricultural uses. Such changes in land use management practices may have multiple effects on microclimate and soil properties, such as soil temperature, water content, and soil C and N fractions, and soil processes, such as C and N mineralization and denitrification. As a result of these changes, soil greenhouse gas (GHG) flux may be affected. Soil surface GHG emissions need to be better quantified in order to assess the total environmental costs of current and possible alternative land uses in the MRF. The objective of this study was to evaluate soil GHG emissions (CO_2, CH_4, N_2O) in MRF floodplain soils under agroforestry, row-crop agriculture, and forested systems in response to differences in soil water content, temperature, land use, and N fertilizer inputs. Intact soil cores (0 – 10 cm depth) were obtained from row crop agricultural (AG), agroforestry (AF) and riparian forest (FOR) sites within the MRF near New Franklin, Missouri and incubated under constant temperature conditions (30 $^{\circ}$C) for a period of 94 days. Cores were subjected to three water regime treatments: flooded (FLD), optimal for CO_2 efflux (OPT), and fluctuating (FLX). Additional N fertilizer treatments (0 or 0.20 g N kg^{-1} soil as KNO_3) for the AG and AF land uses were included during the incubation and designated as AG-N and AF-N, respectively. The soil CO_2,

CH_4, and N_2O fluxes were determined by sampling the head space at three times (0, 20, 40 min. after sealing).

Soil CO_2 emissions were associated with both the MRF land use system and the soil moisture regime. FOR soils had higher cumulative soil CO_2 emission rates than those in AG-N despite no significant differences at $p \leq 0.05$ in initial soil POM or $KMnO_4$-oxidizable C. OPT sample cumulative emissions were significantly higher ($p \leq 0.05$) than those of FLD and FLX. Soil CH_4 emissions showed no influence of land use, but were related to the water regime and length of time under flooding conditions. Flooded (FLD) soil samples showed a significant difference in cumulative soil CH_4 emissions from other samples at Day 51 onward when E_h had significantly decreased. Flooded (FLD) CH_4 efflux rates were significantly ($p \leq 0.05$) higher than under the OPT and FLX soil moisture regimes. Time, land use, and water regime all showed independent and interactive influences on the surface cumulative N_2O emissions. As expected, AG-N and AF-N emitted the most N_2O among the land use and N treatments. Loss of N through soil N_2O emissions decreased as the incubation progressed. These results suggest that land use management, the soil moisture regime, and soil N content can influence the amounts of GHG emitted in the MRF.

2.2 Introduction

Around the world river floodplains have historically been home to large human populations. Often, these settlements manipulate the river and floodplain environment for ideals, such as increased flood protection, river transportation, and agricultural production (Islam and Braden, 2006). Floodplain soils often lack pedological development, but have high native fertility from alluvial sediments. The combination of

the presence of relatively young, fertile soils, frequently Entisols and Inceptisols, and the availability of water in the soil increase the ability for plant growth, and specifically agricultural production.

The lower Missouri River Floodplain (MRF) reflects this global pattern for floodplains. While historically forested, over 90% of the region has been converted to cultivated row crop production representing some of the most productive land in the United States (Galat et al., 1998). To accomplish this, the river was channelized and disconnected from its associated floodplain in a series of measures throughout the 19th and 20th century. The MRF, even under native vegetation, differs from historical ecosystems due to the hydrological separation (Dey et al., 2000).

The conversion of the MRF land use altered natural nutrient cycles through management practices, such as tillage, compaction, removal of biomass, nutrient or other applications, and vegetation changes. These actions modify soil environments previously conducive to storing C and N, to the promotion of their losses, some of which occur through volatilization. Three of the gases emitted from soils are carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O), and are considered to be among the most important greenhouse gases (GHG) on an annual global basis. The MRF is particularly susceptible to these emissions through its high levels of soil organic carbon (SOC), routine application of N fertilizers in agricultural land uses, and recurrent flooding leading to anaerobic soil conditions. Increases in flooding frequency due to changes in precipitation patterns are expected in coming decades across the Midwestern U.S., and may cause increased GHG losses in a feedback relationship since increased levels of

these gases cause additional climate change (U. S. Global Change Research Program, 2009).

Accurate assessment of GHG emissions from the MRF is important to understand local contributions to the cumulative global issue. Currently, a lack of soil GHG emissions data specific to the Midwestern U.S. leads to the use of generalized estimates of the area and land uses as a whole. Increased precision in global models from regions such as the Midwestern U.S. would provide a clear evaluation of the current and predicted levels of GHG. It would also assist in the development of potential alternative management practices to mitigate GHG emissions. Presently, agriculture accounts for 10 to 12% of annual anthropogenic GHG production (Intergovernmental Panel on Climate Change, 2007).

Integrated land use systems, such as agroforestry, produce food, fiber, and fuels and may reduce environmental pollution such as soil GHG emissions. The United States Department of Agriculture (USDA) defines agroforestry as "intentionally combin[ing] agriculture and forestry to create integrated and sustainable land-use systems." Bene et al. (1997) further describes agroforestry as management that increases total production over monoculture systems and also complies with the culture of the local people. While agroforestry has many definitions, several basic characteristics of this type of land management include plant and/or animal integration, plant diversification, increased productivity, and sustainability. Common agroforestry practices include silvopasture, alley cropping, riparian buffers, and forest farming (United States Dept. of Agriculture, 1997). Agroforestry practices occur globally and are relied on by approximately 1.2 billion people, but these practices are most prevalent in the tropics (Sanchez et al., 1997).

Previous research has shown the multiple benefits of agroforestry including the reduction of stream nutrient loading (Palone and Todd, 1997), lessening wind and water erosion of sediments (Correll, 1997), improving soil quality through carbon sequestration (Mutuo et al., 2005), and increasing wildlife diversity (Yahner, 1982), but claims of lowering GHG emissions are still largely untested (Johnson, 1995).

Quantification and comparison of soil GHG emissions among MRF land uses would provide land managers with additional information about the relative environmental impacts of different production and native land use systems. The objective of this study was to evaluate soil GHG emissions (CO_2, CH_4, N_2O) in MRF floodplain soils under agroforestry, row-crop agriculture, and forested systems in response to differences in soil water content, temperature, land use, and N fertilizer inputs.

2.3 Materials & Methods

2.3.1 Site Location

The sampling sites were located at the University of Missouri Horticulture and Agroforestry Center (HARC), a 267 ha research station situated near New Franklin, Missouri (39° 0‘ 0 N, 92° 46‘ 0 W) (Figure 2.1). The HARC research station borders Sulphur Creek on the south and west. This tributary’s catchment covers a large portion of southern Howard County and drains into Bonne Femme Creek shortly before its entry into the Missouri River. The northern portion of HARC sits on loess-capped river hills while the southern edge is in the Missouri River Floodplain, approximately 4 km from its current course. The proximity to both Sulphur Creek and the Missouri River has influenced the diverse soils of the area, encompassing seven soil associations (CAFNR,

2008). Both the floodplain and river hills are high in silt contents due to historic alluvial and aeolian influences. The floodplain has slopes of 0-2% and is situated at approximately 183 meters above sea level (Center for Applied Research and Environmental Systems, 2011). The area has an annual average temperature of 12.6°C and 107 cm precipitation (National Weather Service, 2012)

Three varying land use sites have been selected along Sulphur Creek consisting of a row crop agriculture (corn/soybean/wheat rotation desginated as AG), agroforestry (pecan orchard/hay desginated as AF), and a managed riparian forest site (designated as FOR). The orchard and riparian forest buffer are both owned and managed by the University of Missouri while the row crop agricultural field is privately owned and adjoins HARC property. The soil at all three locations has been surveyed as Nodaway silt loam, (Fine-silty, mixed, superactive, nonacid, mesic Mollic Udifluvents) (Grogger and Landtiser, 1978) . This soil is moderately well drained, but is subject to occasional flooding and ponding up to 14% annually (Natural Resources Conservation Service, 2005). The frequency of such events is affected by the soil's location along stream corridors. The top of the seasonally high water table is at 122 cm. Within Howard County, Nodaway soils cover 6.7%, or 8,200 ha. This soil has predominately been used for agricultural production and as such is characterized by a silt loam Ap horizon (0-20 cm) overlaying a silt loam/silty clay loam C horizon (20-150 cm) (Natural Resources Conservation Service, 2005).

The 2.8 ha orchard consists of mixed pecan varieties planted during autumn 1993 (Appendix A). Four groups of thirty-two trees were planted with four grass fields in a complete randomized block configuration. The grasses in all areas of the block are

primarily tall fescue (*Festuca arundinacea)* with patches of johnson grass (*Sorghum halepense).* Mean tree measurements are the following : total height 10.4 +/- 1.5 m, live crown height 2.4 +/- .3 m, DBH 19.8 +/- 3.6 cm.

The adjacent row crop field has been primarily used for growing corn (*Zea mays* Indenta) and soybeans (*Glycine max* Merr.) during much of the 20th century. Its current area is 30 ha. The current leaser also has planted winter wheat (*Triticum aestivum*) when possible. In 2009 soybeans were planted and due to early spring rains, the manager altered plans to plant corn and again planted no-till soybeans late May 2010.

The approximately 4 ha riparian forest site is primarily comprised of four tree species: silver maple (*Acer saccharinum*), american elm (*Ulmus americana*), sycamore (*Platanus occidentalis*), and cottonwood (*Populus deltoides*) (Appendix B). A fifth species, green ash (*Fraxinus pennsylvanica*), was also found in recent surveying. In 2005-2006, 148 trees were harvested for an approximate volume of 46,164 board feet (102 m^3). At this time there are no plans for further management actions (Walter, 2010). Basal area was determined to be 30.5 m^2 ha^{-1}.

2.3.2 Soil Sampling

Soil sampling was conducted at all three land use sites in early May 2010. Transects were established to determine sampling locations within land use and soil type restrictions. Within the riparian forest and agricultural field samples were taken at five meter intervals. Agroforestry samples were taken two meters from each pecan tree, approximately at the edge of the drip line. Sampling consisted of collecting a soil core, 7.6 cm diameter,10 cm depth with an Uhland Core Sampler and an adjacent one kilogram bulk sample taken. A total of 25 samples were collected from each location and placed in

a cooler for transport back to the University of Missouri. All samples were stored at 5°C to minimize sample degradation.

2.3.3 Incubation Study

Intact soil cores were incubated for a period of 94 days for the measurement of GHG (CO_2, N_2O, & CH_4) production (Figure 2.2). During testing the cores remained intact inside to account for management induced soil structural changes. Three soil moisture regimes (60% WFPS(OPT), flooded or 100% WFPS (FLD), and fluctuation between 100% WFPS and 60% WFPS(FLX)) were used to simulate precipitation and the frequent flooding of Sulphur Creek. To mimic real management conditions N fertilizer was added to the agroforestry and agricultural soils in the form of a potassium nitrate (KNO_3) solution and designated as AF-N and AG-N, respectively. Both management groups received either 0 or 0.20 g N kg^{-1} soil, approximately equivalent to a common field application rate of 180 kg N ha^{-1}. The combination of three moisture regimes and five land use types resulted in the total of 15 treatments, each replicated three times for a total of 45 samples. Due to incubator and sampling space constrictions, replications were divided into three groups within a randomized complete block design. Each sample block was placed in the same location within the incubator and sampled at the same time.

Three days prior to the initiation of the experiment, all samples were equilibrated at the incubation temperature of 30°C, as erroneous gas efflux may occur after long periods of refrigeration. This temperature is near optimal for microbiological activity and GHG efflux, emphasizing differences between the experimental treatments. Also, this surface soil temperature is commonly observed at the sampling locations under typical summer weather conditions. To begin sampling on Day 1, all samples were brought to

60% WFPS and placed into open two-liter plastic chambers. Fertilizer additions for selected treatments occurred on Day 2 and moisture regimes treatments were implemented on Day 3. Greenhouse gas emissions were measured on Days: 1, 3, 5, 8, 10, 13, 15, 16, 19, 22, 24, 29, 31, 36, 43, 50, 57, 65, 71, and 94 for a total of 20 times over the incubation period. Water regime was managed on a weekly basis. If required, water was evenly spread over the soil core with a pipette until the desired WFPS was acquired.

Sampling largely followed the vented field static chamber techniques set forth by the USDA GraceNET protocol (2003), with adaptations to accommodate laboratory measurements (Figure 2.2). At the initiation of each sampling period (Time 0), the chambers were closed and flushed with He to alleviate the issue of diffusion resistance, the result of high concentration of atmospheric gases within the chamber headspace (Rochette and Eriksen-Hamel, 2008). Three 12 ml gas samples were taken at 0, 20, and 40 minutes by syringe and immediately placed in an 8 ml pre-evacuated glass vial. The vials were sealed with gray butyl rubber septas, considered to be minimally reactive to N_2O and CH_4 (Parkin et al., 2003). Samples were stored at room temperature and processed as soon as possible to minimize the possibility of contamination. Gas concentration analysis was carried out using a Shimadzu 2014 Gas Chromatograph (Columbia, MD) fitted with a methanizer, flame ionization detector, and electron capture detector (Table 2.1). The three timed gas samples were used to perform the efflux calculations. The gas efflux was estimated at the ambient gas concentration tangent using linear regression.

2.3.4 Soil Analyses

The corresponding bulk samples were used to analyze initial soil properties. Soil gravimetric water content was measured by oven drying to 105°C. Total organic carbon (TOC) and total nitrogen (TN) were analyzed in duplicate with a TruSpec CN analyzer (LECO, St. Joseph, MI). Particulate organic carbon (POM-C) and particulate organic nitrogen (POM-N) were measured using methods adapted from (Cambardella and Elliott, 1992) The procedure outlined by Weil et al ,(2003), was followed in the analysis of $KMnO_4$-Oxidizable organic C.

Additional tests were conducted corresponding with GHG sampling to measure soil property changes during the incubation period. Soil pH was measured (1:1 water) using an Ag-AgCl electrode (Thermo Fisher Scientific, Waltham, MA). Soil redox potential was examined using an Oakton 300 series meter (Cole-Parmer, Vernon Hills, IL). The instrument has an ORP millivolt range of ±2000 mV, with a resolution of 0.1 to ±-399.9 mV and 1 mV when outside of this range.

2.3.5 Statistical Analysis

Data analyses were completed using SAS 9.2 (SAS Institute, 2009). The data were subjected to the ANOVA procedure PROC MIXED with a repeated measures covariance structure, as the mean values were separated with Fisher's LSD at $p \leq 0.05$. The procedure PROC CORR was used to test for correlations between soil CO_2, CH_4, and N_2O emissions and soil C and N fractions, soil pH, bulk density, and soil redox.

2.4 Results

2.4.1 Land Use Effects on Initial Soil Properties

Preliminary soil tests did not reveal considerable differences in measured soil properties among land uses, and may be attributed to the large natural variation within the alluvial MRF soils (Table 2.2 & 2.3). Despite differences in land disturbance, primarily through tillage and other agricultural field operations, among the land uses, soil bulk density was not significantly different among the land uses and averaged approximately 1.34 g cm^{-3} (Table 2.3). Soil pH in AG soils was 5.9, slightly lower than the AF and FOR pH values of 6.1 and 6.2, respectively. Through particle size analysis all soils were characterized as silt loam, but will fluctuate across the landscape due to depositional banding (Table 2.4). The differences in cation exchange capacity can be attributed to these textural variances and clay content in particular. Soil POM-C values were highest in FOR samples, with the lowest values in AF soils. Differences among TOC and $KMnO_4$ oxidizable C values were not significant among land use treatments.

2.4.2 Land Use Effects on GHG Emissions

Land use effects were mixed among the three GHG emissions (Tables 2.3-2.5). No significance at $p \leq 0.05$ was found among management treatments in soil CH_4 emissions. Each land use treatment showed consistent CO_2 emissions throughout the incubation period. Cumulative FOR soil CO_2 emissions were 17.3 g CO_2 kg^{-1} soil and were significantly higher ($p \leq 0.05$) than the other four treatments (Figure 2.3). An interaction between land use and water regime treatments prompted the individual analysis of each treatment combination (Figure 2.4). Nitrogen applied agriculture and agroforestry treatments had significantly higher soil N_2O emissions, 155.5 and 83.3 mg

N_2O kg soil $^{-1}$, respectively, than non-fertilized treatments. In the FLX treatment, no difference in N_2O flux rates was found between AG and AGN samples, but both were significantly higher than AF, AFN, and FOR treatments. However, the cumulative emissions of FLX-AGN treatment was much lower than FLD-AGN at 34.2 3 mg N_2O kg soil $^{-1}$. OPT land use samples displayed the lowest N_2O emissions of all water regime treatments. OPT-FOR samples had the highest cumulative N_2O emissions at 15.4 mg N_2O kg soil $^{-1}$.

2.4.3 Water Regime Effects on GHG Emissions and Soil Properties

Soil moisture regime treatments had effects on all GHG emissions. OPT samples were significantly higher at $p \leq 0.05$ in CO_2 emissions than OPT and FLX treatments (Figure 2.5). OPT soils cumulatively emitted 16.3 g CO_2 kg^{-1}, while FLD and FLX emitted 9.7 and 5.2 g CO_2 kg^{-1}, respectively. FLD samples emitted significantly more CH_4, 614.1 g CH_4 kg^{-1} than the other two treatments (Figure 2.6). FLX and OPT treatments emitted a minimal amount of CH_4, 5.1 and 2.4 614.1 g CH_4 kg^{-1}, respectively.

Soil redox measurements of E_h ranged between 200 to 400 in both OPT and FLX treatments; however, measurements in FLD treatments decreased throughout the incubation (Figure 2.8). FLD treatments ended the incubation period flooding at a low point of E_h=-284. In comparison, FLX and OPT treatment E_h values were both above 300 at the end of incubation. Soil pH measurements did not show differing trends as values decreased slightly throughout the incubation among all water treatments (Figure 2.8). FLD treatment pH values were consistently one pH unit higher than OPT and FLX treatments.

2.4.4 Relationship of Soil Chemical Properties with GHG Emissions

Initial soil organic C and N values generally were not significantly correlated ($p \leq 0.05$) with CO_2, CH_4, and N_2O (Tables 2.8, 2.9, 2.10). $KMnO_4$-Oxidizable C did correlate with soil CO_2 emissions at p=0.0581. Soil redox values were strongly ($p \leq 0.05$) negatively correlated with soil CH_4 flux, but were not correlated with soil CO_2 or N_2O emissions (Table 2.11). Soil pH_w displayed positive correlation with N_2O and CH_4 emissions at $p \leq 0.05$ (Table 2.12).

2.5 Discussion

2.5.1 Land Use

The majority of the effects on SOC and GHGs from land use changes occur in the years after modification. Franzluebbers and Follett (2005) observed rapid increases in SOC in the range of 15-17 years following a switch to forage grasses from cultivated crop rotations. While GHGs are still emitted during this period, the new management causes a net C gain within the soil, and thus has a mitigation effect on GHG emissions. Research by Cole et al. (1996) supports agroforestry as a viable option to accumulate SOC by adding additional soil inputs, slowing decomposition, lowering the rate of erosion, and maintaining or improving soil structure. At the time of measurement the Pecan agroforestry system was approximately 17 years old. Although the analyses of TOC, $KMnO_4$-oxidizable C, and POM-C varied in contrast to AF than AG treatments, the differences were not enough to be statistically significant. This indicates that there has been no effect of GHG mitigation by the AF managed soils. However, C integrated into AF vegetation may still prove to cause a net reduction in soil-atmospheric C flux. Quantification of vegetation C pools would verify this proposal. The same principles of

the accumulation of SOC through implementation of perennial vegetation inputs and protection against soil erosion applied to agroforestry systems can be applied to the riparian forests. It was expected that the relatively undisturbed residue inputs of riparian forests by the limitation of biomass removal and decomposition would lead to an increased level of SOC compared to the two other more intensively managed land uses. However, the FOR treatment results did not show significantly higher levels of soil total organic C and C fractions (i.e., $KMnO_4$-oxidizable C, and POM-C) than the AG and AF soils.

Generally, soil management practices that have increased SOC stores will subsequently release more GHG through additional biomass in plants, animals, and particularly microbial populations (Franzluebbers and Follett, 2005). The increase in emissions is often offset by C biomass incorporation. The lack of SOC increase in FOR soils observed in this research suggests that another factor may be the cause of the high rates of CO_2 emissions. Mutuo et al. (2005) investigated tropical land use systems and showed that the quality of organic inputs, not just the quantitative measurements, are important for the investigation of soil CO_2 emissions. Agroforestry systems measured by Mutuo had higher SOC levels, but did not increase CO_2 emissions as the inputs were low in N and high in lignin content.

Management practices have long been known to impact soil structure and pore size distribution, and may be a source of GHG variation in these systems. Allaire et al. (2009) suggests that macopores from both soil cracking and faunal pedoturbation can possibly lead to increases in soil gas exchange of more than a level of magnitude. Management with perennial vegetation that supports the macropore development may

result in increased GHG emissions through the increase in gas exchange pathways. Soil samples were collected in the spring and may reflect the influences of spring tillage. This particularity in the methods may reflect on the lower AG bulk density without evidence of mechanical soil compaction. A change in the sampling schedule to another season may reflect differences in soil physical properties, and thus changes to GHG efflux rates. Alternatively, frequent flooding may have negated the direct influence of management upon soil physical properties. Hao et al. (2011) demonstrated an increase in soil bulk density from alluvial agricultural plain soils following a long term flooding treatment.

The results of the soil GHG emissions showed the influence of land use upon two of the three GHG studied in this research. No significant difference was found among the land uses in soil CH_4 emissions as other measured variables such as soil water content had greater influence. Riparian forest soil CO_2 emissions were significantly higher than all other treatments. Agroforestry (AF) managed soils were the second highest in soil CO_2 efflux, unexpectedly significantly greater than AF-N. AG and AG-N CO_2 emissions were not significantly different. This supports the data collected by Bailey (2005) at an upland site in northern Missouri where grass-only and grass-tree vegetated buffers emitted more CO_2 than synthetically N-fertilized agriculture alone. This opposes the findings of Palm et al. (2002) in which high-input cropping emitted 25% more CO_2 than other land uses including agroforestry.

As expected, soils treated with N fertilizer had increased soil N_2O emissions over non-fertilized samples. The additional substrate allowed for an increased conversion of gases through nitrification and denitrification processes. This is in agreement with Hoben et al. (2011) who tested multiple types of N fertilizer in agricultural production

finding an increase in soil N_2O flux increasing exponentially with N application rate across all sites and years.

2.5.2 Soil Moisture Regime

Incubation results of GHG emissions observed in this research largely supported previous literature on the subject. Water regime clearly defined cumulative CH_4 emissions as FLD treatments emitted significantly more than FLX and OPT. This effect became clear at Day 51, when flooding induced a significant decrease in E_h. Oxygen diffusion within the FLX and OPT treatments was never limited for a significant amount of time, if at all, and as such E_h values remained steady throughout the incubation. These results support the study by Koh (2009) that investigated soil emissions in four locations. The resulting emissions increased from locations that were unflooded, occasionally flooded, shallow permanently flooded, and to a deeper-water permanently flooded site. From this information it can be concluded that within the MRF, CH_4 emissions have the potential to occur at high rates during long term flooding only and is not a concern for short term flooding and saturated conditions. Soil CH_4 emissions are directly related to the length of saturation and are not subject to difference among land uses and management practices.

As expected, soil CO_2 emissions were the largest in the OPT treatments, as it was designed to maximize soil microbial activity and emissions. Previous literature draws consensus around maximum soil CO_2 efflux occurring around a median WFPS, although the exact water content figure varies from 30-70% WFPS. Soils with low water contents may have restricted microbial activity due to the limiting diffusion rate of nutrients. Conversely, soil microbial populations may be restricted under high soil water content

conditions due to the lack of oxygen diffusion. Linn and Doran (1984) compiled literature supporting 60% WFPS as an optimal moisture content for maximizing numerous biological activities including quantity of bacteria (Seifert, 1961), organic matter (Gilmour et al., 1977), CO_2 efflux through decomposition (Pal and Broadbent, 1975), ammonification and nitrification (Greaves and Carter, 1920), and denitrification (Bremner and Shaw, 1958). Bailey (2005) measured CO_2 emissions from four water treatments with the highest emissions occurring at 80% WFPS. Schaufler et al. (2010) figures were lower and present a wide range of 20-60% WFPS as optimal conditions with maximum CO_2 emitted at the higher end of the range.

The FLD treatment resulted in the highest N_2O emissions contrary to some previously reported figures. Results from a study by Laville et al. (2011) found maximum N_2O fluxes in conditions between 57-69% WFPS, quickly tailing off outside of this range. Flechard et al. (2006) found high N_2O emissions in two treatment groups, the first being soils between 60-90% WFPS supporting the OPT treatment, and the second being dry soils following precipitation resembling the FLX treatment. In agreement with the dataset is the findings of Pathak and Nedwell (2001), who report higher N_2O emissions in submerged soils than those at field capacity. The reasoning may relate to the prolonged time that FLD treatment samples took to reach lower E_h conditions, as adequate oxygen in the samples remained for some time after the start of incubation.

2.5.3 Soil Chemical Properties

Correlations between GHG emissions and soil C and N concentrations were not evident across all land use and moisture treatments. This is contrary to the hypothesis

that substrate values would show influence on the amount of observed GHG emissions. The similarity in soil C and N concentrations among the land uses may have been the main reason for this lack of correlation. The best correlated data were $KMnO_4$-oxidizable C with CO_2 emissions ($p \leq 0.0581$). In contrast to this are the positive strong correlations between SOC and CH_4 (Koh et al., 2009) and CO_2 (Franzluebbers and Follett, 2005). N_2O correlations were performed with and without N-applied samples with neither showing correlations at a significance level of $p \leq 0.05$.

2.5.4 Soil pH

Soil pH values showed an overall decrease across all H_2O treatments through the incubation. The decrease in the FLD treatment pH diverges from the accepted notion that alkaline and acidic soils will converge to neutral when submerged (Mitsch and Gosselink, 2000). However, it does support the findings by Unger (2008) who performed research on similar soils adjacent to the sample collection sites and also found no neutral convergence. Soil CO_2 and N_2O emissions were positively correlated with pH, while CH_4 had no correlation. This is also in contrast to findings by van der Weerden et al. (1999) and Van den Heuvel et al. (2011). They found a strong negative relationship between soil pH and N_2O emissions due to the decrease in N_2O : (N_2O+N_2) ratio under acidic conditions. Van den Heuvel cites the highest efflux rates at pH=5, and theorizes isolated acidic conditions as the cause of N_2O emission hotspot occurrence.

2.5.5 Soil Redox Potential

Soil redox potential remained stable for FLX and OPT and decreased for FLD treatments. No decrease in E_h value was expected in FLX and OPT samples as open pore space provided for oxygen diffusion throughout the incubation. The small variations in

FLX E_h agree with D'Amore et al. (2004) who indicated a decrease in redox potential under saturation, but an immediate rise upon soil drainage. FLD samples slowly reduced in E_h values becoming significantly different from the other treatments at Day 13. The overall negative trend for FLD E_h continued through the end of the experiment. Research by Mansfeldt (2003) described this same soil redox potential trend being associated with increased saturation periods. A strong negative correlation ($p < 0.0001$) occurred between FLD CH_4 emissions and redox potential. Koh et al. (2009) used a multiple regression model to fit CH_4 emissions to factoring soil properties. In this model soil temperature and soil redox potential explained 65% of the variation measured. Unger (2008) measured the effects of flooding on soil properties in a flood simulation laboratory at HARC adjacent to the field areas studied in this research. Redox values measured were slightly lower than those continuously measured by Unger in five week stagnant and flowing floods. Sumner (2000) described a general scenario where CH_4 production was increased with Eh values lower than -150 mV at a neutral pH. FLD samples approximately reached this redox potential within a three week period, slightly longer than the time period stated by Kirk (2004) However, large soil CH_4 efflux rates were delayed until submerged approximately seven weeks. Soil redox potential proves to be a valuable measurement in studying CH_4 production, but may have less importance in soil CO_2 and N_2O measurements for the environmental conditions studied under this research.

2.6 Conclusions

Both land use management and water regime influenced soil GHG emissions in the MRF soil studied in this research. Soil CO_2 emissions were related to both land use and the water regime treatments. The effects of land use occurred despite few differences

observed in initial soil C and N levels. The OPT treatment soil CO_2 emissions were significantly higher than those of FLD and FLX, supporting a large base of literature on this subject. Soil CH_4 emissions showed no influence of land use, but were associated with water regime treatments. Soil CH_4 emissions increased during the FLD treatment with significant increases in cumulative soil CH_4 emissions after Day 51 due to decreases in E_h. Multiple factors showed strong independent and interactive influences on soil N_2O efflux rates including time of incubation, land use, and water regime. The two N-applied samples, AG-N and AF-N, emitted the highest amount of soil N_2O. The emissions from these fertilized treatments decreased over time possibly as the NO_3-N in the soil was reduced.

Nonsignificant correlation of GHG emission with soil pH measurements contrasted with previous published literature and the reasons for this lack of correlation are unclear. Soil redox potential strongly correlated with soil CH_4 emissions as expected. This measurement can be a valuable tool in estimation of CH_4 emissions across the MRF where analysis of GHG emissions is not practical.

While this analysis has evaluated soil GHG emission traits in the MRF, further research is necessary to characterize the cumulative GHG emissions of the land uses as a whole. Pathways through vegetation are an important source of gas diffusion into and out of soils. An often cited example of plant mediated gas exchange is CH_4, which can account for as much as 80-90% of an agricultural field emissions. Micrometeorological data collection techniques could be utilized to encompass both soil and plant emissions as seen in several research projects including the AmeriFlux Network.

2.7 References

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2.8 Tables and Figures

Table 2.1. Components, settings, and calibration standards used during greenhouse gas testing using Shimadzu GC-2014 Gas Chromatograph.

Gas Chromatograph Components and Settings	
Carrier Gas	Helium
Carrier Gas Flow Rate	31.29 ml/min
Makeup Gas	P-5 (5% CH_4/95% Ar)
Other Gases Used	Air, Hydrogen
Column Temperature	80^0C
FID Temperature	250^0C
ECD Temperature	325^0C
Column Types*	1 m Hayseep T 80/100
	4 m Hayseep D 80/100
	.7 m Shim -Q 100/180
	1.5 m Hayseep N 80/100
	1.5 m Hayseep N 80/100
Sample Loop Size	1 ml

Calibration Standards		
Calibration Level	Component	Concentration (ppm)
1	CH_4	1
	CO_2	200
	N_2O	1
2	CH_4	5
	CO_2	400
	N_2O	10
3	CH_4	12
	CO_2	800
	N_2O	25
4	CH_4	25
	CO_2	1500
	N_2O	50

*All columns are 3.175 mm Inner Diameter and made of stainless steel

Table 2.2. Initial surface soil properties (0-10 cm) of land use areas in the Missouri River Floodplain (MRF).

Land use	pH_w	Exch. acidity	Bray I P	Ca	Mg	K	CEC
		$cmol_c\ kg^{-1}$	---------- kg ha^{-1} ----------				$cmol_c\ kg^{-1}$
Row-crop agriculture	5.9	15	75.04	5036	504	544	152
Riparian forest	6.2	10	62.72	3978	426	269	118
Agroforestry	6.1	15	66.08	4207	524	440	133

Table 2.3. Initial surface soil properties (0-10 cm) of land use areas in the Missouri River Floodplain (MRF). Arithmetic mean and standard error are presented by land use.

Land use	Bulk Density	Total org. C	Total N	POM*		$KMnO_4$ Oxid. C
				C	N	
	- g cm^{-3} -	---------- % of total soil ----------				mg kg $soil^{-1}$
Row-crop agriculture	1.31 ± 0.06	1.19 ± 0.29	0.14 ± 0.03	0.59 ± 0.14	0.05 ± 0.01	663 ± 5
Riparian forest	1.37 ± 0.04	1.44 ± 0.18	0.15 ± 0.02	0.41 ± 0.12	0.03 ± 0.01	688 ± 4
Agroforestry	1.35 ± 0.08	1.51 ± 0.44	0.14 ± 0.03	0.96 ± 0.35	0.06 ± 0.02	719 ± 2

*POM = particulate organic matter

Table 2.4. Soil particle size analysis of land use areas in the Missouri River Floodplain (MRF).

Land use	Sand	Silt	Clay	Texture
	---------- % ----------			
Row-crop Agriculture	17.5	57.5	25.0	silt loam
Riparian forest	17.5	67.5	15.0	silt loam
Agroforestry	10.0	72.5	17.5	silt loam

Table 2.5. ANOVA results for incubation soils based CO_2 emission efflux rates.

Effect	Df	F	Pr > F
Land use	4	5.18	0.003
Water	2	14.82	<.0001
Land use * Water	8	1.94	0.0925
Day	19	2.59	0.0003
Day * Land use	76	1.21	0.1195
Day * Water	38	4.39	<.0001
Day * Land use * Water	152	1.06	0.3079

Table 2.6. ANOVA results for incubation soils based CH_4 emission efflux rates.

Effect	Df	F	Pr > F
Land use	4	0.9	0.4776
Water	2	4.22	0.025
Land use * Water	8	0.91	0.5237
Day	19	1.92	0.0108
Day * Land use	76	1.1	0.2651
Day * Water	38	1.96	0.0007
Day * Land use * Water	152	1.13	0.1637

Table 2.7. ANOVA results for incubation soils based N_2O emission efflux rates.

Effect	Df	F	Pr > F
Land use	4	9.75	<.0001
Water	2	15.32	<.0001
Land use * Water	8	7.28	<.0001
Day	19	4.41	<.0001
Day * Land use	76	2.97	<.0001
Day * Water	38	5.98	<.0001
Day * Land use * Water	152	2.89	<.0001

Table 2.8. Pearson correlation coefficients (r) between cumulative soil CO_2 emissions and soil total organic C and N properties

Soil C and N properties	Soil CO_2 emissions	
	Correlation coefficient (r)[†]	P value
TOC	0.012	0.833
TN	-0.127	0.576
POM-C	0.247	0.824
POM-N	0.124	0.623
$KMnO_4$ Oxid.-C	0.016	0.058

[†]number of observations (n) = 45 for all correlations

Table 2.9. Pearson correlation coefficients (r) between cumulative soil CH_4 emissions and soil total organic C and N properties.

Soil C and N properties	Soil CH_4 emissions	
	Correlation coefficient (r)†	P value
TOC	-0.032	0.9397
TN	-0.086	0.4065
POM-C	-0.034	0.1022
POM-N	-0.075	0.4166
$KMnO_4$ Oxid.-C	-0.285	0.9173

†number of observations (n) = 45 for all correlations

Table 2.10. Pearson correlation coefficients (r) between cumulative soil N_2O emissions and soil total organic C and N properties.

Soil C and N properties	Soil N_2O emissions	
	Correlation coefficient (r)†	P value
TOC	0.107	0.4825
TN	0.131	0.3897
POM-C	-0.126	0.4099
POM-N	-0.133	0.3852
$KMnO_4$ Oxid.-C	0.06	0.6975

†number of observations (n) = 45 for all correlations

Table 2.11. Pearson correlation coefficients (r) between soil redox and soil GHG emissions.

Soil GHG emissions	Soil Redox (E_h)		
	Correlation coefficient (r)	P value	DF
CO_2	0.01584	0.6376	886
CH_4	-0.24675	<.0001	887
N_2O	-0.03321	0.3235	886

Table 2.12. Pearson correlation coefficients (r) between soil pH and soil GHG emissions

Soil GHG emissions	Soil pHw		
	Correlation coefficient (r)	P value	DF
CO_2	0.09358	0.0125	710
CH_4	-0.02663	0.4787	711
N_2O	0.07971	0.0337	710

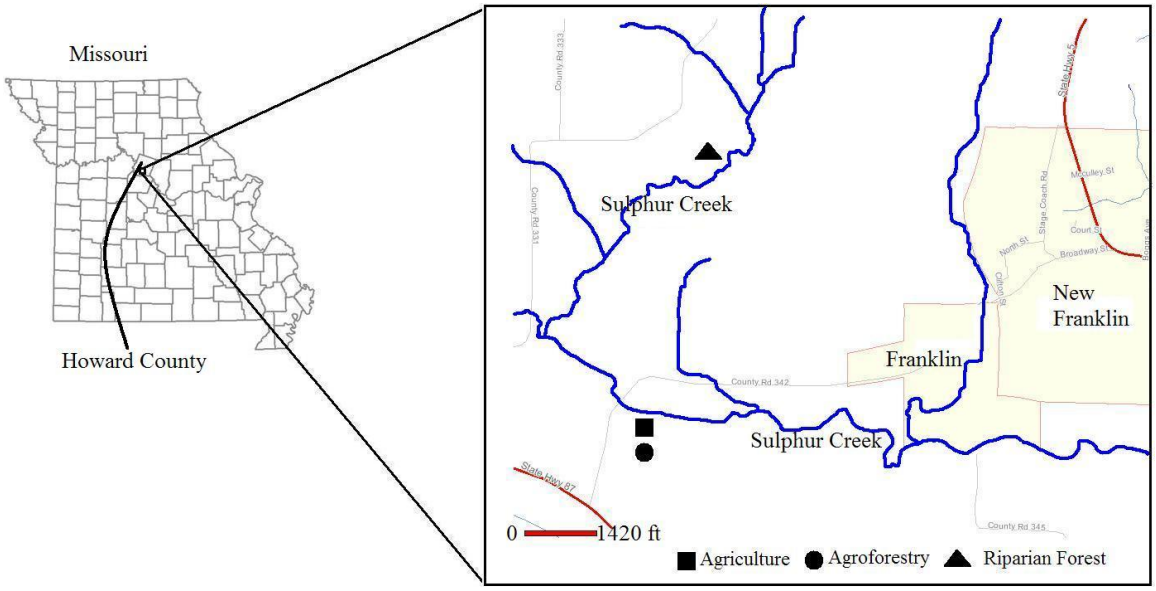

Figure 2.1. Map of sampling sites located near Franklin, MO showing proximity to Sulphur Creek.

Figure 2.2. Laboratory setup of greenhouse gas emission measurements showing gas manifold used to flush the incubation units with He gas and sample vials for storing gas samples.

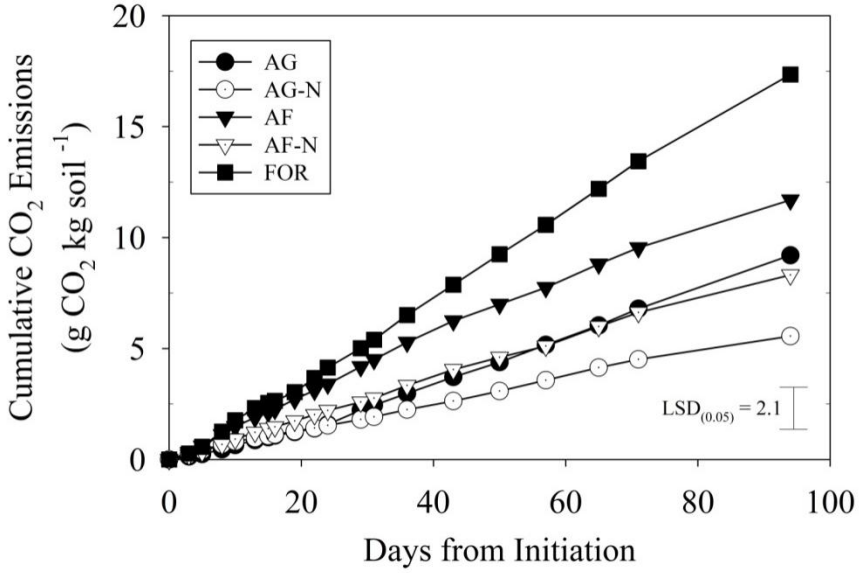

Figure 2.3. Land use effects on cumulative soil CO_2 emissions through incubation period of agricultural (AG), nitrogen fertilized agricultural (AG-N), agroforestry (AF), fertilized agroforestry (AF-N), and riparian forest (FOR) soils.

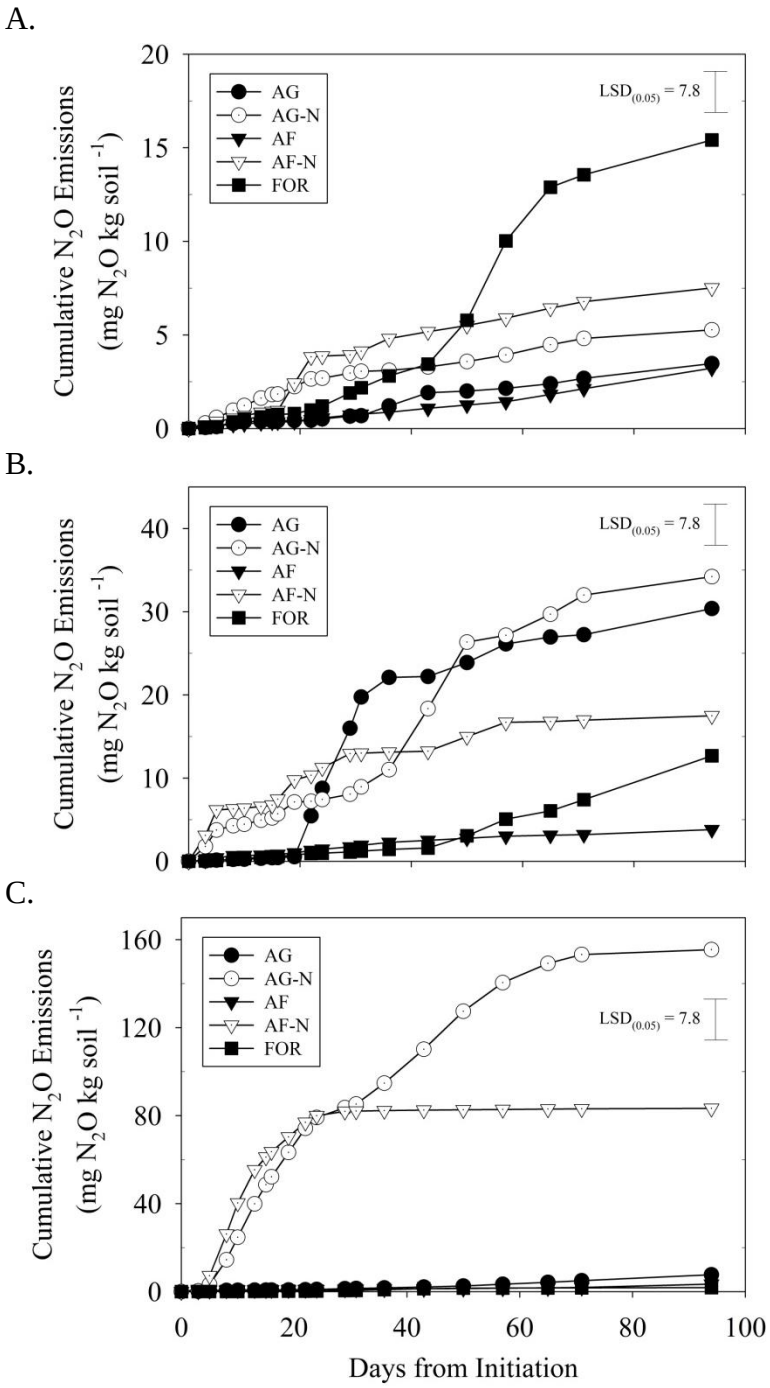

Figure 2.4. Land use effects of agricultural (AG), nitrogen fertilized agricultural (AG-N), agroforestry (AF), fertilized agroforestry (AF-N), and riparian forest (FOR) soils on cumulative soil N_2O emissions among three water regime treatments: A. Optimal (OPT) B. Fluctuating (FLX) C. Flooded (FLD). Note change of scale for emissions values.

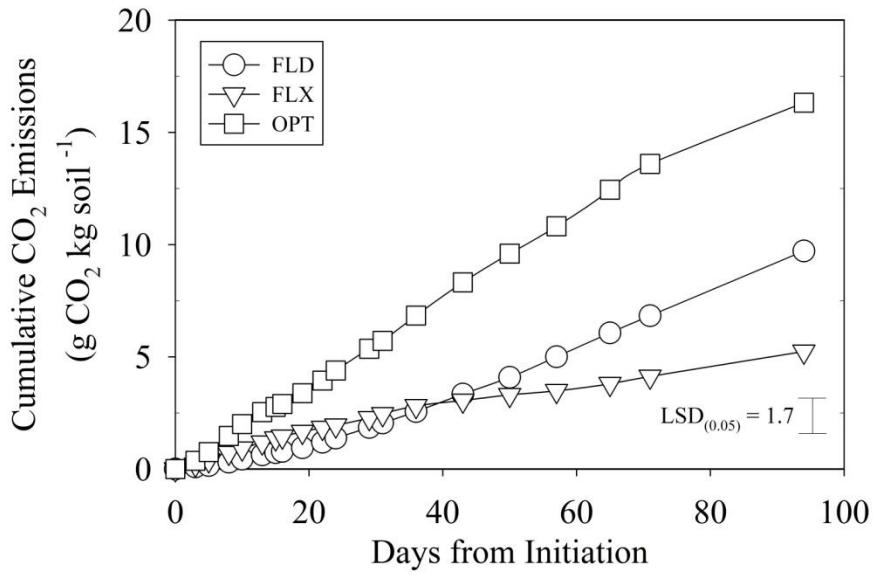

Figure 2.5. Water regime effects on cumulative soil CO_2 emissions through incubation period. The incubation included three water regime treatments: A. Optimal (OPT) B. Fluctuating (FLX) C. Flooded (FLD).

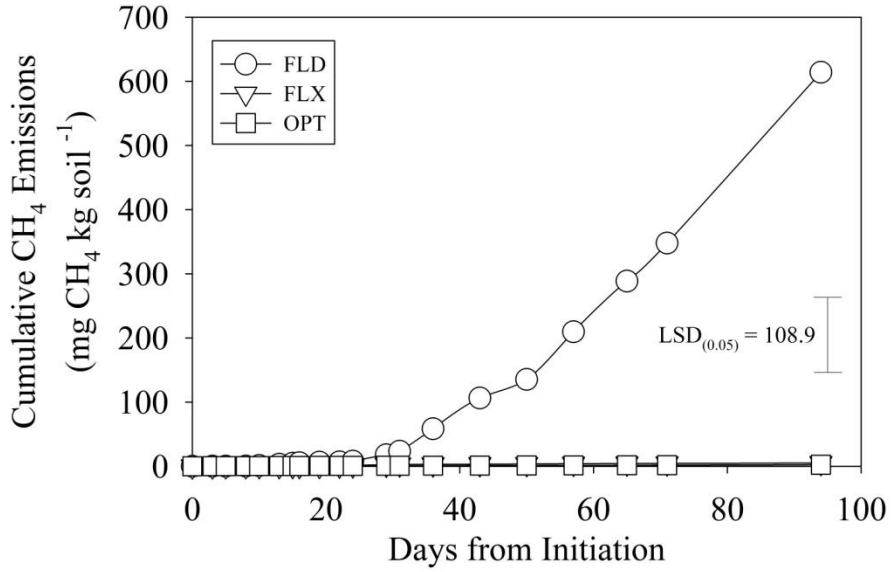

Figure 2.6. Water regime effects on cumulative soil CH_4 emissions through incubation period. The incubation included three water regime treatments: A. Optimal (OPT) B. Fluctuating (FLX) C. Flooded (FLD).

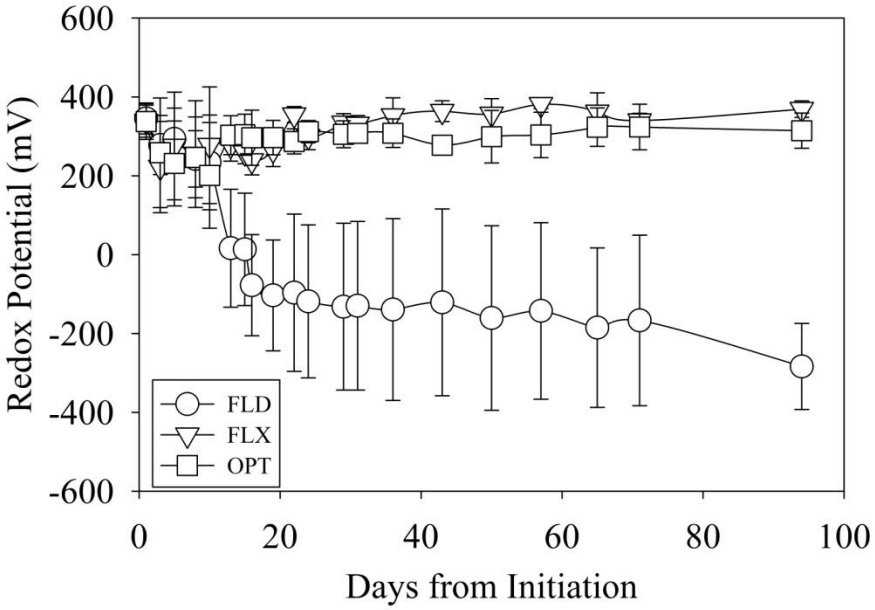

Figure 2.7. Water regime effects on soil redox potential (E_h) through incubation period. Standard error bars are presented per sampling date. The incubation included three water regime treatments: A. Optimal (OPT) B. Fluctuating (FLX) C. Flooded (FLD).

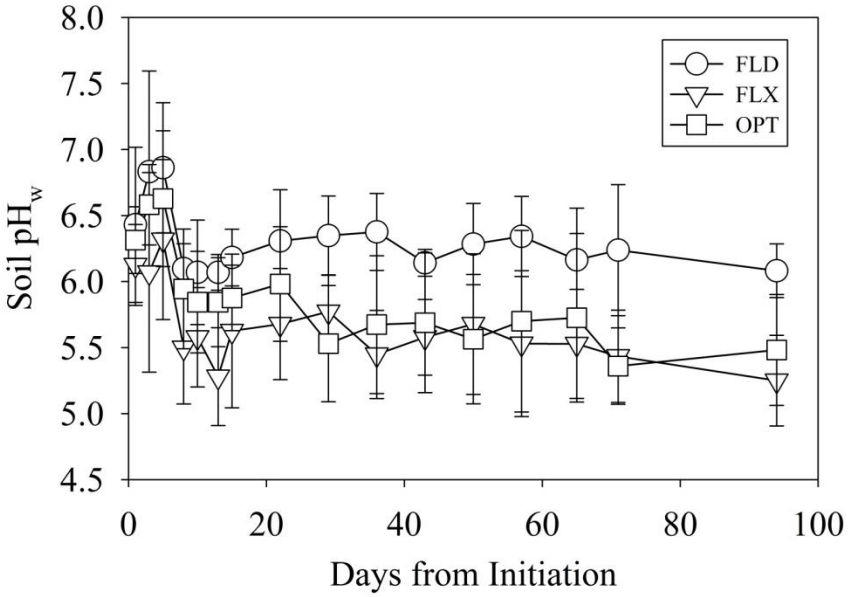

Figure 2.8. Water regime effects on soil pHw through incubation period. Standard error bars are presented per sampling date. The incubation included three water regime treatments: A. Optimal (OPT) B. Fluctuating (FLX) C. Flooded (FLD).

CHAPTER 3: EFFECT OF LAND USE ON SPATIAL VARIATION OF SOIL GREENHOUSE GAS PRODUCTION

3.1 Abstract

Agricultural sources contribute to global greenhouse gas (GHG) emissions. The fertile lower Missouri River Floodplain (MRF) has largely been converted to agricultural land use and is thought to be a large source of GHG, yet few studies have directly investigated GHG emissions in the MRF and provided quantitative analysis of spatial variation in GHG flux. Such research is needed to design optimal sampling strategies for determination of cumulative GHG emissions. This study's objective was to spatially characterize and evaluate soil GHG emissions in floodplain soils in response to differences in soil moisture, temperature, land use, and N inputs. Three intensive GHG samplings occurred during the 2011 growing season to determine GHG flux and changes in soil properties from agriculture, agroforestry, and riparian forested areas. No significant relationships were found between soil properties and gas emissions. Limited significant relationships at $p \leq 0.05$ were found between soil properties and log transformed GHG emissions. Methane efflux rates were low in the majority of samplings, but were high in locations with optimal soil conditions. Carbon dioxide rates fluctuated the least out of all three gases tested, with coefficient of variation (CV) values ranging from 47-133%. Nitrous oxide efflux rates were greatest in the May agricultural soil samples following N fertilization. Spatial variation was wide ranging in some situations having CV values as low as 66, while peaking at 351%. Additional testing is

needed in the lower MRF to infer the dominant soil properties governing the spatial variation of GHG efflux rates.

3.2 Introduction

Global climate change is directly related to the emission of greenhouse gases and has been largely affected in recent centuries by the increase of anthropogenic sources. The spread of agriculture and its associated greenhouse gas (GHG) sources now account for 10-12% of annual anthropogenic GHG production (Intergovernmental Panel on Climate Change, 2007b). The GHGs produced through soil processes with the highest global impact are carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) accounting for 63%, 18% and 6% of total long lived GHG radiative forcing, respectively (Intergovernmental Panel on Climate Change, 2007a). The variation in gas emissions is linked to the interactions between the numerous production and transportation factors of the individual gases (Fang and Moncrieff, 1998). These factors include a number of heterogeneous soil chemical, physical, and biological properties. Consequently, soil GHG emissions frequently have high levels of spatial and temporal variation (Laville et al., 1999) Quantification of GHG spatial variation is needed to design research that is statistically sound, defining the necessary amount of samples to accurately characterize GHG data. Such information will help researchers with bottom up estimation methods measuring field and regional scale GHG-substrate pools and fluxes for extrapolation to larger area estimations (Intergovernmental Panel on Climate Change, 2007a). Current models have issues with transposing scales; global models are imprecise at local levels and opposing field scale models lack the versatility to be relevant at macro levels (Nalley et al., 2011). Better understanding of field scale processes will improve modeling by

reducing spatial uncertainty at the regional, national, and international scales, a necessity for the creation of regulation and policy change (Li, 2007). More research is needed such as that by Ogle et al. (2006) who is improving GHG models and reducing uncertainty by coupling regional estimations with field-scale GHG data within Midwestern US agricultural systems.

The relationships between soil gas emissions and soil properties have been heavily researched yet remain unclear within alluvial environments such as the Missouri River Floodplain (MRF). Factors affecting soil microbial population, such as soil temperature, soil water content, and C and N substrate concentrations have displayed inconsistent GHG emissions effects among recent studies (Linn and Doran, 1984; Machefert et al., 2004; McKenzie et al., 1998; Stehfest and Bouwman, 2006). The large spatial variability of alluvial soil properties can be influenced by management practices (Rao and Wagement, 1985). Changes in vegetation, tillage, and the use of amendments affect the spatial distribution of soil properties and thus may affect the spatial variation of soil GHG emissions (Hefting et al., 2006). For example, the conversion from conventional to no till farming affects multiple GHG factors by leading to a net increase in soil organic carbon (SOC), increasing soil moisture, and often lowering soil surface temperatures. Practices that promote perennial vegetation including agroforestry also demonstrate an increase of GHG emissions through SOC sequestration. Species and arrangement of vegetation dictates soil carbon sequestration as illustrated by Oelbermann and Voroney (2007). The authors demonstrate SOC decreases away from trees in temperate agroforestry systems. The application method of fertilizer can affect GHG emissions from agricultural fields as broadcast applications may create more homogenous

soil N concentrations than subsurface applications of liquid fertilizers or anhydrous ammonia (Ciarlo et al., 2008). Observable influences of individual soil properties on GHG when other properties are not limiting factors (Dobbie et al., 1999). This may explain the inconsistencies of results concerning the relationships between soil properties and GHG emissions.

The MRF is an active agricultural region and as such, has the potential to be a large source of soil GHG emissions. No known studies have investigated these emissions, rather models have relied on extrapolated information to draw inferences (West et al., 2008). An accurate quantitative assessment of the spatial variation of MRF emissions is needed to begin to characterize the region. Spatial data will serve as the basis to conduct management and seasonal specific treatment efflux measurements, as well as to calculate site and regional annual cumulative emission rates. The MRF can support a variety of vegetation, yet has been largely converted to monoculture row crop agriculture (Galat et al., 1998). This study has the potential to initiate future work in search of alternative economically viable management practices mitigating GHG emissions within the MRF. Regionally adapted GHG mitigation projects are essential in reducing global annual emissions (Intergovernmental Panel on Climate Change, 2007b)

The objective of this study was to evaluate and spatially characterize soil GHG emissions (CO_2, CH_4, N_2O) in floodplain soils under agroforestry, row-crop agriculture, and forested systems in response to differences in soil moisture, temperature, land use, and N inputs.

3.3 Materials and Methods

3.3.1 Site Location

The sampling sites were located at the University of Missouri Horticulture and Agroforestry Center (HARC), a 267 ha research station situated near New Franklin, Missouri (39° 0‘ 0 N, 92° 46‘ 0 W) (Figure 2.1). The HARC research station borders Sulphur Creek on the south and west. This tributary’s catchment covers a large portion of southern Howard County and drains into Bonne Femme Creek shortly before its entry into the Missouri River. The northern portion of HARC sits on loess-capped river hills while the southern edge is in the Missouri River Floodplain, approximately 4 km from its current course. The proximity to both Sulphur Creek and the Missouri River has influenced the diverse soils of the area, encompassing seven soil associations (CAFNR, 2008). Both the floodplain and river hills are high in silt contents due to historic alluvial and aolean influences. The area has an annual average temperature of 12.6°C and 107 cm precipitation (National Weather Service, 2012). The floodplain has slopes of 0-2% and is situated at approximately 183 meters above sea level (Center for Applied Research and Environmental Systems, 2011).

Three varying land use types were selected along Sulphur Creek consisting of row crop agriculture (corn/soybean/wheat rotation), agroforestry (pecan orchard/hay), and a managed riparian forest sites (Tables 2.1, 2.2 for initial soil properties). The orchard and riparian forest buffer are both owned and managed by the University of Missouri while the row crop agricultural field is privately owned and adjoins HARC property. The soil at all three locations has been surveyed as Nodaway silt loam, (Fine-silty, mixed, superactive, nonacid, mesic Mollic Udifluvents) (Grogger and Landtiser, 1978) . This

soil is moderately well drained, but is subject to occasional flooding and ponding up to 14% annually (Natural Resources Conservation Service, 2005). The frequency of such events is tied to the soil's location along stream corridors. The top of the seasonally high water table is at 122 cm. Within Howard County Nodaway soils cover 6.7%, or 8,200 ha. This soil has predominately been used for agricultural production and as such is characterized by a silt loam Ap horizon (0-20 cm) overlaying a silt loam/silty clay loam C horizon (20-150 cm) (Natural Resources Conservation Service, 2005).

The 2.8 ha orchard consists of mixed pecan varieties (*Carya illinoinensis)* planted during autumn 1993 (Appendix A). Four groups of thirty-two trees were planted with four grass fields in a complete randomized block configuration. The grasses in all areas of the block are primarily tall fescue (*Festuca arundinacea)* with patches of johnson grass (*Sorghum halepense)*. Mean tree measurements are the following : total height 10.4 +/- 1.5 m, live crown height 2.4 +/- .3 m, DBH 19.8 +/- 3.6 cm. In February 2011 the ochard was thinned at a 50% rate, with the cutting of alternate trees leaving 16 trees. In the second week of June 2011 a broadcast application of urea fertlizer was applied at a rate of 84 kg NH_4NO_3–N ha^{-1} (Hunt, 2011). In the second week of July 0.2 g NH_4NO_3-N was applied to the 1.5 meter radius around each pecan tree.

The adjacent row crop field has been primarily used for growing corn (*Zea mays* Indenta) and soybeans (*Glycine max* Merr.*)* during much of the 20th century and has a current area of 30 ha. The current leaser also has planted winter wheat when possible. In 2009 soybeans were planted and the following year, due to early spring rains, the manager altered plans of planting corn and again planted no-till soybeans late May 2010. The field was conventionally tilled during the fall 2010. During the final week of March

2011 184 kg NH_3-N ha^{-1} was applied through injection to a depth of 15cm. The manager planted corn during the latter part of the 1st week and early into the 2nd week of April 2011. Due to above average precipitation in April a rescue application of N was applied in mid May 2011 consisting of a sidedress application of UAN solution which was dribble applied using a sprayer at the rate of 112 kg NH_4NO_3-N ha^{-1}.

The approximately 4 ha riparian forest site is primarily comprised of four tree species: silver maple (*Acer saccharinum*), american elm (*Ulmus americana*), sycamore (*Platanus occidentalis*), and cottonwood (*Populus deltoides*). A fifth species, green ash (Fraxinus pennsylvanica), was also found in a 2011 survey (Appendix B). Basal area was determined to be 30.5 m^2 ha^{-1}. In 2005-2006, 148 trees were harvested for an approximate volume of 46,164 board feet (102 m^3). At this time there are no plans for further management actions (Walter, 2010).

3.3.2 Sampling Design

Sampling locations were determined using a space filling curve (SFC) algorithm, "cover.design", within the R software package (Furrer et al., 2011). Lister and Scott (2009) demonstrated the usefulness of using a SFC to select unbiased and spatially balanced sampling points in natural resources monitoring. In the forest and agriculture sites, all sampling points were determined using this method (Figure 3.1). In the pecan orchard, 29 points were selected with the SFC and the remaining 16 points were paired with the 16 trees to provide more information about these point sources of variation. The tree paired samples were located 1 meter from the tree, within the approximately 1.5 meter drip line, and directionally distributed by random. August sampling points were selected using the same method with variables adjusted to create a more evenly

distributed sampling pattern, and also for easing collection effort due to the difficulties of accessing each site. The August agricultural sampling area was moved due to land owner concerns of crop damage to an adjacent area approximately 400 meters to the west.

3.3.3 GHG Sampling Chambers

Soil greenhouse gas collection chambers were created in winter 2011 for sampling following guidelines established by Rochette and Eriksen-Hamel (2008). The 42 cm tall polyvinyl chloride (PVC) chamber was made to be inserted 12 cm, having an above ground height of 30 cm. The chambers had an inner diameter of 20 cm for a total headspace of 9730 cm^3. Rubber caps were constructed to seal the chamber during sampling by use of an adjustable metal ring. Each cap was fitted with a 10 cm long, 0.3175 cm diameter, curved copper vent tube to alleviate pressure differences that can develop between the chamber and ambient atmosphere. A second brass fitting with 0.3175 cm plug septa was inserted into the cap for gas sampling by use of a syringe. Sampling chambers were installed at the study sites a minimum of two days prior to gas sampling to minimize the influence of installation disturbance upon the collected GHG samples.

3.3.4 Soil and GHG Sampling

GHG sampling strategies were adopted from the methods recognized by the USDA Agricultural Research Service (Follett et al., 2010) Gas samples were collected from each chamber at Time 0, 30, and 60 minutes following closure by the rubber cap. Technicians followed a routine beginning with the gas collection of a set of gas collection chambers every two minutes in a cycle to condense the sampling time. Twenty ml gas samples were collected and stored in 10 ml vials, sealed with grey butyl rubber stoppers.

Soil surface samples, 0-10 cm, were collected immediately adjacent to the gas collection chambers for lab characterization. Samplings were dispersed throughout the growing season occurring on May 10, June 1, and August 15, 2011.

3.3.5 Laboratory Analyses

Soil analyses were performed on each soil sample to test for relationships with the associated GHG sample. Soil gravimetric water content was measured by oven-drying field soil samples at 105°C in a forced air oven. $KMnO_4$-Oxidizable C was analyzed using a spectrophotometer in triplicate following the procedure outlined by Weil et al (2003). Soil nitrate (NO_3-N) and ammonium (NH_4-N) were measured with a Lachat QuickChem 8500 Flow Injection Analyses System using Lachat QuickChem Method 12-107-04-1-B and the Lachat QuickChem Method 12-107-06-2-A, respectively. Gas concentration analysis was carried out using a Shimadzu 2014 Gas Chromatograph (Columbia, MD) fitted with a methanizer, flame ionization detector, and electron capture detector (Table 2.6). The three timed gas samples were used to perform the soil efflux calculations.

3.3.6 Emissions Calculations and Statistical Analyses

Gas emission flux rates were calculated using both linear and Hutchinson and Mosier (HM) methods (Follett et al., 2010; Hutchinson and Mosier, 1981). The HM method defines the flux rate as:

$$f_0 = ((C_1 - C_0)^2 / [t_1 * (2*C_1 - C_2 - C_0)] * \ln[(C_1 - C_0) / C_2 - C_1)] * (V/A)) \quad [eq\ 1]$$

"where f_0 is the flux at time 0, C_0, C_1, and C_2 are the chamber headspace gas concentrations (ppm(v)) at time 0, 1, 2, respectively, t1 is the interval between gas sampling points, V is the chamber volume (L) and A is the chamber surface area (m2)."

Gas emissions and soil properties were examined per land use and seasonal measurement basis. Statistical analyses were completed using the software program R (R Development Core Team, 2011). Additional geospatial analysis was completed within the program using the package GeoR (Diggle and Ribeiro, 2007). Linear model and correlation functions were employed for the evaluation of soil property influences upon soil GHG emissions.

3.4 Results

3.4.1 Observed Soil Properties

Differences in soil properties were examined among the three samplings and land use treatments. In May, FOR soils had a higher mean soil gravimetric water content, 25%, than that of AG, 14%, and AF, 13% (Table 3.1). The drier land use areas corresponded with warmer temperatures as AG and AF soils were approximately 5°C warmer than the FOR soils. High variation in soil $KMnO_4$ Oxidizable C, NH_4-N, and NO_3-N measurements masked differences between the land uses. In particular, AG soils had much higher NO_3-N levels, but are not statistically different from the other treatments. The patterns of soil water content and temperature continued for the June sampling, but were not as pronounced (Table 3.2). FOR soils were again wetter and cooler (14.9°C) than AF (16.2°C) and AG (23.4°C). No significant differences among land uses were found in $KMnO_4$ Oxidizable C, NH_4-N, and NO_3-N measurements. In August, AG and AF soils had soil water contents of 15%, 5% drier than that of FOR (Table 3.3). AF soils were the warmest (26.4°C) while AG soils were 22.8°C, and FOR soils were the coolest (23.3°C). While AF NH_4-N, and NO_3-N values were much higher, the spatial variation is still too high to find discernable differences between the opposing

treatments. No significant differences were found land use treatments for $KMnO_4$ Oxidizable C measurements.

3.4.2 Soil GHG Efflux Rate Distributions

The distribution of soil CH_4, CO_2, and N_2O gas emission rates can be visually analyzed using a series of boxplot diagrams (Figures 3.2-3.4) and tables (Tables 3.4-3.6). The June AG CH_4 efflux values were large, and its distribution is the widest spread. May AF CH_4 samples also indicated a sizable CH_4 efflux and spread distribution. The remaining CH_4 distributions are centered near zero, and often negative, indicating a weak CH_4 sink. Soil CO_2 efflux distributions are fairly condensed, with each sampling showing outliers, or 'hot spots' areas of higher emission rates. The May AG sample shows many outliers in both CO_2 and N_2O efflux. In each of the three seasonal samplings AF soils had the higher mean CO_2 efflux rates than the respective AG and FOR samplings, with the highest mean sampling being AUG AF CO_2 emissions at 61.0 mL CO_2 m^{-2} h^{-1}. The lowest efflux rates were in the June FOR soils, at an average of 9.1 mL CO_2 m^{-2} h^{-1}, approximately three times lower than that of the next lowest CO_2 sampling.

With the exception of the May AG and AUG AF samplings, the soil N_2O efflux distributions are near zero. Within the May sampling the AG soil mean efflux rate, 178.4 µL N_2O m^{-2} h^{-1}, was by far higher than that of AF, 31.0 µL N_2O m^{-2} h^{-1}, and FOR 4.8 µL N_2O m^{-2} h^{-1}. FOR soils N_2O efflux peaked in June, 22.1 µL N_2O m^{-2} h^{-1}, while AF soils had the highest N_2O efflux in August, 28.3 µL N_2O m^{-2} h^{-1}.

The spatial variation of each seasonal land use sampling was interpolated using Inverse Distance Weighting techniques. May CH_4 efflux interpolations display weak

sink and emissions from each of the land use surfaces (Figure 3.5). Methane sinks occur in contiguous areas on the western side of both FOR and AG samples. Large and small CO_2 efflux rates in the May sampling are adjacent to each other, forming small sections of more consistent rates (Figure 3.6). May N_2O results show low, or negative values in FOR soils, and to some extent higher values in AF, but both are fairly even across the landscape (Figure 3.7). May AG N_2O efflux rates were high across the measured surface peaking in the northwest corner, producing the highest amount of N_2O within the study. The coefficient of variation (CV) values differed among each land use and GHG type (Table 3.7), with the highest variation found in CH_4 rate (172-496%). AF CO_2 efflux rates had the lowest CV, 70%, while the highest variation was in AG soils, 132%. The CV of N_2O efflux rates among the three land uses ranged from 146-219%.

June CH_4 efflux rates continued to be near zero for AG and FOR land uses, and had low to moderate variation across each surface (Figure 3.8). FOR soils had small areas of CH_4 sinks among the larger areas of weak emissions. Microtopographical lows in AF created areas of increased soil water content, and thus contributed towards the necessary conditions for increased soil CH_4 emissions. This circumstance occurred in multiple locations and resulted in very high CH_4 efflux rates in a low spot near the levee in the southwest corner as well as a scour hole in the north side of the field. Among the high emission areas were also two samples noted as CH_4 sinks. June CO_2 efflux rates were very low in FOR soils, with few spread out samples of higher emissions (Figure 3.9). AF and AG soils efflux rates were higher, yet still fairly consistent across space, with a small cluster of samples showing increased soil CO_2 efflux rates. June N_2O emissions were low in each land use type, lacking the major hot spots seen in the May

samplings (Figure 3.10). During the June sampling the GHG efflux rates variation changed among the three land uses as noted by the CV values (Table 3.8). Soil CH_4 efflux rate CV values ranged from 237-282%, while the CO_2 CV values ranged from 55-132%. The highest CV values of each CH_4 and CO_2 are attributed to FOR samples, while the variation in CH_4 was equally low among AF and AG, and the lowest in AF samples, CV=55. FOR land use type displayed the lowest amount of variation within N_2O efflux rates, CV=71%.

August CH_4 efflux rates were regularly around zero, displaying a majority of weak positive fluxes, yet negative fluxes did occur in some areas (Figure 3.11). The maps show the limited interpolation due to the high failure rate of the HM equation. August CO_2 efflux rates show increased spatial variation in each land use (Figure 3.12). High and low soil CO_2 efflux rates occurred close to each other, prompting the irregular interpolation results. In AG soils sections of low emissions formed a contiguous area in the north side of the field, while higher emissions were found in the southern portion. August N_2O efflux rates were low and regular across the AG and FOR surfaces (Figure 3.13). AF soil N_2O flux rates were lower on the eastern end of the field. High CV values of CH_4 efflux rates again are present in the August sampling, and could possibly be attributed to the low number of samples (Table 3.9). The highest CH_4 variation occurred in the AF land use, CV=490%. The variation among the CO_2 rates in AF and FOR samples was lowest of the three seasonal samplings. The range of CV values for CO_2 efflux was 47-77%. The CV of AF N_2O efflux rates was very high, 351%, much higher than AG, 67%, and FOR, 104%.

3.4.3 Soil GHG Efflux and Soil Properties Correlations

Using the Shapiro-Wilkes test null hypothesis of normal distribution for the majority of the HM calculated GHG emission rates were rejected at $p \leq 0.05$ (Table 3.10). Through a log transformation of the GHG emission rate the Shapiro-Wiles normal test normal distribution null hypothesis was not rejected at $p \leq 0.05$ for a larger percentage of the data (Table 3.11). Portions of the results did not fit a normal distribution through any transformation, and thus interpretations through linear models were limited.

Normal distributions of GHG gas efflux and the gas efflux log transformations permitted analysis through multiple regression techniques. Soil temperature, soil gravimetric water content, and $KMnO_4$-oxidizable C were not found to be significant variables for soil CO_2 and CH_4 flux at $p \leq 0.05$ (Tables 3.12-3.18). Soil temperature, soil gravimetric water content, soil NO_3-N, and soil NH_4-N were not found to be significant variables for soil N_2O efflux at $p \leq 0.05$ (Table 3.19). Significance of soil properties within log GHG emission flux models was inconsistent (Tables 3.20-3.34). At $p \leq 0.05$ significance occurred in the following log transformed efflux relationships: soil NO_3-N to May agriculture N_2O, soil $KMnO_4$-C to May agroforestry CH_4, soil temperature to May and August riparian forest CO_2, and soil gravimetric water content to June riparian forest CH_4. As a whole, the coefficient of determination values (R^2) of the log transformed CH_4, CO_2, and N_2O models were low, an indication of the lack of fit of the measured soil properties.

3.5 Discussion

The non-normal distribution of a large quantity of the soil GHG emission rates complicates the interpretation of the study and consequently these violations of regression assumption prevent the usage of linear models. Furthermore, logarithmic transformations failed to normalize many of the gas emission distributions, preventing interpretation in those cases as well. Cambardella et al. (1994) noted similar issues with non-normal and highly skewed distributions, observing that larger temporal sampling improved the issue only slightly. Also in similarity is the detail that log transformations only normalized a portion of their data, constraining their statistical approach. The authors cite management and temporal effects as likely, but unproven causes of these data issues.

The models present show few significant associations at $p \leq 0.05$ with the measured soil properties. Those relationships between soil properties and soil GHG efflux are that are significant are irregular among land use and seasonal influences. The lack of correlation in the literature is not unseen, but does contrast a body of literature supporting the influences of numerous soil properties on soil GHG emissions. Linn and Doran (1984) presented the effects of soil water content upon microbial processes including CO_2 production, with a maximum at 60% WFPS. Strong correlations were also found between SOC and CH_4 (Koh et al., 2009) and CO_2 (Franzluebbers and Follett, 2005). Studies have also identified temperature as a strong influence on soil microbial activity and soil GHG production such as those by McKenzie et al. (1998) and Mielnick and Dugas (2000). The weak relationships found may be due to the narrow data set collected, and could strengthen with increased temporal sampling across a wider range of

the tested variables. For instance, expansion of sampling to the winter would include much cooler soil temperatures and alternative vegetative states, allowing for a broader range for which to draw inference.

The low observed soil CH_4 fluxes were expected due to relatively dry conditions during the sampling times. Flooding and surface ponding during the spring season did not last long enough to induce the anoxic soil conditions required for sizeable CH_4 production. Late season soil and climatic conditions were typically dry, further reducing CH_4 efflux. The CO_2 and N_2O efflux fell within the reported ranges of previous research such as Bailey et al. (2009).

While no error is known, the lack of a normal distribution and poor model fit may indicate an issue within the data collection procedures. The complexity of closed open chamber soil GHG efflux methods increases the opportunity for inaccuracy and a larger volume soil chamber was used in this study due to the potential for flooding. Following testing, contamination and leakage of the sample collection containers has been ruled out. The septa sealed vials consistently held pressure through storage until the completion of the gas chromatograph analysis. This supports previous methods research by Glatzel and Well (2008) that supports the use of this equipment for the collection of air samples. The gas chromatograph used during testing has passed all quality assurance checks, and displayed little drift among samples. All gas samples had concentrations within the range of certified standards and above minimum detection limits (MDL).

A proposed methods issue that has potential to be a source of inaccuracy is the dimensions of the soil gas collection chamber. The ability to accurately quantify each of the extreme low and high GHG efflux situations is difficult with closed chamber

techniques (Conen and Smith, 1998). While following guidelines presented by Rochette and Eriksen-Hamel (2008), the 30 cm chamber height generates a large headspace volume, thus is suited towards the collection of gas samples at higher emission rates. Chamber construction was designed around ensuring sampling during flooding and the associated increased CH_4 efflux. The dry conditions during the 2011 sampling led to limited quantities of CH_4 and N_2O efflux. These small gas concentration values are hard to detect during the maximum one hour sampling period in larger headspace chambers. The chambers had little issue in the collection of the higher concentrated CO_2. Conversely, during optimal conditions the chambers had negative effects upon data collection because of high concentrations of GHG and a feedback relationship.

The result of collecting low GHG emission samples in larger closed chambers is an increase in data noise, masking the true emission rates to a degree. The calculation of GHG emission rates using the HM equation is preferred for accurate estimation, but does require specific conditions for its operation. Flux calculations cannot be calculated if the data diverges into one of the other 16 data patterns (Follett et al., 2010). HM failure principally influenced the data collection of CH_4 emissions, and to a lesser degree, N_2O emissions. The lack of HM accepted CH_4 sample points enlarged the difficulty in establishing spatial GHG patterns. The HM failure of CO_2 emissions was very low, having little impact upon these calculations.

While accepted as an underestimation of gas flux, linear regression methods can be employed when other estimations lack applicability. These estimations were employed for these analyses, but did not result in a pronounced improvement in clarifying spatial and soil property relationships. In recent years alternative methods

have been proposed such as a quadratic method by Wagner et al. (1997), stochastic diffusion model by Pedersen et al. (2001) non-steady state diffusive flux estimator (NDFE) by (Livingston et al., 2006). As three gas samples per chamber were collected for the flux calculations, these three methods were not employed as they are not recommended for calculation with four or less gas samples (2010)

The spatial interpolation maps visually display the variations in gas efflux across each land use surface. These images display the complexity of spatial data, and can help direct future investigations. The interpolations show the 'hot spot' areas of high GHG efflux while surrounding areas can have relatively low emissions. The variation among samplings is also put into great perspective when judged against one another. For example, the August CH_4 efflux rates show variation within each land use, but the variance among samples is relatively low when compared to more optimum conditions in June (Figure 3.10). As tested, the spatial correlation was often weak among the linear models. Even while the sampling was denser than a majority of closed chamber GHG studies, the distance between points was too much. Soil gas emission investigations in the MRF need to be conducted on a finer scale to obtain clear information about spatial dependence. This information should be considered in the interpretation of the spatial interpolation diagrams.

Closed chamber techniques for the measurement of soil greenhouse gases remains viable for researchers with the state of currently available technology. Automated chamber and micrometeorological techniques have been successfully employed in several studies such as Skiba et al. (1996), Arnold et al. (2001), and Merbold et al. (2009) but their cost prohibits their widespread use.

The measurement of additional soil properties may improve the understanding of GHG efflux from the MRF soils. Follett et al. (2010) suggests the emissions could be correlated with particulate organic matter (POM-C, POM-N), water extractable substrates (WEOC, WEON), P, K, and pH. Analysis of the management practice as whole may illustrate land use impacts in a more holistic manner. An example being West and Marland (2002) where the analysis was not only of changes to soil GHG, but also those involving plant respiration as well as energy consumption from machinery and transportation. Another important aspect for investigation is the role of plants in the emissions of soil based GHG. Yu et al. (1997) found a large portion of CH_4 emissions were emitted through plant pathways, rather than directly from the soil surface. Other research has suggested that the type and density of vegetation affects efflux through the alteration of soil physical and chemical properties (Christensen et al., 2003). It is unknown how vegetation transported GHG would have affected the results of this study.

3.6 Conclusions

The results of this study show the challenges associated with determining cumulative GHG emission in the MRF and other similar environments characterized by occasional flooding. Measuring three GHG simultaneously requires special analytical capabilities and limits the numbers of samples that can be analyzed due to the relatively lengthy analysis time. The statistical challenges include the difficulty of using established equations for flux calculations, especially with CH_4, which only has periodic emissions dependent on flooding events and the non-normal distribution of the data. In addition, high spatial variation and the presence of "hot spots" or areas of high flux observed in the research indicates that representative sampling may require very large

numbers of samples across each site to capture this spatial variability. Land management practices, such as land-leveling for agricultural drainage and tree planting for agroforestry, may create microtopographic differences which produce these hot spots. Due to these statistical challenges, few significant relationships between soil properties, such as soil water content, and GHG emissions were observed

Further testing of appropriate chamber designs and flux calculation methods may be needed for the MRF. The larger chamber design used in this research was thought appropriate to be able to measure GHG flux under flooded conditions and was pre-tested, but it may have caused problems under field conditions associated with gas mixing and detection limits during each sampling period. Access to the field when flooded conditions occurred was limited and, therefore, soil GHG emissions were only measured when soil water content was lower, which may not be representative of the MRF. It is suggested that representative cumulative GHG emissions estimates for this region will require a method to determine GHG flux throughout the duration of flooding events.

3.7 References

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3.8 Tables and Figures

Table 3.1. Selected soil properties of soil samples collected in May by land use site. The first number presented indicates the mean value. The second number (in italics) presented represents the standard deviation.

Land Use	Grav. Water Content	Soil Temp.	$KMnO_4$ Oxid. C	NH_4-N	NO_3-N
	- % -	-- °C --	---------- mg kg^{-1} ----------		
Agriculture	14.01	23.53	920.34	46.27	622.16
	1.97	*1.56*	*111.94*	*13.37*	*515.36*
Agroforestry	13.07	17.35	981.43	51.49	278.26
	3.34	*1.73*	*155.14*	*17.64*	*241.97*
Riparian Forest	24.95	18.56	1154.77	38.95	246.28
	3.72	*1.43*	*274.08*	*8.54*	*101.04*

Table 3.2. Selected soil properties of soil samples collected in June by land use site. The first number presented indicates the mean value. The second number (in italics) presented represents the standard deviation.

Land Use	Grav. Water Content	Temp.	$KMnO_4$ Oxid. C	NH_4-N	NO_3-N
	- % -	°-- C --	---------- mg kg^{-1} ----------		
Agriculture	20.70	23.40	1024.75	39.55	356.92
	1.75	*0.49*	*166.95*	*6.80*	*122.45*
Agroforestry	23.28	16.18	1058.79	50.79	267.94
	4.20	*1.31*	*280.80*	*21.86*	*258.56*
Riparian Forest	26.66	14.87	1387.95	37.27	173.31
	2.90	*1.04*	*717.90*	*11.11*	*104.32*

Table 3.3. Selected soil properties of soil samples collected in August by land use site. The first number presented indicates the mean value. The second number (in italics) presented represents the standard deviation.

Land Use	Grav. Water Content	Temp.	$KMnO_4$ Oxid. C	NH_4-N	NO_3-N
	- % -	°- C -	---------- mg kg^{-1} ----------		
Agriculture	14.39	22.75	1200.38	32.07	94.40
	2.02	*1.13*	*230.86*	*10.90*	*34.76*
Agroforestry	14.72	26.37	1065.88	201.88	480.47
	3.14	*0.98*	*250.28*	*701.02*	*636.77*
Riparian Forest	19.94	23.29	1372.64	29.87	113.91
	3.68	*0.83*	*345.24*	*11.51*	*52.32*

Table 3.4. Distribution of soil CH_4 flux (µL m^{-2} h^{-1}) across land use and sampling time. The first number indicates the mean rates of emissions. The second number (in italics) indicates the emissions rate standard deviation.

	Land use		
	AG	AF	FOR
May	-2.68	77.90	-18.96
	12.01	*125.36*	*100.69*
June	318.82	1.56	-3.30
	738.84	*1.37*	*9.11*
August	-3.31	-1.74	-2.87
	7.28	*8.23*	*4.29*

Table 3.5. Distribution of soil CO_2 flux (mL m^{-2} h^{-1}) across land use and sampling time. The first number indicates the mean rates of emissions. The second number (in italics) indicates the emissions rate standard deviation.

	Land use		
	AG	AF	FOR
May	50.25	52.82	44.35
	65.70	*36.07*	*32.70*
June	30.60	30.80	9.08
	19.42	*16.75*	*11.88*
August	26.98	60.95	39.06
	20.58	*28.27*	*19.60*

Table 3.6. Distribution of soil N_2O emissions (µL m^{-2} h^{-1}) across land use and sampling time. The first number indicates the mean rates of emissions. The second number indicates the emissions rate standard deviation.

	Land use		
	AG	AF	FOR
May	178.35	30.99	4.82
	330.96	*44.19*	*9.66*
June	3.04	10.81	22.06
	3.54	*13.33*	*15.81*
August	6.65	28.31	4.35
	4.38	*98.19*	*4.47*

Table 3.7. Coefficient of variation of GHG fluxes from May sampling.

	CH_4	CO_2	N_2O
	---------- % ----------		
Agriculture	457.3	132.3	187.9
Agroforestry	172.0	70.0	146.3
Riparian Forest	496.4	74.6	219.3

Table 3.8. Coefficient of variation of GHG fluxes from June sampling.

	CH_4	CO_2	N_2O
	---------- % ----------		
Agriculture	237.2	64.2	188.5
Agroforestry	237.2	55.0	120.6
Riparian Forest	282.3	132.6	71.4

Table 3.9. Coefficient of variation of GHG fluxes from August sampling.

	CH_4	CO_2	N_2O
	---------- % ----------		
Agriculture	226.5	77.2	66.6
Agroforestry	490.0	46.9	351.0
Riparian Forest	156.9	50.8	104.0

Table 3.10. Shapiro-Wilk Normality test of gas emission data. The first number in each comparison represents the test statistic (w) and the second number represents the P value for the comparison. P values greater than or equal to 0.05 are in bold.

		Gas Emissions		
Month	Land use	CH_4	CO_2	N_2O
May	AG	0.7294	0.7499	0.773
		2.53E-05	*2.78E-07*	*1.55E-06*
	AF	0.958	0.9747	0.7837
		0.7908	***0.8336***	*0.0004994*
	FOR	0.4852	0.8066	0.4553
		1.78E-08	*6.14E-06*	*2.92E-10*
June	AG	0.694	0.8492	0.5504
		1.65E-05	*7.14E-05*	*6.73E-09*
	AF	0.694	0.9618	0.9204
		1.65E-05	***0.1821***	*0.01874*
	FOR	0.9055	0.7531	0.9516
		0.03829	*1.04E-06*	***0.1268***
August	AG	0.8242	0.8664	0.9378
		0.004436	*0.0001639*	*0.02167*
	AF	0.9346	0.9652	0.3235
		0.319	***0.2134***	*1.15E-12*
	FOR	0.9734	0.9684	0.7278
		0.9183	***0.2915***	*3.69E-07*

Table 3.11. Shapiro-Wilk Normality Test of Log transformation of gas flux data. The first number in each comparison represents the test statistic (w) and the second number represents the P value for the comparison. P values greater than or equal to 0.05 are in bold.

		Log (Gas Emissions)		
Month	Land use	CH_4	CO_2	N_2O
May	AG	0.9221	0.9504	0.9819
		0.3366	*0.05684*	***0.746***
	AF	0.9593	0.946	0.9659
		0.8035	***0.2856***	***0.6669***
	FOR	0.9165	0.9862	0.9615
		0.09769	***0.8854***	***0.2694***
June	AG	0.8627	0.9774	0.8085
		0.01684	***0.5794***	*5.92E-05*
	AF	0.9153	0.9678	0.8928
		0.123	***0.2922***	*0.005617*
	FOR	0.9171	0.9794	0.9373
		0.4474	***0.6827***	*0.04622*
August	AG	-	0.9592	0.9197
		-	***0.1377***	*0.005228*
	AF	0.8132	0.9468	0.8345
		0.1034	*0.04573*	*2.64E-05*
	FOR	-	0.8929	0.8751
		-	*0.0008885*	*0.0004604*

Table 3.12. ANOVA results for May agroforestry soil CH_4 emissions.

Effect	Df	Sum of Squares	Mean Square Error	F Value	Pr(>F)
Water	1	48123	48123	7.0701	0.05645
Temp	1	635	635	0.0934	0.77519

$KMnO_4$-Ox. C	1	49741	49741	7.3079	0.05391
Residuals	4	27226	6807		

Table 3.13. ANOVA results for May agroforestry soil CO_2 emissions.

Effect	Df	Sum of Squares	Mean Square Error	F Value	Pr(>F)
Water	1	753840000	753839472	0.6198	0.44196
Temp	1	4460700000	4460686295	3.6675	0.07246
$KMnO_4$-Ox. C	1	1427000000	1426982191	1.1733	0.29386
Residuals	17	20676000000	1216261993		

Table 3.14. ANOVA results for June agroforestry soil CO_2 emissions.

Effect	Df	Sum of Squares	Mean Square Error	F Value	Pr(>F)
Water	1	250790000	250791684	0.6207	0.4358
Temp	1	223540000	223540524	0.5533	0.4617
$KMnO_4$-Ox. C	1	36774000	36773977	0.091	0.7646
Residuals	37	14950000000	404047910		

Table 3.15. ANOVA results for August agroforestry soil CH_4 emissions.

Effect	Df	Sum of Squares	Mean Square Error	F Value	Pr(>F)
Water	1	8.23	8.235	0.1053	0.7516
Temp	1	140.3	140.303	1.7947	0.2074
$KMnO_4$-Ox. C	1	6.77	6.775	0.0867	0.7739
Residuals	11	859.92	78.175		

Table 3.16. ANOVA results for August agroforestry soil CO_2 emissions.

Effect	Df	Sum of Squares	Mean Square Error	F Value	Pr(>F)
Water	1	2105100000	2105079171	2.573	0.1168
Temp	1	349350000	349351505	0.427	0.5173

$KMnO_4$-Ox. C	1	19979	19979	2.442E-05	0.9961
Residuals	39	31907000000	818128439		

Table 3.17. ANOVA results for August riparian forest soil CH_4 emissions.

Effect	Df	Sum of Squares	Mean Square Error	F Value	Pr(>F)
Water	1	6.49	6.49	0.3019	0.5998
Temp	1	0.765	0.765	0.0356	0.8558
$KMnO_4$-Ox. C	1	45.068	45.068	2.0966	0.1909
Residuals	7	150.466	21.495		

Table 3.18. ANOVA results for August riparian forest soil CO_2 emissions.

Effect	Df	Sum of Squares	Mean Square Error	F Value	Pr(>F)
Water	1	12291000	12290800	0.0301	0.8631
Temp	1	7855500	7855515	0.0193	0.8904
$KMnO_4$-Ox. C	1	615470000	615467226	1.5088	0.2269
Residuals	38	15501000000	407915547		

Table 3.19. ANOVA results for June riparian forest soil N_2O emissions.

Effect	Df	Sum of Squares	Mean Square Error	F Value	Pr(>F)
Water	1	184.5	184.49	0.7766	0.3852
Temp	1	54.4	54.4	0.229	0.6357
NH_4-N	1	858.2	858.16	3.6123	0.067
NO_3-N	1	8.8	8.8	0.037	0.8487
Residuals	30	7127	237.57		

Table 3.20. ANOVA results for log transformation of May agriculture soil CH_4 emissions.

Effect	Df	Sum of Squares	Mean Square Error	F Value	Pr(>F)
Water	1	0.51557	0.51557	1.1785	0.3136

Temp	1	0.39079	0.39079	0.8933	0.3761
$KMnO_4$-Ox. C	1	0.05527	0.05527	0.1263	0.7327
Residuals	7	3.06223	0.43746		

Table 3.21. ANOVA results for log transformation of May agriculture soil N_2O emissions.

Effect	Df	Sum of Squares	Mean Square Error	F Value	Pr(>F)
Water	1	0.3329	0.33287	1.2319	0.274395
Temp	1	0.9293	0.92931	3.4392	0.071872
NH_4-N	1	1.0916	1.09157	4.0397	0.051981
NO_3-N	1	2.5359	2.5359	9.3849	0.004127
Residuals	36	9.7276	0.27021		

Table 3.22. ANOVA results for log transformation of May agroforestry soil CH_4 emissions.

Effect	Df	Sum of Squares	Mean Square Error	F Value	Pr(>F)
Water	1	1.6601	1.6601	1.6779	0.2649
Temp	1	0.0253	0.0253	0.0256	0.88068
$KMnO_4$-Ox. C	1	8.7549	8.7549	8.8488	0.04095
Residuals	4	3.9576	0.9894		

Table 3.23. ANOVA results for log transformation of May agroforestry soil CO_2 emissions.

Effect	Df	Sum of Squares	Mean Square Error	F Value	Pr(>F)
Water	1	0.00888	0.008878	0.0793	0.7816

Temp	1	0.25537	0.255374	2.2821	0.1492
$KMnO_4$-Ox. C	1	0.0846	0.084604	0.7561	0.3967
Residuals	17	1.90234	0.111902		

Table 3.24. ANOVA results for log transformation of May agroforestry soil N_2O emissions.

Effect	Df	Sum of Squares	Mean Square Error	F Value	Pr(>F)
Water	1	0.697	0.69698	0.9235	0.3518
Temp	1	0.0365	0.03653	0.0484	0.8288
NH_4-N	1	0.0371	0.03709	0.0492	0.8275
NO_3-N	1	0.6403	0.64032	0.8485	0.3716
Residuals	15	11.3202	0.75468		

Table 3.25. ANOVA results for log transformation of May riparian forest soil CH_4 emissions.

Effect	Df	Sum of Squares	Mean Square Error	F Value	Pr(>F)
Water	1	0.0007	0.00073	0.0014	0.97067
Temp	1	1.4006	1.40057	2.6732	0.12286
$KMnO_4$-Ox. C	1	1.8278	1.82779	3.4886	0.08146
Residuals	15	7.859	0.52393		

Table 3.26. ANOVA results for log transformation of May riparian forest soil CO_2 emissions.

Effect	Df	Sum of Squares	Mean Square Error	F Value	Pr(>F)
Water	1	0.04688	0.04688	1.1172	0.297193

Temp	1	0.43988	0.43988	10.4836	0.002501
$KMnO_4$-Ox. C	1	0.00867	0.00867	0.2067	0.651983
Residuals	38	1.59444	0.04196		

Table 3.37. ANOVA results for log transformation of May riparian forest soil N_2O emissions.

Effect	Df	Sum of Squares	Mean Square Error	F Value	Pr(>F)
Water	1	0.5566	0.55659	1.4617	0.2364
Temp	1	0.0047	0.00471	0.0124	0.9122
NH_4-N	1	0.1998	0.19978	0.5247	0.4747
NO_3-N	1	0.3556	0.35556	0.9337	0.3419
Residuals	29	11.0429	0.38079		

Table 3.28. ANOVA results for log transformation of June agriculture soil CO_2 emissions.

Effect	Df	Sum of Squares	Mean Square Error	F Value	Pr(>F)
Water	1	0.01803	0.018034	0.2665	0.6088
Temp	1	0.04528	0.045278	0.669	0.4186
$KMnO_4$-Ox. C	1	0.02264	0.022636	0.3345	0.5665
Residuals	37	2.5041	0.067678		

Table 3.29. ANOVA results for log transformation of June agroforestry soil CH_4 emissions.

Effect	Df	Sum of Squares	Mean Square Error	F Value	Pr(>F)
Water	1	2.1273	2.12728	0.9336	0.3516

Temp	1	0.0935	0.09345	0.041	0.8426
$KMnO_4$-Ox. C	1	0.2871	0.28714	0.126	0.7283
Residuals	13	29.6209	2.27853		

Table 3.30. ANOVA results for log transformation of June agroforestry soil CO_2 emissions.

Effect	Df	Sum of Squares	Mean Square Error	F Value	Pr(>F)
Temp	1	0.0118	0.011799	0.1287	0.7219
$KMnO_4$-Ox. C	1	0.0973	0.097259	1.0605	0.3098
Residuals	37	3.3931	0.091706		

Table 3.31. ANOVA results for log transformation of June riparian forest soil CH_4 emissions.

Effect	Df	Sum of Squares	Mean Square Error	F Value	Pr(>F)
Water	1	3.1682	3.1682	106.6171	0.001938
Temp	1	0.0581	0.0581	1.9553	0.256457
$KMnO_4$-Ox. C	1	0.2859	0.2859	9.6211	0.053221
Residuals	3	0.0891	0.0297		

Table 3.32. ANOVA results for log transformation of June riparian forest soil N_2O flux.

Effect	Df	Sum of Squares	Mean Square Error	F Value	Pr(>F)
Water	1	0.1149	0.11491	0.69	0.41271
Temp	1	0.0825	0.08245	0.4951	0.48708

NH_4-N	1	0.6653	0.66526	3.9948	0.05477
NO_3-N	1	0.0025	0.00246	0.0148	0.90403
Residuals	30	4.9959	0.16653		

Table 3.33. ANOVA results for log transformation of August riparian forest soil CO_2 emissions.

Effect	Df	Sum of Squares	Mean Square Error	F Value	Pr(>F)
Water	1	0.3251	0.32511	1.7411	0.1949
Temp	1	1.1586	1.15857	6.2045	0.01723
$KMnO_4$-Ox. C	1	0.013	0.01303	0.0698	0.79306
Residuals	38	7.0958	0.18673		

Table 3.34. ANOVA results for log transformation of August agroforestry soil CH_4 emissions.

Effect	Df	Sum of Squares	Mean Square Error	F Value	Pr(>F)
Water	1	0.0022	0.0022	0.0018	0.9728
Temp	1	3.7545	3.7545	3.1388	0.3271
$KMnO_4$-Ox. C	1	0.0024	0.0024	0.002	0.9717
Residuals	1	1.1962	1.1962		

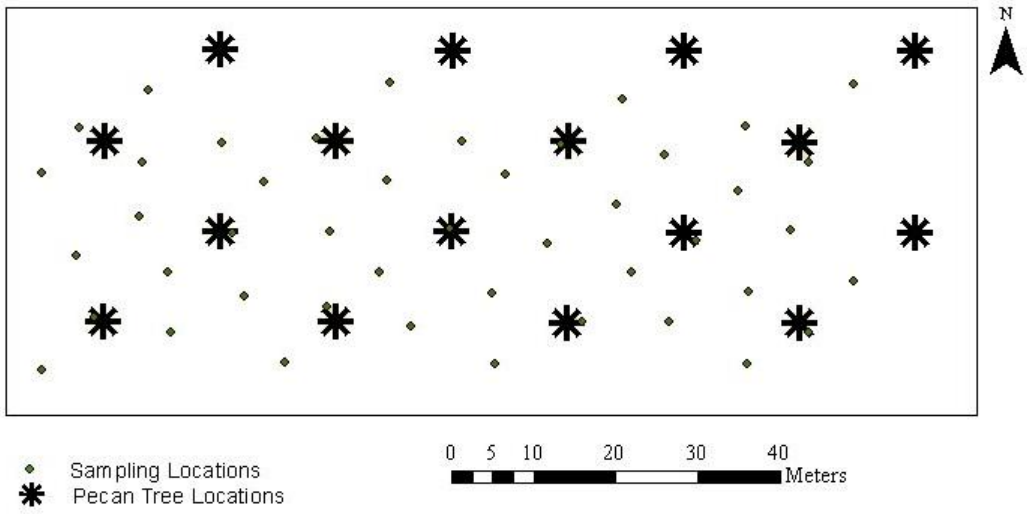


Figure 3.1. Example diagram of soil and gas sampling locations as located by space filling curve algorithm.

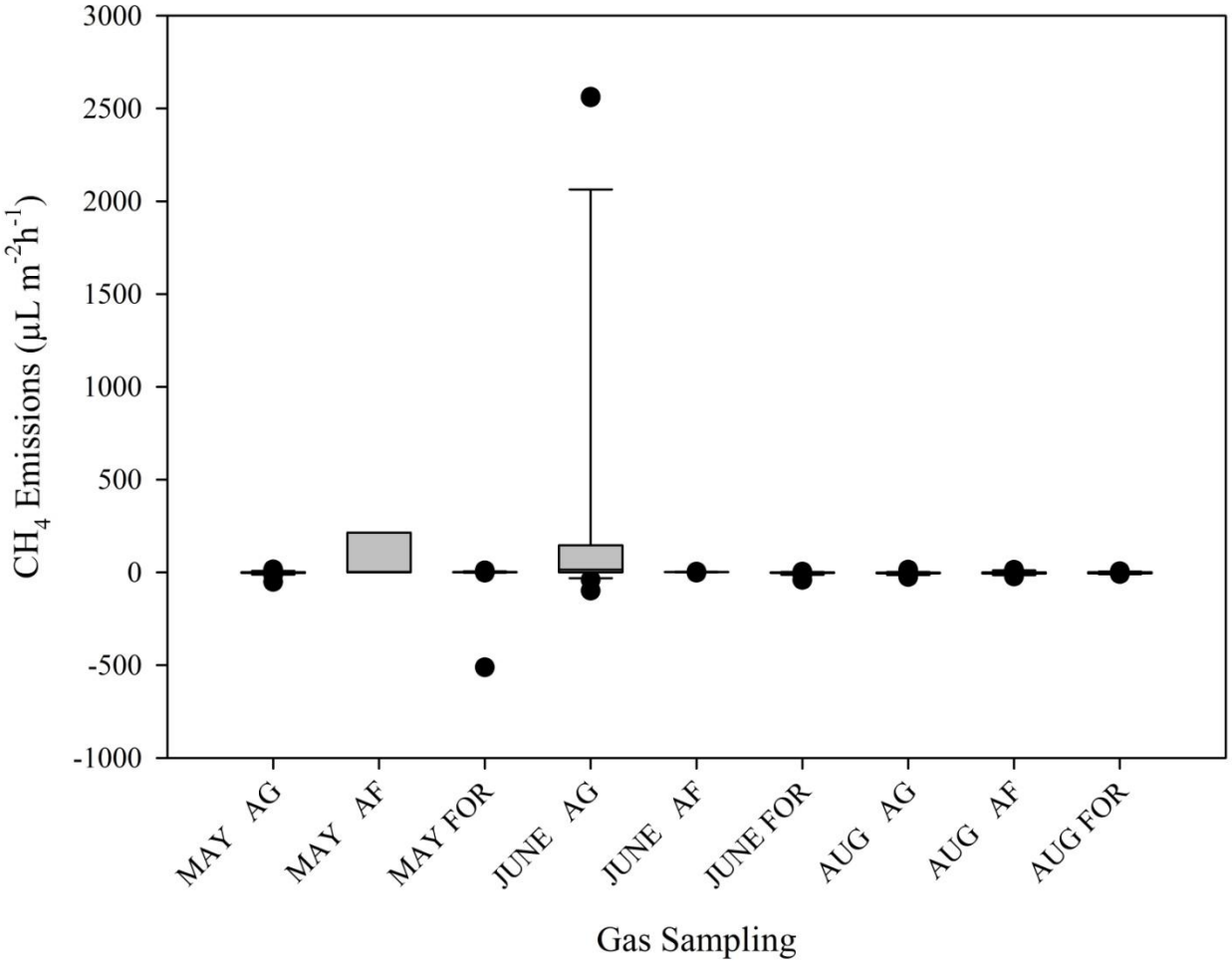


Figure 3.2. Boxplot diagram of soil CH_4 flux from all gas sampling times and locations.

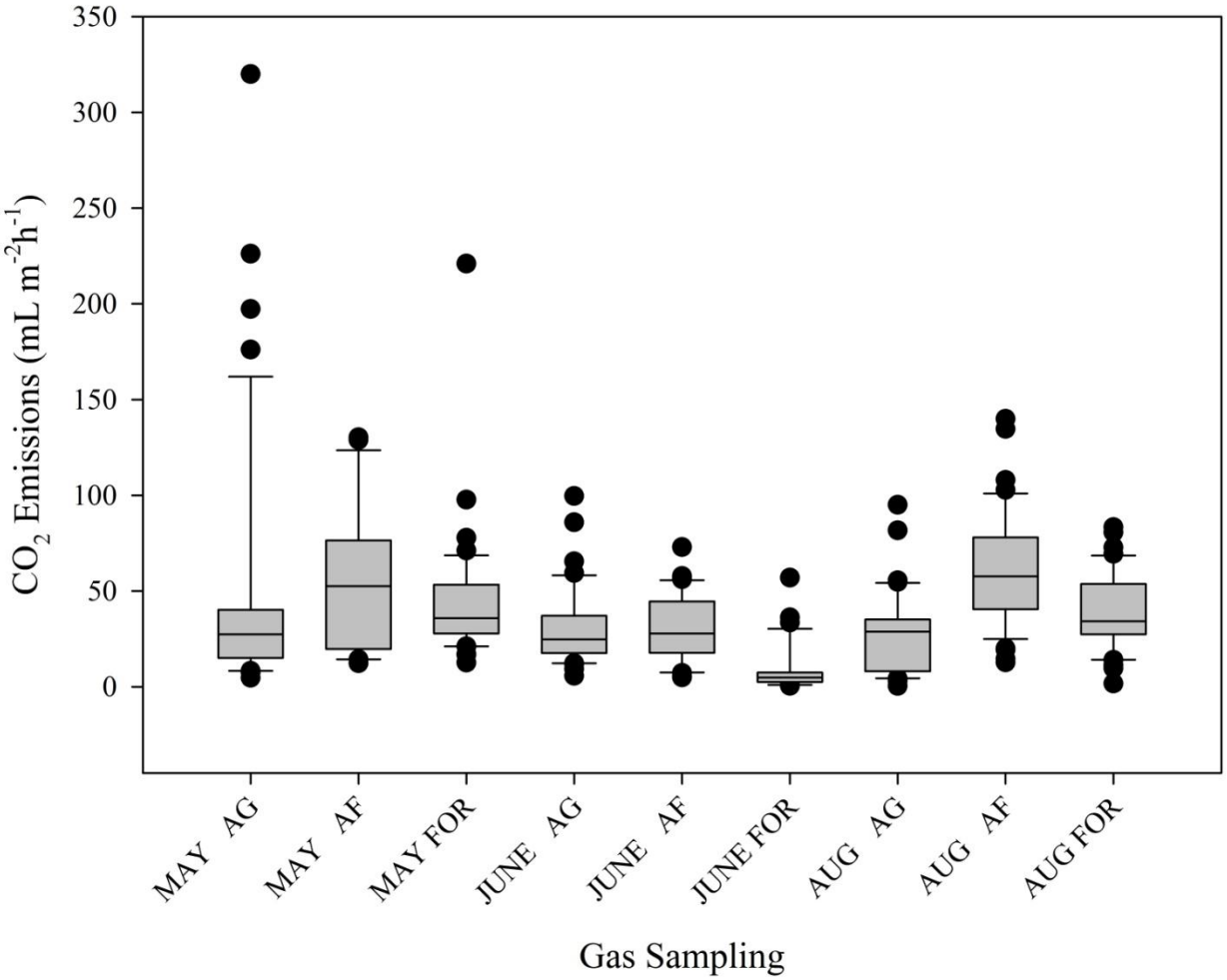


Figure 3.3. Boxplot diagram of CO_2 emissions from all gas sampling times and locations.

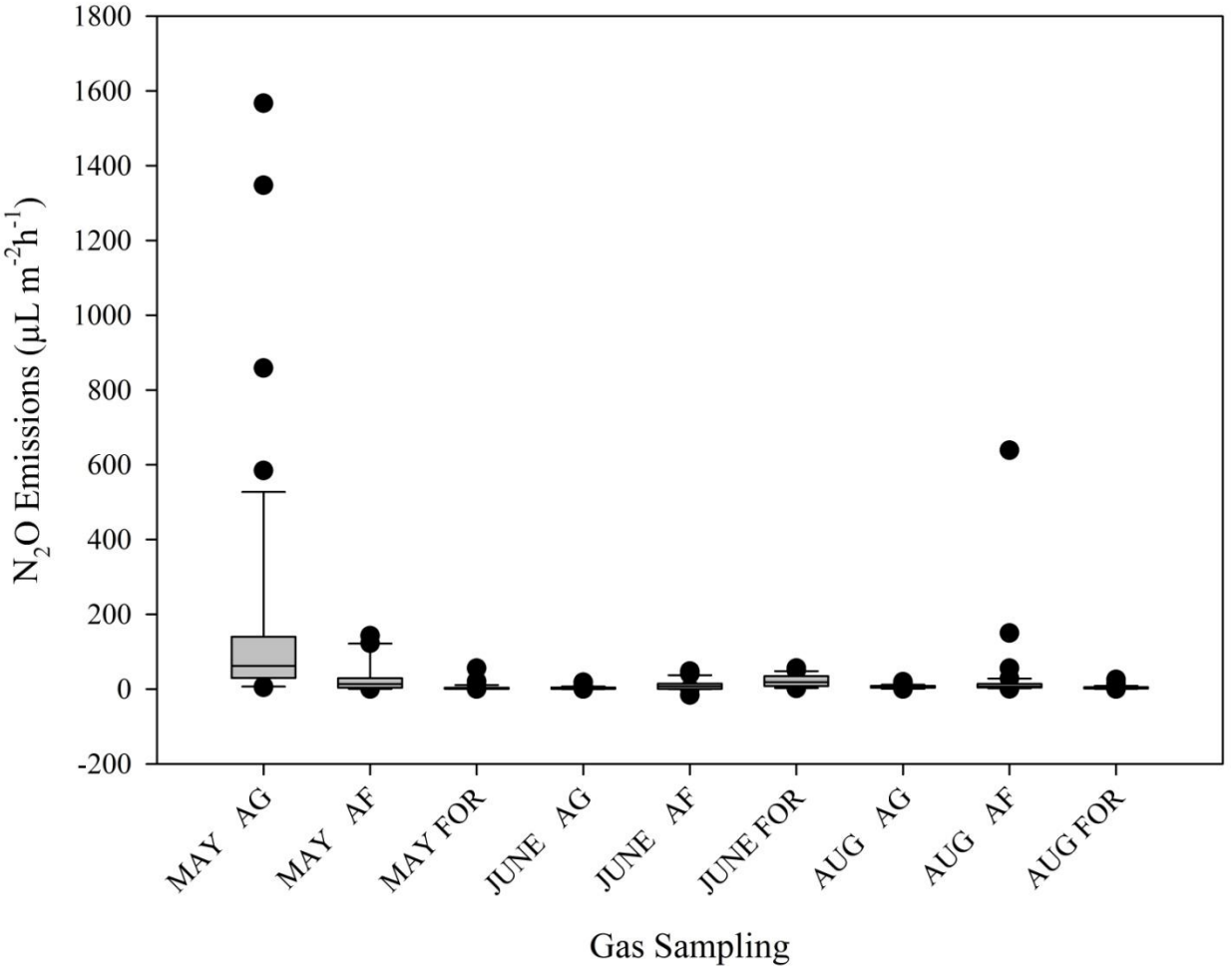
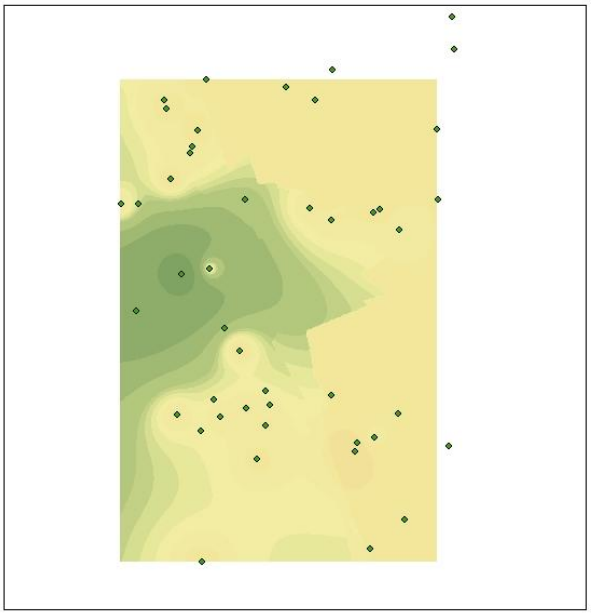
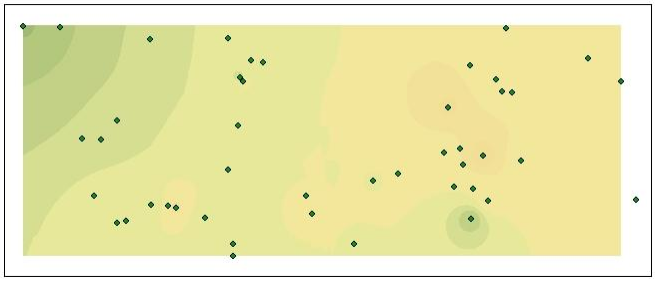


Figure 3.4. Boxplot diagram of N_2O emissions from all gas sampling times and locations.

A.



B.



C.

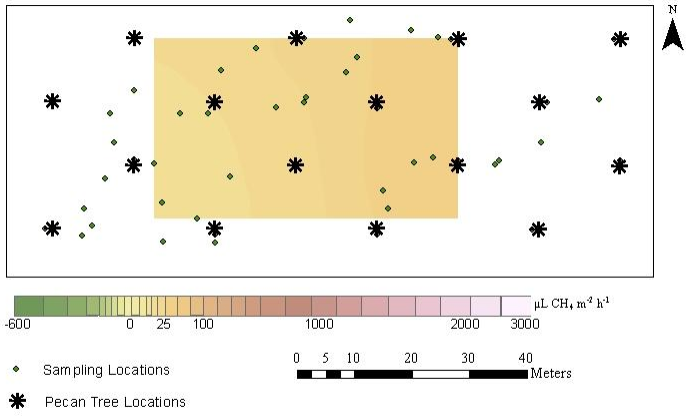
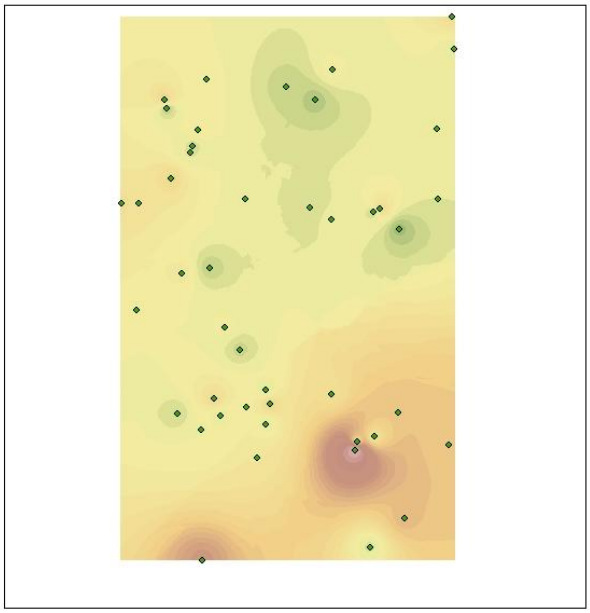


Figure 3.5. Interpolated May soil CH_4 fluxes by Missouri River Floodplain (MRF) land use. A. Riparian Forest, B. Agriculture, and C. Agroforestry.

A.



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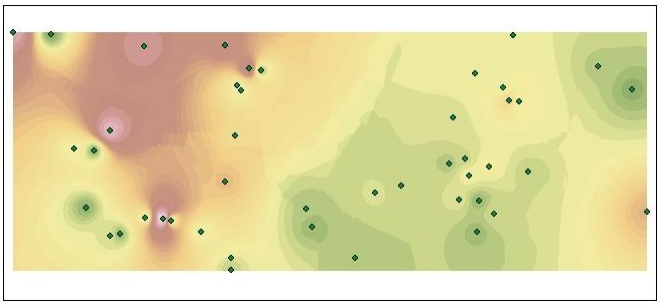


C.

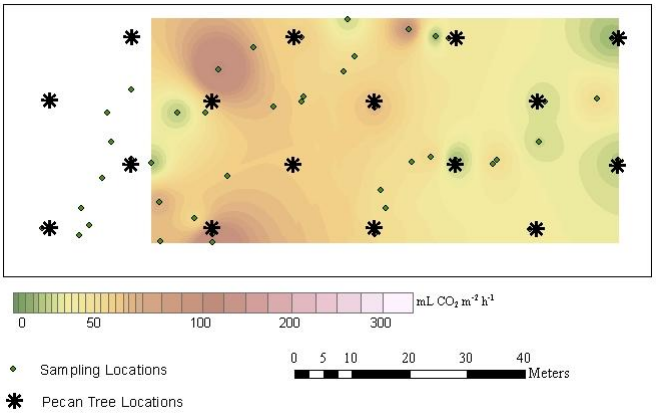
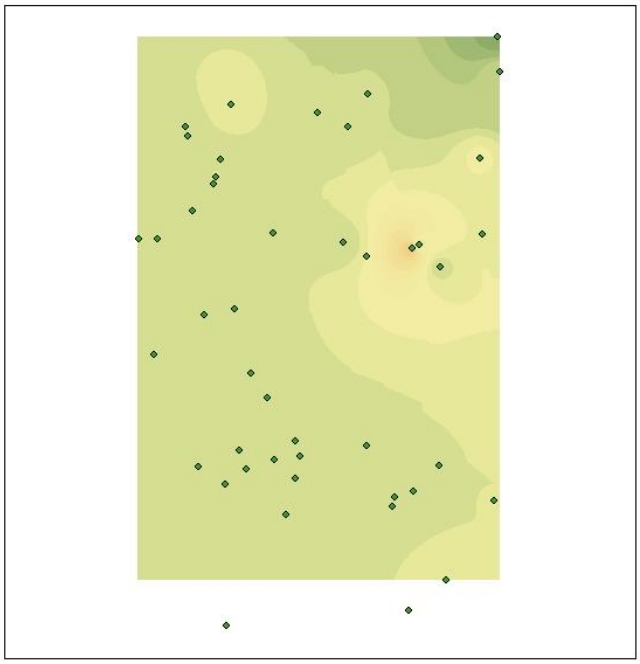


Figure 3.6. Interpolated May soil CO_2 emissions by Missouri River Floodplain (MRF) land use. A. Riparian Forest, B. Agriculture, and C. Agroforestry.

A.



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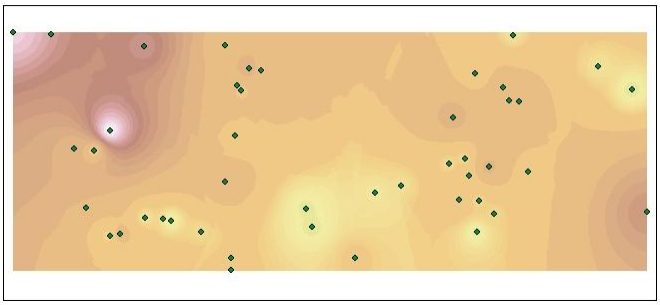

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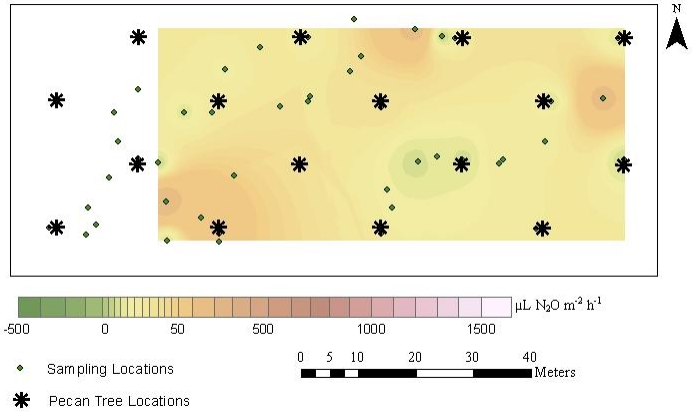
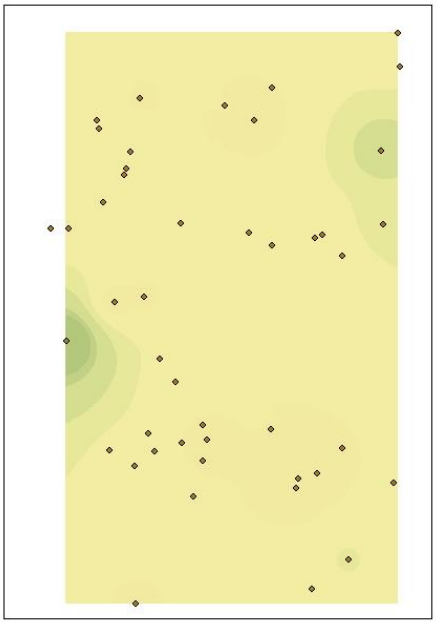


Figure 3.7. Interpolated May N_2O emissions by Missouri River Floodplain (MRF) land use. A. Riparian Forest, B. Agriculture, and C. Agroforestry.

A.



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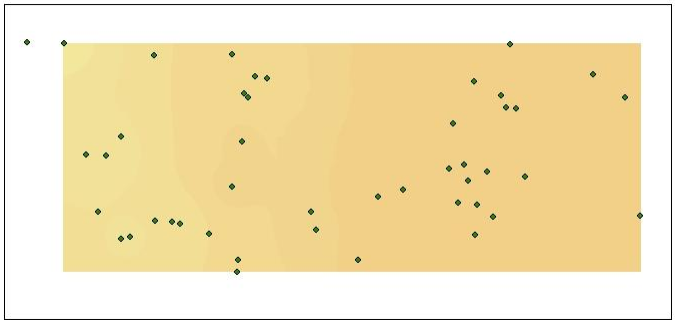

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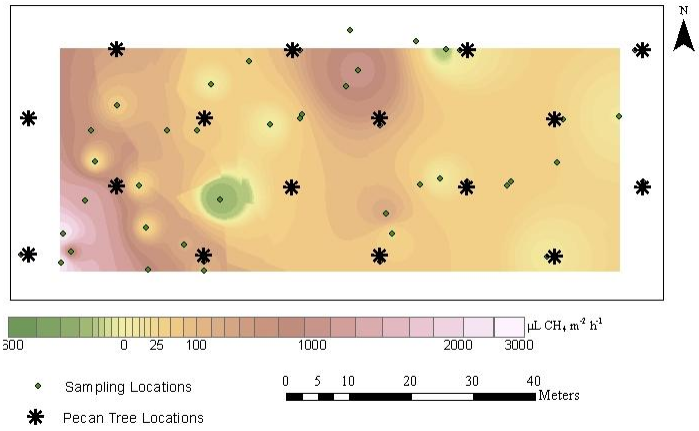
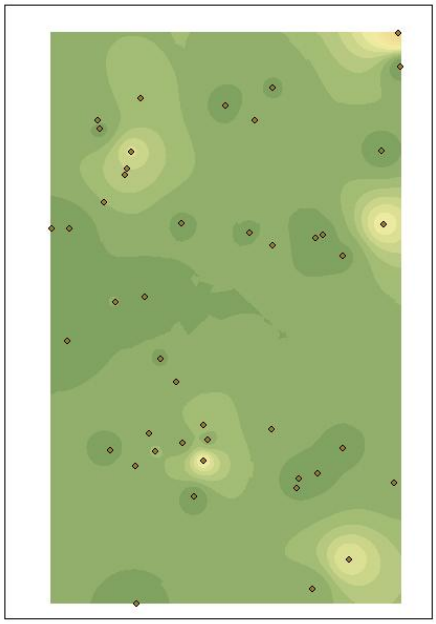


Figure 3.8. Interpolated June CH_4 emissions by Missouri River Floodplain (MRF) land use. A. Riparian Forest, B. Agriculture, and C. Agroforestry.

A.



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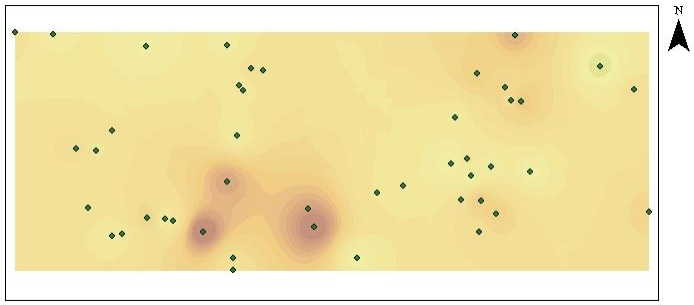

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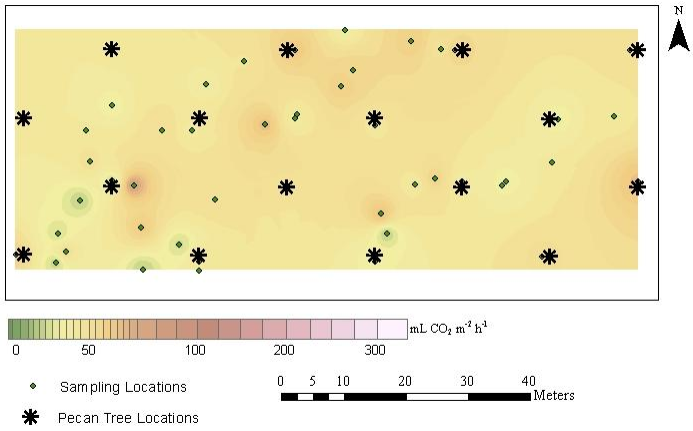
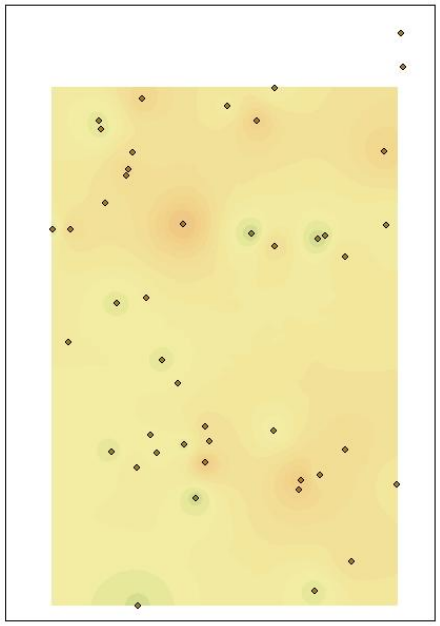


Figure 3.9. Interpolated June CO_2 emissions by Missouri River Floodplain (MRF) land use. A. Riparian Forest, B. Agriculture, and C. Agroforestry.

A.



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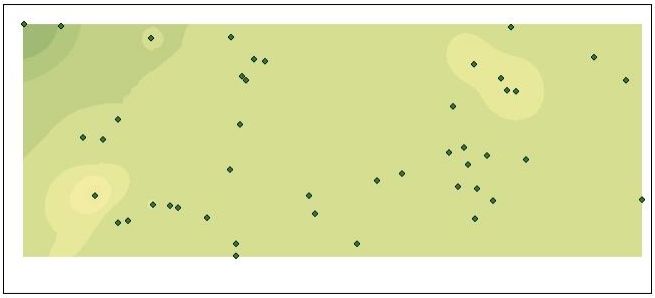

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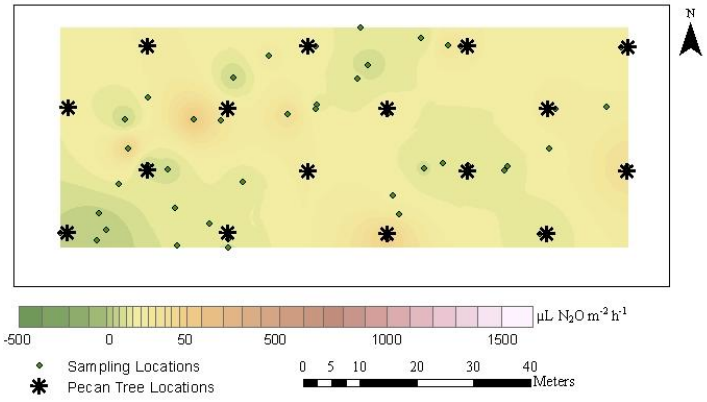
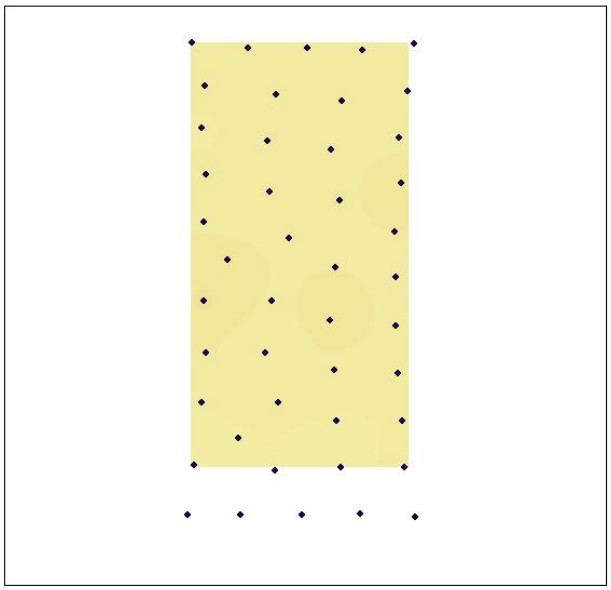


Figure 3.10. Interpolated June N_2O emissions by Missouri River Floodplain (MRF) land use. A. Riparian Forest. B. Agriculture C. Agroforestry.

A.



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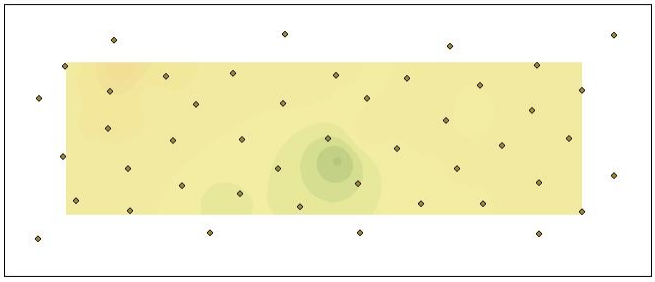

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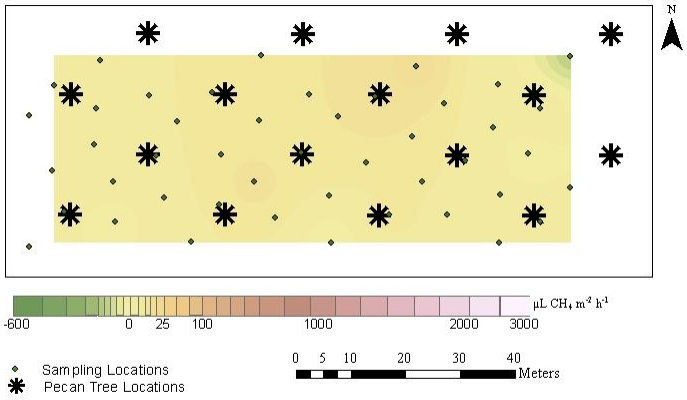
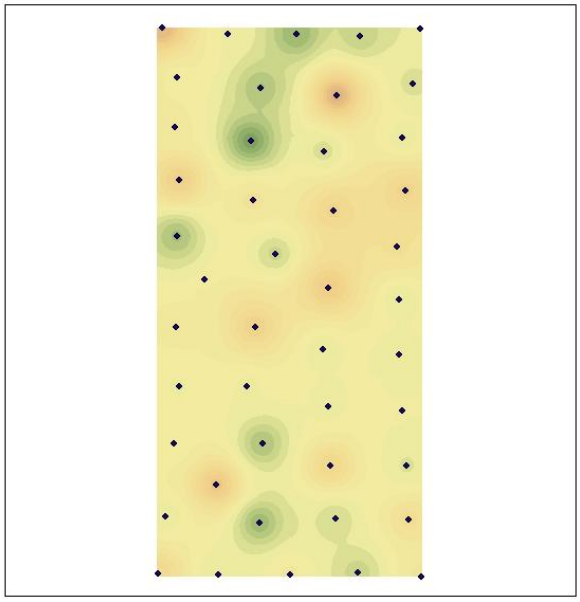


Figure 3.11. Interpolated August CH_4 emissions by Missouri River Floodplain (MRF) land use. A. Riparian Forest, B. Agriculture, and C. Agroforestry.

A.



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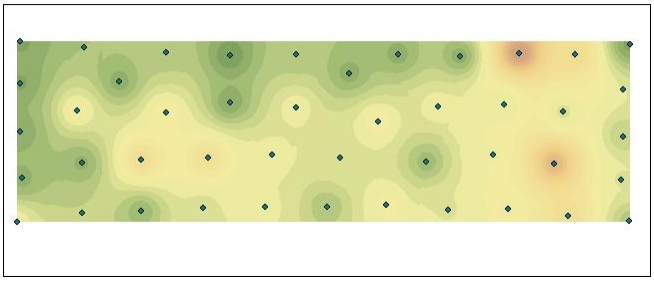


C.

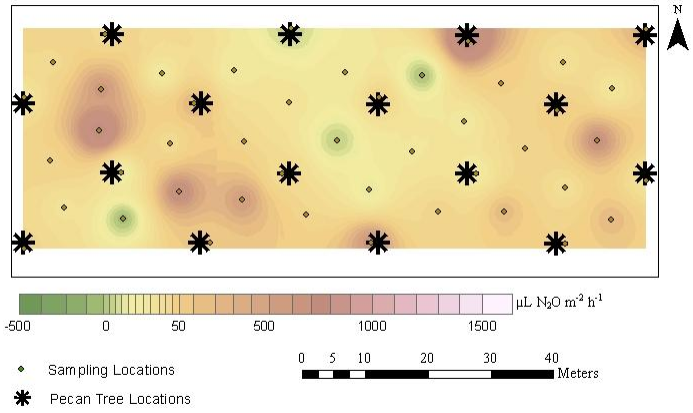
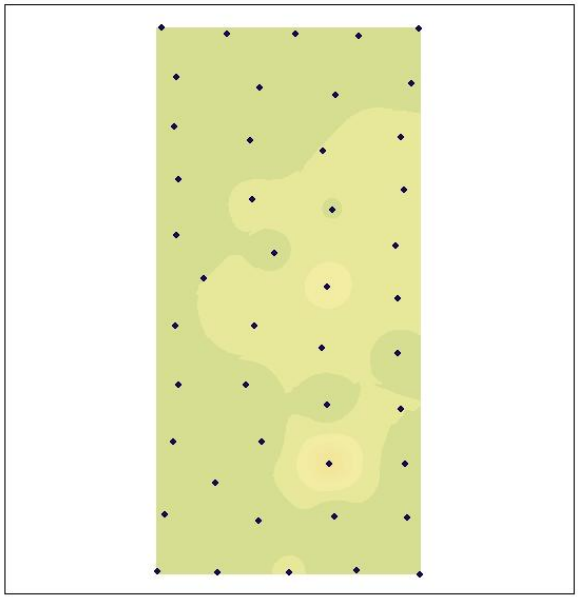
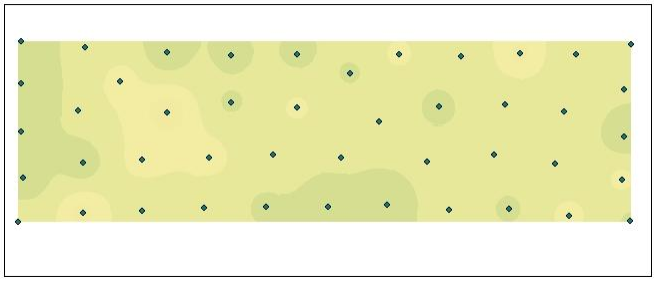


Figure 3.12. Interpolated August CO_2 emissions by Missouri River Floodplain (MRF) land use. A. Riparian Forest, B. Agriculture, and C. Agroforestry.

A.



B.



C.

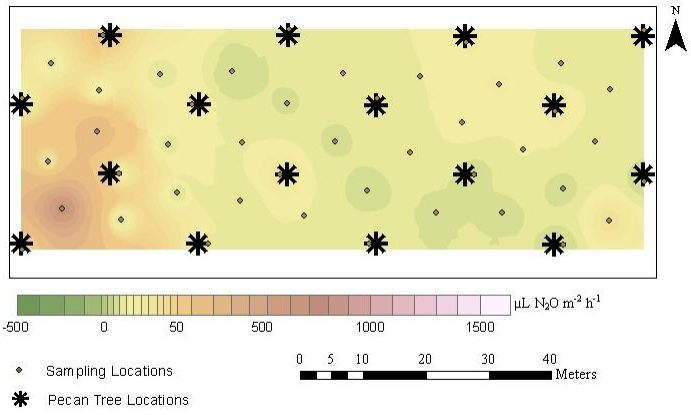


Figure 3.13. Interpolated August N_2O emissions by Missouri River Floodplain (MRF) land use. A. Riparian Forest, B. Agriculture, and C. Agroforestry.

CHAPTER 4: CONCLUSIONS

A major goal of this research was to initiate examination of the effects of different current land use systems in the lower Missouri River Floodplain (MRF) region and a possible alternative agroforestry system on spatial and temporal variations in soil GHG emissions under both controlled laboratory and more highly variable field conditions. Changes in soil properties due to the different land uses and the varied soil moisture regimes observed in the floodplain may influence GHG emissions and also possibly contribute to possible spatial variation in soil GHG emissions in this environment.

In the controlled laboratory study, several important soil relationships between land use, soil water regime and time were examined. Soil water regime was identified as the essential limiting factor for soil CH_4 emissions. Logically, the emissions required anoxic environmental conditions commonly created through saturated conditions. It was unexpected that cumulative soil CH_4 emissions of the flooded samples would take over seven weeks to be statistically larger than the other soil water regime treatments. Due to river management and hydrologic changes, long-term flooding on the lower MRF is limited which may reduce the magnitude of cumulative soil CH_4 emissions from this region. Channelization and water control structures protects the MRF from natural flood patterns and promotes agricultural practices that further homogenize large quantities of land. Prospects for soil CH_4 efflux remain in select areas where microtopographical features still are created through scouring and depositional processes. These features were noted in the riparian forest study site and were observed holding water in small pools throughout the spring season. The data also suggests that soil CH_4 emissions may

have been significant in long-term flooded conditions along the Missouri River in northwest Missouri.

The interpretation of the soil CO_2 and N_2O emission results is more complicated, as interactive effects were noted between land use, water regime, and time. The results suggest than any number of soil or environmental conditions can serve as GHG limiting factors. For example, the different water regime treatments effects on cumulative soil CO_2 emissions may have been multifold. The 60% water filled pore space samples contain a sufficient amount of water and oxygen for microbial populations reducing the limitations set forth by extreme water conditions. The air filled pores of the samples also provide pathways for the produced gas to be emitted into the atmosphere. Filled or reduced pore space can often be a limiting factor in GHG efflux. As expected, the cumulative soil N_2O emissions from fertilized land uses were higher than the non-fertilized counterparts confirming the higher risk for N_2O emissions associated with use of N fertilizers in lower MRF land use systems where flooding is a high probability during the spring months close to when fertilizer is applied. As the incubation progressed the amount of available soil NO_3^- is assumed to have been lost to denitrification involving other N gases (e.g., N_2), corresponding with the eventual decrease in N_2O emission rates.

The focus of the field study was to quantify the spatial and temporal variability in soil GHG flux among the different land use systems and analyze any relationships between observed variation in soil properties and observed soil GHG emissions. Emission rates form from a complex series of soil, plant and environmental factors, any of which can be limiting in nature. A large percentage of the GHG emission distributions from individual samplings were non-normal and required a log transformation for linear

regression analysis. Resulting models showed limited spatial correlation at the sample distances tested, but may be stronger at smaller intervals. Soil temperature, soil water content, soil NO_3-N, and soil $KMnO_4$-C were significant at $p \leq 0.05$ to log GHG efflux in individual circumstances, but lacked consistent significance among all data sets.

Prospective research on the subject of MRF soil gas emissions should utilize this information in its approach and statistical design. The effects of soil properties seen in the laboratory incubation may be supported by additional sampling in field conditions. Measurement of additional soil physical, chemical, and biological parameters may provide further explanation of observed soil GHG emissions. The spatial variation of GHG emissions should be investigated at much smaller intervals. The current scale proved to be too large to elucidate this variation bringing questions other recent studies that have relied on much sparser data set. By utilizing the information learned in this study the opportunity exists to improve our understanding of soil GHG emission processes not only within the MRF, but in science as a whole.

APPENDIX

APPENDIX A

Pecan tree measurements taken September 2010

Tree Number	Live Crown Height	Total Height	Breast Height Diameter
	----------m----------		--cm--
1	1.94	8.64	17.8
2	2.12	7.88	14
3	1.94	8.46	16.2
4	2.47	8.64	21.2
5	2.47	8.9	24.1
6	2.3	8.9	19
7	2.65	9.74	25.5
8	2.3	9.45	24.1
9	2.12	9.98	24.2
10	2.29	10.16	23.4
11	2.47	8.99	18.2
12	2.12	8.28	18.3
13	2.29	10.16	24.2
14	2.65	11.13	24
15	1.59	10.16	26
16	2.47	9.98	24
17	2.29	11.13	22.4
18	2.83	10.79	22
19	2.29	10.29	23.1
20	2.83	12.02	26.3
21	2.29	9.68	18
22	1.94	8.9	19.5
23	2.47	11.83	26
24	3.01	13.4	30
25	2.47	12.58	26.7
26	2.65	12.2	26.1
27	3.01	10.47	18.6
28	2.47	8.13	15.7
29	2.47	10.47	22.9
30	2.12	10.79	20.7
31	2.3	10.34	17.7
32	2.29	11.47	21

Tree Number	Live Crown Height	Total Height	Breast Height Diameter
	----------m----------		--cm--
33	2.65	9.45	21.6
34	2.65	9.45	25.3
35	2.65	10.97	24.7
36	2.65	8.9	18.7
37	2.47	9.85	22.9
38	2.47	9.27	24.7
39	2.47	9.45	24.6
40	1.94	9.27	23.8
41	2.65	11.47	27
42	2.65	11.65	25
43	1.76	9.98	22
44	1.94	9.09	22.7
45	1.94	9.85	18.5
46	2.83	10.65	20.7
47	1.94	8.2	18.5
48	2.12	9.56	21
49	2.83	12.58	27.1
50	2.29	12.41	23.5
51	2.65	12.02	24.4
52	2.65	13.22	27.1
53	2.47	12.02	23
54	1.41	4.31	17.1
55	2.29	12.2	28.3
56	2.47	10.79	24.8
57	2.47	10.47	22.7
58	2.47	13.16	28.4
59	2.29	10.47	20.9
60	2.47	8.99	22
61	2.12	12.95	22.9
62	2.47	11.83	23.9
63	2.47	12.58	19.7
64	2.29	11.47	26.6

APPENDIX B

Riparian forest species sampling and measurements taken October 2010

Plot	Tree Number	Species	Diameter	Tree Area	Notes
			---cm---	---m^2---	
1	1	Silver Maple	44.8	0.16	
1	2	Silver Maple	34.9	0.1	Dead
1	3	American Elm	39.1	0.12	
1	4	Silver Maple	25.3	0.05	
2	1	Silver Maple	51.7	0.21	
2	2	Silver Maple	28.7	0.06	
2	3	American Elm	12.3	0.01	
2	4	Sycamore	60.3	0.29	
2	5	Cottonwood	98	0.75	
3	1	Silver Maple	24.7	0.05	
3	2	Silver Maple	38.2	0.11	
3	3	American Elm	27.7	0.06	
3	4	American Elm	28.7	0.06	
3	5	American Elm	18.3	0.03	
4	1	Silver Maple	46.9	0.17	Dead
4	2	American Elm	43.3	0.15	
4	3	American Elm	24.1	0.05	
4	4	American Elm	15.2	0.02	
4	5	Sycamore	53.9	0.23	
4	6	Sycamore	38.4	0.12	
4	7	American Elm	32.6	0.08	
5	1	Silver Maple	35.6	0.1	Dead
5	2	Silver Maple	39.9	0.12	
5	3	Silver Maple	32.3	0.08	
5	4	Silver Maple	35.6	0.1	
5	5	American Elm	14.6	0.02	
5	6	American Elm	16	0.02	
5	7	American Elm	13	0.01	
5	8	Silver Maple	35.8	0.1	
5	9	Silver Maple	7.9	0	

Plot	Tree Number	Species	Diameter	Tree Area	Notes
			---cm---	---m^2---	
5	10	American Elm	10.2	0.01	
5	11	American Elm	13.2	0.01	
5	12	American Elm	17.3	0.02	
5	13	Silver Maple	40	0.13	
5	14	Silver Maple	24.4	0.05	
5	15	Green Ash	8.4	0.01	
5	16	Green Ash	8	0.01	
5	17	Silver Maple	26.4	0.05	
5	18	Silver Maple	25.9	0.05	
5	19	Silver Maple	46	0.17	
7	1	American Elm	18.3	0.03	
7	2	American Elm	18	0.03	
7	3	Silver Maple	36.1	0.1	
7	4	American Elm	15.4	0.02	Dead
8	1	American Elm	8.6	0.01	
8	2	American Elm	15.5	0.02	
8	3	Green Ash	9	0.01	
8	4	American Elm	20.6	0.03	
8	5	American Elm	19.7	0.03	
8	6	American Elm	8.1	0.01	
8	7	American Elm	21.8	0.04	
8	8	American Elm	7.6	0	
8	9	American Elm	9.5	0.01	
9	1	Silver Maple	36.6	0.11	
9	2	Cottonwood	71.6	0.4	
9	3	Sycamore	26.4	0.05	
9	4	Silver Maple	13.2	0.01	
9	5	Silver Maple	21.8	0.04	
9	6	American Elm	18	0.03	
9	7	Silver Maple	26.2	0.05	
9	8	Silver Maple	17.8	0.02	
9	9	American Elm	8.6	0.01	
9	10	American Elm	11.4	0.01	

Plot	Tree Number	Species	Diameter	Tree Area	Notes
			---cm---	---m^2---	
9	11	Silver Maple	24.1	0.05	
9	12	Silver Maple	34.8	0.1	
9	13	Silver Maple	26.2	0.05	
9	14	Silver Maple	34.7	0.09	
9	15	Silver Maple	22.4	0.04	
9	16	Silver Maple	10.9	0.01	
9	17	Silver Maple	11.2	0.01	
10	1	Silver Maple	20.6	0.03	
10	2	Silver Maple	27.4	0.06	
10	3	Silver Maple	25.9	0.05	
10	4	Silver Maple	34.8	0.1	
10	5	Silver Maple	30.7	0.07	
10	6	Silver Maple	21.8	0.04	
10	7	Silver Maple	41.1	0.13	
10	8	Silver Maple	35.4	0.1	
10	9	Silver Maple	16.8	0.02	
10	10	Sycamore	54.1	0.23	

*No Trees were found in Plot 6