THESIS

SIMULATED CARBON AND NITROGEN DYNAMICS IN TURFGRASS SYSTEMS USING THE DAYCENT MODEL

Submitted by

Yao Zhang

Department of Horticulture and Landscape Architecture

In partial fulfillment of the requirements

for the Degree of Master of Science

Colorado State University

Fort Collins, Colorado

Fall 2012

Master's Committee:

Advisor: Yaling Qian

William J. Parton
Anthony J. Koski

ABSTRACT

SIMULATIONS OF CARBON AND NITROGEN DYNAMICS IN TURFGRASS SYSTEMS USING THE DAYCENT MODEL

Ecosystem modeling offers an opportunity to better understand the carbon and nitrogen dynamics in a certain ecosystem. Modeling provides a way for researchers to expand their research to larger scales or other situations where field measurements are difficult or costly to conduct. In this study, the DAYCENT ecosystem model was parameterized and validated under home lawn conditions. Long-term effects of irrigation and fertilization on turfgrass quality, soil carbon and nitrogen sequestration, and nitrous oxide (N_2O) emissions were investigated. The DAYCENT model was also used as a tool to develop best management practices (BMPs) for a Kentucky bluegrass lawn.

Clipping yields, evapotranspiration (ET), deep percolation, nitrate leaching, and soil temperature of a Kentucky bluegrass lawn were simulated and compared with the measured values from a three-year lysimeter study. Parameters that control damping factors of soil temperature and nitrate leaching rate were modified to reflect the unique properties of turfgrass ecosystems. The prediction of weekly ET and deep percolation of the three years was acceptable ($r > 0.6$). The simulated clipping yield was improved compared to the monthly time step CENTURY ecosystem model, with the r value increased from -0.32 to 0.74.

Modeled N_2O emissions were validated for Kentucky bluegrass (*Poa pratensis* L.) and perennial ryegrass (*Lolium perenne* L.). The annual cumulative N_2O emissions predicted by the DAYCENT model were close to the measured emission rates of Kentucky bluegrass sites in

Colorado (within 16% of the observed values). For the perennial ryegrass site in Kansas, the DAYCENT model overestimated the N_2O emissions for all treatments by about 200% (urea and ammonium sulfate at high rate and urea at low rate). After including the effect of biological nitrification inhibition (BNI) in the root exudate, the DAYCENT model properly simulated the N_2O emissions for all treatments (within 8% of the observed values).

After calibration and validation, the DAYCENT model was further used to predict best management practices (best irrigation and nitrogen fertilization rates) for a Kentucky bluegrass lawn. Irrigation that decreases from 100% potential evapotranspiration (PET) to 60% PET is predicted to reduce 50-percent of annual net production in the semi-arid region. The model simulation suggested that gradually reducing fertilization as the lawn ages from 0 to 50 years would significantly reduce long-term nitrate leaching and N_2O emissions when compared to applying nitrogen at a constant rate (at 150 kg N ha^{-1} yr^{-1}). Our simulation indicates that a Kentucky bluegrass lawn could change from a sink to a weak source of greenhouse gas (GHG) emissions about 20 to 30 years after establishment.

ACKNOWLEDGMENTS

I would like to thank my adviser Dr. Yaling Qian, who gave me immense intellectual support during my Master’s degree study. Without her advice and comments, I cannot image how I could have gotten this research accomplished. Not only supporting my study, Dr. Qian is also a respectable and trustworthy friend, who has taught me many things about life, career, and family. Also, I would like to thank Dr. William Parton and Dr. Anthony Koski for serving on my committee. The critical comments they provided are very helpful. Thanks are also due to Mr. Brent Mecham, Dr. Dale Bremer, and Dr. Jason Kaye for sharing their field study data with me. I would also like to thank Ms. Sarah Wilhelm and Ms. Cindy Keough for providing technical support.

TABLE OF CONTENTS

LIST OF TABLES

Table

LIST OF FIGURES

Figures

CHAPER 1. DEVELOPMENT OF BEST TURFGRASS MANAGEMENT PRACTICES USING THE DAYCENT MODEL

1.1 SUMMARY

To predict the best management practices for lawns in Colorado, the DAYCENT ecosystem model was parameterized and applied on a turfgrass ecosystem. In this study, the daily time step DAYCENT model was parameterized and validated using field-measured data on clipping yields, evapotranspiration (ET), deep percolation, nitrate leaching, and soil temperature from a three-year lysimeter study. The prediction of ET and deep percolation was acceptable for the three years ($r > 0.6$). The simulation result for clipping yield was improved compared to the monthly time step CENTURY ecosystem model, with the r value increased from -0.32 to 0.74. The long-term irrigation effect on Kentucky bluegrass (*Poa pratensis* L.) biomass and soil carbon and nitrogen was also examined. We predicted a 50-percent drop in the annual net production as irrigation decreases from 100% potential evapotranspiration (PET) to 60% PET in this semi-arid region. The simulation result suggests that the annual fertilization rates should be gradually reduced to approximately half of the initial rates after 10 years for both moderately and highly managed lawns, and that the rates could be further reduced with increasing age of the turfgrass stand.

1.2 INTRODUCTION

Turfgrasses are extensively used in United States urban landscapes, including in residential, commercial, and institutional lawns, parks, sports fields, and golf courses. For example, in Larimer County (montane forests excluded), Colorado, urban lawns occupy 6.4 percent of the land area (Kaye et al., 2004). In the continental United States, turfgrass area is estimated to be 163,800 km^2, which is three times larger than that of any irrigated crop (Milesi et al., 2005). As a result of urbanization, which has proceeded at unprecedented rates and extents during the past few decades in the United States, large areas of crop, forest, and native vegetation are being converted into urbanized landscapes (Milesi et al., 2005; Golubiewski, 2006). Urbanization is predicted to continue over the next 15 years (Alig et al., 2004). Accompanying this growth and development is likely to be a rapid increase in urban areas of turfgrass.

Turfgrass ecosystems require intensive management; and to maintain a high-quality lawn, individual homeowners and landscape managers must provide both irrigation and fertilizer input. As such, outdoor water use accounts for about 55 percent of the residential water use in urban areas along the Colorado Front Range each year, most of which is used in landscapes (Waskom and Neibauer, 2010). In addition, the fertilizer used on urban turfgrass is estimated at rates similar to or exceeding those of cropland systems (Law et al., 2004). These human inputs—water and fertilizer—appear to alter the storage and fluxes of carbon and nitrogen, with possible influence on carbon sequestration, greenhouse gas budget, nitrate leaching, and air and water quality in urban and suburban areas, as well (Petrovic, 1990; Qian et al., 2003; Kaye et al., 2004). Computer modeling, as research illustrates, is one of the best ways to study the middle- to long-term (10 to >100 years) carbon and nitrogen dynamics as affected by the impact of irrigation and fertilization management.

1.2.1 CENTURY and DAYCENT Models

The CENTURY model is a monthly step ecosystem model that has been parameterized and used to simulate turfgrass ecosystems in golf course and home lawn conditions (Bandaranayake et al., 2003; Qian et al., 2003). Originally, the CENTURY model was designed for simulations of medium- to long-term changes in soil organic matter, plant productivity, and other ecosystem parameters. In the golf course study, the CENTURY simulated data compared well with measured, long-term soil organic carbon (SOC) data, with age ranging from one to 45 years with a coefficient of determination (R^2) of 0.67 for fairways and 0.78 for putting greens (Bandaranayake et al., 2003). Regarding home lawn conditions, the CENTURY model correctly simulated annual cumulative clipping yields over three years, although the seasonal trend was mis-timed (Qian et al., 2003). Despite its usefulness, monthly time step has its limitations in simulating turfgrass management. In contrast to crop management practices, turfgrass management practices, such as mowing and irrigation, are usually conducted on a weekly or daily basis. To develop useful best management practices (BMPs) for turfgrass, a daily time-step model is therefore ideal; a finer time-step model could also provide short-term (<10 years) predictions, which are more desirable for turfgrass managers and homeowners.

The DAYCENT model (Parton et al., 1998; Del Grosso et al., 2001) is the more recently developed, daily time-step version of the CENTURY model (Parton et al., 1987; Parton et al., 1993; Parton et al., 1994). The DAYCENT model uses a finer time scale than the CENTURY model in modeling decomposition, nutrient flows, soil water, and soil temperature, and has increased spatial resolution for soil layers. The key submodels of the DAYCENT model are plant production, soil organic matter (SOM) decomposition, soil water and temperature dynamics, and trace gas fluxes. The DAYCENT model has been well-validated and successfully applied to various ecosystems and locations in the world (Del Grosso et al., 2005; Pepper et al., 2005; Li et al., 2006). The newer, advanced features of the DAYCENT model now give us an opportunity to simulate and predict daily carbon and nitrogen dynamics in turfgrass ecosystems.

The goal of this study is to evaluate different management strategies to improve sustainability in human-dominated turfgrass ecosystems using the DAYCENT model. The objectives of this study were:

1) To parameterize and validate the DAYCENT model under home lawn conditions;
2) To predict the long-term impacts of different management practices (irrigation and fertilization) on primary productivity, carbon sequestration, and nitrogen leaching.
3) To develop BMPs for a Kentucky bluegrass lawn to reduce fertilization requirements, water requirements, and nitrate leaching.

1.3 MATERIALS AND METHODS

1.3.1 Field Experiment

Field data for model parameters modification and validation came from a three-year study of an eight-year-old Kentucky bluegrass (*Poa pratensis* L.) site in the Colorado Front Range. In this experiment, three on-site bucket lysimeters were established a year before the experiment to measure turfgrass water use and nitrate leaching.

The lysimeters were made from polyvinyl chloride (PVC) tubes and measured 30.5 cm in diameter and 80 cm deep. A 1.3-cm thick PVC drainage plate was installed, which separated the tube into two compartments. The compartment at the top of the lysimeters measured 60 cm in depth, and the compartments at the bottom measured 20 cm in depth. Below each drainage plate was an air space connected to a one-cm diameter PVC air inlet tube open to the air at the top of the lysimeter. Soil cores, which fit the lysimeters, were excavated and carefully filled into the top compartment. Lysimeters were buried into the soil, and the turfgrass canopies in the lysimeters were maintained at the same level as turfgrass surrounding the lysimeters. A soil temperature sensor was placed at 15 cm below the soil surface outside each of the lysimeters. Adjacent to each lysimeter, two precipitation gauges were recessed into the ground to measure the amount of precipitation each location received from rainfall and from applied irrigation.

Irrigation was applied through an automatic sprinkler system, based on the estimated reference evapotranspiration for mowed grass using the Kimberly-Penman equation (Allen et al., 1989) with a coefficient of 0.8 for turfgrass, which was recommended by “Reference Evapotranspiration Calculator version 2.0” (Allen, 1990). Mixtures of urea and sulfur-coated urea were applied at 187, 122, and 164 kg N ha^{-1} yr^{-1}, respectively, in the three years (Table 1.1). The turfgrass was mowed weekly at 5.1 cm using a mower with a mulching deck to return clippings. The clippings from three 1 m × 18.3 m strips were collected each week and weighed. Sub-samples from each of these collections were oven-dried, and dry weight was determined; the dried clippings were subsequently analyzed for nitrogen content using the Kjeldahl nitrogen method (Watkins et al., 1987). On a similar site with the same management, verdure biomass was measured in June and July.

Cumulative deep percolation was collected from lysimeters and measured weekly. All samples were refrigerated at 4 °C and subsequently analyzed for nitrates using a Cd-Reduction Method (U.S. EPA, 1979). The lysimeters were lifted from the soil and weighed each week during the growing season, both before and after deep percolations were drained off into a graduated cylinder. Weekly cumulative ET was calculated using the weight difference between the current and preceding weeks for each bucket lysimeter and corrected for precipitation, irrigation applications, and leachate.

$$ET = Precipitation+Irrigation-Percolation+(Weight_{t-1}-Weight_t)$$

$Weight_{t-1}$ is the weight of lysimeters at the beginning of a measurement week.

$Weight_t$ is the weight of lysimeters at the end of a measurement week following collecting percolation.

1.3.2 Simulation of the Field Study

We parameterized the DAYCENT model to simulate the field study. Parameters, including the ratio of carbon allocation above ground and below ground; lignin content; belowground carbon to nitrogen ratio; and decomposition rates of soil organic matter, were obtained from Qian et al. (2003). Minimum aboveground C/N ratio has been set as 9.6, which was measured from clippings of lawns in Colorado (Golubiewski, 2006). Our field observed soil temperature and nitrate leaching data were used to modify the DAYCENT model. Based on the field study, the damping factor coefficient for calculating soil temperature by layer was increased to 0.0045. The parameters controlling nitrate leaching were reduced (fleach1 and fleach3 were set 0.1 and 0.2, respectively). Observed clipping yield, leaf nitrogen content, ET and deep percolation rates, which were not used for parameterization, were compared with simulated results to validate the model. To assess the model performance, Pearson product-moment correlation coefficient (r) was calculated for comparisons.

Information on weather, soil, and management practices was needed for conducting simulations. Daily maximum/minimum temperatures and precipitation in our study were obtained from the weather station on site. The soil was a Fort Collins, Colorado, loam (fine-loamy, mixed, superactive, mesic Aridic Haplustalf; 29% clay, 54% sand, and 17% silt).

In the study, we simulated turfgrass management practices of fertilization, irrigation, and mowing. The turfgrass site received consistent management both before and during the period of the field experiment; we simulated the previous land use as cropland and the turfgrass growth for eight years with high maintenance. To model the effect of slow-release nitrogen, sulfur-coated urea was simulated as several applications of quick-release fertilizer, according to the release rate described by Salman, et al. (1989). Urea was modeled as the input of NH_4^+ into soil; ammonia volatilization loss in the process of converting urea to NH_4^+ by urease was not considered since volatilization loss is usually small under common conditions (Petrovic, 1990). Irrigation was assumed to be rain events in this simulation as the version of the DAYCENT model we used in this study cannot otherwise schedule irrigation water on a daily basis. Mowing is a unique management practice for turfgrass and was treated as a harvest event in our DAYCENT model (Qian et al., 2003). In our simulation, mowing was scheduled at the same date as that in the field experiment. Each mowing was simulated as leaving approximately 3366 kg ha^{-1} aboveground live biomass, which is the average of observed verdure biomass, and removing the rest.

1.3.3 Long-term Simulations

After the simulation for the three years, long-term (50 years since turfgrass establishment) simulations were then conducted for the field experimental site using fifty-year daily weather data (from 1961 to 2010) recorded for Fort Collins, Colorado (National Climatic Data Center, weather station no. 53005). In these simulations, we predicted the influences of long-term management practices (irrigation and fertilization) on the carbon and nitrogen dynamics. Firstly, we compared the effect of three irrigation levels while applying fertilization at a same constant rate of 90 kg N ha^{-1} yr^{-1}. Irrigation rates were based on model predicted potential evapotranspiration (PET), which was calculated by using a modified Penman-Monteith equation, and at 60%, 80%, 100% PET replacement rates. Irrigation was scheduled every three days. Secondly, we predicted the effect of long-term fertilization at two constant rates with same irrigation of 100% PET replacement. Fertilization rates of 90 kg N ha^{-1} yr^{-1} and 150 kg N ha^{-1} yr^{-1} for medium- and high-quality turfgrasses, as suggested by Koski and Skinner (2011), was applied equally in April, May, and October for 50 years.

To estimate the long-term effect of management on turfgrass quality, aboveground net primary productivity (ANPP), which has been suggested to indicate turfgrass quality, was used as indication in the present study (Qian et al., 2003; Walker et al., 2007). Kaye et al. (2005) measured annual ANPP of 1800 kg C ha^{-1} yr^{-1} for low- to medium-quality lawns in Colorado. Aboveground net primary productivity of a medium- to high-quality lawn was found to be 2800 kg C ha^{-1} yr^{-1} by Qian et al. (2003). We also added leaf nitrogen content as another indicator for turfgrass quality. Leaf nitrogen content was found closely related to the greenness of turfgrass (Rodriguez and Miller, 2000). Rodriguez and Miller (2000) suggest that a leaf with three percent to four percent nitrogen content corresponds to a quality rating of 4 to 7 out of 10, which represents a medium-quality turfgrass. The DAYCENT model outputs include the monthly average shoot carbon to nitrogen ratio. We converted this ratio to nitrogen content, assuming the carbon content of dried biomass is 43 percent (Kaye et al., 2005; Golubiewski, 2006).

1.3.4 Developing BMPs

Based on the results of long-term simulations, irrigation and fertilization rates were adjusted to select BMPs for the Kentucky bluegrass experimental site. Best irrigation and fertilization rates were selected to maintain turfgrass at high and medium turf quality. First, best levels of irrigation were selected. Then, we conducted many runs using different levels of nitrogen input and picked up the minimal nitrogen rate, which results in the desired annual productivity and leaf nitrogen content. The criteria for a medium-quality lawn included an annual ANPP of at least 1800 kg C ha^{-1} yr^{-1} and a growing season mean leaf nitrogen content of at least 3 percent (Kaye et al., 2005). A high-quality lawn was defined as a lawn with an annual ANPP of at least 2800 kg C ha^{-1} yr^{-1} and leaf nitrogen content above 4 percent (Rodriguez and Miller, 2000; Qian et al., 2003). The best nitrogen rates were predicted by the DAYCENT model for two scenarios: clipping fully returned and 50-percent returned. Nitrogen fertilizer was simulated as NH_4^+ type and applied three times each year.

1.4 RESULTS AND DISCUSSION

1.4.1 Measured and Simulated Results

Measured data showed that soil temperature at 15 cm depth closely related to the change of air temperature but with less fluctuation (Fig. 1.1). Measured soil temperature was used for DAYCENT parameterization. After modifying the damping factor coefficient for calculating soil temperature, the DAYCENT model properly simulated the observed trends in soil temperature with a Pearson product-moment correlation coefficient (r) 0.68 (Fig. 1.1). The characteristics of the soil under turfgrass cover changed after land conversion from agricultural use. The modification of the damping factor for soil temperature was because of the thatch layer, which has the effect of insulation. Thatch, a tightly intermingled layer of dead and living stems and roots, is known to develop between the soil surface and the zone of green vegetation of Kentucky bluegrass (Beard, 1973). The value of damping factor coefficient for calculating soil temperature for turfgrass was similar to that used for forests with a floor organic layer.

Our field data showed very low nitrate leaching for the three years (0.03, 0.08, and 0.03 g N m^{-2} yr^{-1} in Year 1, 2, and 3, respectively). This is in agreement with previous studies that have shown that turfgrass has a low potential for leaching, considering the rate of fertilization applied every year (Petrovic, 1990; Miltner et al., 1996; Easton and Petrovic, 2004; Barton and Colmer, 2006). Since the default parameters controlling nitrogen leaching were derived from cropland experiments, we used our field experiment results to modify these parameters. Simulated nitrate leaching was reduced to the level of measured values after the modification of leaching parameters (Fig. 1.2). One source of uncertainty, which was difficult to assess, was the effect of soil disturbance caused by the transfer of soil into the lysimeters.

Field observed ET, deep percolation, clipping yield, and clipping nitrogen content were used to validate the DAYCENT model. Measured ET in growing seasons totaled 73.8, 73.8, and 69.4 cm for the three years, respectively. The simulated total ET rates are close to the measured values, 73.7, 71.7, and 68.0, respectively. In all three years, highest weekly ET in each growing season was observed in June and July (Day 150 to 200, Fig. 1.3). An unusually high weekly ET rate of approximately 5 cm was observed in May of Year 2, close to the highest rate in June and July of that year. The DAYCENT model predicted similar trends for weekly ET; the correlation coefficients were 0.85, 0.69 and 0.61, respectively (Fig. 1.3). In addition, the DAYCENT model predicted deep percolation with an overall Pearson’s r value of 0.84 (Fig. 1.4).

Clipping yield reflects the aboveground growth of turfgrass. We observed substantially higher clipping yield in Year 1 than the other two years (Fig 5a). The high yield was a result of the vigorous growth of Kentucky bluegrass in the spring of Year 1; clippings of 1990 kg ha^{-1} were collected in May, which are 235% and 150% of those in the same month of Year 2 and

Year 3, respectively. The DAYCENT model simulated the annual clipping yields close to measured values with deviation of no more than nine percent (Fig.5a). The DAYCENT model also predicted the general seasonal trend of growth with an overall correlation coefficient of r= 0.74 (Fig. 1.5b, c, and d) compared to a correlation coefficient of -0.32 of the simulation using the CENTURY model (Qian et al., 2003). This is partially because, unlike the CENTURY model, the DAYCENT model included photoperiod effects on growth, which indicates that growth will slow in the fall as the day length decreases. One weakness of this simulation is the assumption that verdure biomass is constant during the growing season. However, variation in verdure biomass has been observed in different months of the same year, with fertilization and irrigation as factors in variation (Falk, 1980).

The observed nitrogen content of shoots appeared to increase from spring to fall during the growing season (Fig. 1.6). The low nitrogen content in the spring can be explained by the high biomass produced in this period, or the dilution of leaf nitrogen (Kaye et al., 2005). Simulated annual average nitrogen contents were 4.3 percent, 4.4 percent, and 4.4 percent, which are comparable to the measured averages of 3.6 percent, 4.3 percent, and 4.3 percent for the three years, respectively.

The DAYCENT model properly simulated the seasonal change of turfgrass ET, percolation, and clipping yield and provided acceptable annual rates. After validation, we used the DAYCENT model to predict long-term effect of irrigation and fertilization management on turfgrass quality, carbon and nitrogen sequestration, and nitrate leaching.

1.4.2 The Influence of Different Management Regimes in Long–term Simulations

Irrigation plays an important role in turfgrass management, especially in semi-arid areas, such as Colorado, where water is the primary limiting factor for turfgrass growth. The effect of replacing 100 %, 80 %, and 60 % PET on ANPP is shown in Fig.7. The output indicated that irrigation that replaced 100 % PET results in the highest ANPP. Irrigation at 60 % PET reduced ANPP, which can be an indicator of turfgrass quality, by about half in most years, compared to 100 % PET.

Fu et al. (2004) found that the visual quality of Kentucky bluegrass was greatly reduced as irrigation was decreased from 100 % ET to 20 % ET in Kansas. Within the same level of irrigation, the year-to-year difference of ANPP was mainly due to air and soil temperatures, since high temperatures significantly limit the growth of cool-season grass (Watschke et al., 1972; Youngner and Nudge, 1976; Aldous and Kaufmann, 1979). The ANPP of medium-quality lawns in Colorado is reportedly about 1800 kg C ha^{-1} yr^{-1} (Kaye et al., 2005), while the ANPP of a medium- to high-quality lawn is approximately 2800 kg C ha^{-1} yr^{-1} (Qian et al., 2003). Our simulation suggests that irrigation at the amount of 60 % to 100 % PET irrigation is required for Kentucky bluegrass turfgrass grown in Colorado to achieve an acceptable ANPP. In years with more favorable temperatures and rainfall for turfgrass growth, 60 % PET irrigation could support medium- to high-quality lawns with ANPP of approximately 2700 kg C ha^{-1} yr^{-1}.

Nitrogen fertilization is also critical for maintaining high turfgrass quality. With an irrigation level of 100 % PET, high nitrogen fertilization typically results in higher productivity (up to 23 %) in 25 years following turfgrass establishment (Fig. 1.8a). However, as turfgrass ages, the requirement for nitrogen fertilizer gradually reduces. There is no difference of predicted ANPP for the two nitrogen rates after 25 years. Leaf nitrogen content was predicted to be significantly affected by nitrogen fertilization. In the first 10 years after establishment, 90 kg N ha^{-1} yr^{-1} resulted in average nitrogen content of 3.2 percent, corresponding to low to medium turf quality (Fig. 1.8b). Annual average nitrogen content increases to 4percent if turfgrass was fertilized for 20 years with 90 kg N ha^{-1} yr^{-1}. In contrast, fertilized with 150 kg N ha^{-1} yr^{-1}, turfgrass was predicted to be of high quality (leaf nitrogen content above 4 percent) about 3 years after establishment.

The DAYCENT model predicted that increased irrigation results in higher soil carbon sequestration rates (Fig. 1.9a). Soil carbon sequestration is a function of plant productivity and SOM decomposition (Parton et al., 1987). About 8.8 Mg C ha^{-1} more carbon was sequestered when irrigation was increased from 60 % PET to 100 % PET across 50 years. Similarly, more soil organic nitrogen was sequestered under long-term irrigation at 100 % PET (Fig. 1.9b). Compared to low nitrogen input, high nitrogen input did not dramatically increase carbon sequestration, but significantly increased the soil organic nitrogen content (Fig. 1.9a and b).

With continuous fertilization of 90 kg N ha^{-1} yr^{-1}, the annual nitrate leaching prediction was very low for 35 years under all irrigation scenarios (Fig. 1.9c). After 35 years, nitrate leaching began to increase. The total nitrogen leaching amounts across 50 years were 97, 107, and 341 kg N ha^{-1} in our simulation for 60 %, 80 %, and 100 % PET irrigation, respectively, when fertilized with 90 kg N ha^{-1} yr^{-1} . Using 150 kg N ha^{-1} yr^{-1} constantly with 100 % PET irrigation, we predicted the nitrate leaching to remain low for 10 to 15 years, then to rise substantially (Fig. 1.9c).

1.4.3 Best Management Practices Generated by DAYCENT Model

Based on the findings in the long-term simulations, we developed BMPs for the lawn to maintain high quality and medium quality. Irrigation levels of 80 % PET and 100 % PET were selected for medium and high turf quality, respectively. The best nitrogen rates, as the DAYCENT model predicts, for high quality and medium quality are described for clipping fully-returned (Fig.10a) and clipping 50-percent returned (Fig. 1.10b) scenarios. A high-quality lawn required approximately twice the amount of fertilizer used to maintain medium quality every year for the first two decades. A newly established lawn required nitrogen fertilizer of 240 and 120 kg N ha^{-1} yr^{-1} to maintain high quality and medium quality with clippings fully returned, respectively. The simulation indicated that two to three years after establishment, the fertilization rate could be reduced to 140 kg N ha^{-1} yr^{-1} and 80 kg N ha^{-1} yr^{-1}. In addition, the rate could be further reduced to about 50 kg N ha^{-1} yr^{-1} at 40 to 50 years. Clipping removal represented a large of amount of nitrogen loss from soil (Heckman et al., 2000; Qian et al., 2003). Under the scenario of returning 50 percent of clippings, lawns required 30 to 80 kg N ha^{-1} yr^{-1} more nitrogen fertilizer than that of the clippings fully-returned scenario (Fig. 1.10b). Studies have suggested that lawns may act as a nitrogen sink, because large amounts of nitrogen were sequestrated into SOM (Porter et al., 1980; Higby and Bell, 1999). Under turfgrass cover, nitrogen immobilization would likely continue for decades until reaching equilibrium between immobilization and mineralization; the soil organic nitrogen (SON) sequestration rate decreased to a minimum after decades as equilibrium was reached (Porter et al., 1980; Higby and Bell, 1999). Our simulations showed a decreasing trend of SON sequestration rate, but it would take more than 50 years for this lawn to reach equilibrium (Fig. 1.11b).

Model output suggested that a highly managed lawn had higher carbon sequestration rate (1.13 Mg C ha^{-1} yr^{-1}) than a medium-quality lawn, which showed a rate of 0.78 Mg C ha^{-1} yr^{-1} in the first 10 years after conversion from agricultural land. Approximately 10 Mg ha^{-1} more carbon (37 percent) can be sequestered by high-quality turfgrass than medium-quality turfgrass in 50 years (Fig. 1.11a). Although we did not have long-term data to validate the effect of different management regimes on SOC, these rates are within the reported range for turfgrass ecosystems. Our simulated carbon sequestration rate was 1.13 Mg C ha^{-1} yr^{-1} for highly managed turfgrass within 10 years after conversion from cropland, which is similar to the rate of 1.0 Mg C ha^{-1} yr^{-1} in golf courses in Colorado as Qian and Follett reported (2002). This is also consistent with the rate of 0.69 Mg C ha^{-1} yr^{-1} over 40 years in golf courses in New Zealand (Huh et al., 2008). However, Selhorst and Lal (2011) estimated the mean rate of carbon sequestration in fairways in Ohio as 3.55 Mg ha^{-1} yr^{-1}, which is more than two times higher than the other two golf course studies. It is likely that this study included thatch layer as SOC, since the measured SOC content in the 0 to 2.5 cm soil layer reached 16 percent. Currently, different studies vary regarding whether or not to include thatch layer in determining SOC. In studies of Qian and Follett (2002) and Qian et al. (2010), the thatch layer was not included in the soil carbon sequestration calculation. Regarding home lawns, Golubiewski (2006) has studied the carbon pools of 53 sites in Colorado’s Front Range. The SOC in 0-20 cm soil that was established with turfgrass for 50 years averaged 48 Mg ha^{-1}(i.e. 0.96 Mg ha^{-1} yr^{-1}) which is similar to our prediction. Regarding nitrate leaching, the DAYCENT model predicted that that the BMP for high-quality turfgrass resulted in a higher annual nitrate leaching rate (due to high nitrogen input) than the BMP for medium-quality turfgrass (Fig. 1.11c). Nevertheless, this rate of nitrate leaching was significantly lower than the conventional constant fertilization of 150 kg ha^{-1} yr^{-1} for high-quality turfgrass (Fig.8c).

To apply the predicted best nitrogen rates for broader use, we should consider site-to-site differences. Nitrogen fate is affected by soil property; we expect best nitrogen rates would vary for soils with different texture and SOM levels. Ammonia volatilization nitrogen loss is difficult to assess since loss rate could vary due to type of fertilizer, management, and environmental conditions; reported loss ranges from nearly zero to 60% of the total applied nitrogen (Titko et al., 1987; Petrovic, 1990; Wood et al., 2007). In our simulations, ammonium-type fertilizer was used and assumed to be properly managed without significant nitrogen loss through volatilization. Additionally, our experimental site was not planted with trees; more nitrogen and water could be needed for sites with trees present, as trees have extensive root systems and compete for nutrients and water with grasses. In summary, as turfgrass management practices vary and there are differences in climate and soil properties, we suggest that the best nitrogen input rates can be determined by simulations with site-specific information.

1.5 CONCLUSIONS

Compared to the CENTURY model, turfgrass management simulated by the daily time-step DAYCENT model was more in line with actual practices. The performance of the DAYCENT model on Kentucky bluegrass lawns is acceptable and reliable. Our results show that the DAYCENT model has the potential for use as a tool to predict best management practices by using site-specific information. The DAYCENT model predicted that 80 % PET irrigation can maintain a medium-quality lawn in Colorado with average precipitation, and this rate can be further reduced to 60 % PET in years with more favorable temperature and rainfall for cool-season grass growth. The nitrogen fertilization rates should be reduced to approximately half of the initial rates after 10 years of establishment for both medium- and high-quality Kentucky bluegrass lawns. Our simulation showed that the best DAYCENT-predicted nitrogen fertilization rates could greatly reduce nitrate leaching, compared to conventional, constant nitrogen fertilization rates over a long term. To apply the DAYCENT model to broader use on turfgrass, we suggest the model be tested in various other turfgrass ecosystems and climates.

Table 1.1. Fertilizer application schedule in field experiment.

	April	**May**	**June**	**July**	**August**	**October**	**Total**
	kg N ha^{-1}						
Year 1	52†		40	43		52	187
Year 2		31			43	48	122
Year 3	82				82		164

†Urea and sulfur-coated urea were in a ratio of 1:1, except in Year 3, which was a ratio of 1:2.

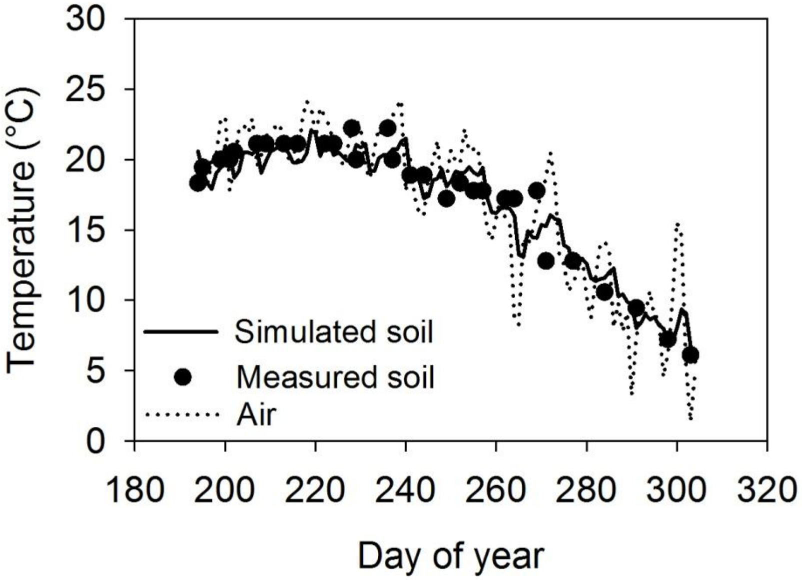

Fig. 1.1: Comparison of measured and simulated soil temperature at 15-cm depth.

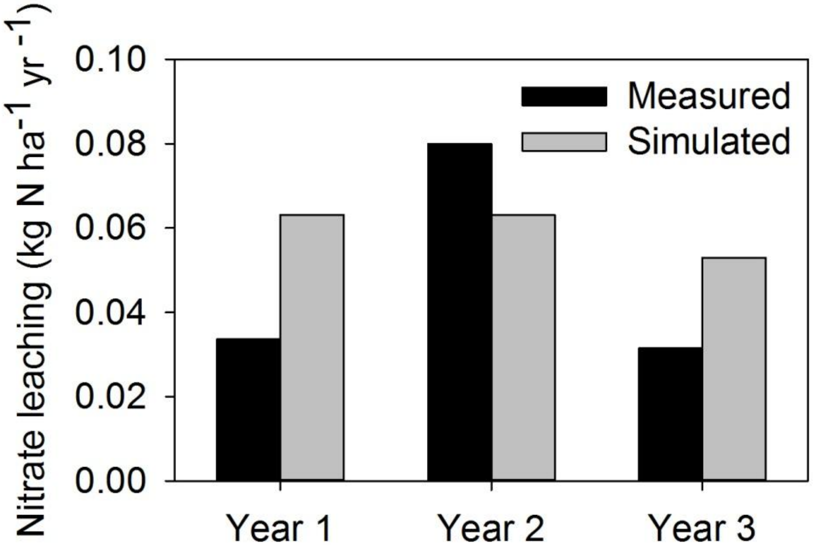

Fig. 1.2: Comparison of measured and simulated annual nitrate leaching after the modification of the parameters that control nitrate leaching.

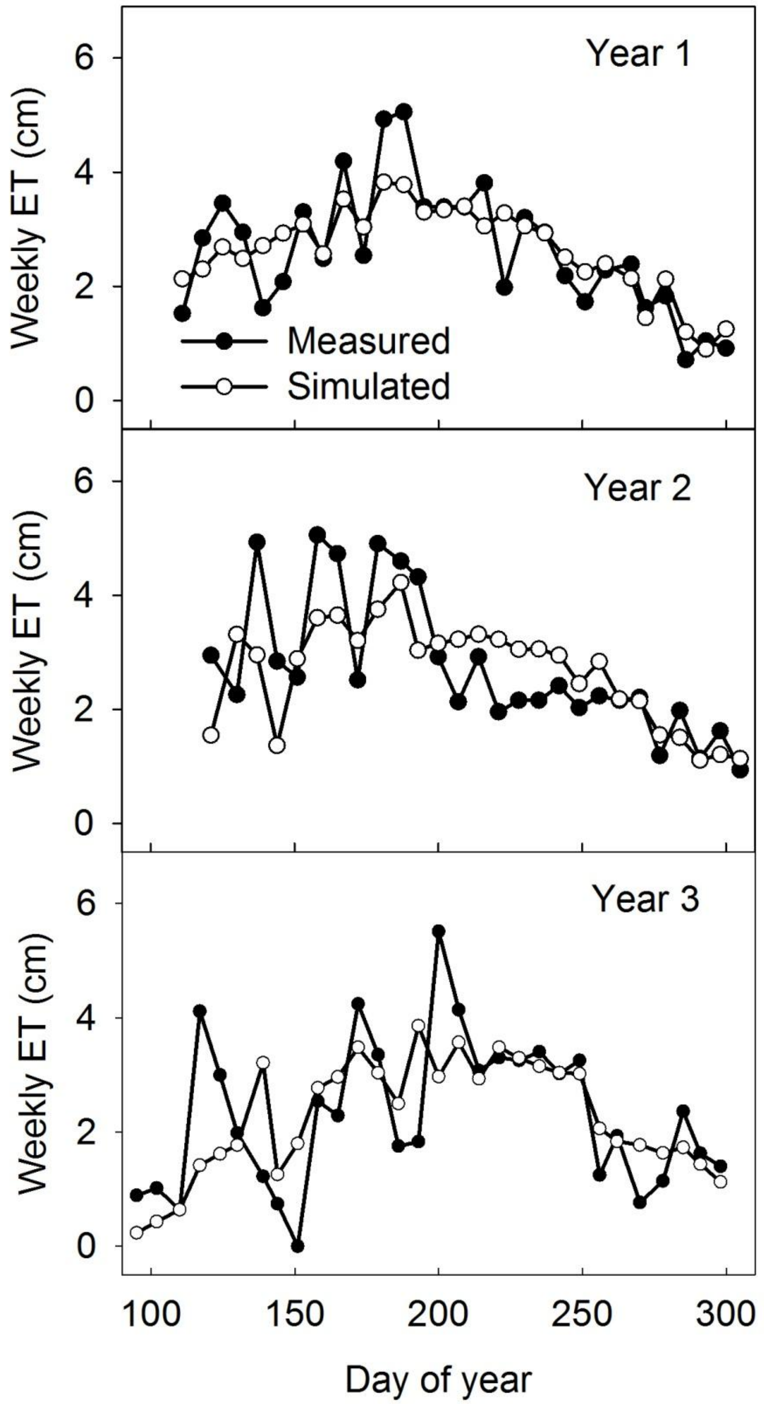

Fig. 1.3: Comparison of measured and simulated weekly evapotranspiration (ET) for three years.

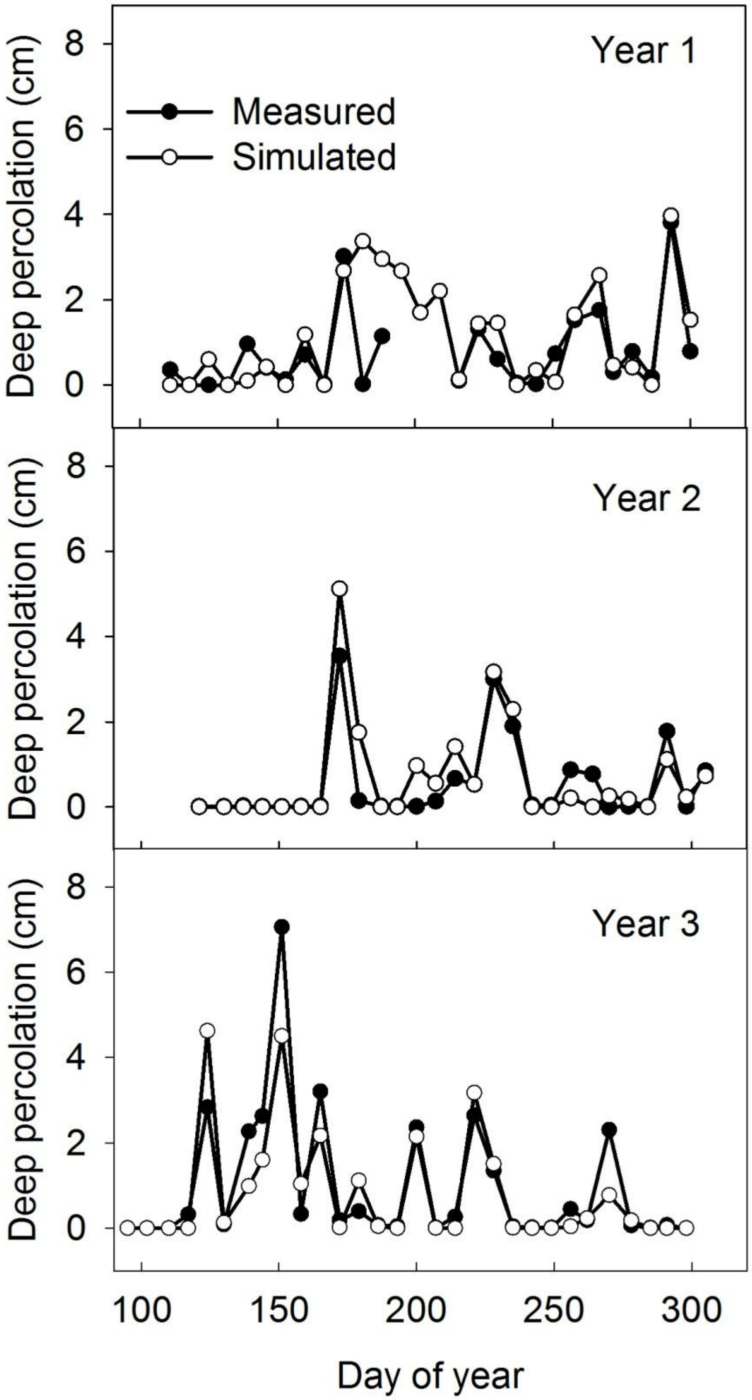

Fig. 1.4: Comparison of measured and simulated weekly deep percolation for three years.

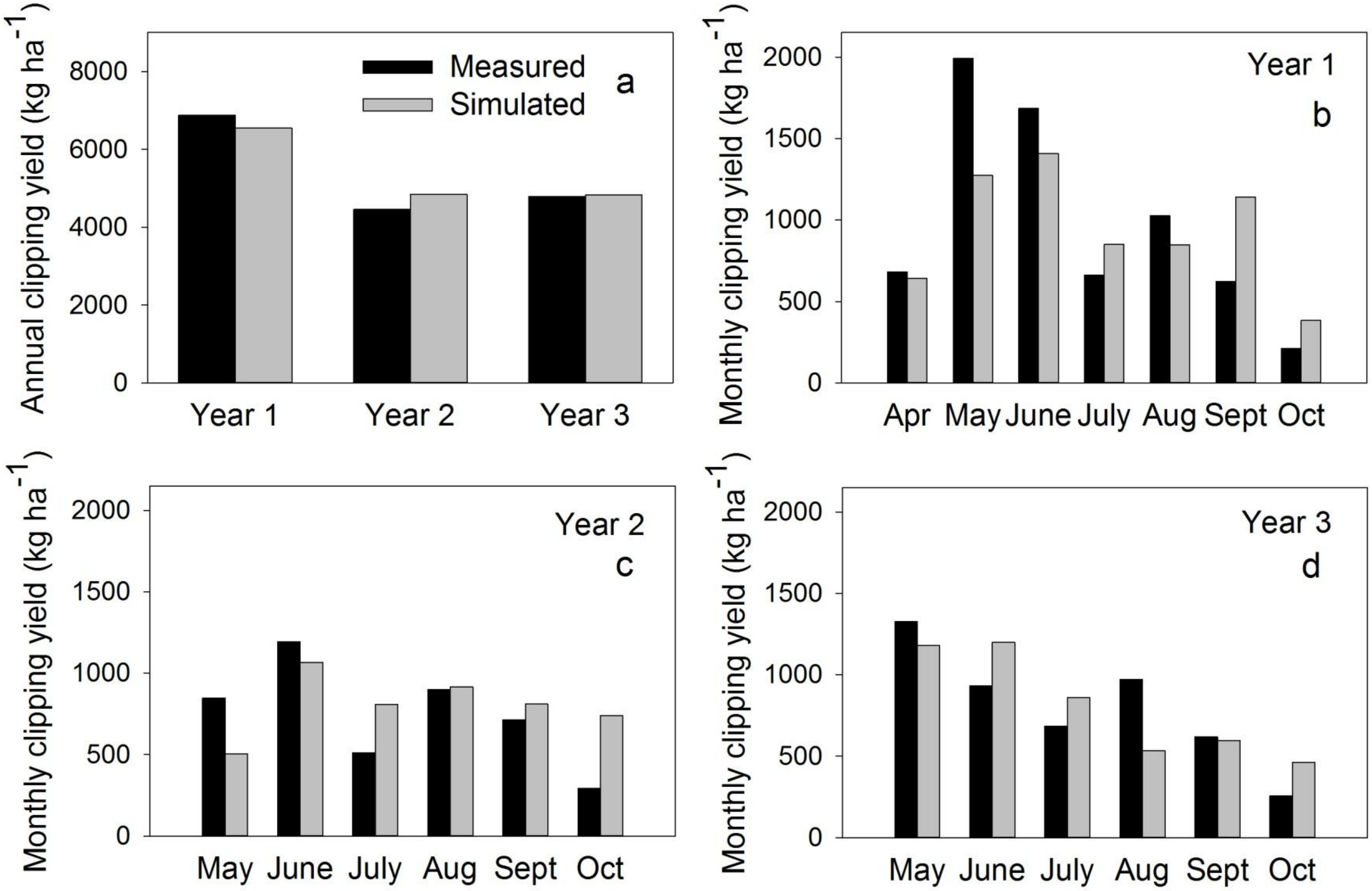

Fig. 1.5: Comparison of measured and simulated annual and monthly cumulative clipping yields.

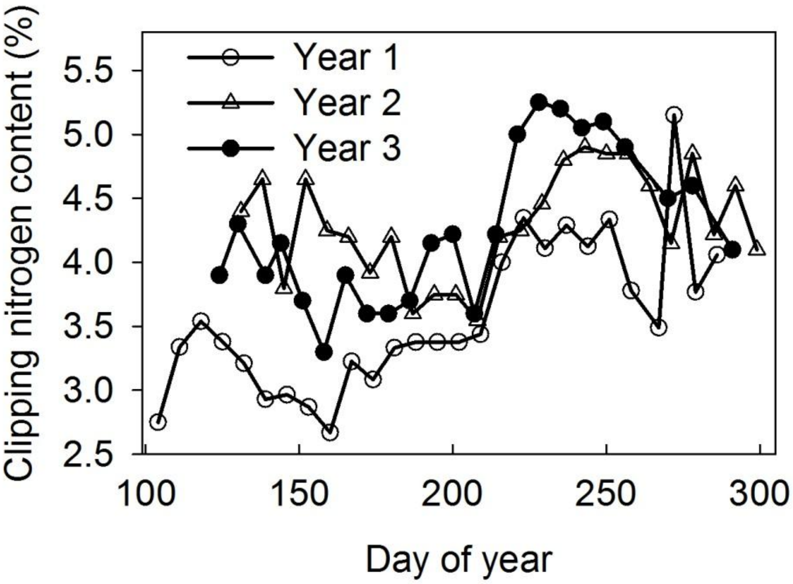

Fig. 1.6: Measured nitrogen content of mown clippings.

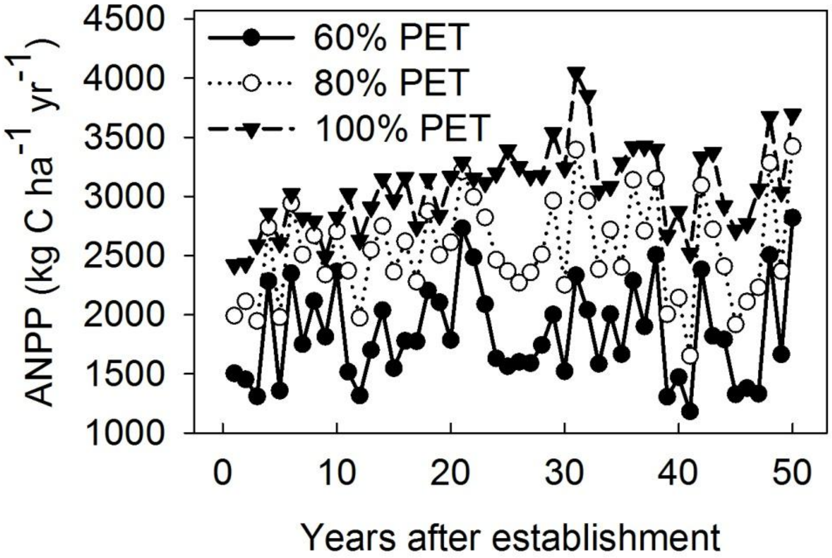

Fig. 1.7: Model simulated annual aboveground net primary productivity (ANPP) for three irrigation levels. Fertilizer was applied at 90 kg N ha^{-1} yr^{-1}. Irrigation was applied through growing season.

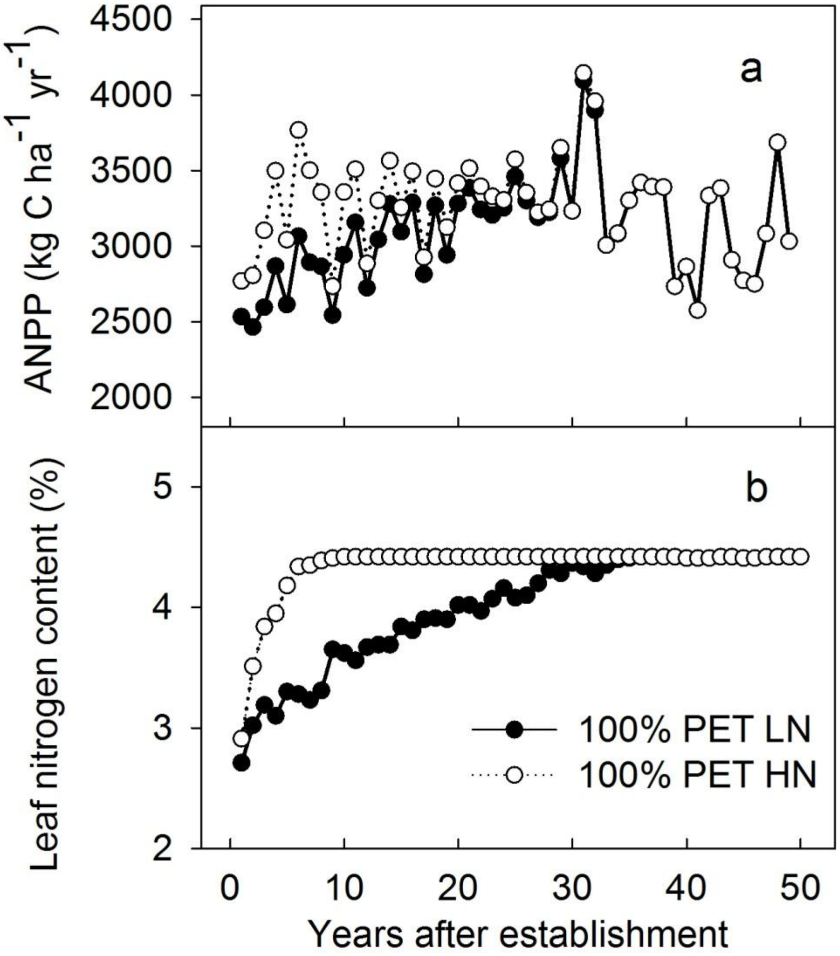

Fig. 1.8: Model simulated (a) ANPP and (b) leaf nitrogen content for a lawn receiving fertilizer of 150 kg N ha^{-1} yr^{-1} (HN) and 90 kg N ha^{-1} yr^{-1} (LN).

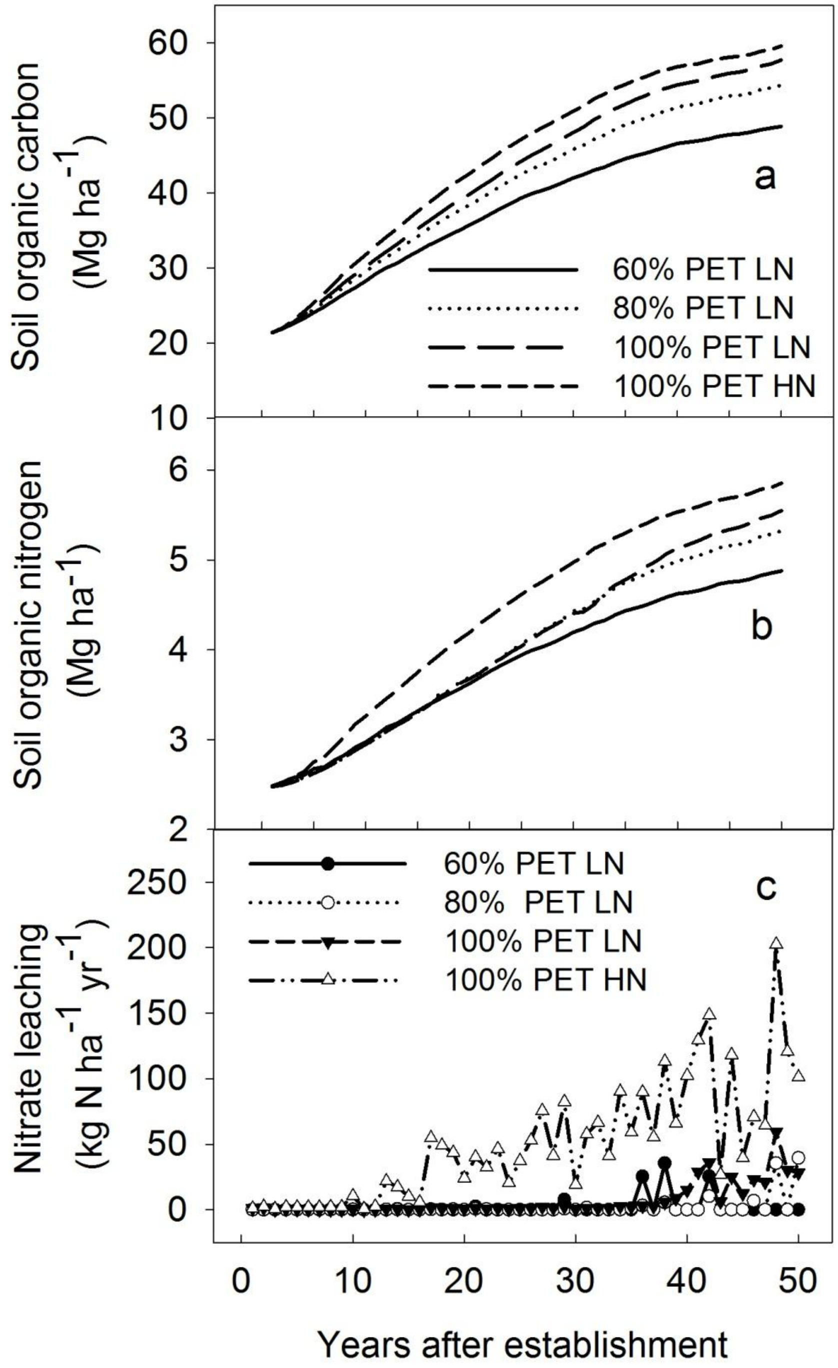

Fig. 1.9: Model simulated (a) soil organic carbon, (b) soil organic nitrogen, and (c) nitrate leaching for three irrigation levels with fertilizer of 90 kg N ha^{-1} yr^{-1} (LN) and 100 % potential evapotranspiration (PET) irrigation with fertilizer of 150 kg N ha^{-1} yr^{-1}N (HN). Fertilizer was applied for 50 years with clippings returned.

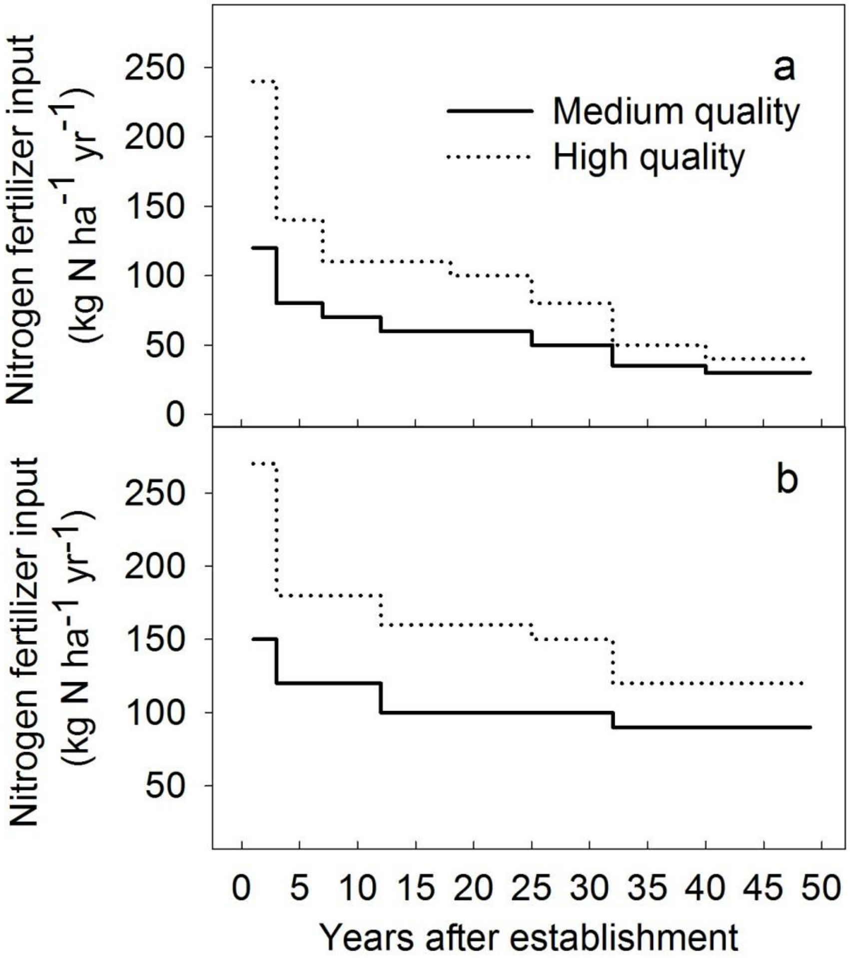

Fig. 1.10: Model predicted best nitrogen input rates for medium- and high-quality lawns with (a) clippings fully returned and (b) 50-percent clippings returned.

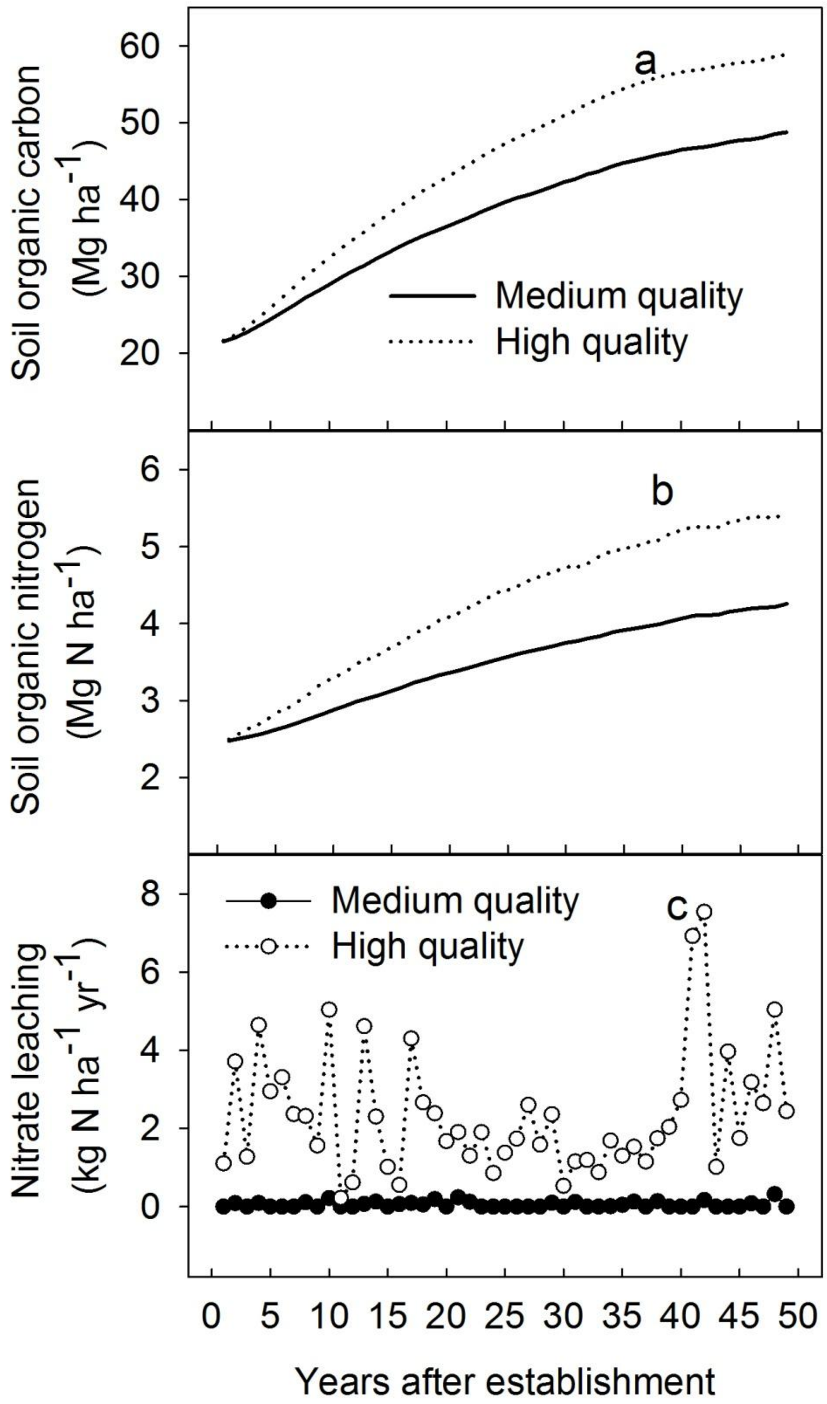

Fig.1.11: Model predicted (a) soil organic carbon, (b) soil organic nitrogen, and (c) nitrate leaching of a lawn using model-predicted best nitrogen input rates.

REFERENCES

Aldous, D.E., and J.E. Kaufmann. 1979. Role of root temperature on shoot growth of two Kentucky bluegrass cultivars. Agron. J. 71:545-547.

Alig, R.J., J.D. Kline, and M. Lichtenstein. 2004. Urbanization on the US landscape: looking ahead in the 21st century. Landscape and Urban Planning. 69:219-234.

Allen, R.G. 1990. REF-ET: Reference Evapotranspiration Calculator version 2.0. Utah State University, Logan, Utah.

Allen, R.G., M.E. Jensen, J.L. Wright, and R.D. Burman. 1989. Operational estimates of reference evapotranspiration. Agron. J. 81:650-662.

Bandaranayake, W., Y.L. Qian, W.J. Parton, D.S. Ojima, and R.F. Follett. 2003. Estimation of soil organic carbon changes in turfgrass systems using the CENTURY model. Agron. J. 95:558-563.

Barton, L., and T.D. Colmer. 2006. Irrigation and fertilizer strategies for minimising nitrogen leaching from turfgrass. Agricultural Water Management. 80:160-175.

Beard, J.B. 1973. Turfgrass: Science and culture. Prentice Hall, Englewood Cliffs, NJ. p. 658.

Del Grosso, S.J., A.R. Mosier, W.J. Parton, and D.S. Ojima. 2005. DAYCENT model analysis of past and contemporary soil N_2O and net greenhouse gas flux for major crops in the USA. Soil Tillage Res. 83:9-24.

Del Grosso, S.J., W.J. Parton, A.R. Mosier, M.D. Hartman, J. Brenner, D.S. Ojima, et al. 2001. Simulated interaction of carbon dynamics and nitrogen trace gas fluxes using the DAYCENT model. In: Shaffer M.J., Ma L. and H. S., editors, Modeling carbon and nitrogen dynamics for soil management. Lewis Publishers, Boca Raton, FL, USA. p. 303-332.

Easton, Z.M., and A.M. Petrovic. 2004. Fertilizer source effect on ground and surface water quality in drainage from turfgrass. J. Environ. Qual. 33:645-655.

Falk, J.H. 1980. The primary productivity of lawns in a temperate environment. J. Appl. Ecol. 17:689-695.

Fu, J., J. Fry, and B. Huang. 2004. Minimum water requirements of four turfgrasses in the transition zone. HortScience 39:1740-1744.

Golubiewski, N.E. 2006. Urbanization increases grassland carbon pools: effects of landscaping in Colorado's Front Range. Ecological Applications. 16:555-571.

Heckman, J.R., H. Liu, W. Hill, M. DeMilia, and W.L. Anastasia. 2000. Kentucky bluegrass responses to mowing practice and nitrogen fertility management. Journal of Sustainable Agriculture. 15:25-33.

Higby, J.R., and P.F. Bell. 1999. Low soil nitrate levels from golf course fairways related to organic matter sink for nitrogen. Commun. Soil Sci. Plant Anal. 30:573-588.

Huh, K.Y., M. Deurer, S. Sivakumaran, K. McAuliffe, and N.S. Bolan. 2008. Carbon sequestration in urban landscapes: the example of a turfgrass system in New Zealand. Soil Research. 46:610-616.

Kaye, J.P., I.C. Burke, A.R. Mosier, and J. Pablo Guerschman. 2004. Methane and nitrous oxide fluxes from urban soils to the atmosphere. Ecological Applications. 14:975-981.

Kaye, J.P., R.L. McCulley, and I.C. Burke. 2005. Carbon fluxes, nitrogen cycling, and soil microbial communities in adjacent urban, native and agricultural ecosystems. Global Change Biology. 11:575-587.

Koski, T., and V. Skinner. 2011. Lawn care. Colorado State University Extension. http://www.ext.colostate.edu/pubs/Garden/07202.html (accessed 20 Sept. 2012).

Law, N., L. Band, and M. Grove. 2004. Nitrogen input from residential lawn care practices in suburban watersheds in Baltimore county, MD. Journal of Environmental Planning and Management. 47:737-755.

Li, X., T. Meixner, J. Sickman, A. Miller, J. Schimel, and J. Melack. 2006. Decadal-scale dynamics of water, carbon and nitrogen in a California chaparral ecosystem: DAYCENT modeling results. Biogeochemistry. 77: 217-245.

Milesi, C., S. Running, C. Elvidge, J. Dietz, B. Tuttle, and R. Nemani. 2005. Mapping and modeling the biogeochemical cycling of turf grasses in the United States. Environ. Manage. 36:426-438.

Miltner, E.D., B.E. Branham, E.A. Paul, and P.E. Rieke. 1996. Leaching and mass balance of ^{15}N-labeled urea applied to a Kentucky bluegrass turf. Crop Sci. 36:1427-1433.

Parton, W.J., M. Hartman, D. Ojima, and D. Schimel. 1998. DAYCENT and its land surface submodel: description and testing. Global and Planetary Change. 19:35-48.

Parton, W.J., D.S. Ojima, C.V. Cole, and D.S. Schimel. 1994. A general model for soil organic matter dynamics: sensitivity to litter chemistry, texture and management. SSSA special publication. 39:147-167.

Parton, W.J., D.S. Schimel, C.V. Cole, and D.S. Ojima. 1987. Analysis of factors controlling soil organic matter levels in great plains grasslands. Soil Sci. Soc. Am. J. 51: 1173-1179.

Parton, W.J., J.M.O. Scurlock, D.S. Ojima, T.G. Gilmanov, R.J. Scholes, D.S. Schimel, et al. 1993. Observations and modeling of biomass and soil organic matter dynamics for the grassland biome worldwide. Global Biogeochem. Cycles. 7: 785-809.

Pepper, D.A., S.J. Del Grosso, R.E. McMurtrie, and W.J. Parton. 2005. Simulated carbon sink response of shortgrass steppe, tallgrass prairie and forest ecosystems to rising CO_2, temperature and nitrogen input. Global Biogeochem. Cycles. 19:GB1004.

Petrovic, A.M. 1990. The fate of nitrogenous fertilizers applied to turfgrass. J. Environ. Qual. 19:1-14.

Porter, K.S., D.R. Bouldin, S. Pacenka, R.S. Kossack, and C.A. Shoemaker. 1980. Studies to assess the fate of nitrogen applied to turf: Part I: Research project technical complete report. Project A-086-NY report to Office of Water Research and Technology. U.S. Department of the Interior, Washington, D.C., USA.

Qian, Y., W. Bandaranayake, W.J. Parton, B. Mecham, M.A. Harivandi, and A.R. Mosier. 2003. Long-term effects of clipping and nitrogen management in turfgrass on soil organic carbon and nitrogen dynamics: the CENTURY model simulation. J. Environ. Qual. 32:1694-1700.

Qian, Y., and R.F. Follett. 2002. Assessing soil carbon sequestration in turfgrass systems using long-term soil testing data. Agron. J. 94:930-935.

Qian, Y., R.F. Follett, and J.M. Kimble. 2010. Soil organic carbon input from urban turfgrasses. Soil Sci. Soc. Am. J. 74:366-371.

Rodriguez, I.R., and G.L. Miller. 2000. Using a chlorophyll meter to determine the chlorophyll concentration, nitrogen concentration, and visual quality of St. Augustine grass. HortScience 35:751-754.

Salman, O.A., G. Hovakeemian, and N. Khraishi. 1989. Polyethylene-coated urea. 2. Urea release as affected by coating material, soil type and temperature. Ind. Eng. Chem. Res. 28:633-638.

Selhorst, A.L., and R. Lal. 2011. Carbon budgeting in golf course soils of Central Ohio. Urban Ecosystems. 14:771-781.

Titko, S., J.R. Street, and T.J. Logan. 1987. Volatilization of ammonia from granular and dissolved urea applied to turf grass. Agron. J. 79:535-540.

U.S. EPA. 1979. Methods for chemical analysis of water and wastes. U.S. Environmental Protection Agency, Washington, D.C.

Walker, K.S., C.A. Bigelow, D.R. Smith, G.E. Van Scoyoc, and Z.J. Reicher. 2007. Aboveground responses of cool-season lawn species to nitrogen rates and application timings. Crop Sci. 47:1225-1236.

Waskom, R., and M. Neibauer. 2010. Water conservation in and around the home. Colorado State University Extension. http://www.ext.colostate.edu/pubs/consumer/09952.html (accessed 20 Sept. 2012).

Watkins, K.L., T. Veum, and G.F. Krause. 1987. Total nitrogen determination of various sample types: a comparison of the Hach, Kjeltec, and Kjeldahl methods. Journal of the Association of Official Analytical Chemists. 70:410-412.

Watschke, T.L., R.E. Schmidt, E.W. Carson, and R.E. Blaser. 1972. Some metabolic phenomena of Kentucky bluegrass under high temperature. Crop Sci. 12:87-90.

Wood, C., E.C. Knight, and E.A. Guertal. 2007. Mowing and nitrogen source effects on ammonia volatilization from turfgrass. Crop Sci. 47:1628-1634.

Youngner, V.B., and F.J. Nudge. 1976. Soil temperature, air temperature, and defoliation effects on growth and nonstructural carbohydrates of Kentucky bluegrass. Agron. J. 68:257-260.

CHAPTER 2. SIMULATION OF N_2O EMISSIONS AND ESTIMATION OF GLOBAL WARMING POTENTIAL IN TURFGRASSES USING THE DAYCENT MODEL

2.1 SUMMARY

Nitrous oxide (N_2O) emissions are an important component of the greenhouse gas (GHG) budget for turfgrasses. To estimate N_2O emissions and the global warming potential (GWP), the DAYCENT ecosystem model was parameterized and applied to turfgrass ecosystems. The annual cumulative N_2O emissions predicted by the DAYCENT model were close to the measured emission rates of Kentucky bluegrass (*Poa pratensis* L.) sites in Colorado (within 16% of the observed values). For the perennial ryegrass (*Lolium perenne* L.) site in Kansas, the DAYCENT model initially overestimated the N_2O emissions for all treatments by about 200% (urea and ammonium sulfate at high rate and urea at low rate). After including the effect of biological nitrification inhibition (BNI) in the root exudate of perennial ryegrass, the DAYCENT model correctly simulated the N_2O emissions for all treatments (within 8% of the observed values). After calibration and validation, the DAYCENT model was further used to simulate carbon sequestration and N_2O emissions of a Kentucky bluegrass lawn under a series of management regimes. The model simulation suggested that gradually reducing fertilization as the lawn ages from 0 to 50 years would significantly reduce long-term N_2O emissions by approximately 40% when compared to applying nitrogen at a constant rate (at 150 kg N ha^{-1} yr^{-1}). Our simulation indicates that a Kentucky bluegrass lawn could change from a sink to a weak source of GHG emissions about 20 to 30 years after establishment.

2.2 INTRODUCTION

Global warming is predicted to continue in the next several decades as currently there is no effective way for mitigation (IPCC, 2007). The increase in anthropogenic greenhouse gas (GHG) concentrations is likely the main cause of observed increase in global average temperatures (IPCC, 2007). Ecosystems have been found to play important roles in GHG emissions. For example, agriculture, in whole, is estimated to account for 13.5% of the total global GHG emissions by IPCC (2007). However, some ecosystems, such as native vegetation could serve as a sink for GHG emissions (Del Grosso et al., 2005). Turfgrass, which is a unique highly managed ecosystem, occupies large area in urbanized land. The role of turfgrass in GHG budget is still not clear. In Larimer County, Colorado turfgrass occupies 6.4 percent of the land area (Kaye et al., 2004). In the continental United States, turfgrass area is estimated to be three times larger than that of any irrigated crop (Milesi et al., 2005). Since urban area is expected to expand rapidly in the USA in the next a few decades, turfgrass may contribute more to the total GHG budget (Alig et al., 2004; Kaye et al., 2004).

Few studies have been conducted on calculating the net global warming potential (GWP) for turfgrasses. The major components of GWP of lawns may include: a) GHGs from soil; b) energy costs associated with turfgrass maintenance, and c) soil carbon sequestration. Greenhouse gases emitted from soil are mainly carbon dioxide (CO_2), nitrous oxide (N_2O), and methane (CH_4) (IPCC, 2007). Energy costs of turfgrass maintenance include manufacturing and transporting fertilizer and pesticides, using electricity for treating and transporting irrigation water, and combusting fuel in mowing. Soils could mitigate GHG emissions by carbon sequestration. Currently, it is not clear whether a turfgrass ecosystem plays a role as a net source or sink of GHG emissions. Intensive management is reported to increase soil carbon sequestration rate (Qian et al., 2003). However, in the meantime, management increases the energy cost from the use of irrigation water and fertilizer. Additionally, applying nitrogen fertilizers leads to increased nitrous oxide (N_2O) emissions (Bremer, 2006; Groffman et al., 2009; Zirkle et al., 2011). One study conducted in California has shown that turfgrasses serve as either sources or sinks of global warming depending on fertilization rates (Townsend-Small and Czimczik, 2010). The estimated GWPs for ornamental lawns with low (100 kg N ha^{-1} yr^{-1}) and high (750 kg N ha^{-1} yr^{-1}) fertilizer input were -108 g CO_2 m^{-2} yr^{-1} and 286 g CO_2 m^{-2} yr^{-1}, respectively. However, the fertilization rate of 750 kg N ha^{-1} yr^{-1} in this experiment is considered extremely high and rarely used in turfgrass industry (Law et al., 2004).

Nitrous oxide is a GHG, which is estimated to have 298 times the GWP of CO_2 over a 100 year period (IPCC, 2007). Total global N_2O emissions were estimated to account for 7.9% of the anthropogenic GHGs in terms of CO_2-equivalent (IPCC, 2007). Regarding agriculture, 2.1 (0.4-3.8) Tg N of N_2O was emitted directly from agricultural soils each year, which is approximately 12% of the total N_2O emissions from the biosphere (Mosier et al., 1998; Albritton et al., 2001). The recorded emissions from frequently fertilized turfgrasses range from 0.5 to 6.4 kg N ha^{-1} yr^{-1} (Guilbault and Matthias, 1998; Kaye et al., 2004; Bremer, 2006; Groffman et al., 2009; Livesley et al., 2010; Townsend-Small and Czimczik, 2010). In a previous study, urban lawns in Colorado were reported to emit greater amounts of N_2O than those of wheat and native grassland (Kaye et al., 2004). Lawns in California were observed to emit N_2O at rates comparable to corn fields and greater than those of vegetable fields (Townsend-Small et al., 2011).

N_2O fluxes result from microbial activities in soil. Both soil nitrification and denitrification processes can emit N_2O. Nitrification is the oxidation of NH_4^+ to NO_2^- and the oxidation of NO_2^- to NO_3^- by microbial populations. Nitrification is optimal in aerobic conditions since this oxidation requires O_2 as the terminal electron acceptor (Bateman and Baggs, 2005). Denitrification rises dramatically when soil is under anaerobic conditions; heterotrophic bacteria and fungi reduce NO_3^- and NO_2^- to N_2O or N_2 when O_2 is limiting (Bateman and Baggs, 2005). Nitrous oxide emissions are closely related to several environmental factors. One is soil nitrogen availability. Nitrogen availability can be increased by applying nitrogen fertilizer, which provides substrate (NH_4^+ or /and NO_3^-) for nitrification and denitrification. Fluxes of N_2O rise significantly after nitrogen fertilization (Kaye et al., 2004; Bremer, 2006; Townsend-Small and Czimczik, 2010). Soil moisture, which could alter the oxygen status of soil, is another factor that plays an important role in the composition of N_2O from nitrification and denitrification (Smith et al., 1998; Dobbie et al., 1999). Additionally, soil temperature and labile organic carbon have been also identified to influence N_2O fluxes (Parton et al., 2001).

In aerated soils, the majority of N_2O emissions are from nitrification. As nitrification coverts immobile NH_4^+ to mobile NO_3^-, which is susceptible to leaching out of the soil profile, some species of plants have been found to suppress nitrification by releasing nitrification inhibitory compounds from their roots (Munro, 1966). Recently, this phenomenon was intensively studied and termed biological nitrification inhibition (BNI). Using sensitive bioassay, BNI was successfully identified and quantified (Subbarao et al., 2009). The BNI capacity appears to be a relatively widespread phenomenon in tropical pasture grasses but also was found in C3 plants (Subbarao et al., 2007). Compared to tropical grass *Brachiaria* *humidicola*, which showed a near total suppression of nitrification, relatively low degree of BNI was found in root exudates of Italian ryegrass (*Lolium multiflorum* Lam.) in the study. In an earlier soil incubation experiment, Moore and Waid (1971) have observed that root washing of perennial ryegrass (*Lolium perenne* L.) exhibited pronounced and persistent effects in reducing nitrification (80% reduction). Reduction in nitrification could result in decrease in N_2O emissions from the process. If fertilized with NH_4^+ type of fertilizer, soil with BNI present is expected to emit less N_2O from both nitrification and denitrification, since the substrate of denitrification (NO_3^-) is the product from the process of nitrification. In a field experiment over 3 years, *Brachiaria* pastures were reported to suppress soil N_2O emissions by >90%; two other pasture grasses that have a low to moderate level of BNI capacity suppressed N_2O emissions by about 50% (Subbarao et al., 2009).

Methane (CH_4) is another GHG that has 21 times the GWP of CO_2 over a 100 year period (IPCC, 2007). Soil could either uptake or emit $CH_{4,}$ depending on soil properties and conditions (Brady and Weil, 2008). Aerobic soils have been found to be sinks of CH_4 because bacteria oxidize CH_4 as an energy and carbon source in non-saturated soils (Dutaur and Verchot, 2007). Water-saturated systems like wetlands and paddy soils (rice fields) that facilitate microbial production of CH_4 are sources (Ojima et al., 1993; Chan and Parkin, 2001). There are few studies conducted on CH_4 emissions in turfgrass ecosystems. Kaye et al. (2004) found that lawns in Colorado could uptake CH_4 at a rate of 0.15 g C m^{-2} yr^{-1}, which is half that of native grasslands. Groffman and Pouyat (2009) observed lawns as either weak net sinks or weak sources of CH_4 in a five-year experiment; the uptake rates of CH_4 in lawns were significantly lower than those in forests.

Soil carbon sequestration has been seen as a way to mitigate GHG emissions (IPCC, 2007). Compared to croplands, turfgrass receives few disturbances like tillage which facilitate the decomposition processes and thus has high rate of carbon sequestration (Qian and Follett, 2002). Turfgrass ecosystems are highly managed; both root and shoot productivity increase after the conversion of native grassland or agricultural land to urban turfgrass (Falk, 1976; Falk, 1980). The residue of the dead biomass would add to soil organic carbon (SOC) pools soil. Qian and Follett (2002) reported that golf fairways could sequestrate SOC at a rate of 1.0 Mg ha^{-1} yr^{-1} during the first 0 to 25 years after turfgrass establishment in Colorado. After examining 53 lawns in Colorado's Front Range, Golubiewski (2006) found that lawn grass produced more biomass and stored more C than local prairie or agricultural fields. Lawns within Denver (>25 years of age) were reported to have almost 2-fold higher SOC densities than in shortgrass steppe soils (Pouyat et al., 2009).

In recent years, experiments of estimating net GWPs from various ecosystems have been conducted. The difficulty of assessing GWPs by conducting short-term field experiments is that there are many components in GWP calculation and large spatial and temporal variations. One easier way to estimate trace gas emissions and GWPs for ecosystems is computer modeling. One of the widely used models is DAYCENT. The DAYCENT model is an ecosystem model, which incorporates the most recent improvements in the understanding of soil carbon and nitrogen dynamics (Parton et al., 2001). The DAYCENT model has been used to simulate trace gas emissions for major crops in the United States and to estimate the net GWP (Del Grosso et al., 2005).

The DAYCENT model has the ability to simulate fluxes of N trace gases. Although the correlation between simulated and observed N_2O flux was found poor on a daily basis in some simulations, the DAYCENT model was able to reproduce soil textural and treatment differences and the observed seasonal patterns of gas flux emissions (Parton et al., 2001). The trace gas submodel has been validated using data from various ecosystems and locations in the world (Del Grosso et al., 2002; Stehfest and Müller, 2004; Del Grosso et al., 2005; Pepper et al., 2005; Li et al., 2006; Adler et al., 2007; Del Grosso et al., 2008). Currently no research is available to compare the DAYCENT-predicted N_2O flux with measured data in turfgrass lawns.

Previously, the DAYCENT model has been adjusted to simulate turfgrass ecosystems and validated using data of biomass, evapotranspiration, leaching, and soil temperature (Chapter 1). The DAYCENT model could model the seasonal trend of turfgrass growth and soil water dynamics and gives fairly good annual outputs on these parameters. The CENTURY ecosystem model, which is the previous version of the DAYCENT model, has been used to simulate SOC in both golf course and home lawn conditions (Bandaranayake et al., 2003; Qian et al., 2003). The SOC submodel has been tested using soil carbon sequestration data from golf courses (Bandaranayake et al., 2003).

In this study, we used the DAYCENT model to predict the carbon and nitrogen fluxes in a turfgrass ecosystem and estimated the GWPs by using model outputs and literature information. The objectives of this study are:

1) To validate the DAYCENT model on N_2O emissions from turfgrasses.
2) To simulate the impact of different management practices on N_2O emissions.
3) To predict the GWP for a Kentucky bluegrass lawn in Colorado.

2.3 MATERIALS AND METHODS

2.3.1 DAYCENT Model Description

The DAYCENT model was developed based on the CENTURY model, which has been widely used in simulations of medium to long term (10 to >100 years) changes in soil organic matter (SOM), plant productivity, and other ecosystem parameters for the major ecosystems in the world (Parton et al., 1987; Parton et al., 1993; Parton et al., 1994). The DAYCENT model uses daily time scale in modeling decomposition, nutrient flows, soil water, and soil temperature and has increased spatial resolution for soil layers. The main inputs of the DAYCENT model are: (1) soil texture, (2) daily weather data (maximum/ minimum air temperature and precipitation), (3) plant type, and (4) management practices (e.g., amount and timing of fertilizer applied).

N_2O emissions from both nitrification and denitrification are modeled. Modeled N_2O fluxes from nitrification are a function of soil NH_4^+ concentration, water-filled pore space (WFPS), temperature, and texture. N_2O emissions from denitrification are a function of soil NO_3^- concentration, WFPS, heterotrophic respiration, and texture. It should be noted that, in the version of the DAYCENT model used in this study, denitrification is assumed to occur only when WFPS is above ~0.55 and the rate increases exponentially when WFPS increases from ~0.55 to ~0.90. In the DAYCENT model, ammonium is assumed to be distributed only in 0-15 cm soil because of its immobility while nitrate is distributed throughout the soil profile. The SOC simulated by the DAYCENT model is within 0 to 20 cm of soil profile. The labile C availability is approximated by simulated heterotrophic respiration.

2.3.2 Simulation of the Field Experiments

In this study, measured N_2O fluxes from experiments conducted by Kaye et al. (2004) and Bremer (2006) were used to evaluate the performance of the trace gas submodel. We simulated turfgrass management including irrigation, fertilization, and mowing. Parameters for turfgrass were from Chapter 1. Soil information was obtained from soil analysis reports. Previous land use before each experiment was simulated according to land use history.

In the experiment of Kaye et al. (2004), N_2O fluxes were measured for one-year period on three turfgrass sites (one institutional lawn and two home lawns) dominated by Kentucky bluegrass (*Poa pratensis* L.) in Fort Collins, Colorado. Fluxes of N_2O from soil were estimated by using static soil chambers (Mosier et al., 1991; Mosier et al., 1997). Sampling dates were approximately twice per month during the growing season and monthly during the winter. Additional samples were collected before and after fertilization and irrigation events. Fluxes were measured between 9:00 and 13:00 which is used to represent the average flux value for the day. The annual emission rates are calculated using linear interpolation between measurement dates (Kaye et al., 2004). The soil analysis results of the sites are shown in Table 2.1. Soil temperature and soil moisture were measured on gas-sampling dates. Fertilizer applied in June and October totaled 110 kg N ha^{-1} yr^{-1}. The institutional lawn was fertilized with urea (46-0-0). The two home lawns used commercial fertilizer (25-5-5) (Jirdon Agri Chemcials, Inc, Morrill, NE). The nitrogen form in this commercial fertilizer is NH_4^+. Irrigation schedule was not documented in this experiment. The total amount of irrigation applied during the growing season is 54±4 cm/yr.

To simulate the irrigation of the three Kentucky bluegrass lawns, we divided the total amount of sprinkler irrigation from May to October into each month according to the monthly evapotranspiration predicted by the DAYCENT model to mimic the common irrigation management in this area. All three sites were converted to lawns about 60-100 years ago. To estimate the soil property affected by long-term management, turfgrass maintenance practices were modeled for 80 years. The intensity of management in previous years was adjusted according to the management history and the SOC levels measured at the time of the experiment (Table. 2.2). The daily weather data were obtained from the online data base of Colorado Climate Center (Station number: 53005, Fort Collins).

The experiment of Bremer (2006) was carried out on a perennial ryegrass (*Lolium perenne* L.) turf in Manhattan, Kansas with three fertilization treatments: urea at two rates (250 and 50 kg N ha^{-1} yr^{-1}) and ammonium sulfate at 250 kg N ha^{-1} yr^{-1}. Measurement method of nitrous oxide fluxes was similar to the Colorado experiment (Kaye et al., 2004). Samples were collected weekly and more frequent measurements were taken after fertilizer applications. The soil texture was 32% sand, 44% silt and 24% clay with pH 7.2. Total soil carbon was 3% in top 15 cm soil and we assume the total soil carbon was equal to organic carbon since pH is less than 7.4 (Schumacher, 2002). The estimated SOC in 0-20 cm soil was 7.2 Mg ha^{-1}. Mowing was conducted once or twice weekly at 7.5 cm. Irrigation was applied three times weekly to keep turfgrass from drought stress. Soil moisture and temperature at 5 cm were measured daily. Ammonium and nitrate concentrations in the top 10 cm soil were measured 4 times during growing season.

The ryegrass site has been established with turfgrass since 1960. We simulated moderate turfgrass management for 45 years. Urea and ammonium sulfate are both NH_4^+ type of fertilizers; the DAYCENT model simulated both of them as NH_4^+ added into the soil. The applications of herbicide and fungicide were not simulated. To estimate the effect of BNI, we decreased the nitrification coefficient (a multiplier on the nitrification rates; range 0 to 1.0) from 0.8 (default value) to 0.1 in a 0.1 step and compared the annual cumulative emissions with the measured values. Weather data were provided by Kansas State University Research and Extension (Website: http://www.ksre.ksu.edu/wdl/; Station ID: Manhattan).

To evaluate the model simulation effectiveness, correlation analysis was performed by comparing measured vs. simulated N_2O emissions for both Colorado and Kansas experiments. Pearson product-moment correlation coefficient (r) was calculated for the simulated daily fluxes. Annual cumulative emission rates were compared with measured values.

2.3.3 Long-term Predictions for a Kentucky Bluegrass Lawn

After validation of the DAYCENT model, we predicted the impact of long-term turfgrass management (fertilization and irrigation) on N_2O emissions on a Kentucky bluegrass lawn near Fort Collins, CO. The soil was a Fort Collins loam (54% sand, 29% clay, and 17% silt). The lawn was converted from agriculture at Year 0 of our simulation. Weather data from 1961 to 2010 were used to drive the simulations, which were obtained from National Climatic Data Center (NCDC, weather station no. 53005). First, we used 3 different levels of potential evapotranspiration (PET) replacement (60%, 80%, and 100% PET) to predict the effect of irrigation under constant fertilization of 90 kg N ha^{-1} yr^{-1} for 50 years. Irrigation was scheduled every 3 days. Then, to predict the effects of fertilization, we compared two fertilization scenarios: constant nitrogen rate of 150 kg N ha^{-1} yr^{-1} and the best nitrogen rates developed in Chapter 1 for the lawn to achieve high quality; both of the scenarios are under 100% PET irrigation. The best nitrogen rates were the predicted minimal nitrogen rates which result in a lawn with the annual aboveground net primary productivity (ANPP) of at least 2800 kg C ha^{-1} yr^{-1} and leaf nitrogen content above 4% (Rodriguez and Miller, 2000; Qian et al., 2003). The DAYCENT-generated best management practice (BMP) of nitrogen fertilization was 240 kg N ha^{-1} yr^{-1} in first 3 years after establishment, 140 kg N ha^{-1} yr^{-1} in year 3 to 6, 110 kg N ha^{-1} yr^{-1} year 7 to 17, and 50 kg N ha^{-1} yr^{-1} at 40 to 50 years after establishment. The fertilizers used in the simulation were NH_4^+ type and all clippings were left on site.

2.3.4 Global Warming Potential

We calculated GWP for the Kentucky bluegrass lawn in Fort Collins, CO in three management scenarios: 1) BMPs predicted by DAYCENT to achieve high turf quality, 2) BMPs predicted by DAYCENT to achieve medium turf quality (Chapter 1), and 3) conventional practices for high quality lawns (100% PET irrigation and fertilization of 150 kg N ha^{-1} yr^{-1}).

The output from the long-term simulations, including N_2O emission rates and net carbon sequestration rates, were converted to CO_2 equivalents and used for GWP estimation for the Kentucky bluegrass lawn. Our calculation also included the energy cost from maintenance (mowing, irrigation, fertilization, and pesticide application), which were estimated by using published data, as described below.

The Outdoor Power Equipment Institute (Sahu, 2008) estimated fuel use of 0.00094 L m^{-2} for a typical walk-behind mower. The emission of gasoline combustion is 2347.4 g CO_2 L^{-1}

(U.S. EPA, 2011). Mowing was conducted weekly in this area from April to October (28 times yr^{-1}). The annual GWP from mowing was calculated to be 61.6 g CO_2 m^{-2} yr^{-1}.

GWP_{mowing}= Gasoline use by mower *Emission from unit gasoline combustion*Number of mowing events

The irrigation water used in home lawns is mainly from city potable water supply system in the city of Fort Collins, CO. The energy cost of Fort Collins' water supply was 0.125 kWh m^{-3}, including water treatment and distribution (Tellinghuisen, 2009). The carbon dioxide emissions coefficient for electric utilities for Colorado is 929 g CO_2 kWh^{-1} (U.S. EIA, 2001). The annual total emission from treating and distributing irrigation water is estimated at 116.0 g CO_2 m^{-3}. The total amount of irrigation water was predicted by the DAYCENT model.

$GWP_{irrigation}$=Energy intensity of water supply*CO_2 emissions coefficient*Total irrigation

Estimates of GHG emissions from manufacture and transportation of nitrogen fertilizer are 3.3 to 6.6 g CO_2 g^{-1} (Lal, 2004). The conversions for phosphorus and potassium are 0.37 to 1.1 and 0.37 to 0.73 g CO_2 g^{-1} respectively (Lal, 2004). We use means of 4.8, 0.73, and 0.55 g CO_2 g^{-1} for N, P, and K in our calculation. A common lawn fertilizer ratio is 5-1-2 (Zirkle et al., 2011). Regarding pesticide application, we used the averages of annual GWP of pesticide on turfgrass (5.5 and 11.7 g CO_2 m^{-2} yr^{-1} for medium and high quality turfgrass) which are estimated by Zirkle et al. (2011).

2.4 RESULTS AND DISCUSSION

2.4.1 Simulated N_2O Emissions from Kentucky Bluegrass Lawns in Colorado

The predicted annual cumulative N_2O emissions from Kentucky bluegrass lawns were within 16% of the observed values (Fig. 2.1). The observed annual rate of N_2O emissions of the institutional lawn was approximately 42% higher than the other two lawns, which is probably due to its higher SOM content (Christensen and Christensen, 1991; Merino et al., 2004; Li et al., 2005). The simulated trends of daily fluxes are acceptable; the Pearson's r equals 0.57, 0.60, and 0.53 for Home lawn A, Home lawn B, and Institutional lawn on daily basis, respectively (Fig. 2.2). The observed and simulated results both showed that N_2O fluxes increased dramatically right after fertilization, although the DAYCENT model underestimated these peaks. The observed high fluxes in February 2001 were not simulated by the DAYCENT model, which likely resulted from the failure of simulating high soil water content at the time of soil thawing.

Although the simulation of daily N_2O fluxes needs improvement, the DAYCENT model is able to predict annual cumulative emissions from Kentucky bluegrass lawns in Northern Colorado. The DAYCENT model is an intermediate complexity biogeochemical model which only requires inputs that are relatively easy to obtain. As described by Del Grosso et al. (2008), the accuracy of the DAYCENT model on simulating daily fluxes of N_2O might be not very high compared to that of more complex mechanistic models which require much more detailed inputs. An example of highly mechanistic model is *ecosys* (Grant et al., 2006). However, it is more practical to use the DAYCENT model to assess cumulative N_2O emissions without detailed input data (Del Grosso et al., 2008).

In our simulation, nitrification is modeled as the main source of N_2O emissions for the three lawns (>93% of total N_2O). The proportion of N_2O from nitrification and denitrification is a function of O_2 availability that is affected by soil water status. Nitrification was active when soil water content is relatively low and denitrification becomes the main source when soil is under anaerobic conditions. With frequent and relative light irrigation, the three Kentucky bluegrass lawns (medium-textured soils) should emit most of N_2O through nitrification process as the soils are usually under aerobic conditions in the semi-arid area.

2.4.2 Simulated N_2O Emissions from Perennial Ryegrass Lawns in Kansas

In the perennial ryegrass experiment, the observed annual N_2O emissions showed little difference between treatments of urea and ammonium sulfate at rate of 250 kg N ha^{-1} yr^{-1}. Approximately 50% more N_2O emissions were found in high nitrogen rate than low nitrogen rate. Initially, the DAYCENT model overestimated annual emission rates by 218%, 210%, and 189% for treatments of ammonium sulfate at high rate, urea at high rate, and urea at low rate, respectively. We then modified the nitrification rates to simulate the effect of BNI. The annual emissions for three nitrogen treatments can be approximated within 8% of the observed values by setting the nitrification coefficient in the model to 0.3 (Fig. 2.3). The Pearson's r of predicted daily fluxes of N_2O was 0.50, 0.65, and 0.78 for treatments of ammonium sulfate at high rate, urea at high rate, and urea at low rate, respectively (Fig. 2.4a and b). Our simulation indicated that soil nitrification was reduced by 50 to 64% after the modification of nitrification coefficient. This reduction of N_2O emissions is similar to that found in a tropical grass *Panicum maximum* (approximately 50%), which was detected the same level of BNI capacity as ryegrass (Subbarao et al., 2007; Subbarao et al., 2009). Although nitrification was suppressed, the DAYCENT model simulated that N_2O emitted from nitrification was still the major component of the total emissions (66% for the high N treatment and 80% for the low N treatment).

Soil temperature was correctly simulated in the growing season (Fig. 2.5a and b). The underestimation of soil temperature in winter time (up to 7.8 °C) likely results from the underestimation of the insulating effect of thatch and snow cover. Soil water content was not well simulated in the winter period by the DAYCENT model as well (Fig. 2.5c). Soil ammonium and nitrate availability plays important roles in simulating nitrification and denitrification. Measured soil ammonium and nitrate concentrations for four measurement days in 2004 were compared with simulated results (Fig. 2.6). The simulated soil ammonium and nitrate concentrations in the high nitrogen rate treatments were overestimated but acceptable considering the high rate of fertilization.

Negative N_2O flux or N_2O uptake has been found in various other ecosystems (Chapuis-Lardy et al., 2007), which has also been observed in this ryegrass study. However, it is still in debate whether negative values should be treated as errors or measurement 'noise' (Chapuis-Lardy et al., 2007). The DAYCENT model does not simulate the uptake of N_2O since the mechanism of soil uptake of N_2O is unclear.

2.4.3 Long-term Effects of Different Irrigation Levels on N_2O

Soil water status and soil nitrogen dynamics are closely related to irrigation. The long-term prediction of N_2O emissions for a Kentucky bluegrass lawn irrigated with 60% PET, 80% PET, and 100% PET replacement are shown in Fig. 2.7a. Our simulation results indicate that 60% PET irrigation results in the highest annual N_2O emissions in 30 years, which is approximately twice as high as that of 100% PET irrigation applied with the same amount of fertilizer in most of the years. In contrast, in the last 10 years of our simulation, annual emissions were slightly higher in 100% PET irrigation. The reason is likely that 60% PET replacement results in retardant growth of turfgrass and a reduction in nitrogen uptake, thus more mineral nitrogen accumulating in the top soil layer (Fig. 2.7b). When there is abundant mineral nitrogen in the soil (>40 years in Fig. 2.7b), annual N_2O emission rates are not very different for the three irrigation levels. The result may be explained by simulated soil WFPS, which was found to closely relate to N_2O emissions (Smith et al., 1998; Dobbie et al., 1999). Daily average water-filled pore space in our simulation of 100% PET irrigation was in a range from 0.4 to 0.5 (WFPS of 0.5 corresponds to the field capacity) for most of the time in growing season. Water-filled pore space for 60% PET irrigation mainly fluctuated between 0.25 and 0.5. According to Parton et al. (2001), in the DAYCENT model, the effect of WFPS on nitrification is highest when WFPS is approximately 0.4 in medium-textured soil. The effect drops from 1.0 to 0.8 when WFPS decrease from 0.4 to 0.3. Changing irrigation from replacing 60% to 100% PET probably resulted in little difference on nitrification rate in the sandy clay loam soil when NH_4^+ is not a limiting factor. The majority of N_2O emissions are modeled to be emitted through nitrification as denitrification was not simulated when WFPS is less than 0.55 (Parton et al., 2001). The highest annual N_2O emissions in our simulation were predicted to reach 6.1 kg N ha^{-1} yr^{-1} at Year 46 in 100% PET scenario, which is comparable to the estimated annual rate of 6.4 kg N ha^{-1} yr^{-1} in golf course fairways in Arizona with high nitrogen input in preceding years (Guilbault and Matthias, 1998).

2.4.4 Long-term Effects of Different Nitrogen Fertilization Regimes on N_2O

Using constant rate for nitrogen fertilization for many years is commonly found in the management of lawns. However, constant nitrogen rates may result in over- or under-fertilization and pose potential risks to turfgrass quality and may cause environmental threats (e.g. nitrate leaching). In our long-term prediction, with fertilization of 150 kg N ha^{-1} yr^{-1} constantly, N_2O emissions increased dramatically in the first 15 years and leveled off at an average rate of approximately 5 kg N ha^{-1} yr^{-1} after 15 years (Fig. 2.8). By applying the model-generated best nitrogen rates, annual N_2O emissions were maintained at the range of 0.6 to 3.1 kg N ha^{-1} yr^{-1} from year 10 to 50, which is approximately half of the emission rate of using conventional constant nitrogen rate. The N_2O emissions were predicted to be higher in the first ten years using the model-generated best nitrogen rates because larger amount of nitrogen was applied to help turfgrass to establish and exhibit high quality. In 50 years, gradually reducing fertilization as the lawn ages would significantly reduce long-term N_2O emissions by approximately 40% when compared to applying nitrogen at the constant rate (150 kg N ha^{-1} yr^{-1}).

2.4.5 Global Warming Potential

Total energy cost from turfgrass maintenance (the sum of GWPs for mowing, fertilization, irrigation, and pesticide application) accounts for the largest proportion of the total emissions (65-80 %) in the first decade for the three scenarios. Mowing, irrigation and fertilization individually contribute to nearly equal proportions of emissions in first decades in the scenario of conventional management (Fig. 2.9a). With constant 150 kg N ha^{-1} yr^{-1} fertilization, the GWP of N_2O emissions increases to around 2250 g CO_2 m^{-2} $decade^{-1}$ (half of the total positive GWP), while other components of positive GWPs remain the same level. In the scenario of best management practices predicted by the DAYCENT model for a high quality lawn, the total emissions tend to decrease though decades (Fig. 2.9b). As fertilization rate is reduced through time, the energy cost of fertilization decreases while the GWP of N_2O emissions is kept stable at an average rate of 1210 g CO_2 m^{-2} $decade^{-1}$.

The GWPs of carbon sequestration vary slightly for the scenarios of constant nitrogen rate and model predicted BMPs for a high quality lawn. It is predicted that nearly 4000 g CO_2 m^{-2} carbon could be sequestrated in the first decade. The sequestration rates gradually decrease as the lawn ages. The soil used in our simulation is a sandy clay loam with 29 percent clay. Higher clay content soil could probably be able to sequestrate substantially more carbon in a long-term since clay particles provide greater protection to SOM (Bandaranayake et al., 2003). Maintaining a lawn with medium quality instead of high quality is predicted to significantly reduce total emissions by more than 30 percent (Fig. 2.9c). However, the carbon sequestration rates will be also reduced due to lower fertilizer and irrigation input; only 2700 g CO_2 m^{-2} could be sequestrated in the first decade.

The model predicted that the amount of water needed for irrigation range approximately from 40 to 80 cm per year, which resulted in GWPs of 46 to 93 g CO_2 m^{-2} yr^{-1}. Since Fort Collins Water Utility relies on gravity-fed surface supplies; global warming potential of irrigation is likely just half or less than half of those of some cities in the South Metro area of Colorado, which use substantial amounts of energy for pumping groundwater from the Denver Basin aquifers (Tellinghvuisen, 2009). Methane uptake was not included in our calculation. Lawns in Colorado were found to be sinks of CH_4 with annual uptake rate of 0.15 g C m^{-2}; that equals to 4.2 g CO_2 m^{-2} yr^{-1}, which is negligible compared to N_2O emissions (Kaye et al., 2004).

In summary, both high and medium quality Kentucky bluegrass lawns of medium-textured soils were predicted to serve as a sink of carbon for at least two decades.

2.5 CONCLUSIONS

Computer modeling provides a relatively easy way for assessing the GHG budget for ecosystems. Our study showed that the DAYCENT model can properly simulate the annual N_2O emissions for Kentucky bluegrass lawns. The simulation for perennial ryegrass indicates that BNI might play an important role in reducing N_2O emissions. Over a long term, a Kentucky bluegrass lawn was predicted to emit more N_2O when irrigation was applied by replacing 60% PET than that of 100% PET. The model suggests that gradually reducing fertilization as the lawn ages from 0 to 50 years would significantly reduce long-term N_2O emissions. The long-term interaction of fertilization and irrigation should be further studied in different soil conditions. In all three scenarios for estimating GWPs, GHG emissions from maintenance accounts for 50% to 80% of the total emissions in each decade. Keeping a constant high fertilization rate of 150 kg N ha^{-1} yr^{-1} could substantially increase GWP of N_2O to half of the total emissions. Our DAYCENT model generated best nitrogen rates for a high quality lawn could help reduce the amounts of the positive GWP by 25% in 50 years compared to those of the conventional nitrogen rates but maintain a similar soil carbon sequestration rate. Further studies are needed to assess the GWPs for lawns of different soil textures and in different climates.

Table 2.1. Soil site properties of 0-15 cm in the experiment conducted by Kaye et al. (2004).

Site	**Sand**	**Clay**	**Silt**	**Bulk density**	**pH**
	-----------%---------			g cm^{-3}	
Home lawn A	47	30	24	1.15	7.5
Home lawn B	74	12	14	1.21	7.7
Institutional lawn	53	22	26	1.21	7.8

Table 2.2. Soil organic carbon (SOC) of three Kentucky bluegrass sites in the experiment conducted by Kaye et al. (2004).

Site	**0-15 cm**	**15-30 cm**	**0-20 cm (estimated†)**
	------------------------------g C m^{-2}	------------------------------	
Home lawn A	4226	2452	5044
Home lawn B	4918	990	5248
Institutional lawn	5128	3108	6164

†Estimated SOC in 0-20 cm of soil was the sum of SOC in top 15 cm and 1/3 of SOC in 15-30 cm of soil. The DAYCENT model simulates SOC in the top 20 cm soil

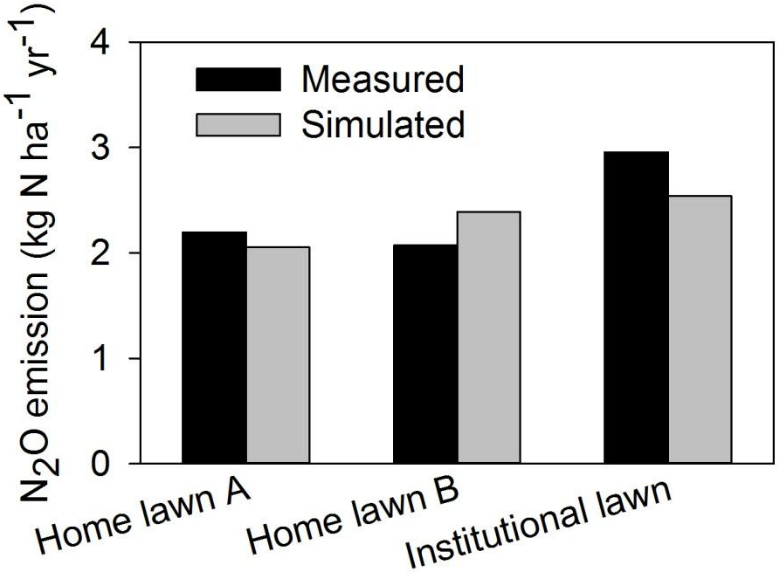

Fig. 2.1: The comparison of measured and simulated annual N_2O emissions from three lawns in Colorado.

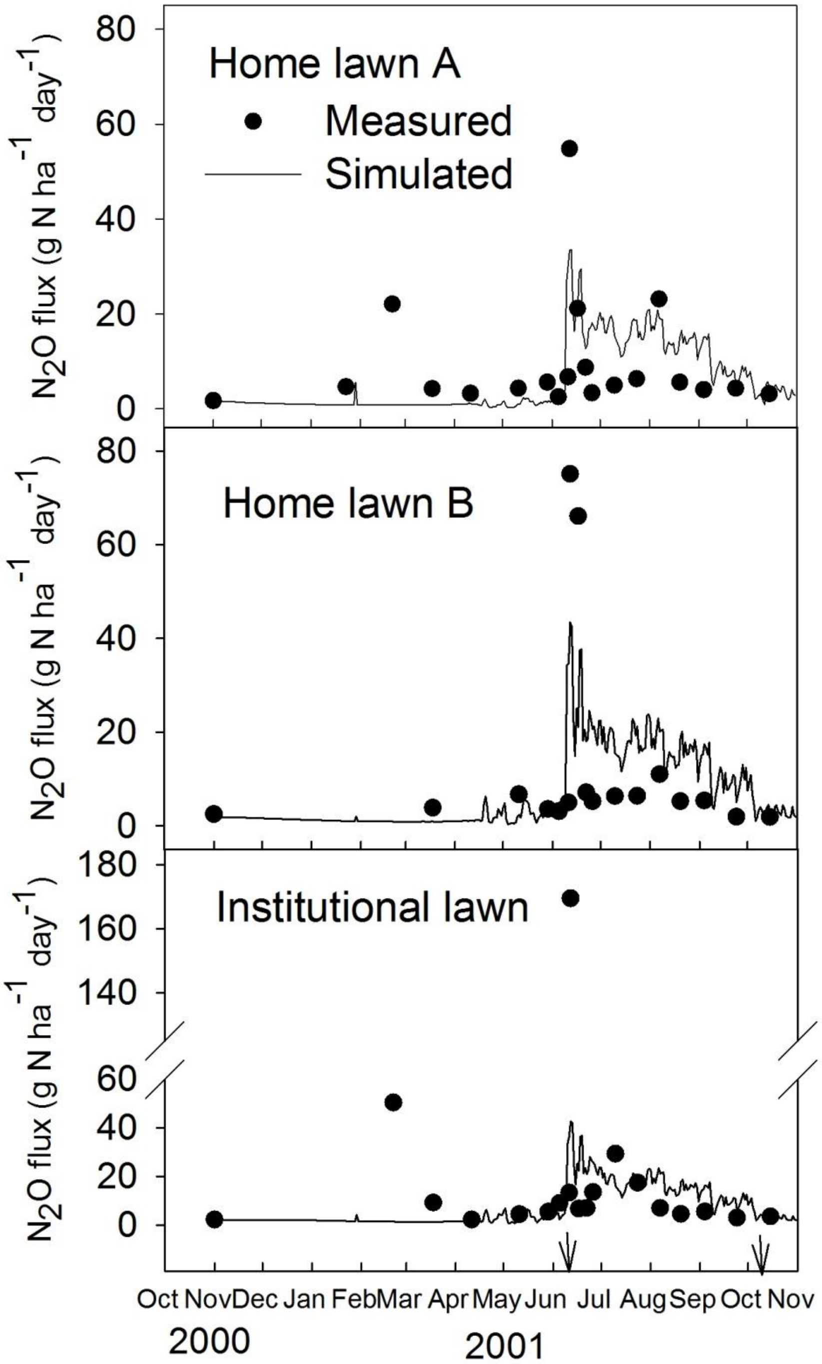

Fig. 2.2: The comparison of measured and simulated daily N_2O fluxes from three lawns in Colorado. Measured data were from the experiment conducted by Kaye et al. (2004). Arrows indicate the fertilization dates.

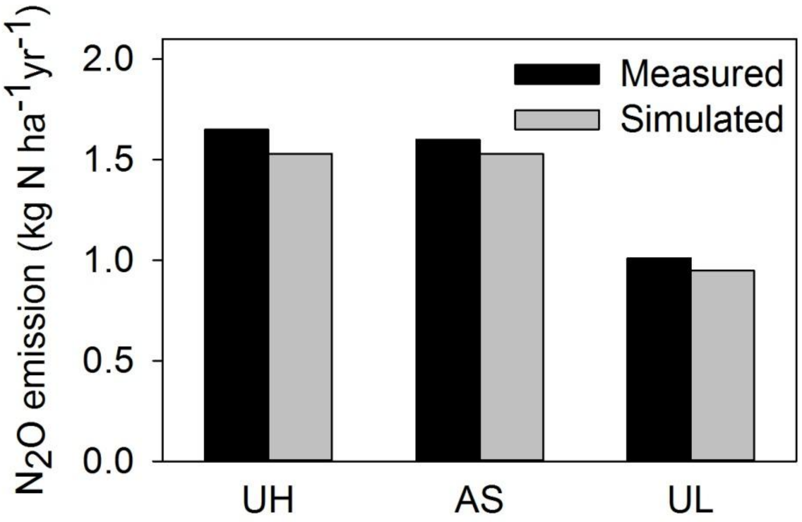

Fig. 2.3: The comparison of measured and simulated annual N_2O emissions from a perennial ryegrass site with three nitrogen treatments (UH: urea, 250 kg N ha^{-1} yr^{-1}; AS: ammonium sulfate, 250 kg N ha^{-1} yr^{-1}; UL: urea, 50 kg N ha^{-1} yr^{-1}). Measured data were from an experiment in Kansas (Bremer, 2006).

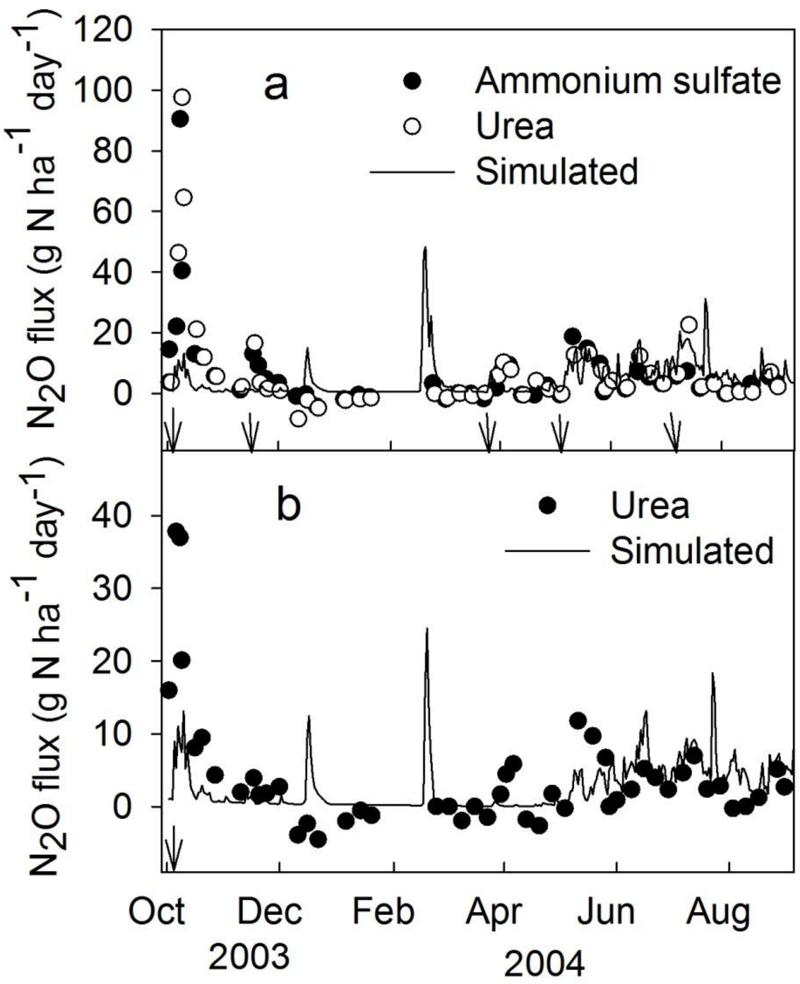

Fig. 2.4: The comparison of measured and simulated daily N_2O fluxes. Perennial ryegrass was fertilized at nitrogen rates of (a) 250 kg N ha^{-1} yr^{-1} or at (b) 50 kg N ha^{-1} yr^{-1} using ammonium sulfate or urea. Arrows indicate the fertilization dates.

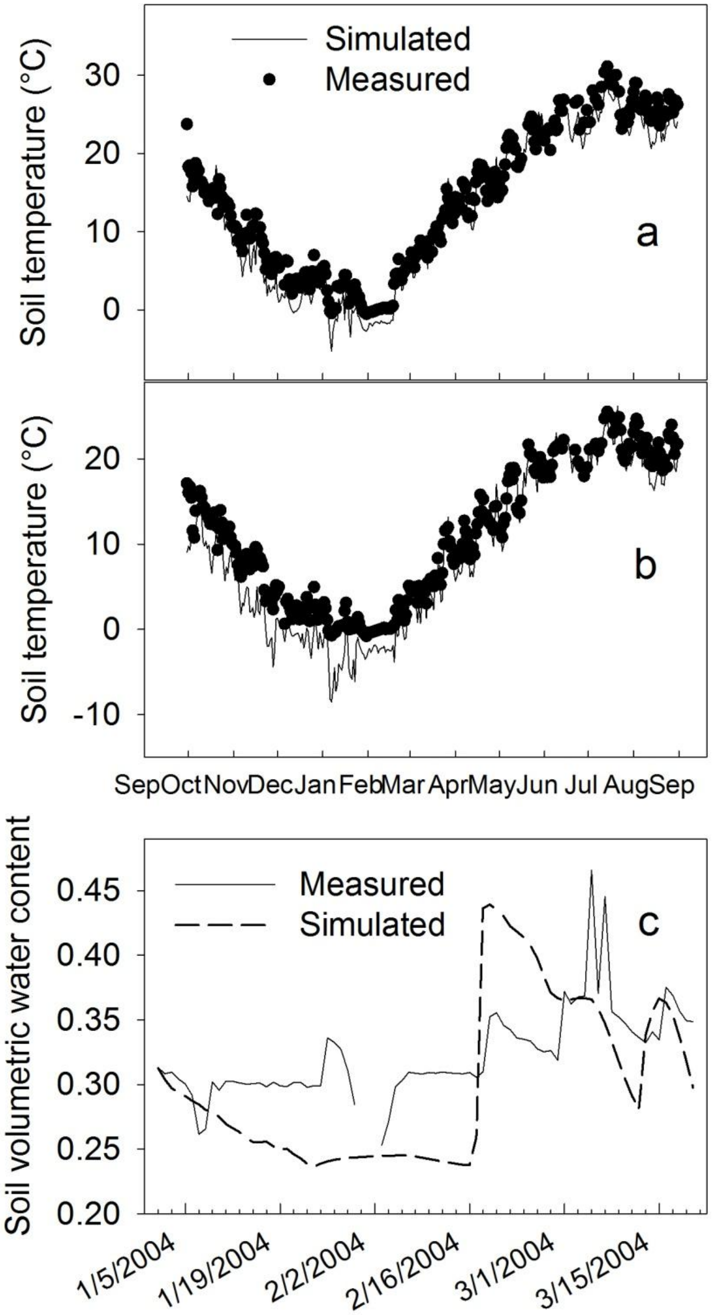

Fig. 2.5: The comparison of measured and simulated (a) daily maximum, (b) minimum soil temperature, and (c) soil volumetric water content. Soil temperature and water content was measured at 5 cm depth. Simulated results were the model output of the 2 to 5 cm soil layer.

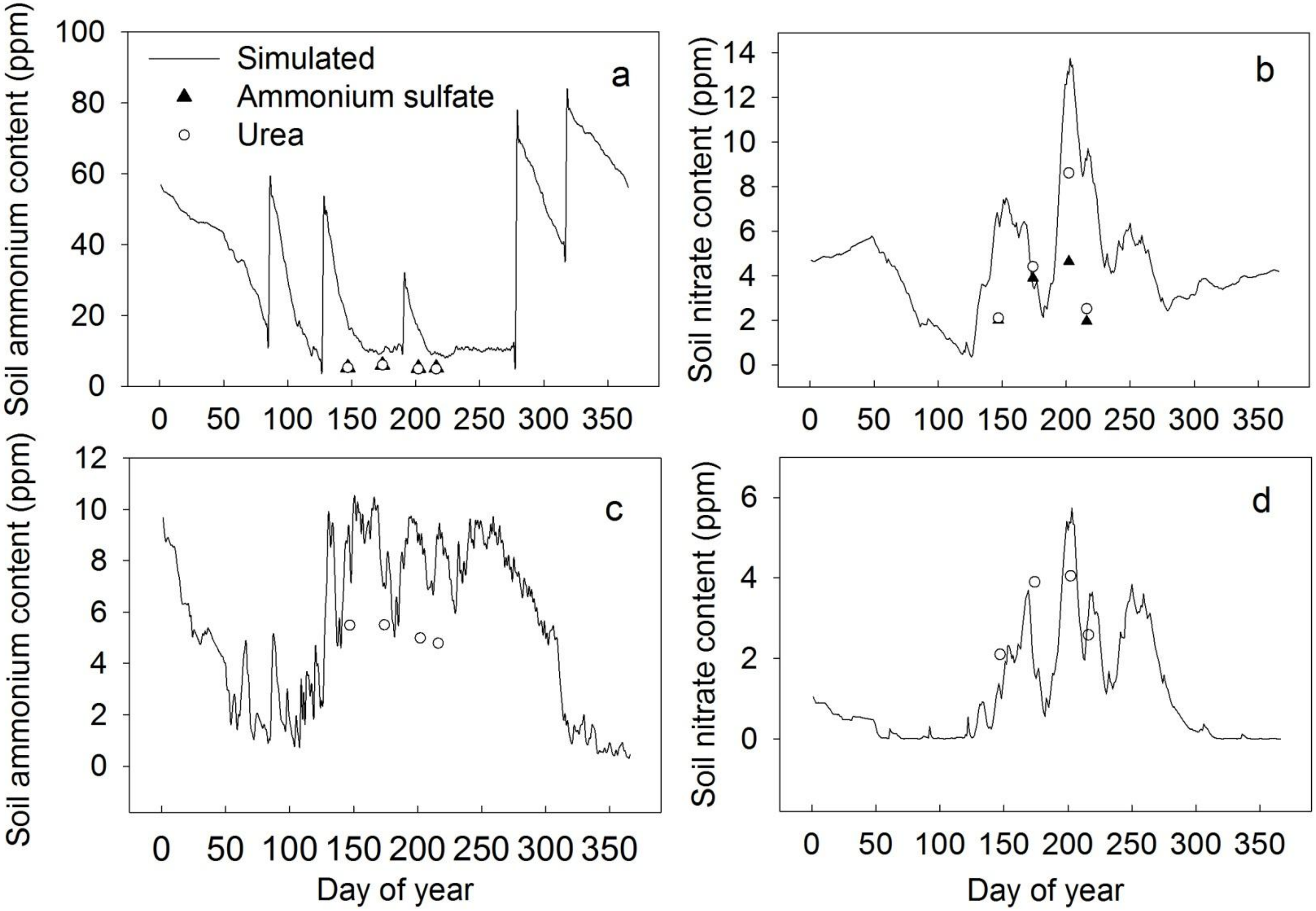

Fig. 2.6: The comparison of measured and simulated soil ammonium and nitrate concentrations. Fertilizer was applied at rates of (a and b) 250 kg N ha^{-1} yr^{-1} or (c and d) 50 kg N ha^{-1} yr^{-1}.

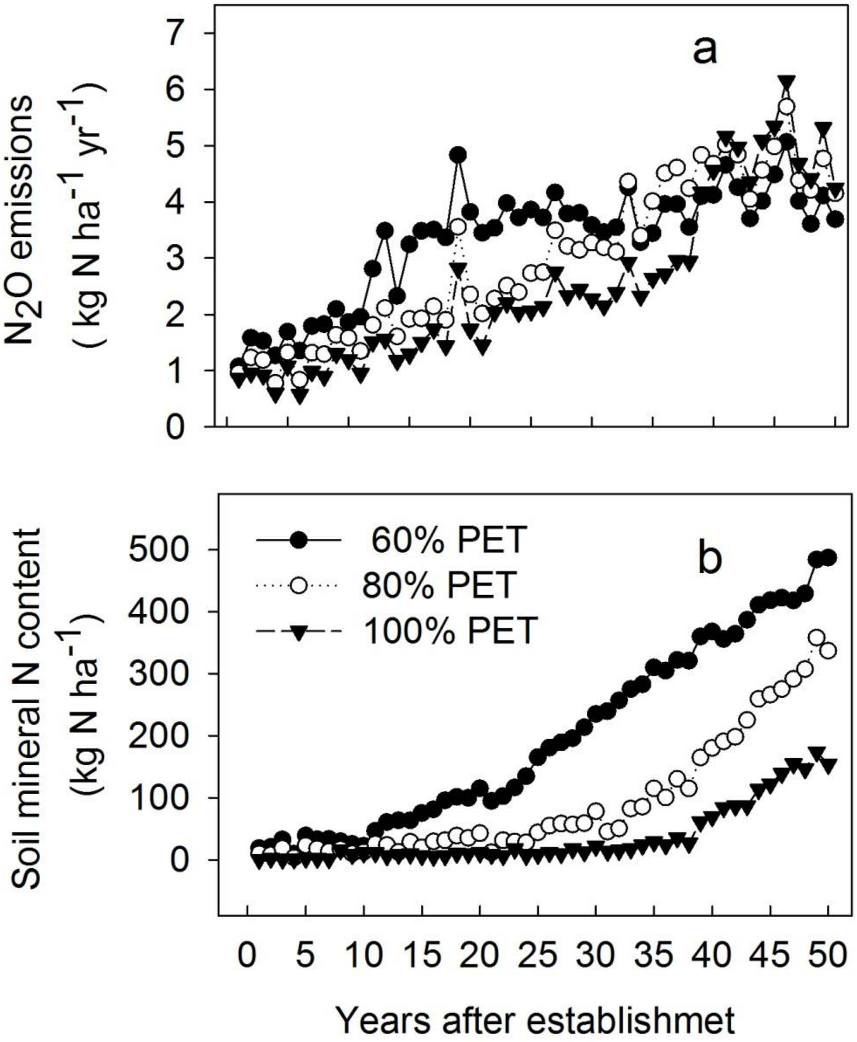

Fig. 2.7: Model predicted (a) annual N_2O emissions and (b) annual average soil mineral nitrogen content in top 10 cm soil for 3 irrigation levels for a Kentucky bluegrass lawn in Colorado.

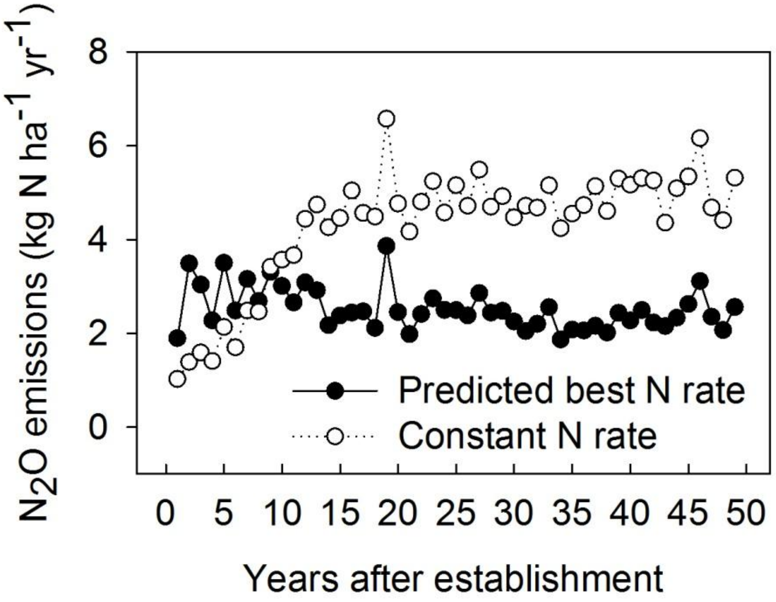

Fig. 2.8: Model predicted annual N_2O emissions for two management scenarios (Predicted best nitrogen rate: using the predicted best N rate for a high quality lawn in Chapter 1; Constant nitrogen rate: fertilizer applied at 150 kg N ha^{-1} yr^{-1}).

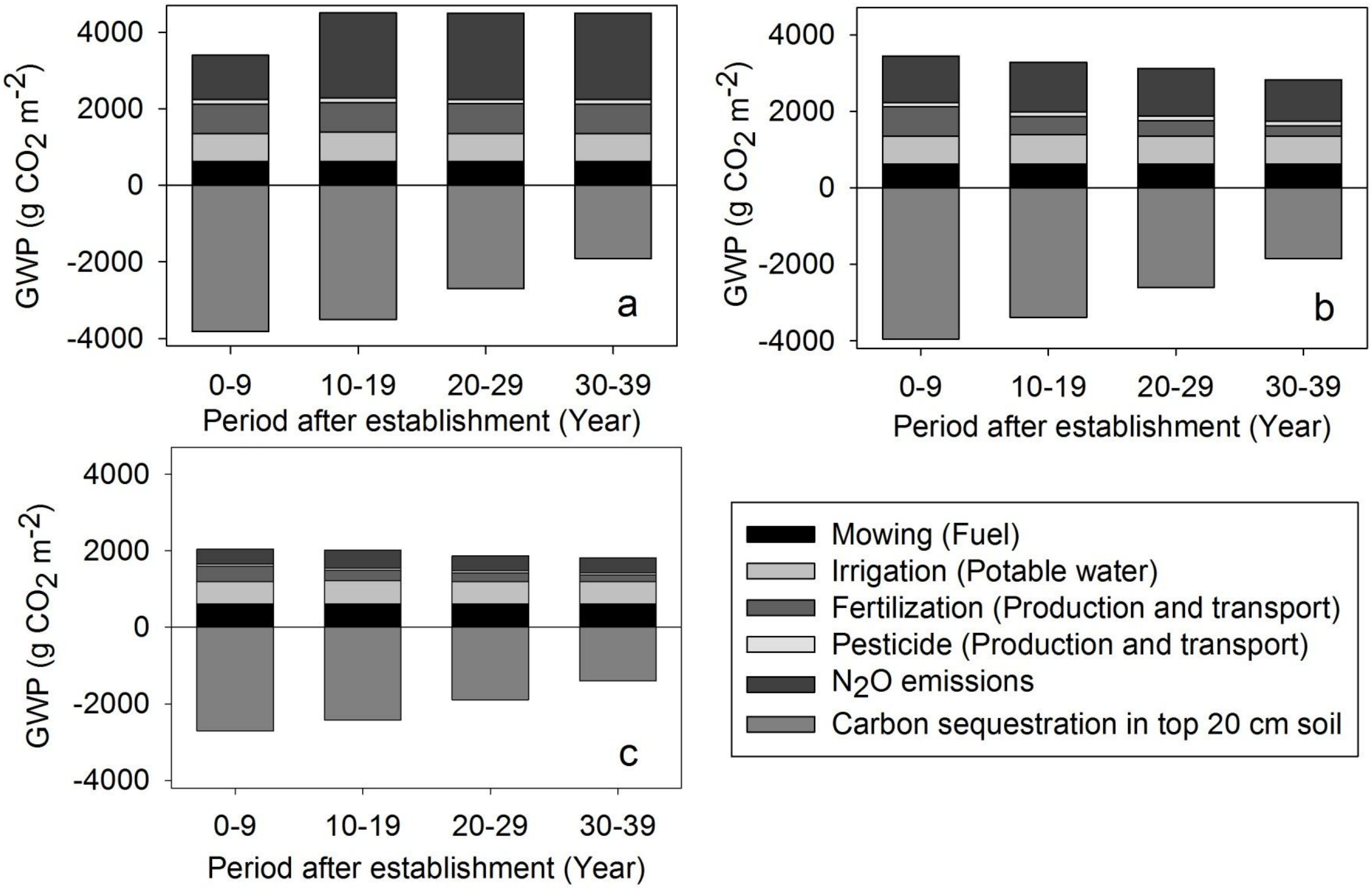

Fig. 2.9: Estimated global warming potentials (GWPs) for lawns using (a) conventional management, BMPs generated by the DAYCENT model for a lawn to maintain (b) high quality and (c) medium quality.

REFERENCES

Adler, P.R., S.J. Del Grosso, and W.J. Parton. 2007. Life-cycle assessment of net greenhouse-gas flux for bioenergy cropping systems. Ecological Applications 17:675-691.

Albritton, D., L. Meira Filho, U. Cubasch, X. Dai, Y. Ding, B. Hewitson, et al. 2001. Technical Summary. In: J. I. Houghton, Y. Ding, D. J. Griggs, M. Nogur, P. J. van der Linden, X. Dai, K. Maskell and C. A. Johnson, editors, Climate Change 2001: The Scientific Basis. Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press., Cambridge, United Kingdom and New York, USA.

Alig, R.J., J.D. Kline, and M. Lichtenstein. 2004. Urbanization on the US landscape: looking ahead in the 21st century. Landscape and Urban Planning 69:219-234.

Bandaranayake, W., Y.L. Qian, W.J. Parton, D.S. Ojima, and R.F. Follett. 2003. Estimation of soil organic carbon changes in turfgrass systems using the CENTURY model. Agron. J. 95:558-563.

Bateman, E.J., and E.M. Baggs. 2005. Contributions of nitrification and denitrification to N_2O emissions from soils at different water-filled pore space. Biol. Fertil. Soils 41:379-388.

Brady, N.C., and R.R. Weil. 2008. The nature and properties of soils. 14th ed. Pearson Prentice Hall, Upper Saddle River, NJ.

Bremer, D.J. 2006. Nitrous oxide fluxes in turfgrass: effects of nitrogen fertilization rates and types. J. Environ. Qual. 35:1678-1685.

Chan, A.S.K., and T.B. Parkin. 2001. Methane oxidation and production activity in soils from natural and agricultural ecosystems. J. Environ. Qual. 30:1896-1903.

Chapuis-Lardy, L., N. Wrage, A. Metay, J.-L. Chotte, and M. Bernoux. 2007. Soils, a sink for N_2O? A review. Global Change Biology 13:1-17.

Christensen, S., and B.T. Christensen. 1991. Organic matter available for denitrification in different soil fractions: effect of freeze/thaw cycles and straw disposal. Eur. J. Soil Sci. 42:637-647.

Del Grosso, S., D. Ojima, W. Parton, A. Mosier, G. Peterson, and D. Schimel. 2002. Simulated effects of dryland cropping intensification on soil organic matter and greenhouse gas exchanges using the DAYCENT ecosystem model. Environ. Pollut. 116 Suppl 1:S75-S83.

Del Grosso, S.J., A.D. Halvorson, and W.J. Parton. 2008. Testing DAYCENT model simulations of corn yields and nitrous oxide emissions in irrigated tillage systems in Colorado. J. Environ. Qual. 37:1383-1389.

Del Grosso, S.J., A.R. Mosier, W.J. Parton, and D.S. Ojima. 2005. DAYCENT model analysis of past and contemporary soil N_2O and net greenhouse gas flux for major crops in the USA. Soil Tillage Res. 83:9-24.

Dobbie, K.E., I.P. McTaggart, and K.A. Smith. 1999. Nitrous oxide emissions from intensive agricultural systems: Variations between crops and seasons, key driving variables, and mean emission factors. J. Geophys. Res. 104:26891-26899.

Dutaur, L., and L. Verchot. 2007. A global inventory of the soil CH_4 sink. Global Biogeochem. Cycles 21:GB4013.

Falk, J.H. 1976. Energetics of a suburban lawn ecosystem. Ecology 57:141-150.

Falk, J.H. 1980. The primary productivity of lawns in a temperate environment. J. Appl. Ecol. 17:689-695.

Golubiewski, N.E. 2006. Urbanization increases grassland carbon pools: effects of landscaping in Colorado's Front Range. Ecological Applications 16:555-571.

Grant, R.F., E. Pattey, T.W. Goddard, L.M. Kryzanowski, and H. Puurveen. 2006. Modeling the effects of fertilizer application rate on nitrous oxide emissions. Soil Sci. Soc. Am. J. 70:235-248.

Groffman, P.M., and R.V. Pouyat. 2009. Methane uptake in urban forests and lawns. Environ. Sci. Technol. 43:5229-5235.

Groffman, P.M., C.O. Williams, R.V. Pouyat, L.E. Band, and I.D. Yesilonis. 2009. Nitrate leaching and nitrous oxide flux in urban forests and grasslands. J. Environ. Qual. 38:1848-1860.

Guilbault, M.R., and A.D. Matthias. 1998. Emissions of N_2O from Sonoran Desert and effluent-irrigated grass ecosystems. Journal of Arid Environments 38:87-98.

IPCC. 2007. Climate change 2007: Synthesis report. Summary for policy makers. Intergovernmental Panel on Climate Change.

Kaye, J.P., I.C. Burke, A.R. Mosier, and J. Pablo Guerschman. 2004. Methane and nitrous oxide fluxes from urban soils to the atmosphere. Ecological Applications 14:975-981.

Lal, R. 2004. Carbon emission from farm operations. Environ. Int. 30:981-990.

Law, N., L. Band, and M. Grove. 2004. Nitrogen input from residential lawn care practices in suburban watersheds in Baltimore county, MD. Journal of Environmental Planning and Management 47:737-755.

Li, C., S. Frolking, and K. Butterbach-Bahl. 2005. Carbon sequestration in arable soils is likely to increase nitrous oxide emissions, offsetting reductions in climate radiative forcing. Climatic Change 72:321-338.

Li, X., T. Meixner, J. Sickman, A. Miller, J. Schimel, and J. Melack. 2006. Decadal-scale dynamics of water, carbon and nitrogen in a California chaparral ecosystem: DAYCENT modeling results. Biogeochemistry 77:217-245.

Livesley, S.J., B.J. Dougherty, A.J. Smith, D. Navaud, L.J. Wylie, and S.K. Arndt. 2010. Soil-atmosphere exchange of carbon dioxide, methane and nitrous oxide in urban garden systems: impact of irrigation, fertiliser and mulch. Urban Ecosystems 13:273-293.

Merino, A., P. Pérez-Batallón, and F. Macías. 2004. Responses of soil organic matter and greenhouse gas fluxes to soil management and land use changes in a humid temperate region of southern Europe. Soil Biol. Biochem. 36:917-925.

Milesi, C., S. Running, C. Elvidge, J. Dietz, B. Tuttle, and R. Nemani. 2005. Mapping and modeling the biogeochemical cycling of turf grasses in the United States. Environ. Manage. 36:426-438.

Moore, D.R.E., and J.S. Waid. 1971. The influence of washings of living roots on nitrification. Soil Biol. Biochem. 3:69-83.

Mosier, A., D. Schimel, D. Valentine, K. Bronson, and W. Parton. 1991. Methane and nitrous oxide fluxes in native, fertilized and cultivated grasslands. Nature 350:330-332.

Mosier, A., W.J. Parton, D.W. Valentine, D.S. Ojima, D.S. Schimel, and O. Heinemeyer. 1997. CH_4 and N_2O fluxes in the Colorado shortgrass steppe: 2. Long-term impact of land use change. Global Biogeochem. Cycles 11:29-42.

Mosier, A., C. Kroeze, C. Nevison, O. Oenema, S. Seitzinger, and O. van Cleemput. 1998. Closing the global N_2O budget: nitrous oxide emissions through the agricultural nitrogen cycle. Nutr Cycl Agroecosyst 52:225-248.

Munro, P.E. 1966. Inhibition of Nitrite-Oxidizers by Roots of Grass. J. Appl. Ecol. 3:227-229.

Ojima, D.S., D.W. Valentine, A.R. Mosier, W.J. Parton, and D.S. Schimel. 1993. Effect of land use change on methane oxidation in temperate forest and grassland soils. Chemosphere 26:675-685.

Parton, W.J., E.A. Holland, S.J. Del Grosso, M.D. Hartman, R.E. Martin, A.R. Mosier, et al. 2001. Generalized model for NO_x and N_2O emissions from soils. J. Geophys. Res. 106:17403-17419.

Parton, W.J., D.S. Ojima, C.V. Cole, and D.S. Schimel. 1994. A general model for soil organic matter dynamics: sensitivity to litter chemistry, texture and management. SSSA special publication 39:147-167.

Parton, W.J., D.S. Schimel, C.V. Cole, and D.S. Ojima. 1987. Analysis of factors controlling soil organic matter levels in great plains grasslands. Soil Sci. Soc. Am. J. 51:1173-1179.

Parton, W.J., J.M.O. Scurlock, D.S. Ojima, T.G. Gilmanov, R.J. Scholes, D.S. Schimel, et al. 1993. Observations and modeling of biomass and soil organic matter dynamics for the grassland biome worldwide. Global Biogeochem. Cycles 7:785-809.

Pepper, D.A., S.J. Del Grosso, R.E. McMurtrie, and W.J. Parton. 2005. Simulated carbon sink response of shortgrass steppe, tallgrass prairie and forest ecosystems to rising CO_2, temperature and nitrogen input. Global Biogeochem. Cycles 19:GB1004.

Pouyat, R., I. Yesilonis, and N. Golubiewski. 2009. A comparison of soil organic carbon stocks between residential turf grass and native soil. Urban Ecosystems 12:45-62.

Qian, Y., W. Bandaranayake, W.J. Parton, B. Mecham, M.A. Harivandi, and A.R. Mosier. 2003. Long-term effects of clipping and nitrogen management in turfgrass on soil organic carbon and nitrogen dynamics: the CENTURY model simulation. J. Environ. Qual. 32:1694-1700.

Qian, Y., and R.F. Follett. 2002. Assessing soil carbon sequestration in turfgrass systems using long-term soil testing data. Agron. J. 94:930-935.

Rodriguez, I.R., and G.L. Miller. 2000. Using a chlorophyll meter to determine the chlorophyll concentration, nitrogen concentration, and visual quality of St. Augustinegrass. HortScience 35:751-754.

Sahu, R.N.D. 2008. Technical Assessment of the Carbon Sequestration Potential of Managed Turfgrass in the United States. Report for the Outdoor Power Equipment Institute.

Schumacher, B.A. 2002. Methods for the determination of total organic carbon (TOC) in soils and sediments. United States Environmental Protection Agency, Ecological Risk Assesment Support Center.

Smith, K.A., P.E. Thomson, H. Clayton, I.P. McTaggart, and F. Conen. 1998. Effects of temperature, water content and nitrogen fertilisation on emissions of nitrous oxide by soils. Atmospheric Environment 32:3301-3309.

Stehfest, E., and C. Müller. 2004. Simulation of N_2O emissions from a urine-affected pasture in New Zealand with the ecosystem model DayCent. J. Geophys. Res. 109:1-7.

Subbarao, G., K. Nakahara, M. Hurtado, H. Ono, D. Moreta, A. Salcedo, et al. 2009. Evidence for biological nitrification inhibition in Brachiaria pastures. Proc. Natl. Acad. Sci. USA 106:17302-17307.

Subbarao, G., M. Rondon, O. Ito, T. Ishikawa, I. Rao, K. Nakahara, et al. 2007. Biological nitrification inhibition (BNI)—is it a widespread phenomenon? Plant Soil 294:5-18.

Tellinghuisen, S. 2009. Water conservation = energy conservation. A report for the CWCB. Western Resource Advocates.

Townsend-Small, A., and C.I. Czimczik. 2010. Correction to "Carbon sequestration and greenhouse gas emissions in urban turf". Geophys. Res. Lett. 37:L06707.

Townsend-Small, A., D.E. Pataki, C.I. Czimczik, and S.C. Tyler. 2011. Nitrous oxide emissions and isotopic composition in urban and agricultural systems in southern California. J. Geophys. Res. 116:G01013.

U.S. EIA. 2001. Updated state-level greenhouse gas emission factors for electricity generation. Energy Information Administration. http://tonto.eia.doe.gov/FTPROOT/environment/e-supdoc.pdf (accessed 20 Sept. 2012).

U.S. EPA. 2011. Greenhouse Gas Equivalencies Calculator Calculations and References. U.S. Environmental Protection Agency. http://www.epa.gov/cleanenergy/energy-resources/refs.html (accessed 20 Sept. 2012).

Zirkle, G., R. Lal, and B. Augustin. 2011. Modeling carbon sequestration in home lawns. HortScience 46:808-814.