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GRASSLAND SUSTAINABILITY IN KENTUCKY: CASE STUDIES QUANTIFYING THE EFFECTS OF CLIMATE CHANGE ON SLUG HERBIVORY IN PASTURES AND DIFFERENT HOME LAWN SYSTEMS ON TURF GREENHOUSE GAS EMISSIONS

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GRASSLAND SUSTAINABILITY IN KENTUCKY: CASE STUDIES
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GREENHOUSE GAS EMISSIONS

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THESIS

A thesis submitted in partial fulfillment of the requirements for the degree of Master of Science in the College of Agriculture, Food, and the Environment at the University of Kentucky

By

Daniel Adam Weber

Lexington, Kentucky

Director: Dr. Rebecca McCulley, Associate Professor of Plant and Soil Sciences

Lexington, Kentucky

2014

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Grasslands comprise the greatest biome by land area, are sensitive to environmental factors affected by climate change, and can impact future climate change through their ability to store and release greenhouse gasses (GHGs). I performed two studies: 1) evaluated the effects of increased temperature and precipitation on slug herbivory/abundance and pasture forage production; 2) quantified different homeowner lawn system effects on soil-to-atmosphere GHG emissions. Climate change will likely affect pasture forage production, with implications for slug herbivory and abundance. I found little evidence that slugs have or will have significant effects on pasture production or plant community. Warming altered the abundance of slugs and modified seasonal trends, increasing slug abundance in spring/winter and reducing it in late-summer/fall, through both direct effects and changes in plant community and forage quality. Home lawns vary in levels of management, influencing the exchange of GHGs. I quantified the effects of three common home lawn systems of central Kentucky on GHG emissions, but found no significant differences in CO₂, N₂O, and NH₃ fluxes. My research suggests that slug herbivory is not a dominant ecological process in Kentucky pastures and that common home lawn systems have similar soil-to-atmosphere GHG emissions.

KEYWORDS: Climate change, Greenhouse gasses, Pasture, Slug herbivory, Turfgrass
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To my friends and family:

Thank you for your continued help and support…
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Chapter 1: The Role of Grasslands in Ecological Sustainability

1.1 The Grassland Ecosystem

Grasslands, land with <15% tree cover and dominated by herbaceous species, account for 40.5% of land area (52,530,000 km²) in the world (White et al., 2000). The United States has 3,384,086 km² of grassland, which ranks fourth among countries worldwide (White et al., 2000). Common ecosystem services provided by grassland systems include the provision of forage for animals, carbon storage, maintenance of hydrologic function, conservation of biodiversity, and recreational opportunities (White et al., 2000). Reductions in grassland area can lead to losses of these important ecosystem services (Foley et al., 2005; Rashford et al., 2011). Over the past 200 years, naturally occurring grasslands worldwide have been largely converted for human uses, such as row crop agriculture, managed pastures, and/or urban development (Rashford et al., 2011; White et al., 2000). The majority of remaining grasslands are considered unsuitable for alternate land uses due to either insufficient precipitation or poor soil quality (O'Mara, 2012). The vast majority of all non-tundra grasslands are either actively or passively managed by humans for livestock production, recreation, or aesthetics, e.g. turf (Taube et al., 2013; White et al., 2000). As the single largest land-area type (White et al., 2000), the future of grasslands and their ability to persist and maintain ecosystem functions under climate change conditions, will have direct implications for humans and Earth system processes.
1.2 Human Modification of Biogeochemical Processes

Humans have modified earth system processes in a variety of ways over the past 250 years. For example, the agricultural revolution of the 18th century allowed people to vastly expand the total area under cultivation, thereby facilitating the conversion of many native forests and grasslands into row crop agriculture with resulting effects on global biogeochemical and hydrologic cycles (Allen, 1999; Smil, 1999, 2001). These changes were followed by the discovery in the 20th century by Fritz Haber of the reaction process to convert diatomic nitrogen and hydrogen gas into ammonia, and Carl Bosch developed the technology to implement the process at an industrial scale, leading to what is known as the Haber-Bosch method - the most common way fertilizer is produced world-wide today (Smil, 2001). Advancements in cultivation technology and selective breeding, along with the Haber-Bosch produced vast quantities of biologically available forms of nitrogen, led to the era of immense crop productivity increases, known today as the “Green Revolution” (Evenson and Gollin, 2003). The Haber-Bosch method of converting atmospheric nitrogen to biologically available ammonia requires creating conditions of substantial heat and pressure which are achieved through the consumption of considerable energy from fossil fuel combustion (Hungate et al., 2003). The use of Haber-Bosch fixed nitrogen, as well as the burning of fossil fuels, has greatly disrupted the natural nitrogen cycle (Smil, 1997). Anthropogenic sources of nitrogen fixation have increased total fixation by 45% above natural and biological fixation to a total of 445.2 Tg N y⁻¹ (Voroney and Derry, 2008). Furthermore, the increased use of mechanical cultivation and synthetic fertilization have allowed increasing land use change for both
cropland and novel uses, further altering both nitrogen and carbon cycles on a global scale (Houghton, 1995; Vitousek, 1994).

By altering land cover and land use, humans have altered many global biogeochemical cycles. For example, a meta-analysis of soil carbon stock changes found that conversion of many types of natural forests and grasslands to tree plantations or croplands can cause anywhere from 10-60% of soil carbon to be lost to the atmosphere (Guo and Gifford, 2002). Guo and Gifford (2002) also found that reverting to previous land uses (e.g. cropland back to grassland; tree plantation back to natural forest) can increase carbon stocks by 8-53%. In the 1980s, 90% of the net increase in atmospheric carbon was as a direct result of land use change, primarily deforestation in the tropics (Watson, 2000). Human decisions to employ reduced or no-tillage systems in croplands can improve carbon sequestration and storage, along with implementing nitrogen inputs that tend to increase organic matter production, which was illustrated in a meta-analysis showing that conversion from conventional tillage to no-till can result in up to a 23% increase in carbon storage (Ogle et al., 2005; Watson, 2000). In systems that leave crop residues on the soil, such as no-till cropping, the use of fertilization has been shown to bolster carbon storage (Alvarez, 2005). Human land use and management decisions have the ability to significantly affect both short and long-term biogeochemical systems.

The application of synthetic fertilizers on agricultural lands, while augmenting the ability to store carbon, is not without negative environmental impacts. Nitrogen based fertilizers increase the rate of nitrogen flow through microbial processes, stimulating nitrogen availability and uptake by plants, as well as increasing nitrogen losses to the atmosphere and leaching to groundwater (Galloway et al., 2003). Nitrogen losses to the
atmosphere associated with fertilization are predominantly in the form of NH$_3$, N$_2$O, and NO$_x$ gaseous fluxes (Galloway et al., 2003), and are also relevant to climate change science (IPCC, 2007b; Martikainen, 1985). Along with CO$_2$, they are major contributors to greenhouse gas accumulation (Galloway et al., 2003; IPCC, 2001; Martikainen, 1985). Human decisions can alter the flux of these gases to and from the atmosphere with direct effects on global phenomenon such as climate change.

Increasing levels of these gases in the atmosphere have given rise to what is known as the “greenhouse effect,” which refers to the retention of heat that occurs when solar radiation reflected by earth excites greenhouse gases that have accumulated in the atmosphere (IPCC, 2001). Greenhouse gases (such as water vapor, CO$_2$, N$_2$O, CH$_4$, and O$_3$) are defined by their ability to convert certain spectrums of solar energy to heat energy in the upper troposphere and have both natural and anthropogenic sources. Carbon dioxide has been the greatest greenhouse gas of concern, as its concentration has increased to just over 380 ppm, or 32% above pre-industrial levels (Keeling and Whorf, 2005). The primary source of CO$_2$ has been the burning of fossil fuels, such as coal, natural gas, and oil, for production of energy, and anthropogenic sources have been augmented by land use changes that result in stimulated fluxes of CO$_2$ and other greenhouse gases from the soil to the atmosphere (IPCC, 2001). Other important greenhouse gases include methane (CH$_4$) and nitrous oxide (N$_2$O), which are not emitted or present in the atmosphere in such large of quantities as CO$_2$, but have disproportionately larger impacts on atmospheric warming that can be quantified by the unit “Global Warming Potentials” (IPCC, 2007b). Global warming potentials are a measure of the ability of a gas, relative to CO$_2$, to persist in the atmosphere and emit heat.
energy when struck with long-wave solar radiation (IPCC, 2007b). Current projections of climate change impacts call for increases in average global temperatures of +1.4 to +5.8°C by the year 2100 depending on what levels of greenhouse gas mitigation are obtained (IPCC, 2007b). Depending on location, climate change may increase or decrease annual precipitation amounts (IPCC, 2007b). For example, most of the eastern US is currently projected to experience 10-30% increases in mean annual precipitation (IPCC, 2007b; Jones et al., 2001).
1.3 Climate Change and Grasslands

Grasslands can mitigate climate change via their ability to sequester high quantities of carbon in belowground pools (Lal, 2004). Increased temperature and moisture could stimulate photosynthesis and sequestration by some plants (Hatfield et al., 2011), but these conditions may also increase microbial turnover and loss of carbon and nitrogen, which could further contribute to additional climate change, depending on the forms lost and their fate in the environment (Bardgett et al., 2008). Whether a particular grassland functions as a source or sink for greenhouse gasses depends on the location of the grassland, the type of grassland, the management, and other limiting factors (Scurlock and Hall, 1998a). The effects of climate change on different types of grasslands will vary, and the response of grasslands, whether they mitigate or contribute to additional change, will be determined by many factors (Grime et al., 2000; Parton et al., 1995). Whether or not grasslands will be a source or sink for greenhouse gasses will be dependent on the interaction of many processes that occur within them, including changes in species composition (Suttle et al., 2007), selective herbivory (Izaurralde et al., 2011), and nutrient cycling (Fay et al., 2008).

Plant Community

Climate change is expected to influence the plant community composition of grasslands through increased atmospheric CO₂ (Hatfield et al., 2011) and changes in average mean temperature and precipitation (Fay et al., 2008). Atmospheric CO₂ concentrations are predicted to increase to ~450ppm by the year 2050 from current levels of ~385ppm (IPCC, 2007b). Physiologically, grassland species respond positively to increased CO₂ by increasing the efficiency of photosynthesis and reducing the impact of...
water stress related stomatal closure (Ainsworth and Long, 2005). However, elevated CO₂ does not benefit all plants the same, with C₃ plants improving productivity significantly more than C₄ plants (Ziska and Bunce, 1997). In addition, C₃ annuals, including many weedy invasive species, become much more competitive under elevated CO₂ conditions (Ziska, 2004). In fact, in field cropping systems, C₃ weeds have been shown to have significantly greater herbicide resistance under elevated CO₂ exposure (Ziska et al., 1999), which does not bode well for managed grasslands such as forage pastures and aesthetic turf systems, where weed control can already be challenging (Heap, 2014). Elevated CO₂ does not benefit all grassland species, as some work has shown that several species of clover show no increase in photosynthesis rates (Newton et al., 2006). Increasing CO₂ concentration will alter the competitive abilities of species, which may also be affected by increasing temperatures and altered precipitation.

Elevated temperatures can increase the length of the growing season by several weeks, by warming up faster in the spring and/or delaying the onset of winter, which can provide competitive advantages for certain species resulting from changes in growing conditions (IPCC, 2001; Izaurralde et al., 2011). Some studies have found that elevated temperatures caused increases in overall productivity of rangelands (native grasslands) by extending the growing season (Wan et al., 2005); however, increased temperature can also decrease productivity of grasses and annuals during the peak of summer, when extreme temperatures can cause reduced growth or even injury (Schlenker and Roberts, 2009). The effects of temperature changes on species competition and ecosystem function will depend on the interactions of future changes in precipitation, as grasslands
are sensitive to water availability (Sala et al., 1988) and changes in temperature often result in changes in soil moisture (Wan et al., 2007).

In general, climate change models predict increases in global precipitation, but the effects are expected to vary locally, with some locations receiving less annual rainfall and others receiving more variation in rainfall patterns (IPCC, 2007b). For example, Fay et al. (2003) found that by increasing the time between rainfall events, without changing the total rainfall applied, grassland productivity was reduced by 10%, which was specifically related to the underperformance of the less abundant grass species. The effects of altered precipitation will also depend on the plant communities’ adaptation to previous climate. In cool season pastures and grasslands, inconsistent winter recharge and spring rainfall can lead to reductions in both forb and overall productivity, and thus carbon sequestration (Heitschmidt and Haferkamp, 2003). Warmer and drier climates, such as southwestern scrub savannas, evolved under specific seasonal rainfall regimes (i.e., monsoons), and could be greatly disturbed by alterations to the seasonal rains (Weltzin and McPherson, 2003). Uncertainty in future local precipitation alterations greatly complicates climate predictions for grasslands, but interactions between temperature, moisture, CO₂, and management intensity are likely to produce different plant communities in the future.

Management is important in determining the response of grasslands to climate change. For example, a low input, native grassland in the UK was found to be very resistant to long term (>10 years) field climate manipulations, under moderate grazing, with variation in productivity and plant species relative abundance more driven by inter-annual variation in climate than by the climate change treatments themselves (Grime et al., 2008). Grime et al. (2008) argue that increasing the grazing in these grasslands may
be of greater concern than climate change. Vogel et al. (2012) found that species richness was positively correlated with aboveground production under experimental warming and drought regardless of the level of fertilization or mowing. However, Zavaleta et al. (2003) found that the application of nitrogen fertilizer to a California grassland significantly decreased the species richness and did not stimulate production. Grasslands’ productivity and diversity is dependent on the plant community response to elevated CO₂, warming, altered precipitation, and management.

*Herbivory*

Herbivory is an important ecological process in grasslands (Borer et al., 2014), with many modern grasses and grasslands having co-evolved with large grazing ungulates (McNaughton, 1979; Stebbins, 1981). Grazing is likely to be affected by climate change through resulting changes in the plant community and productivity shifts that may impact selectivity and the carrying capacity of grazing animals (Collins et al., 1998; Newton et al., 2006; Newton et al., 2014; White et al., 2014). For example, a simulated climate change and grazing experiment in Canada found that warming and reduced precipitation significantly decreased the quality of the forage available, and the researchers suggested that the effects of climate change on forage availability may be exacerbated by reductions in forage quality (White et al., 2014). Elevated CO₂ alone has been shown to significantly alter the plant species community in tall grass prairies by reducing the abundance of cool season perennial grasses (Owensby et al., 1999). Newton et al (2014) found that grazing reduced the climate change induced plant community alterations by selectively grazing the legume species that prospered most under elevated CO₂. Animal grazing also interacts with the invertebrate grazers that are present, often...
reducing the diversity and quantity of invertebrates with increasing animal grazing intensity (Kruess and Tscharntke, 2002).

While many grasslands are used for grazing of livestock, the ‘ungrazed’ grasslands, which include hay pastures, some natural grasslands, and urban grasslands, can receive significant herbivory pressure from invertebrates (Hulme, 1994; Stein et al., 2010). Little is known about the interaction of climate change with the effects of invertebrate herbivores on grassland ecology. Invertebrate herbivory has been defined by Stein et al. (2010) as direct consumption of plant material by organisms within the phylums Mollusca and Arthropoda. Research has shown that invertebrate exclusion from grasslands has no general effect on overall plant production, but is instead modified by plant species richness. In grasslands with high plant species richness, the effect of herbivory was negative, but the effect was opposite in grasslands with low plant species richness (Stein et al., 2010). Research by Stein et al. (2010) showed that exclusion of invertebrate herbivores decreases plant species richness by allowing grasses to outcompete erosulate forbs, suggesting that invertebrate herbivores are important for maintaining grassland diversity. Fabian et al. (2012) also showed that slug herbivory is negatively affected by increased plant species richness. Climate change is likely to have both direct and indirect effects on invertebrate herbivores (Sternberg, 2000). Hulme (1994) found that mollusk (more specifically slug) herbivory in grasslands can be as intense as rodent herbivory, and as such, could be important for determining the response of grasslands to climate change. Direct effects of climate change stem from changing precipitation amounts and temperature increases impacting slug abundance, biomass, herbivory, and fecundity. For example, Baur and Baur (1993) found that, due to the
urban heat island effect, several populations of slugs in urban and suburban areas were eradicated. Similarly, in a cool season grassland, Sternberg (2000) found that warming decreased slug population densities, but did not impact the plant communities. To date, very little research has been conducted examining the interactions of multiple climate change factors and slug herbivory within grassland communities.

*Nutrient Cycling*

Whether a grassland is a source or sink depends on the management and environmental conditions of the site (Kaye et al., 2005). Previous research has estimated that up to 0.5 Pg of carbon can be sequestered by the world’s grasslands per year (Scurlock and Hall, 1998b). The rates of carbon sequestration are affected by overall productivity as well as human use and management (Wan et al., 2005). Due to the depth and density of root infiltration into soils, grasslands can exhibit net sequestration of carbon (Joffre and Gren, 2001). Grasslands that are more productive are found to sequester more carbon through greater below ground growth and biomass turnover (Wan et al., 2005). Improving fertility, increasing irrigation, altering grass and legume species composition, and changing grazing practices have been shown to improve carbon storage in rangelands and pastures (Conant et al., 2001). Unfortunately, large portions of grasslands are overgrazed, which has been shown to reduce or even negate the ability of grasslands to sequester carbon, as grazing causes grasses to increase carbon allocation above ground where sequestration capacity is relatively limited (Conant and Paustian, 2002). Mowing in aesthetic grasslands, such as home lawns and athletic turfs, and can impact carbon sequestration rates of these systems, presumably through similar mechanisms (Townsend-Small and Czimczik, 2010). Elevated fertility has been shown
to increase turf carbon sequestration rates (Qian et al., 2010), while damage, such as daily clipping on golf greens (Bartlett and James, 2011) and foot traffic on athletic fields (Townsend-Small and Czimczik, 2010), can reduce or eliminate net positive carbon sequestration. Grassland carbon sequestration is, in part, governed by the intrinsic location, soil properties, and human management, but it is also very likely to be modified by climate change in the coming decades.

Climate change could theoretically improve carbon sequestration rates in grasslands, though in order to attain increased productivity, additional nitrogen would probably be required (Izaurralde et al., 2011). In areas with predictions for increased rainfall, carbon sequestration is expected to increase, though organic decomposition may increase simultaneously (Qian et al., 2010). Increased heat may produce positive effects on carbon sequestration through an overall increase in productivity (Parton et al., 1995); however, the local effects on grasslands are still mixed (Wan et al., 2005; White et al., 2014). Until the relative contributions of inputs to and outputs from grassland soil C pools are known, predicting the net outcome and carbon sequestration under future climatic conditions will remain challenging.
1.4 Sustainability of Kentucky Grasslands

Two regionally and globally relevant types of actively managed perennial grasslands currently present in Kentucky include pasture/haylands and turfgrasses. Half of the 4,226,540 ha of actively managed agricultural lands in Kentucky are forage pastures and hayland, which accounts for roughly 20% of total land area (USDA, 2013). Only forest has more land cover in Kentucky (USDA, 2013). Turfgrass is generally considered an intensively managed type of grassland, and is clustered in and around urban areas (Milesi et al., 2005). Turfgrasses are the main vegetative cover in urban and suburban areas and account for nearly 25% of land area in these locations, which cover 852,000 ha in Kentucky, making them the next largest grassland ecosystem in Kentucky (Robbins and Birkenholtz, 2003; USDA, 2013).

Forage pastures in Kentucky typically consist of a mixture of cool season grass species, including tall fescue (*Schedonorus arundinaceus* Steud.), Kentucky bluegrass (*Poa pratensis* L.), orchard grass (*Dactylis glomerata* L.), and timothy-grass (*Phleum pratense* L.), as well as 1-3 legume species, including white clover (*Trifolium repens* L.), red clover (*Trifolium pratense* L.), and alfalfa (*Medicago sativa* L.) (Henning et al., 2000). Maintaining the species diversity of these haylands is important for continuing to sustainably produce quality forage (Henning et al., 2000). Climate change is expected to alter the interactions of grassland plant species, and their interactions with invertebrate herbivores, which are vital to both maintaining hayland productivity and diversity (Izaurralde et al., 2011; Stein et al., 2010). Slugs are considered major pests of pasture and haylands in New Zealand and the United Kingdom (Barker, 1989), and their herbivory has been shown to significantly alter the species composition of pastures.
Barker, 1991). The relationship between slugs and pasture productivity and species composition has not, to date, been examined as extensively in the United States; thus, further research is needed to explore these relationships and assess how they might change in response to climate change. Therefore, I performed an experiment to evaluate the effects of warming and increased rainfall on central Kentucky pasture plant community dynamics, slug herbivory, and slug abundance (Chapter 2).

The sustainability of turfgrass ecosystems has been defined by its ability to mitigate climate change by sequestering carbon and reducing both carbon and nitrogen-based greenhouse gas emissions (Selhorst and Lal, 2013). Turfgrasses have been shown to be relatively sustainable depending on the types of maintenance they receive, as well as the location where they occur (Bartlett and James, 2011; Koerner and Klopatek, 2002). Higher maintenance turfs, such as golf courses, tend to have higher sequestration rates, but also higher emissions of nitrogen-based greenhouse gases and fossil fuel exhaust (Bartlett and James, 2011). Sequestration rates and greenhouse gas emissions have also been shown to vary with previous land use and local climate, mainly growing season length and irrigation requirements (Qian et al., 2010; Selhorst and Lal, 2013). Kentucky is in a transitional climatic zone, experiencing both continental northern and subtropical influences, which allows a wide variety of turf species to be grown, that also vary in maintenance requirements (Powell, 1976). Unfortunately, there is no research to suggest what lawn types are most environmentally sustainable given Kentucky’s climate. Therefore, I performed an experiment examining the effects of home lawn systems on greenhouse gas emissions from several common central Kentucky homeowner lawn types (Chapter 3).
Chapter 2: Influence of Climate change on Slug Herbivory in a Central Kentucky Tall Fescue Pasture

2.1 Introduction

Herbivory is an important ecological process in most ecosystems (Borer et al., 2014; Metcalfe et al., 2014), but especially in grasslands, which typically co-evolved with significant levels of mammalian herbivory (Jaremo et al., 1999; Milchunas and Lauenroth, 1993; Muthoni et al., 2014). A lesser studied, but potentially significant, herbivore of grass dominated ecosystems, is the slug (Loranger et al., 2014; Loranger et al., 2012). Slugs have long been considered agricultural pests in pasture and field-crop settings, especially in cool and wet climates such as the United Kingdom and New Zealand (Barker, 1989; Martin, 1991). They are also particularly abundant and problematic in cereal crops, where dense stands of grasses are grown in wet climates (South, 1992). With the adoption of no-till agriculture in the United States, slugs have become a much more prominent pest, as they perform better in undisturbed soil with greater residues (Douglas and Tooker, 2012). While slugs can have a large, negative effect on the yield of row crops (Douglas and Tooker, 2012), it is not always as simple to determine the effects of slug herbivory in pastures or perennial grasslands, as they are more heterogeneous systems characterized by high inter-annual variation in plant biomass and species relative abundances (Knapp and Smith, 2001; Sternberg, 2000). Slugs can have a measureable positive or negative effect on plant biomass production, depending on a myriad of environmental and biological factors (Moise and Henry, 2012; Rodriguez and Brown, 1998).
Not only can slug herbivory have variable impacts on plant biomass (Buschmann et al., 2005), but because slugs are selective in what they eat (Barlow et al., 2013), they can also alter the plant species composition of a stand. For example, in pasture and grassland settings, slugs have been reported to selectively graze seedlings (Buschmann et al., 2005). Slug seedling selection can reduce the success of over-seeding other grasses and legumes into pastures (Byers et al., 1985; Hanley, 1998); however, slug preference has been shown to also depend on seedling age and species (Hanley et al., 1995). Nitrogen content is another important factor determining grazing preference for slugs, with plant material containing higher nitrogen being preferred when all else is equal (Mattson, 1980). However, as shown by Mattson (1980), seedling age appears to be the dominant factor governing palatability for slugs, due, in part, to young shoots having less physical barriers to herbivory, as well as increased succulence. While seedling age governs grazing selection in monocultures, species identity and abundance can be greater drivers of grazing selection in polycultures (Hanley et al., 1995). Hanley et al. (1995) found that when multiple species of rosette forbs were grown in polyculture (versus monoculture), rates of herbivory were more dependent on species than seedling age. Due to the high species richness of most grasslands, slug herbivory may either increase or decrease plant diversity depending on if the more palatable species are the most or least abundant (Buschmann et al., 2005).

The effects of slug herbivory are likely to be impacted by climate change through both direct and indirect interactions (Moise and Henry, 2012; Sternberg, 2000). Climate change is expected to increase global average annual temperatures by 1.4 to 5.8°C depending on the effectiveness of mitigating greenhouse gas emissions (IPCC, 2007b).
The eastern US is also expected to experience anywhere from 10-30% increases in mean annual rainfall, primarily in the form of more intense summer storm events. Changes in both abiotic parameters, temperature and water availability, could directly and significantly affect the ecology of slugs in pastures (Barker, 1991; Douglas and Tooker, 2012; Jones et al., 2001). Slugs are known to be sensitive to abiotic factors, strongly preferring cool, moist conditions (Barker, 1991; Baur and Baur, 1993; Sternberg, 2000). For instance, the urban heat island effect has been shown to decrease slug survival in otherwise suitable environments (Baur and Baur, 1993), and while slugs have been shown to prefer wetter places in fields, with densities being determined primarily by soil moisture, the effect of moisture availability has been shown to be highly seasonal and to change dramatically from year to year (Barker, 1991; Sternberg, 2000).

Climate change may also have indirect effects on slugs if it alters plant community composition and quantity, quality, or nitrogen concentration of the plant material. In an experimental grassland in the UK, climate manipulations of winter warming and added or reduced summer precipitation were shown to affect slug relative abundance, but this result was attributed primarily to changes in vegetation (Sternberg, 2000). However, other research has shown that in some systems, slugs can influence the composition of vegetation through seedling herbivory, suggesting that direct effects of climate change on the slug community may result in alterations to the plant community (Hanley, 1998). In a factorial study of warming, nitrogen addition, and slug exclusion, Moise and Henry (2012) found that long term warming (3 yrs) in an old field pasture reduced overall plant productivity, but that slug exclusion in warmed plots negated the negative effect on biomass of warming alone. The results of Moise and Henry (2012)
suggest that the negative effect of warming on grassland productivity was mostly a direct
effect of warming increasing slug herbivory, but the effects of slugs in mediating plant
community response to climate change may depend on the species of slug itself.

The cosmopolitan slug *Deroceras reticulatum* (O. F. Müller) has been shown to adapt well to changing environments, including agricultural fields with frequent disturbances (Barker, 1991). *D. reticulatum* has shown preference for pasture plant species compared to some native grassland species (Barlow et al., 2013), and therefore, herbivory by this organism may influence pasture plant community dynamics depending on grazing preferences for the plant species present. Slug herbivory has been shown to influence the competitive abilities of co-occurring plants, thus altering community composition, of mature *Poa annua* (L.) and *Arabidopsis thaliana* (L.) in a mixed species greenhouse setting (Rodriguez and Brown, 1998). Slug herbivory can influence pasture plant community dynamics through both selective grazing and by altering plant competitive abilities, but the ultimate effect of slug herbivory on a system is likely to depend on its type.

Little research on slug herbivory or ecology has been conducted in pastures in North America, although in no-till wheat crops, slugs have been shown to be a significant pest in areas of the United States (Douglas and Tooker, 2012). As slugs have generally been considered major pests in cool and wet climates, they have not been investigated in the transitional climatic zone of the United States, which favors both cool and warm season plants. There have been no studies determining the effect of slug herbivory on biomass yield in tall fescue pastures of the transitional climatic zone. Furthermore, there have been only a few investigations into the effects of climate change on the slug
community and its interaction with the plant community of pastures (Moise and Henry, 2012; Sternberg, 2000). As previously stated, the southeastern United States is expected to experience increases in annual mean temperature of 1.4-5.8°C and receive an additional 10-30% of mean annual rainfall (Jones et al., 2001). There are 14 million hectares of tall fescue pasture in the United States, which are mostly concentrated to the east of the Mississippi River (Hill et al., 1990), and in Kentucky alone, there are over 2 million hectares of pasture (USDA, 2013). Unfortunately, not enough is known about slug herbivory and ecology in pastures of the transitional climatic zone to begin predicting responses to climate change. To address this gap in knowledge, I measured slug abundance, biomass, and herbivory in a manipulative, field climate change project in central Kentucky pasture, where temperatures were raised +3°C year-round and growing season precipitation was increased by +30% of the long-term mean.

I hypothesized that slug herbivory would have a measurable negative impact on aboveground plant biomass. I also hypothesized that climate change factors would affect slug herbivory, as well as slug abundance and biomass. As others have shown (Barker, 1991; Sternberg, 2000), I predicted that slug numbers and biomass would vary over the course of the year, and slug response to the climate treatments would be seasonally dependent. For example, warming might increase slug abundance, biomass, and herbivory during cool spring and fall months, but suppress them during hot summer months. Added precipitation might increase slug numbers, biomass, and herbivory during hot and dry months of the summer, but have limited effect during the rest of the year. I predicted that the combination of warming and additional precipitation would allow slugs to persist and exhibit significant herbivory during the hot summer months.
compared to warmed only plots where they would be much less abundant during this stressful time of the year. Co-occurring changes in plant community composition, both seasonally and resulting from the climate change treatments, could modify these responses.
2.2 Materials and Methods

Research Site

The experiment was conducted at the University of Kentucky’s Spindletop Research Farm in Lexington, Kentucky (38°10’N, 84°49’W). The site was established in an existing climate change project as described in Brosi (2011) and Bourguignon (2013). After a spring application of glyphosate (Roundup Pro; Monsanto, St Louis, MO, U.S.A.) to remove existing vegetation, the research field was established in the summer of 2008 with ‘Kentucky-31’ tall fescue (Schedonorus arundinaceus, seeded at a rate of 11.2 kg ha\(^{-1}\) pure live seed), ‘Ginger’ Kentucky bluegrass (Poa pratensis, 7.8 kg ha\(^{-1}\)), ‘Freedom’ red clover (Trifolium pretense, 6.7 kg ha\(^{-1}\)), and ‘Patriot’ white clover (Trifolium repens, 2.2 kg ha\(^{-1}\)). Bermudagrass (Cynodon dactylon) was plugged into the field later in the summer at a rate of 10.76 plugs m\(^{-2}\).

During the winter of 2008-2009, twenty 3m diameter hexagonal plots were established, and divided into five replicates of four climate treatments (Figure 2.1). The climate treatments were ambient conditions (control), +heat (+3°C above ambient), +precipitation (+30% of the long term mean annual rainfall was applied during the growing season from rainwater collected on site), and a combination of +heat and +precipitation. The climate treatments began on May 1\(^{st}\), 2009 and ran continuously until November 11\(^{th}\), 2013. Each plot was surrounded with aluminum flashing buried 0.5m deep in the soil. Soil temperature and moisture at 5cm below the soil surface were recorded in each plot at 15 minute intervals for the duration of the project. Heated plots were consistently maintained on average 2.8-3°C above non-heated plots, and added precipitation plots were consistently wetter than non-irrigated plots by anywhere from 3-
15% volumetric water content, with differences between treatments in soil moisture being
greatest during the summer and fall (Figure 2.2). The field was managed as a hayfield, and harvested (to 6cm) three times each year. Over the time period of this particular experiment, harvests occurred on October 5, 2011; May 21, July 30, October 8, 2012; and May 21, August 5, and October 14, 2013.

**Exclosure Design**

To quantify the effects of slug herbivory on pasture productivity, areas of slug exclusion were constructed and paired with control structures that did not exclude slugs in each experimental plot (Figure 2.3). The exclosures were modeled after those described by Strauss et al. (2009). Strauss et al. used 50 cm wide rings made of 20.3 cm aluminum flashing. In order to accommodate two rings in each of the 3 m diameter plots, it was necessary to reduce the size of the rings. The height was reduced from 20.3 cm to 15.2 cm, as that was the next size down aluminum flashing (8” to 6”). The diameter reduction was calculated proportionally to the height decrease of 25%; thus the diameter used was 37.5 cm. Slug exclosure rings were rimmed at the top with a 5cm wide band of copper tape, which has a repellant effect on slugs. The control rings lacked the copper tape and had three 4x8 cm sections removed from the base, in order to allow movement in and out of the ring. The rings were installed in the plots on August 5, 2011. The rings were fastened flush to the ground with several stakes made from baling wire. There were two rings per plot (one exclosure and one control) for a total of 40 rings. The above ground plant biomass was harvested from within the rings during the tri-annual harvests of the rest of the field. Slug herbivory was quantified as the difference in above ground plant biomass between each set of paired collars within a plot. If slug exclosures had less
plant biomass measured than the adjacent control ring, then slug herbivory had occurred at the measured level of consumption. If the two rings were equal in plant biomass, then no slug herbivory was detected.

Slug Abundance

Live traps were set at regular intervals during the second year of the study to determine if there were differences in the abundance of slugs between the climate treatments. As the result of several preliminary trials (unpublished data), it was determined that the best method of live trapping was to place approximately 10g of wet cat food (Friskies Classic Paté, Nestle Purina Petcare Company, St. Louis, MO, USA) in the center of a 100mm diameter petri dish, and the dish was left in a plot for 15 minutes just after dawn. This method attracted the greatest number of slugs consistently. Trapping was conducted every two weeks from January 3, 2013 and ended on October 10, 2013.

Starting at sunrise, baited dishes were placed randomly in each of the four plots within the first of five blocks, and left there for 15 minutes. It was determined through six months of methods testing (unpublished data) that it was important to finish the trapping by 45 minutes after sunrise, as the trapping success decreased after full sun was present. At the end of the 15 minute period, each dish was examined, and the length, number, and species of each slug found on the dish was recorded. Baited dishes were placed in the second block 5 minutes after dishes were placed in the first block. At any time, two blocks were being trapped so that their 15 minute periods were on a staggered schedule, which allowed for all 5 blocks to be trapped within 45 minutes. The order in
which blocks were trapped was determined by a random sequence generated by PROC PLAN in SAS the day of each trapping event.

Additional slug trapping occurred outside the experimental plots from August 15 to October 15, 2013. These slugs were taken to the lab and used to establish a relationship between slug length and dry biomass. In the lab, slug length was measured in the same way as occurred during plot trapping, and then slugs were dried in a 105°C forced air drying oven for 1 day and weighed. The length versus biomass curve was used to estimate the biomass of slugs caught during trapping, so that it was only necessary to measure the length of slugs when live-trapping, as it was a time sensitive process. The curve was determined using the model ln(mass) = A + B*ln(length) in PROC REG as discussed in Hawkins, Lankester, and Lautenshlager (1997). The model yielded the equation: 

$$mass (mg) = 0.007 \times length (mm)^{2.92},$$

and had an R^2 value of 0.5855 (P=0.0099, n=10, Figure 2.4). The majority of slugs captured (84%) had lengths that fell within the range of the curve used to estimate mass (Figure 2.5).

**Forage Quality**

Above ground plant biomass carbon to nitrogen ratio (C:N) and plant functional clade relative abundance were recorded at a plot level from every tri-annual harvest, as part of the larger project. C:N analysis was performed on a random sample of forage taken from each plot after it had been mowed. Functional clade relative abundance was determined from permanent quadrat biomass harvests, which were sorted to the species level. The functional clades were C_3 grasses, C_4 grasses, legumes, and weedy forbs. Visual cover estimates were also recorded by species (to 1%) for each slug exclosure and control ring (n=40) immediately preceding the final harvest in October 2013.
**Statistical Analysis**

In order to compare the degree of slug herbivory across the climate treatments, differences in plant biomass between the harvested control and slug exclosure rings were analyzed as a 2x2 climate treatment factorial (Heat*Precipitation) within PROC MIXED with repeated measures analysis (SAS Institute Inc. 9.3, Cary, NC). The slug exclosure and control rings were treated as split-plots within the 2x2 climate treatment factorial, with fixed effects as slug presence/removal, heat, precipitation, and time. Block was used as the random effect. Above ground plant biomass data was normally distributed as determined by the Anderson-Darling test on four of seven harvest dates. The three harvests that were not normally distributed were determined to be within tolerances of an ANOVA model, so no transformations were performed.

Slug count and biomass were analyzed over time using a repeated measures PROC MIXED analysis with fixed effects of heat, precipitation, and time. Block was the random effect. The count of slugs trapped was square root transformed. The square root of count and the estimated mass of slugs trapped were not normally distributed, but were determined to be appropriate for analysis by ANOVA by visually assessing a histogram of the data. Pairwise comparisons were performed with lsmeans.

Plant species cover data taken from the slug exclosure and control rings prior to the October 2013 harvest were analyzed using non-metric multidimensional scaling ordination in PCORD (NMS; PCORD version 6.0, MjM Software, Gleneden Beach, OR). Pairwise comparisons of the climate treatment plant species cover estimates were performed with MRPP analysis, and p-values were corrected using the Bonferroni-adjustment for the number of comparisons ($\alpha=0.05/6 : \alpha_{adj}=0.008$). Plant biomass C:N
and %N were analyzed using the same factorial climate model described above in PROC MIXED.
2.3 Results

Effects of Slug Exclosure

Over the entire two and a half year duration of the study, slug removal had no effect on harvested above ground plant biomass (Table 2.1; Figure 2.6). However, on one harvest date (May 2013), there was a significant effect of slug removal in the heated plots, with a greater amount of plant biomass harvested from the slug exclosure rings than the control rings, suggesting significant slug herbivory occurred in this climate treatment at that point in time. This was also the only point in time during the study when heated plots had significantly more total plant biomass than the other climate treatments (Figure 2.6). There were significant effects of time, time*heat, time*precipitation, and time*heat*precipitation on plant biomass, but there were no significant interactions with the slug exclosure treatment (Table 2.1). Seasonal differences in harvested plant biomass were not consistent across years, with the lowest occurring in July 2012 (11.3 ± 1.1g average plant biomass per ring) and the highest in August 2013 (45.5 ± 3.3g plant biomass; Figure 2.7). There were no main effects of precipitation, though a significant time*precipitation interaction was apparent in the fall of 2013, where +precipitation plots had significantly greater plant biomass than ambient precipitation plots.

Slug Abundance

Both slug count and biomass were significantly reduced in heated vs. non-heated treatments, but this climate treatment effect was modified by time (Tables 2.2 and 2.3). In the winter, heated plots had higher counts and biomass than non-heated plots, but in the late summer and early fall, the heat effect was reversed (Figure 2.8). A significant
time*precipitation interaction was also apparent for slug counts: added precipitation plots had more slugs than plots that received no precipitation but only in late summer and early fall (Figure 2.8). Overall time effects were apparent, as numbers and biomass of slugs trapped were much higher in late spring and early fall than throughout the rest of the experiment (Figure 2.9). Slug species was not considered in the analysis as only 4 of 325 slugs trapped in 2013 (1.2%) were not *Deroceras reticulatum*. The other species found was *Arion hortensis* (Férussac), and was present in both control and precipitation plots.

**Forage Quality**

Ordination analysis of the slug exclosure and control ring plant cover estimates showed two main groupings, which were defined by the heat treatment. Heated plots had distinctly different vegetative communities compared to the non-heated plots at the time of the October 2013 harvest (Figure 2.10). Pairwise comparison analysis using multi-response permutation procedures (MRPP) confirmed this result, yielding significant differences in all comparisons between heated and non-heated treatments (Table 2.4). There was no grouping that suggested differences between control or slug exclosure rings, illustrating that slug removal had no detectable effect on the plant community composition which is consistent with the overall lack of measurable slug herbivory at the site. The only significant effect of slug exclosure on plant biomass was observed in May 2013 (Figure 2.6); however, there were relatively few differences in plant community composition at this time, as all plots were dominated by C₃ grasses (Figure 2.11).

Similar to the species cover ordination, plant biomass data from the harvests showed that heated plots became dominated by C₄ grasses at the end of the growing season, whereas the non-heated treatments remained dominated by C₃ grasses throughout
the year (Figure 2.11). While the number and biomass of slugs trapped were significantly
affected by heat and precipitation treatments, the effects were seasonally dependent and
may reflect both trends in the quantity of plant biomass and the variation in plant
community composition (Figures 2.7, 2.8, and 2.11). For example, the stimulation of
slug number and biomass in the added precipitation treatments during Fall 2013 might be
the result of increased forage availability (as evidenced by greater plant biomass; Figure
2.7). However, the depressive effects of the heat treatment on slug numbers and biomass
at this same point in time does not appear to be related to reduced forage quantity (as
plant biomass was similar between heated and non-heated plots; Figure 2.7) but could be
related to the quality of the forage present, as heated plots were dominated by C4 grasses
(in contrast to C3 dominated non-heated plots), which are known to have lower forage
quality due to lower %N (Figure 2.11).

Differences in plant species composition across treatments appeared to influence
the quality of available forage, as significant effects of heat, heat*precip, time, and
heat*time on C:N ratios of grab-sampled forage for the year of slug trapping were
identified (Table 2.5). Heated plots had higher average C:N values (± S.E.) than non-
heated plots (30.3±1.0 vs. 27.3±0.7, respectively), but this effect of heat was only
significant in summer and fall harvests, when the plots were dominated by C4 grasses.
The addition of precipitation to +heat and ambient plots affected C:N differently. In
+heat+precip treatments, C:N was subtly augmented compared to +heat alone, while in
+precip treatments, C:N was slightly lower than in ambient plots (Figure 2.12). In the
spring, when plant community composition was more similar across climate treatments,
C:N ratios did not differ (Figure 2.12). Changes in C:N over the course of the season
appeared to be driven primarily by changes in %N, as the increases and decreases in %N almost directly mirror the changes and significant differences in C:N (Table 2.5; Figure 2.13).
2.4 Discussion

Despite prior work illustrating that slug herbivory can significantly reduce plant production in pastures (Borer et al., 2014; Buschmann et al., 2005) and my hypothesis that it would do so at my site, I found no strong effect of slug exclosure on plant biomass over the 2.5 year study, with the exception of a positive effect of slugs exclusion in +heat plots at the May 2013 harvest date (+39% plant biomass in slug exclosures of this treatment). Analysis of the plant communities within the rings at the end of the 2013 growing season did not show any significant separation between slug exclosure and control rings, suggesting slug herbivory had minimal effects on plant community dynamics. Overall, these results indicate that slug herbivory is relatively limited at this site, despite the fact that I caught 325 slugs over the 2013 season.

In addition to eating above ground plant material, slugs are also known to consume litter and detritus (Goble, 2006), which is a possible reason why I was unable to capture a significant effect on above ground plant biomass. Slugs in my project may have been primarily eating detritus, which I did not measure. Research on the effects of slug exclusion has focused mainly on establishment and development of plants, and has found that slugs can influence the development of pastures and other ecosystems through seedling herbivory (Asplund and Gauslaa, 2008; Borer et al., 2014; Byers et al., 1985; Hanley, 1998). As the experimental pasture in which this project was conducted was established more than three years prior to the beginning of this project, it is possible that the primary influence of slugs on seedling survival and resulting plant community dynamics were missed. However, several plant species present in the pasture were
annuals (comprising at times 1.8%-6.9% of the biomass), sprouting new seedlings each spring and fall, and still, no effects of slug herbivory were observed.

It is also possible that the slug exclosures did not operate as effectively as the rings described in Strauss et al. (2009), which they were modeled after, since I could not insert them 5 cm into the soil as they were there. However, great care was taken to pin the rings as flush to the ground as possible during installation to keep slugs from invading the exclosure, but it was probably not as optimal as having the rings installed below the surface. This design issue would merit from further investigation in the field.

I hypothesized that seasonal trends in moisture and temperature would interact with the imposed climate treatments to alter effects of slug herbivory and slug abundance and biomass. I found that seasonal differences in slug abundance and biomass were modified by the climate treatments, in ways that I anticipated. When moisture was limiting (in late summer), +heat treatments were hotter and drier than non-heated plots (Figure 2.2) and had lower numbers of slugs (Figure 2.8). During the same time period, added precipitation significantly increased the abundance of slugs. Slug abundance and biomass varied significantly across the year. Slugs were only present above ground during the early summer, early fall, during an unseasonably warm week in February, and an unseasonably cool and wet period between mid-June and early July, illustrating, as others have found, that they are sensitive to the temperature and moisture levels in the field.

I also hypothesized that seasonal changes would interact with the climate treatments to alter plant community composition, forage quality, and biomass. I found evidence that warming significantly modified the seasonal plant community dynamics.
and forage quality. While the abundance of C₄ species and C:N ratio increased overall, as the season progressed for all treatments, +heat treatments had significantly greater increases in both parameters, which is consistent with research suggesting that warming will reduce overall forage quality (White et al., 2014). Heat, precipitation, and their interaction modified the seasonal effects on plant biomass. Plant biomass seasonal changes and climate modifications were inconsistent across the years, which suggests that the effects of climate change are subject to inter-annual variation. For example, during the drought of 2012, the +precip treatments had higher biomass, but during the relatively wet year of 2013, the +heat treatments had significantly higher biomass. Climate change may make inter-annual variation in harvestable biomass more variable, which is consistent with predictions by White et al. (2014).

I hypothesized that added heat, precipitation, and their combination would allow persistence of slugs during times of the year when temperature or moisture were limiting. Consistent with this hypothesis, I found that slugs were able to persist in +heat treatments during the winter, and in +precip treatments during the summer. The effect is particularly evident in the winter of 2013, when slugs were significantly more abundant in +heat plots, which likely resulted in the significant effect of slug exclusion observed in the spring of 2013. I found that heat effects observed in biomass and slug abundance were also observed in forage quality metrics. While the relative plant species cover estimates were only recorded for the final forage harvest, ordination analysis showed a statistically significant effect of heat promoting dominance by C₄ vegetation that was also observed in the relative abundance of plant species. Interestingly, slug abundances being higher in non-heated plots at the end of the summer coincides with the largest difference in C:N
between heated and non-heated plots, though there were no significant effects of slug exclusion during this period. During the late summer and fall, heated plots were majority C₄ grasses and ambient plots were majority C₃ grasses, which explains the large difference in C:N of the bulk forage for those plots. The changes in C:N were mostly driven by changes in N%, which is an indicator of forage quality for slugs (Mattson, 1980). More specifically, research has shown that invertebrate herbivores prefer C₃ grasses to C₄ grasses (Barbehenn and Bernays, 1992; Barbehenn et al., 2004), so with the combination of lower N% and lower overall forage quality, it is possible that the significantly higher abundance of slugs in non-heated plots was related to the significantly higher quality of available forage, though it should be noted that no significant effect of slug exclusion (i.e., herbivory) was measured at this point in time.

It appears that while the climate treatments altered both slug abundance and the forage quality in late summer and early fall, the ecological role of the slugs as grazers remained unchanged. These findings illustrate that the climate treatments can indirectly influence slug abundance through changes in plant community dynamics and forage quality; however, it is important to note that there was a significant effect of slug herbivory in the heated plots during the spring of 2013, when the differences in plant community and forage quality were minimal, which suggests that climate treatments were also directly influencing slug activity.

An important factor to consider when interpreting this work is that plot level climate manipulations have been shown to augment trends in herbivore differences (Moise and Henry, 2010). Because herbivores can artificially congregate or avoid climate treatments depending on preference, it is possible that the differences in slug
abundance between climate treatments could be artificially exaggerated relative to values that might be observed in an entire field or region subject to climate change. However, I saw little evidence of preferential congregation in my experiment, as I rarely caught large numbers of slugs in any treatment. Considering that the most abundant slug in my pasture was *D. reticulatum*, which is known for its ability to adapt to widely varying climates (Barker, 1991), it is possible that the slug community at the site has adjusted to the climate manipulations imposed over the three year period or that the magnitude of the climate manipulations was too low to significantly alter slug population dynamics.

**Conclusions**

Slugs do not appear to have a significant effect on plant biomass production and community dynamics in fescue pastures of central Kentucky under current conditions or with warming and additional precipitation, except potentially in the early growing season when slug herbivory and abundance was directly stimulated by increased temperatures. It is also possible that a significant effect of slugs on biomass was missed due to the timing of the study, as seedling emergence and stand establishment have been shown to be significant periods of slug herbivory. Later in the season, effects of the climate treatments on slug abundance appeared to be related to both direct and indirect effects. Increased precipitation buffered the negative effects of hot and dry conditions in late summer and early fall on slug abundance and biomass, but differences at this time of the year between heated and non-heated plots may have been related to a direct negative effect of heat on slugs and/or a decrease in plant %N and forage quality, associated with heat-driven changes in the plant community. Although slug abundance was sensitive to abiotic and biotic changes resulting from the climate change treatments, the limited effect...
of slug herbivory measured with the exclosures suggests that this ecological process has limited importance on pasture production in established, mixed species pastures of the transition zone of the central U.S. under current or future climatic conditions.
Table 2.1: ANOVA results illustrating the effect of slug exclusion on above ground plant biomass within the climate treatment plots over time. Degrees of freedom are shown as numeratorDF, denominatorDF. Bolding indicates significant main effects and interactions.

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<th>P-Value</th>
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</tr>
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<td>heat<em>exclos</em>time</td>
<td>6,180</td>
<td>0.21</td>
<td>0.9734</td>
</tr>
<tr>
<td>precip<em>exclos</em>time</td>
<td>6,180</td>
<td>0.31</td>
<td>0.9299</td>
</tr>
<tr>
<td>heat<em>prec</em>excl*time</td>
<td>6,180</td>
<td>0.52</td>
<td>0.7924</td>
</tr>
</tbody>
</table>
Table 2.2: Average number and estimated biomass (± S.E.) of slugs caught per plot by climate treatment across all 20 trapping events. Letters indicate significant differences across climate treatments.

<table>
<thead>
<tr>
<th>Treatment</th>
<th>Number of Slugs Trapped</th>
<th>Biomass of Slugs Trapped (mg)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Control</td>
<td>0.92 ± 0.19 a</td>
<td>13.23 ± 3.18 ab</td>
</tr>
<tr>
<td>Precipitation</td>
<td>1.19 ± 0.28 a</td>
<td>22.81 ± 5.25 a</td>
</tr>
<tr>
<td>Heat</td>
<td>0.66 ± 0.21 b</td>
<td>11.88 ± 3.84 b</td>
</tr>
<tr>
<td>Heat*Precipitation</td>
<td>0.48 ± 0.13 b</td>
<td>8.83 ± 2.79 b</td>
</tr>
</tbody>
</table>
Table 2.3: ANOVA table for the average count and biomass of slugs trapped per plot for 2013 in the climate change treatments. Degrees of freedom are shown as numeratorDF, denominatorDF. Bolding indicates significant P-values.

<table>
<thead>
<tr>
<th>Effect</th>
<th>Slug Count</th>
<th>Slug Biomass</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>DF</td>
<td>F</td>
</tr>
<tr>
<td><strong>Heat</strong></td>
<td>1,12</td>
<td>13.41</td>
</tr>
<tr>
<td>Precip</td>
<td>1,12</td>
<td>0.17</td>
</tr>
<tr>
<td>Heat*Precip</td>
<td>1,12</td>
<td>1.03</td>
</tr>
<tr>
<td><strong>Time</strong></td>
<td>19,304</td>
<td>20.62</td>
</tr>
<tr>
<td>Time*Heat</td>
<td>19,304</td>
<td>3.57</td>
</tr>
<tr>
<td>Time*Precip</td>
<td>19,304</td>
<td>2.01</td>
</tr>
<tr>
<td>Time<em>Heat</em>Precip</td>
<td>19,304</td>
<td>1.56</td>
</tr>
</tbody>
</table>
Table 2.4: Pairwise comparisons of the plant species cover estimates for the different climate treatments done with MRPP analysis in the ordination, showing that the plant community of all heated plots differed significantly from that of non-heated plots. Bolded P-values indicate significance.

<table>
<thead>
<tr>
<th>Comparison</th>
<th>P-value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Heat+Precip vs. Precip</td>
<td>0.000012830</td>
</tr>
<tr>
<td>Heat+Precip vs. Heat</td>
<td>0.061953540</td>
</tr>
<tr>
<td><strong>Heat+Precip vs. Control</strong></td>
<td><strong>0.000007520</strong></td>
</tr>
<tr>
<td>Precip vs. Heat</td>
<td>0.000044430</td>
</tr>
<tr>
<td>Precip vs. Control</td>
<td>0.450131430</td>
</tr>
<tr>
<td><strong>Heat vs. Control</strong></td>
<td><strong>0.000020670</strong></td>
</tr>
</tbody>
</table>
Table 2.5: ANOVA table for C:N ratio and %N of bulk forage from the 2013 harvests. Degrees of freedom are shown as numeratorDF, denominatorDF. Bolding indicates significance.

<table>
<thead>
<tr>
<th>Effect</th>
<th>C:N</th>
<th>%N</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>DF</td>
<td>F</td>
</tr>
<tr>
<td>Heat</td>
<td>1,32</td>
<td>14.08</td>
</tr>
<tr>
<td>Precip</td>
<td>1,32</td>
<td>0.31</td>
</tr>
<tr>
<td>Heat*Precip</td>
<td>1,32</td>
<td>7.23</td>
</tr>
<tr>
<td>Time</td>
<td>2,32</td>
<td>47.70</td>
</tr>
<tr>
<td>Time*Heat</td>
<td>2,32</td>
<td>7.16</td>
</tr>
<tr>
<td>Time*Precip</td>
<td>2,32</td>
<td>0.78</td>
</tr>
<tr>
<td>Time<em>Heat</em>Precip</td>
<td>2,32</td>
<td>0.15</td>
</tr>
</tbody>
</table>
Figure 2.1: Experimental plot layout of the University of Kentucky pasture climate change project where this work was conducted. There were five blocks (replicates) of four climate treatments (C = ambient control; H = +3°C; P = +30% of the long-term mean annual precipitation; and HP = the combination of those individual treatments)(credit: Jim Nelson).
Figure 2.2: Soil moisture (blue lines) and soil temperature (red lines; both at 5cm depth) in all climate treatments during 2013, as well as ambient precipitation received at the site and amounts of added precipitation applied to the +precipitation treatments.
Figure 2.3: Paired slug control (left) and exclosure (right) rings within an experimental climate treatment plot (personal photo).
Figure 2.4: Relationship between slug length and mass developed from *Deroceras reticulatum* caught on-site, and used to convert slug abundance data from trapping in the treatments to mass estimates.

![Graph showing the relationship between slug length and mass](image)

- \( R^2 = 0.5855 \)
- \( P = 0.0099 \)
- \( \text{mass(mg)} = 0.007 \cdot \text{length(mm)}^{2.92} \)

Figure 2.5: Histogram of the number of slugs of each size that were trapped during the 2013 season. Sizes range from 2.5mm to 35mm.

![Histogram showing slug sizes](image)
Figure 2.6: Average (±S.E.) above ground plant biomass harvested from the control and slug exclosure rings for each climate treatment over the seven harvest dates. The asterisk denotes a significant difference in plant biomass between the control and slug exclosure rings in the heated plots only in May of 2013.
Figure 2.7: Average (±S.E.) plant biomass harvested in each climate treatment from October 2011 to October 2013. The capital letters denote significant differences across the harvests, and the lowercase letter denote climate treatment significant effects within each harvest.
Figure 2.8: Trends in average (± S.E.) number of slugs trapped in heated vs. non-heated plots (A) or in added precipitation vs. ambient precipitation plots (B), and average (± S.E.) estimated biomass of slugs trapped in heated vs. non-heated plots (C). Points in time where significant effects between the climate treatments were observed are indicated by asterisks.
Figure 2.9: Average (± S.E.) number (A) and biomass (B) of all slugs trapped in each climate treatment over the sampling period, January – October 2013.
Figure 2.10: Ordination of plant species cover in the slug exclosure and control rings from each climate treatment experimental plot. Square points are slug exclosure rings and circular points are control rings. The plant species noted at each end of the axes indicate the positive and negative drivers of the ordination separation and had an R value $\geq 0.65$. 
Figure 2.11: Average (± S.E.) relative abundance of four plant functional groups (C$_3$, C$_4$ grasses; legumes; weedy forbs) harvested from permanent quadrats in each climate treatment during the period of the slug exclosure treatment.
Figure 2.12: Average C:N ratio (± S.E.) of bulk forage harvested from the four climate treatments over the 2013 growing season. Letters indicate significant differences within a given harvest date. There were no significant differences across climate treatments on the first harvest date.

Figure 2.13: Average of %N (± S.E.) of bulk forage harvested from the four climate treatments over the 2013 growing season. Letters indicate significant differences within a given harvest date. There were no significant differences across climate treatments on the first harvest date.
Chapter 3: Soil-to-Atmosphere Greenhouse Gas Emissions from High- and Low-Input Turf Systems of Central Kentucky

3.1 Introduction

Urban land use, which includes turfgrass systems, is one of the fastest growing land use types in the United States, largely derived from conversion of crop land or natural areas (Foley et al., 2005). Currently, there are 160,000 km² of turfgrass in the United States, which is three times more land area than the next most abundant irrigated crop - corn (Milesi et al., 2005). The majority of turfgrass systems are within urban areas (Figure 3.1), and can have associated environmental impacts, including changes in nutrient cycling (Selhorst and Lal, 2013), pesticide and fertilizer runoff (Robbins and Birkenholtz, 2003), increased water usage (Mayer et al., 1999), and maintenance-related fossil fuel usage (Witteveen and Bavier, 2013). Many of these environmental impacts can alter greenhouse gas (GHG) emissions, such as CO₂, N₂O, and CH₄, both positively and negatively relative to the current trends in GHG production (IPCC, 2007a).

Quantifying and reducing the emission of major GHGs, such as CO₂, N₂O, and CH₄, has been designated as necessary for understanding and limiting the impact of land use change on global climate change (IPCC, 2007a). Greenhouse gasses are important factors in climate change due to their ability: 1) to persist and break down ozone (O₃), which allows significantly more ultraviolet light to enter the atmosphere; and 2) to absorb and re-emit long wave energy into the upper atmosphere, causing what is known as the “Greenhouse Effect” (IPCC, 2007a). The ability of any specific GHG to contribute to warming in the upper troposphere is determined by both its efficiency of reemitting long wave energy absorbed from the sun and its longevity in the atmosphere (IPCC, 2007a).
In order to compare the potential warming effects of various GHGs, ‘100 year Global Warming Potentials’ (GWPs) were adopted as metrics (IPCC, 2007a). A GWP is defined as the amount of heat a certain mass of GHG would trap in the atmosphere relative to the amount of heat the same mass of CO$_2$ would trap in the 100 years after release into the atmosphere (IPCC, 2007a). Nitrous oxide is currently the greatest contributor to tropospheric warming when accounting for both the amount released into the atmosphere and the GWP of N$_2$O (Ravishankara et al., 2009), as it is has a GWP of 298, indicating that a given amount of N$_2$O can trap 298 times more heat than the same amount of CO$_2$ (IPCC, 2007a). Other major GHGs include a myriad of hydrocarbons and chlorofluorocarbons (IPCC, 2007a).

Homeowner lawns and golf courses in urban areas have been shown to be substantial contributors to GHG loads in these environments (Kaye et al., 2004; Selhorst and Lal, 2013). In turf systems, the major greenhouse gasses measured or emitted are CO$_2$ and N$_2$O (Townsend-Small and Czimczik, 2010). Carbon dioxide is released from turf systems through plant respiration as well, as from general microbial activity in the soil (Sylvia et al., 2005). Nitrous oxide is mainly released as a product of the denitrification pathway performed by soil microbes, but is also an intermediate in most nitrogen cycle pathways (Sylvia et al., 2005). Therefore, N$_2$O emissions are often stimulated by the addition of nitrogen fertilizers (Bijoor et al., 2008; Kaye et al., 2004). Nitrogen applications also tend to increase microbial and plant productivity, which can stimulate CO$_2$ fluxes from the soil and plants (Milesi et al., 2005). In addition, fertilization can produce gaseous ammonia as a result of the natural breakdown of urea,
the most common form of fertilizer applied (Sylvia et al., 2005). While ammonia is not a GHG, it can be transformed into N$_2$O in the atmosphere (Martikainen, 1985).

Greenhouse gas emissions from soils vary considerably across the globe. The production of many GHGs is controlled by biological processes but is also affected by environmental factors as well (Iqbal et al., 2013; Parkin and Kaspar, 2003; Parkin and Kaspar, 2004). Greenhouse gas fluxes often differ across plant cover and land management/use types (Kaye et al., 2004; Kaye et al., 2005). Recent research has found that spatial variation in environmental factors is important in governing GHG flux across a landscape, and that this variation was lower in grass dominated systems that also had higher correlations between GHG flux and soil moisture and temperature than adjacent cropland (Kreba et al., 2013). Turfgrass systems can experience significant spatial variability in local environmental conditions, depending on their placement in the landscape and intended use and resulting management strategies, all of which can influence their GHG production and mitigation potential.

Turf is largely concentrated around urban and suburban areas, where it can make up the majority of green-space land-use, which gives it the unique ability to both contribute to and mitigate urban-associated GHG production (Milesi et al., 2005; Robbins and Birkenholtz, 2003). Turf has the potential to produce more GHGs per ha than other crops (Kaye et al., 2004; Kaye et al., 2005). In Fort Collins, CO, Kaye et al. (2004; 2005) found that urban turf contributed 24% and 30% of regional CO$_2$ and N$_2$O fluxes, despite making up only 6.4% of land area. However, turf is also touted for its ability to sequester carbon (Bartlett and James, 2011; Qian et al., 2010). Turf systems can exhibit net sequestration of carbon, though the magnitude depends on the environment, level of
fertility input, management, and stand age (Selhorst and Lal, 2013). For example, Huyler et al. (2014) found that home lawns in southern Alabama were sequestering carbon (C) in the top 15cm of soil, though at much lower rates than reported by other researchers in cooler climates, possibly due to the warm and moist conditions in Alabama leading to faster microbial turnover. The cumulative effects of carbon sequestration and GHG emission determine if a system either net produces or sequesters GHGs. As previously stated, reducing GHGs is important for mitigating and reducing the effects of climate change, thus systems that net sequester GHGs are considered beneficial and perhaps even sustainable.

Variability in turf species, local environmental factors, and management practices determine the GHG production and C sequestration potential of turf systems (Maggiotto et al., 2000; Pouyat et al., 2007; Townsend-Small and Czimczik, 2010). Unlike many row crops, there are a wide range of fertilizer, irrigation, and cultivation levels for turfs (Bartlett and James, 2011; Milesi et al., 2005; Townsend-Small and Czimczik, 2010). Golf course turf is the most management intensive turf, and areas such as greens and fairways, which receive the highest fertility and almost daily mowing, are predicted to be net producers of GHGs (Bartlett and James, 2011). Less management intensive areas of golf courses, such as the fringes and roughs, can exhibit net C sequestration (Bartlett and James, 2011). Bartlett and James (2011) concluded that the golf course under study exhibited net sequestration of C; however, they were only able to offset the high emissions of N₂O resulting from fertilizer applications and CO₂ from fossil fuels associated with fuel from mowing and application of fertilizer when including the C sequestration of trees in non-playing areas of the course. The turf of the course was not
able to sequester enough C to offset the significant emission of CO₂ and N₂O during maintenance of grassy playing areas, implying that such high levels of turf maintenance are not sustainable without some associated offsetting carbon sink.

Reducing inputs of fertilizer and irrigation has been shown to lower both GHG emission and C sequestration potential (Milesi et al., 2005; Selhorst and Lal, 2013). In Townsend-Small and Czimiczik’s study of public athletic fields and ornamental areas (2010), under similar fertility management, athletic fields were unable to sequester any C compared to significant sequestration measured in ornamental turfs. The authors hypothesized this was the result of the athletic turf having to allocate C resources to aboveground plant regrowth required from the constant and concentrated foot traffic injury, and therefore, allocating less C to belowground C pools where sequestration typically occurs in these systems. They also found that applications of fertilizer caused significant pulses of N₂O for up to eight days following the fertilization event, although, in ornamental turf fields, soil C sequestration far outweighed the GWP of N₂O emissions under high or low fertility regimes. However, when C emissions from fossil fuel consumption related to mowing, fertilizer manufacturing, and irrigation were factored into the net equation, both ornamental and athletic turfs under high or low input maintenance regimes were all net emitters of GHGs. Selhorst and Lal (2013) sampled soils from homeowner lawns across the country to estimate potential for C sequestration and found that the rates of C sequestration varied widely depending on the level of maintenance, with the highest rates of sequestration being associated with the highest levels of maintenance. Milesi et al. (2005) modeled the net carbon exchange for turf systems in the lower 48 states under both high and low input scenarios and found that in
order for turf systems to reach full C sequestration potential, significant increases in irrigation would be necessary. Milesi also found that, according to their model, very few places in the United States exist where a high quality, mono-culture turfgrass lawn is possible without irrigation or fertilization. Most of those locations occur in a band from Iowa and Minnesota eastward to New England.

While this prior work has demonstrated that GHG emissions and C sequestration of turf systems depend on the climate of the site and the use and management of the stand, little work has been done to directly compare the GHG emissions of different home owner lawn management systems. Townsend-Small and Czimiczik’s (2010) study comparing public park fields showed that low traffic and low maintenance turfs, which are analogous to many home lawns, can be more sustainable than comparable athletic fields. Given that home owner lawns are by far the most common (cover the largest area) in most urban centers, and have the potential to contribute and mitigate GHG, it is important to create a better understanding of the sustainability of local homeowner lawns. Selhorst and Lal (2013) calculated the C sequestration potential for soils collected from many lawns across the United States and found that lawns will only net sequester carbon for a finite number of years after establishment and will become unsustainable in 66-199 years. When modeling the length of time a home lawn could be considered sustainable, they found that the potential use of lower input management increase the length of time until a stand reaches the point where it becomes a net emitter of C and GHG. Selhorst and Lal found that the ability of soils to sequester C varied significantly by location, which implies that the management levels will be most important in determining the sustainability of lawns within locations. Given that residential lawns comprise the
majority of urban turf systems (Milesi et al., 2005; Robbins and Birkenholtz, 2003) and likely experience a wide range of management types, depending on homeowner preference, it is important to compare common local lawn and maintenance types to determine the emissions of GHGs in order to improve future practices.

The primary objective of this study was to quantify GHG emissions for common homeowner lawn and management systems in central Kentucky. Kentucky has approximately 2400 km² of turfgrass, including golf courses, public parks, and home lawns (Milesi et al., 2005). As Kentucky is located in a transitional climatic zone between cold, continental and warm, sub-tropical influences, it is possible to grow both cool season turfs that are common farther north and warm season turfs common throughout the southeast. The common home lawn types in this study were tall fescue (*Schedonorus arundinaceus* Steud.), Kentucky bluegrass (*Poa pratensis* L.), zoysiagrass (*Zoysia japonica* Steud.), and an endemic polyculture dominated by fescue, bluegrass, and white clover (*Trifolium repens* L.). Fescue and bluegrass lawns are often kept as close to monoculture as possible and require fertilization, irrigation, and pesticide applications to maintain a high quality lawn, which is a subjective measure defined by the National Turf Evaluation Program as stands that have dark green color, high density, fine leaf texture, and high degrees of uniformity. Zoysiagrass is a C₄, warm season grass that is a low maintenance lawn that retains a high quality. The endemic polyculture is a low maintenance lawn in which the homeowner has not attempted to retain high quality as classically defined and accepted. Given that the four turf systems vary in dominant plant species, I hypothesized that there will be differences in GHG fluxes between all treatments over the entire season, as asserted by Pouyat, Yesilonis, and Nowak (2006). I
also hypothesized that the turf systems receiving irrigation, pest management, and fertilizers would have significantly higher soil-to-atmosphere CO₂, N₂O, and NH₃ fluxes due to higher soil moisture, urea additions, and increased productivity compared to the turf systems not receiving any of those maintenance additions. I also expected that the treatments receiving fertilizers would exhibit short-term increases in GHG emissions that would differ depending on the dominant plant species present.
3.2 Materials and Methods

Research Site

The experiment was conducted at the A. J. Powell, Jr. Turfgrass Research Center located at University of Kentucky’s Spindletop Research Farm in Lexington, KY (38°10’N, 84°49’W). In order to assess the C footprint of several common turfgrass systems for the area, twelve experimental plots were established in the spring of 2012. The plots were 18.2m x 18.2m with 18.2m runways between them (Figure 3.2). Irrigation was installed in the field such that any plot could be irrigated individually, depending on the treatment. There were four turfgrass systems or treatments, with three replications of each arranged in a randomized block design. The four treatments were common homeowner turfgrass/management systems in central Kentucky (Powell, 1976). Each treatment is a common turf type paired with the dominant management system, as described below.

Experimental Design

The four turf treatments were: tall fescue [*Schedonorus arundinaceus* (Schreb.)] receiving a high maintenance regime (described below); Kentucky bluegrass [*Poa pratensis* (L.)] also receiving high maintenance; Zoysia [*Zoysia japonica* (Steud.)] with a low maintenance regime; and an endemic polystand also receiving low maintenance. Bluegrass and fescue sods were purchased from a local supplier (Sodworks Inc., Bourbon County, KY). Kentucky bluegrass sod contained a 1:1:1:1 mixture of Everglade, Total Eclipse, Everest, and NuGlade cultivars; while the fescue sod was a 1:1 mixture of Inferno and Quest cultivars (Jacklin® Seed, J.R. Simplot Company Boise, ID). The zoysia sod was harvested from another experiment at the A.J Powell Jr. Research Center.
and was the Compadre cultivar. The endemic polystand was sodded from existing runways at the field site, and contained a diverse mixture of C₃ and C₄ grasses, as well as broadleaf weeds. The dominant grasses were bluegrass, fescue, nimblewill (\textit{Muhlenbergia schreberi} (J.F. Gmel)), Bermuda grass (\textit{Cynodon dactylon} (L.) Pers.), common crabgrass (\textit{Digitaria sanguinalis} (L.) Scop.), and creeping bentgrass (\textit{Agrostis stolonifera} (L.)). Common broadleaf weeds were dandelion (\textit{Taraxanum officinale} (F.H. Wigg.)), white clover (\textit{Trifolium repens} (L.)), and violet (\textit{Viola pratincola} (Greene)).

High maintenance treatments (bluegrass, fescue) were treated five times a year by a local lawn care company (Burley Belt Lawn Service Inc., London, KY) with fertilizer (total of 146.5 kg N ha⁻¹ y⁻¹) and pesticides (herbicides and insecticides) as deemed necessary by the applicators. High maintenance treatments also received irrigation based on evapotranspiration, as estimated by an onsite evapotranspiration gauge (ETgage Model E, ETgage Company, Loveland, CO). Fescue and bluegrass plots were irrigated to replace 60 and 80% of evapotranspiration minus rainfall (recorded onsite), respectively (Ervin and Koski, 1998; Samples, 2008). Irrigation was applied to fescue and bluegrass plots concurrently between July 13 and October 27, 2013, and totaled 110mm for the fescue plots and 150mm for the Bluegrass plots (Cropper, 2013). Low maintenance plots received no irrigation, fertilizers, or pesticides.

All treatments were maintained at a 7.6cm height of cut, and were mowed with a Toro Z Master Commercial Mower (Toro Company, Bloomington, MN) when the average height of all three replicates of a given treatment reached at least 10.2cm in height. The turfgrass maintenance treatments began in March of 2013, and the total number of mowing occurrences was 26 for the endemic polystand, 24 for bluegrass, and
22 for fescue (Cropper, 2013). During the spring of 2013, the zoysia plots were resodded with Meyer zoysia due to poor establishment and heavy pressure from non-desired Bermuda grass. Therefore, this treatment was removed from this particular study, leaving three turfgrass systems (two high maintenance – bluegrass and fescue; and one low maintenance – endemic).

**Greenhouse Gas Measurements**

In each of the twelve plots, two 30cm diameter by 15cm tall PVC rings were randomly placed and pressed into the ground, so that only ~2cm of the ring remained above ground, for a total of 24 rings across the field. To measure GHGs, a PVC ‘cap,’ lined with Teflon© tape (Bytac©, Saint Gobain Performance Planstics), was placed on top of a ring to create a sealed chamber that was connected as a closed loop to a photoacoustic spectroscopy gas analyzer (Innova Air Tech Instruments Model 1412, Ballerup, Denmark) via Teflon© tubing. Ammonia (NH₃), nitrous oxide (N₂O), and carbon dioxide (CO₂) concentrations (ppm) were recorded simultaneously every minute for ten minutes at each PVC ring during a measurement period. Fluxes were calculated using the calculations described by Iqbal et al. (2013). Concurrently with flux measurements at each ring, air temperature, soil temperature, and soil moisture were recorded. Soil temperature (at a depth of 5cm) and air temperature were recorded using a Taylor digital pocket thermometer (Model 9878E, Taylor Precision Products, Oak Brook, IL). Soil moisture was measured at a depth of 5cm with a DELTA-T HH2 moisture meter using a ML2x 6cm theta probe (Delta-T Devices, Cambridge, England). All plots were sampled consecutively on the same day between 10am and 3pm. Starting position and order of plots was changed every sampling date in the field, in a random fashion.
**Sampling Scheme**

Seasonal measurements started on February 10, 2013 and continued until December 11, 2013. From February 10 until April 30 and August 14 to October 25, measurements were taken every two weeks. From April 30 until August 14, measurements were taken every week. Measurements between October 25 and December 11 were recorded monthly.

As fertilization events can have a strong effect on GHG fluxes (Townsend-Small and Czimczik, 2010), the frequency of GHG flux measurements increased in the turfgrass treatments receiving fertilizer (high maintenance bluegrass and fescue) when fertilization occurred. There were five applications of N fertilizer in these treatments over the course of the 2013 growing season, of which, four were monitored closely for effects on GHG emissions. The first three applications monitored were on March 29, May 22, and September 13, and were applications of 12.21kg N ha⁻¹ as dissolved urea. An application was made in July that was not captured due to a scheduling error. The final fertilization event was 97.64kg N ha⁻¹ and was applied on October 2 as slow release urea. For every fertilizer event monitored, GHGs were measured each day for the two days immediately preceding the application and daily for at least 7 days post application until fluxes appeared similar to non-fertilized treatments.

**Statistical Analysis**

The three turfgrass treatments were compared over the course of a year (February 2013 – January 2014) for differences in GHG fluxes (NH₃, CO₂, and N₂O) using a repeated measures proc MIXED analysis in SAS (SAS Institute Inc. 9.3, Cary, NC). Fixed effects were treatment and time, and the random effect was replicate. The same
analysis was used to determine any significant effects of treatment on soil temperature and moisture. Annual estimates were calculated for CO₂ and N₂O fluxes by interpolating between recording dates and calculating the area under the curve with the trapezoidal rule. Data was recorded for 305 days, and annual estimates were scaled to one year with the following equation:

\[
\text{annual total flux} (gX \text{ m}^{-2} \text{ y}^{-1}) = \frac{\text{total flux for measured period} (gX \text{ m}^{-2})}{\frac{305}{365} \text{ days}}
\]

Annual estimates were then compared with a simple ANOVA model using treatment as the main effect and replication as the random effect.

The GHG data collected while monitoring fertilization events were analyzed with the same statistical model as the full season, but were performed on each event separately. Analysis of fertilization events only included the fescue and bluegrass treatments in the comparison, as these were the only treatments that received fertilizer. Again, soil temperature and moisture during fertilization were analyzed using the same model. The data were examined to see if GHG fluxes changed in response to fertilizer application and whether this response varied between the two turf systems. Specific comparisons between the day prior and each subsequent day after fertilization were also examined with the use of lsmeans for gasses and fertilization events that had significant time effects. Additionally, the data were also examined with a similar proc MIXED model that replaced the time main effect with a binary “pre/post” fertilization main effect in order to compare the average flux before and after application.
3.3 Results

Turf Treatment and Seasonal Effects

Turfgrass treatment and interaction with time were not significant for any GHG measured. Carbon dioxide fluxes varied significantly over time (Table 3.1), with highs in late June (363±33mg C-CO$_2$ m$^{-2}$ h$^{-1}$) and lows during December (47±7mg C-CO$_2$ m$^{-2}$ h$^{-1}$; Figures 3.3-3.5), corresponding to seasonally significant temperature maxima and minima of 26.3±0.6°C and 3.3±0.2°C, respectively (Figure 3.6). Nitrous oxide fluxes also varied significantly seasonally (221±47 to -7±164µg N-N$_2$O m$^{-2}$ h$^{-1}$; Table 3.1), often peaking with spikes in soil moisture from rainfall or irrigation (Figures 3.3-3.6). Ammonia had no significant time trends (Table 3.1), but fluxes varied from a low of 0.08±0.23 to a high of 1.57±0.93mg N-NH$_3$ m$^{-2}$ h$^{-1}$.

Soil moisture exhibited significant effects of time, treatment, and time*treatment (Figure 3.6). Reflecting the fact that the site experienced a wetter than average spring (2013), the spring was wettest (41.3±0.6% VWC), while early fall was driest (20.6±0.6% VWC), across all treatments. Averaged across the year, all treatments differed significantly from each other (35.8±0.4%, 32.4±0.4%, and 30.4±0.7% VWC for fescue, bluegrass, and endemic treatments, respectively). Soil moisture in the endemic treatment decreased significantly throughout the year, especially during the dry summer and fall months, compared to the irrigated treatments, resulting in the significant time*treatment interaction.

Soil temperature also varied significantly over time: highest in July (26.2°C) and lowest in December (3.2°C), as is consistent with the continental climate of the site. However, soil temperature was only marginally significant across treatment (P=0.057;
Figure 3.6), with average soil temperatures of 17.7°C, 17.4°C, and 17.2°C for bluegrass, endemic, and fescue treatments, respectively. Unlike soil moisture, there was no significant interaction between time and treatment.

There were no significant treatment effects on annual estimates of CO₂ (F₂=10.40; P=0.0877) and N₂O fluxes (F₂=1.85; P=0.3504). Carbon dioxide flux averaged across all three treatments was estimated to total 17.7kg C-CO₂ m⁻² y⁻¹, and N₂O flux totaled 9.8kg N-N₂O m⁻² y⁻¹ (Table 3.7).

**Fertilization Events**

Fluxes of CO₂ and N₂O varied over time during the majority of the fertilization events (Table 3.2). Analysis comparing average fluxes before and after fertilization found significant differences in CO₂ fluxes for all fertilization events, N₂O fluxes in the first two fertilization events, and NH₃ fluxes in only the second fertilization event (Table 3.4). CO₂ and N₂O fluxes increased in response to fertilization for the first event, but significantly declined or had no response during all other events. Ammonia fluxes decreased significantly after the second fertilization, but had no response to any of the other events.

Soil temperature and moisture varied significantly over time during all four fertilization events (Table 3.4). Soil moisture had significant treatment effects for all fertilization events, except for the second, while soil temperature had a significant treatment effect only during the third fertilization (Table 3.3). For all significant treatment differences in soil moisture and temperature, fescue had higher moisture than bluegrass and bluegrass plots had higher temperature than fescue plots. Often, CO₂ and N₂O fluxes mirrored soil moisture or temperature during a fertilization event, depending
on the time of year (Figures 3.7 and 3.8). For example, during the first fertilization, both 
N₂O and CO₂ fluxes followed the trend of soil temperature, perhaps due to the sampling 
period coinciding with the first warm period of the season; however, during the fourth 
fertilization event, while CO₂ fluxes still reflected soil temperature trends, N₂O fluxes 
tended to mirror soil moisture trends over time (Figures 3.7 and 3.8).

During each fertilization event no differences between bluegrass and fescue 
treatment responses were observed, with the exception of the final fertilization event on 
October 3, 2013 (97.64kg N ha⁻¹ applied) (Table 3.2, Figure 3.7), when the fescue plots 
had significantly higher CO₂ flux (217±8 mgC-CO₂ m⁻² h⁻¹) than bluegrass plots (190±8 
mgC-CO₂ m⁻² h⁻¹) following fertilization. Significant treatment*time interactions were 
only present in soil moisture during the first fertilization and soil temperature during the 
third fertilization (Table 3.3). During the first fertilization, soil moisture declined faster 
in bluegrass plots than in fescue plots, and during the third fertilization, soil temperature 
declined more in fescue plots than in bluegrass plots.
3.4 Discussion

Despite predicting that differences in dominant species and management regimes would create differences in GHG emissions from the three lawn types studied, I found there were no significant turf treatment effects for soil-to-atmosphere CO$_2$, N$_2$O, or NH$_3$ fluxes. It is possible that treatment differences were less pronounced than they would normally be due to the unusually cool and wet spring and early summer experienced at the site (2013). However, the fact that I measured significant differences in soil moisture and marginally significant differences in soil temperature between the high and low management types illustrated that treatment differences were strong enough to effect these abiotic factors, which are known to be important in determining GHG flux differences (Dobbie and Smith, 2003). As the season became dryer and warmer in August and September, high maintenance treatments receiving irrigation had greater soil moisture than low maintenance treatments, and there were similar apparent differences in CO$_2$ and N$_2$O fluxes between these treatments at this time of year, but they were not strong enough to be significant. If the year had been drier and warmer, as is typical for the site, differences between fluxes of these GHGs from the high and low maintenance plots may have emerged. It is also possible that treatment differences have not had enough time to become apparent, as my study occurred in the first year post-establishment for the stands. Previous work has shown turfgrass establishment effects on GHG emissions and C and N cycling change over time and vary by management type (Bartlett and James, 2011; Bijoor et al., 2008; Qian et al., 2010). Additional data collected over time from this site will assess these possibilities.
Despite the use of urea based fertilizers in high maintenance plots in this experiment, and significant evidence from the literature that additions of nitrogen both increase productivity as well nitrogen emissions (Bijoor et al., 2008; Townsend-Small and Czimczik, 2010), I only measured a stimulation of CO₂ and N₂O fluxes post-fertilizer application during the first event, and temporal trends during this event for both gasses appeared to strongly reflect co-occurring increases in temperature (Figures 3.5 and 3.6). At all other fertilization events, fluxes of CO₂, N₂O, or NH₃ either did not change over time or decreased. It is likely that the vast majority of applied ammonia was rapidly taken up by the grass, as there was visual evidence that they responded quickly to liquid urea fertilization (D. Weber personal observation). Fast plant uptake combined with relatively low rates of urea application (12kg N ha⁻¹) for the first three fertilizer applications may have contributed to the general lack of stimulation in GHG emissions. However, the larger granular application of 98kg N ha⁻¹, which occurred at the fourth application (in October), would still be expected to stimulate N₂O and NH₃ fluxes. The application date coincided with an extreme rainfall event, which I expected would increase the responsiveness of CO₂, N₂O, and NH₃ fluxes, but there was only a single day post rainfall and fertilizer application where N₂O spiked significantly and all other measured gasses failed to demonstrate a response (Figure 3.7). I hypothesize that the lack of flux response to the final fertilization was likely due to the urea being applied in a slow release granular form instead of as solubilized urea.

Annual total flux estimates for CO₂ and N₂O also did not differ significantly across the treatments. Although for CO₂ fluxes treatment differences were borderline significant (P=0.08), pairwise comparisons were not significantly different from each
other (P>0.3 in all cases). The marginal difference between treatments may be the cumulative result of a long term difference between the bluegrass and endemic plots that occurred over the course of the late summer and early fall when soil moisture differences were most pronounced. It was surprising that the annual N₂O flux totals were not significantly different between high and low maintenance levels, since high maintenance plots received applications of fertilizer that totaled an additional 146kg N ha⁻¹. Previous work has shown that common fertilization regimes can increase N₂O fluxes significantly (Maggiotto et al., 2000). As the rates applied at any given application date in my study were much lower than in other studies (12kg N ha⁻¹ vs. 50kg N ha⁻¹), it is possible that the application rate explains the lack of significant flux for the first three fertilization events, though not the final application of 96kg N ha⁻¹. The same study by Maggiotto et al. (2000) found that N₂O flux rates were lower when using urea based fertilizers compared to ammonium nitrate, and fluxes were further lowered when using coated urea, which was utilized for the 96kg N ha⁻¹. The type of fertilizers used in this study may have resulted in the general lack of GHG response.

Comparison of my annual estimates to those made in similar systems reveals that for both CO₂ and N₂O my annual estimates for all three turf systems are high compared to what others have measured (Table 3.5). The annual estimates of CO₂ calculated in this study most closely resemble those calculated by Koerner and Klopatek (2002), with values from my study falling between their estimates for home lawns and golf courses in Phoenix, AZ (Table 3.5). Koerner and Klopatek calculated their annual fluxes in the same way as in this study (i.e. single point in time measurements extrapolated between measurement dates using the trapezoidal rule), with the exception being their
measurements were recorded on a monthly basis (less frequently than my measurements). More recent research has found that even bi-weekly extrapolations tend to significantly over estimate CO₂ fluxes by neglecting to account for diurnal variation in fluxes, as well as the significant temporal variation that most likely occurs between measurement dates (Kreba et al., 2013; Parkin and Kaspar, 2004). This same sampling issue is present within my annual estimates of N₂O emissions, as the values from this project are 10x higher than other estimates from turf systems. The only other study that reports values in the same range as my estimates for N₂O fluxes was that of Dobbie and Smith (2003) measured in a hay pasture in the United Kingdom (Table 3.5). The high values reported by Dobbie and Smith may stem from their similarly careful sampling around fertilization events, as they applied large applications of fertilizer (60-120kg N-NH₄NO₃ ha⁻¹) that resulted in significant losses of N₂O. In the future, more temporally extensive measurements, aimed at capturing diurnal variation and irrigation and rain events, would help refine annual flux estimates at my site and may further illustrate significant differences between the treatments.

Conclusions

The goal of this study was to quantify GHG emissions from common home lawn maintenance systems in Kentucky. I expected there would be differences in CO₂, N₂O, and NH₃ fluxes between the three tested lawn systems attributable to both the plant species cover types, as well as the addition of fertilizer and irrigation. However, I found that there were few differences between CO₂, N₂O, and NH₃ emissions from fescue, bluegrass, and endemic lawns, despite maintenance regimes varying considerably. I also expected that the application of fertilizers would cause short-term increases in CO₂, N₂O
and NH$_3$ fluxes; however, I did not find strong evidence for such a response. It appears that the low rates of solubilized urea and moderate rates of slow release urea were not large enough to consistently stimulate GHG losses in these systems. My results show that GHG emissions from home lawns in central Kentucky were significant but were relatively insensitive to turf species or maintenance regime. As previously stated, the 2013 growing season was unusually wet and cool and the turf stands were newly established. More data, collected across multiple years, as the stands age, and incorporating finer temporal dynamics, will verify whether turf species and management regime really is not important in governing GHG emissions at this site. However, the results of this study regarding the negligible stimulation of gaseous N$_2$O and NH$_3$ losses resulting from fertilization suggest that homeowners and turf professionals are currently applying rates and forms of fertilizer that minimize these negative environmental impacts. Clearly, these types of home lawn and turf systems have the capacity to produce significant soil-to-atmosphere greenhouse gas emissions; however, the ultimate assessment of their sustainability will require additional measurements determining changes over time in both emissions and sequestration, and quantification of fossil fuel consumption contributions from mowing, fertilizer, and irrigation.
Table 3.1: ANOVA results for turfgrass treatment, time, and their interactive effects on carbon dioxide, nitrous oxide, and ammonia fluxes measured over the 2013 growing season. Bold values indicate significance. Degrees of freedom are presented as: numerator DF, denominator DF.

<table>
<thead>
<tr>
<th>Effect</th>
<th>CO₂</th>
<th></th>
<th></th>
<th>N₂O</th>
<th></th>
<th></th>
<th>NH₃</th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>DF</td>
<td>F</td>
<td>P</td>
<td>DF</td>
<td>F</td>
<td>P</td>
<td>DF</td>
<td>F</td>
</tr>
<tr>
<td>Treatment</td>
<td>2,4</td>
<td>0.44</td>
<td>0.6713</td>
<td>2,4</td>
<td>2.12</td>
<td>0.2353</td>
<td>2,4</td>
<td>0.61</td>
</tr>
<tr>
<td>Time</td>
<td>28,166</td>
<td>42.72</td>
<td>&lt;0.0001</td>
<td>27,160</td>
<td>4.66</td>
<td>&lt;0.0001</td>
<td>27,160</td>
<td>1.20</td>
</tr>
<tr>
<td>Time*Treatment</td>
<td>56,166</td>
<td>1.07</td>
<td>0.3719</td>
<td>54,160</td>
<td>0.78</td>
<td>0.8489</td>
<td>54,160</td>
<td>0.90</td>
</tr>
</tbody>
</table>
Table 3.2: ANOVA results for the effect of turfgrass treatment, time, and their interaction on carbon dioxide, nitrous oxide, and ammonia fluxes as measured over the four recorded fertilization events. Bolding indicates significance. Degrees of freedom are presented as: numerator DF, denominator DF.

<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>DF</td>
<td>F</td>
<td>P</td>
<td>DF</td>
</tr>
<tr>
<td>Treatment</td>
<td>1,2</td>
<td>0.06</td>
<td>0.8293</td>
<td>1,2</td>
</tr>
<tr>
<td>CO₂</td>
<td>Time</td>
<td>10,40</td>
<td>12.26</td>
<td>&lt;0.0001</td>
</tr>
<tr>
<td></td>
<td>Time*Trt</td>
<td>10,40</td>
<td>1.26</td>
<td>0.2841</td>
</tr>
<tr>
<td>N₂O</td>
<td>Treatment</td>
<td>1,2</td>
<td>0.07</td>
<td>0.8195</td>
</tr>
<tr>
<td></td>
<td>Time</td>
<td>10,40</td>
<td>7.81</td>
<td>&lt;0.0001</td>
</tr>
<tr>
<td></td>
<td>Time*Trt</td>
<td>10,40</td>
<td>1.23</td>
<td>0.3028</td>
</tr>
<tr>
<td>NH₃</td>
<td>Treatment</td>
<td>1,2</td>
<td>0.34</td>
<td>0.6205</td>
</tr>
<tr>
<td></td>
<td>Time</td>
<td>10,40</td>
<td>0.65</td>
<td>0.7589</td>
</tr>
<tr>
<td></td>
<td>Time*Trt</td>
<td>10,40</td>
<td>1.19</td>
<td>0.3256</td>
</tr>
</tbody>
</table>
Table 3.3: ANOVA results of the effects of turfgrass treatment, time, and their interaction on soil temperature and moisture during the four recorded fertilization events. Bolding indicates significance. Degrees of freedom are presented as: numerator DF, denominator DF.

<table>
<thead>
<tr>
<th></th>
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<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>DF   F   P</td>
<td>DF   F   P</td>
<td>DF   F   P</td>
<td>DF   F   P</td>
</tr>
<tr>
<td>Temperature</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Treatment</td>
<td>1,2  0.78  0.4715</td>
<td>1,2  1.23  0.3824</td>
<td>1,2  27.18  0.0349</td>
<td>1,2  13.85  0.0652</td>
</tr>
<tr>
<td>Time</td>
<td>10,106 413.05 &lt;0.0001</td>
<td>9,96 15.84 &lt;0.0001</td>
<td>7,76 204.73 &lt;0.0001</td>
<td>7,76 268.58 &lt;0.0001</td>
</tr>
<tr>
<td>Time*Trt</td>
<td>10,106 1.59 0.1189</td>
<td>9,96 0.46 0.8977</td>
<td>7,76 3.44 0.0030</td>
<td>7,76 0.98 0.4547</td>
</tr>
<tr>
<td>Moisture</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Treatment</td>
<td>1,4  86.01 0.0114</td>
<td>1,2  12.69 0.0705</td>
<td>1,2  86.92 0.0113</td>
<td>1,2  18.67 0.0496</td>
</tr>
<tr>
<td>Time</td>
<td>10,106 154.76 &lt;0.0001</td>
<td>8,55 43.20 &lt;0.0001</td>
<td>7,76 7.40 &lt;0.0001</td>
<td>7,76 105.54 &lt;0.0001</td>
</tr>
<tr>
<td>Time*Trt</td>
<td>10,106 1.94 0.0472</td>
<td>8,55 1.43 0.2062</td>
<td>7,76 0.50 0.8317</td>
<td>7,76 0.87 0.5305</td>
</tr>
</tbody>
</table>
Table 3.4: ANOVA results of the effects of turfgrass treatment, pre vs. post fertilizer application, and their interaction on CO₂, N₂O, and NH₃ during the four recorded fertilization events. Bolding indicates significance. Green cells indicate a positive change in fluxes between pre- and post-fertilizer application, and red cells indicate a negative change in fluxes. Degrees of freedom are presented as: numerator DF, denominator DF.

<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>DF, F, P</td>
<td>DF, F, P</td>
<td>DF, F, P</td>
<td>DF, F, P</td>
</tr>
<tr>
<td>Treatment</td>
<td>1,2 0.32 0.6305</td>
<td>1,2 0.08 0.8011</td>
<td>1,2 2.61 0.2476</td>
<td>1,2 5.27 0.1487</td>
</tr>
<tr>
<td>Pre/post</td>
<td>1,58 8.62 <strong>0.0048</strong></td>
<td>1,52 15.08 <strong>0.0003</strong></td>
<td>1,40 32.69 &lt;0.0001</td>
<td>1,40 8.08 <strong>0.007</strong></td>
</tr>
<tr>
<td>Pre/post*Trt</td>
<td>1,58 0.36 0.5519</td>
<td>1,52 0.14 0.7107</td>
<td>1,40 0.10 0.7533</td>
<td>1,40 0.01 0.9108</td>
</tr>
<tr>
<td>Treatment</td>
<td>1,2 0.52 0.5445</td>
<td>1,2 0.08 0.8002</td>
<td>1,2 0.46 0.5674</td>
<td>1,2 0.83 0.4592</td>
</tr>
<tr>
<td>Pre/post</td>
<td>1,58 5.33 <strong>0.0245</strong></td>
<td>1,52 7.44 <strong>0.0087</strong></td>
<td>1,40 0.11 0.7395</td>
<td>1,40 3.73 0.0606</td>
</tr>
<tr>
<td>Pre/post*Trt</td>
<td>1,58 0.93 0.3395</td>
<td>1,52 0.04 0.8395</td>
<td>1,40 0.00 0.9458</td>
<td>1,40 0.12 0.7270</td>
</tr>
<tr>
<td>Treatment</td>
<td>1,2 0.01 0.9343</td>
<td>1,2 0.11 0.7726</td>
<td>1,2 2.85 0.2337</td>
<td>1,2 0.07 0.8178</td>
</tr>
<tr>
<td>Pre/post</td>
<td>1,58 0.30 0.5832</td>
<td>1,52 4.89 <strong>0.0315</strong></td>
<td>1,40 0.14 0.7149</td>
<td>1,40 0.46 0.5036</td>
</tr>
<tr>
<td>Pre/post*Trt</td>
<td>1,58 0.37 0.5436</td>
<td>1,52 0.01 0.9176</td>
<td>1,40 3.43 0.0716</td>
<td>1,40 0.38 0.5418</td>
</tr>
</tbody>
</table>
Table 3.5: A comparison of previously measured annual estimates of CO$_2$ and N$_2$O fluxes from similar turf systems and grasslands.

<table>
<thead>
<tr>
<th>Author</th>
<th>Location</th>
<th>System</th>
<th>CO$_2$ Fluxes (kg C-CO$_2$ ha$^{-1}$ y$^{-1}$)</th>
<th>N$_2$O Fluxes (kg N-N$_2$O ha$^{-1}$ y$^{-1}$)</th>
</tr>
</thead>
<tbody>
<tr>
<td>(Weber, <em>this study</em>)</td>
<td>Lexington, KY</td>
<td>Irrigated/fertilized tall fescue</td>
<td>17,800±1400</td>
<td>10.4±1.2</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Irrigated/fertilized Kentucky bluegrass</td>
<td>18,800±100</td>
<td>10.9±0.4</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Mowed only mixed stand</td>
<td>16,700±1300</td>
<td>8.2±0.4</td>
</tr>
<tr>
<td>(Koerner and Klopatek, 2002)</td>
<td>Phoenix, AZ</td>
<td>Residential lawns</td>
<td>11,125</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Golf courses</td>
<td>69,145</td>
<td>-</td>
</tr>
<tr>
<td>(Townsend-Small and Czimczik, 2010)</td>
<td>Irvine, CA</td>
<td>Low fertilization public park turf</td>
<td>-</td>
<td>1</td>
</tr>
<tr>
<td></td>
<td></td>
<td>(estimate)</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>High fertilization public park turf</td>
<td>-</td>
<td>3</td>
</tr>
<tr>
<td></td>
<td></td>
<td>(estimate)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>(Groffman et al., 2009)</td>
<td>Baltimore, MD</td>
<td>Low management, McDonogh School</td>
<td>~700</td>
<td>3.5</td>
</tr>
<tr>
<td></td>
<td></td>
<td>High management, University of Maryland</td>
<td>~1000</td>
<td>0.8</td>
</tr>
<tr>
<td>(Kaye et al., 2005)</td>
<td>Fort Collins, CO</td>
<td>Irrigated/fertilized Kentucky bluegrass</td>
<td>1,330±40</td>
<td>1.13±0.05</td>
</tr>
<tr>
<td>(Kaye et al., 2004)</td>
<td>Fort Collins, CO</td>
<td>Irrigated/fertilized Kentucky bluegrass</td>
<td>-</td>
<td>2.4±0.3</td>
</tr>
<tr>
<td>(Dobbie and Smith, 2003)</td>
<td>Aberystwyth, UK</td>
<td>Hay field</td>
<td>-</td>
<td>27.6</td>
</tr>
<tr>
<td></td>
<td>Grange-over-sands, UK</td>
<td>Hay field</td>
<td>-</td>
<td>4.7</td>
</tr>
<tr>
<td></td>
<td>Boxworth, UK</td>
<td>Winter wheat</td>
<td>-</td>
<td>1.2</td>
</tr>
</tbody>
</table>
Figure 3.1: Map showing the areas with the greatest concentrations of turfgrass as shaded areas. Reproduced from Milesi et al. (2005).
Figure 3.2: Layout (not to scale) of low maintenance (Zoysia, endemic polystand - light green) and high maintenance (tall fescue, Kentucky bluegrass – dark green) plots. Due to poor establishment, the Zoysia treatment was removed from this study (and re-established for future work). The circles indicate the location of GHG rings. Each plot is 18.2m square, with 18.2m runways in between.
Figure 3.3: Average soil-to-atmosphere fluxes (± S.E.) of carbon dioxide for three turfgrass systems (tall fescue, Kentucky bluegrass, and a polyspecies endemic stand) measured across the 2013 growing season.
Figure 3.4: Average soil-to-atmosphere fluxes (± S.E.) of nitrous oxide for three turfgrass systems (tall fescue, Kentucky bluegrass, and a polyclimate endemic stand) measured across the 2013 growing season.
Figure 3.5: Average soil-to-atmosphere fluxes (± S.E.) of ammonia for three turfgrass systems (tall fescue, Kentucky bluegrass, and a polyppecies endemic stand) measured across the 2013 growing season.
Figure 3.6: Average soil temperature (A) and soil moisture (B) (± S.E.) for the three turfgrass systems (tall fescue, Kentucky bluegrass, and a polyspecies endemic stand) that were measured concurrently with GHG flux measurements. Precipitation values were recorded by the UK Agricultural Weather Center (2013) at a weather station proximate to the experimental field. Bold values indicate significance in the ANOVA tables.
Figure 3.7: Average (± S.E.) carbon dioxide, nitrous oxide, and ammonia fluxes across the four fertilization events that occurred in the tall fescue and Kentucky bluegrass turf systems. The vertical red lines indicate the date of fertilizer application. Ticks on the x-axis indicate days prior to and post fertilization events. Asterisks indicate days that have significantly different fluxes from the day just before application for the average of both treatments except for CO₂ fluxes during the last fertilization where double asterisks are for bluegrass and single asterisks for fescue, as there was a treatment effect for this gas on this date.
Figure 3.8: Average (± S.E.) soil temperature and moisture, across the four fertilization events that were measured. The vertical red lines indicate the date of fertilizer application. Missing soil moisture data from May 22 was due to an equipment malfunction. Ticks on the x-axis indicate days prior to and post fertilization event.
Chapter 4: Conclusions & Future Directions

Grasslands are important in Kentucky as they support animal production, provide recreation, and conserve biodiversity. Understanding their response to climate change and potential to mitigate future changes in climate is important to the state and the nation. This thesis examines two components but additional work remains. I performed experiments to assess the effects of climate change on pasture slug herbivory and the contribution of home lawns systems to soil-to-atmosphere greenhouse gas fluxes.

I found that climate change, specifically warming and increased precipitation, did not significantly modify the plant herbivory by slugs in hayed pastures of central Kentucky, despite altering their abundance. There were very few significant effects of slug exclusion on plant biomass or plant community dynamics. It appears that climate change will alter the seasonal pattern of slug abundance, as well as potentially overall abundance, through both direct and indirect effects. Warming had a negative effect on slug abundance during summer and fall, in part through changes in the plant community that reduced forage quality for slugs, but warming significantly increased slug abundance and, moderately, increased herbivory in the winter and early spring, most likely through direct effects of increased temperature on the slugs themselves. The dominant slug species was *Deroceras reticulatum*, which is a cosmopolitan species known for its wide range of adaptability. The low levels of slug herbivory measured in this study suggest that the primary ecological role of this organism may be something other than grazing above ground plant material. Hay producers in the region may be more impacted by warming-related changes in plant community dynamics and forage quality than climate
change alterations in slug herbivory. Future research could identify what role, if any, slugs play in these hay pastures.

Despite predicting that high maintenance lawns would have greater soil-to-atmosphere GHG emissions than low maintenance due to inputs of fertilizer and irrigation, as well as species differences, I found that three common central Kentucky home lawn systems, with differing management regimes, did not differ in greenhouse gas emissions. I also found that the addition of irrigation and fertilization did not significantly stimulate the total annual emission of CO₂, N₂O, and NH₃ over those lawns that did not receive these inputs. The magnitude of total GHG emissions from these systems was greater than similar systems, which was likely due to methodology bias. There were also minimal greenhouse gas emission increases associated with individual applications of fertilizer, which was likely due to the low rates of summer urea application and the use of slow release urea for the fall application. It is possible that the overall effects of lawn type and management regime were muted by the unusually wet year of the study and/or the fact that the treatments have only been in place for one year. This work has shown that current homeowner fertility types and rates likely minimize the emission of greenhouse gasses associated with fertilization events. Future work in the same experiment will further explore the effects of lawn type and management on the overall sustainability of home lawns in Kentucky and assess the carbon balance of all three lawn types to better determine their ultimate contribution towards sustainability.

This research examined two grassland systems and specific factors within them that interact with climate change with the broad goal of improving the understanding of climate change effects and improving sustainability. I found that, in general, the
responses to climate and management alterations in pasture and turf systems respectively were more resistant to change than I anticipated. As noted in other studies, inter-annual variation can be a greater source of variation than treatment differences, so it is imperative that studies be continued for at least several years to account for natural variation in the local climate. Ultimately we hope to be able to understand the responses of these grassland systems to climate and management manipulations in order to improve the climate change mitigation potential of grasslands in Kentucky and worldwide.
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Published Abstracts
“Influence of Climate change on Slug Herbivory in a Central Kentucky Tall Fescue Pasture” with Dr. Rebecca McCulley and Dr. James Harwood (coauthors) – presented at:
   - University of Kentucky Invasion Biology, Ecology, and Management Conference, Spring 2013
   - Tracy Farmer Institute for Sustainability Annual Research Showcase, Fall 2013

“Soil-to-Atmosphere Greenhouse Gas Emissions from High- and Low-Input Turf Systems of Central Kentucky” with Dr. David Williams and Dr. Rebecca McCulley (coauthors) – presented at:
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