GREENHOUSE GAS ABATEMENT FROM

HOME LAWN CARE MANAGEMENT PRACTICES

by

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DEDICATION

To my parents

Gene and Beverly Garrison
&
Trudy and Gary Smith

Thank you for your love and support!

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INTRODUCTION

The research presented in my thesis focuses on greenhouse gas abatement from home lawn care by investigating alternative practices affecting N losses, changes in soil carbon following establishment, and economic viability of management practices to reduce carbon emissions. Urban and suburban turf areas, including home lawns, cover approximately 16 million ha in the United States. Lawn turf has received an increasing amount of attention from environmental advocates and legislators in order to reduce potentially negative environmental impacts from lawn care. The attention on home lawn care is partly a result of increasing social awareness towards the identification of environmental pollution sources and the mitigation of global warming caused by anthropogenic greenhouse gases (GHG) emissions, such as CO_2 and N_2O. Regulations aimed to limit the negative effects of turfgrass management on the environment can impact the lawn care decisions of 80 million households in the United States.

The identification of best management practices that may reduce the environmental impact of turf management has become imperative because regulations are increasingly restricting the use of synthetic fertilizers and pesticides. Chapter one presents a study that compared professional and do-it-yourself (DIY) approaches to home lawn care that utilized conventional or alternative products for fertilization and weed control. Turf stands established by seeding or mature sod were also included as treatments in the comparison. The experimental design consisted of a five by two factorial arrangement, representing five management programs and two establishment methods. This study evaluated NO_3 leaching, N_2O emissions, and aesthetic qualities of the treatments for two years following establishment. Though the alternative approaches provided turf quality equal to the conventional, NO_3 leaching and N_2O

emissions were not reduced when alternative products were utilized. Chapter one was formatted for submission to Crop Science.

Perennial grass swards, such as turfgrass, may help mitigate global warming by fixing atmospheric CO_2 through photosynthesis and accumulating the carbon as organic residues in the soil. The establishment of turfgrasses and other ornamental plants has replaced agricultural production as suburban and urban areas in the United States have expanded. During new home construction, the topsoil has often been removed and there is an option of establishing the turfgrass on the subsoil. Establishing turfgrasses on subsoil has often been viewed as undesirable; however, the subsoil lacks soil organic carbon (SOC) and may be capable of storing more atmospheric carbon than topsoil. The primary objective of the study presented in chapter two was to compare the change in SOC and soil organic nitrogen (SON) when turfgrasses were established on topsoil and subsoil. The second objective was to compare the effects of N applications and irrigation on SOC and SON change in the two soil types. Treatments were arranged in a split plot design with four replications, irrigation was the whole plot factor and fertilizer rate and soil type were subplot factors. Soil organic carbon and SON declined in the topsoil during the first three years following establishment. The SOC and SON of the subsoil increased during the same period. Nitrogen fertilization increased SOC and SON when compared with the non-fertilized treatments, although irrigation did not affect either SOC or SON. Chapter two was formatted for submission to HortScience.

Strategies are sought to mitigate the threat of global warming and reduce the emissions of CO_2 and other greenhouse gases. As a consequence, the routine maintenance practices employed for home lawn care have gained attention as a possibility to reduce carbon emissions. The goal of the study presented in chapter three was to assess the cost effectiveness of reducing carbon emissions from lawn care practices by comparing conventional lawn care with an alternative strategy. As a result of the geographical variances in the availability of resources and emissions from power generation, the lawn care strategies were also compared at two locations that emphasized these differences. To accomplish the comparisons, data summaries were constructed and then utilized to conduct consumption-based estimations of the carbon emissions from conventional and alternative N fertilizer production, gasoline and electric lawn mowing, and irrigation. The cost effectiveness of the alternative practices to reduce carbon was determined through assessments of both annual cost and the marginal abatement cost of carbon reduction. Though the marginal abatement cost of the alternative N fertilizer was an expensive option to make reductions, the electric lawn mower provided a cost effective method to reduce carbon emissions. Irrigation of home lawns was the largest source of carbon emissions at both locations. Reductions of lawn care emissions can be accomplished from the use of alternative practices, however, the alternatives were sometimes more expensive. Chapter three was formatted for submission to Environmental Management.

CHAPTER ONE

Nitrous Oxide Emissions and Nitrate Leaching from Conventional and Alternative Turfgrass Management Programs

ABSTRACT

The identification of best management practices that reduce the environmental impact of turf management has become imperative as consumer demand for alternative products increases and regulations restrict the use of synthetic products for turf. The current study consisted of a five by two factorial arrangement, representing five management programs and two grass establishment methods. Management programs evaluated a no input control and four approaches to lawn care of seeded or sodded turf. Lawn care approaches included do-it-yourself (DIY) and professional programs with either conventional or alternative products for fertilization and weed control. Nitrate leaching, N_2O emissions, and aesthetic qualities were evaluated for two years following establishment. Professional and DIY programs had statistically similar NO_3 leaching, N_2O emissions, and aesthetic qualities. The amount of NO_3 leaching was greater during the first year following establishment ($P < 0.05$) and decreased as the stands matured. The use of a conventional N fertilizer did not reduce or increase emissions of N_2O when compared with the alternative N fertilizer. Nitrous oxide emissions were greater for the fertilized treatments than the control on two of the six application dates. Herbicides were necessary more frequently for the seeded, than the sodded treatments. Turf quality was significantly lower for seeded treatments in the first year, but was similar to the sodded treatments during the second year of the study. Though the alternative programs provide turf quality equal to the conventional programs a majority of the time, NO_3 leaching or N_2O emissions were similar.

Conventional amenity turf care practices that rely on synthetic fertilizers and pesticides have gained public attention for their possible impact on the environment (Robbins and Birkenholtz, 2003; Robbins and Sharp, 2003; Townsend-Small and Czimczik, 2010; Milesi et al., 2009). Several US states have restricted the use of fertilizers and pesticides for maintaining turfgrasses in an attempt to protect environmental quality (WDNR, 2004; State of Minnesota, 2006; Williamson, 2007; State of Connecticut, 2009; State of Michigan, 2010; Hochmuth et al., 2011). Synthetic (conventional) products allow for relatively inexpensive and effective means to improve the aesthetic qualities of turfgrass, however, the convenience of using these products has raised concerns that their misuse may be polluting the environment (Robbins and Sharp, 2003; Bertin and Westin, 2004). To address public concerns and reduce the need for government regulation, identifying best management practices for turfgrass has become increasingly imperative. Alternatives to the conventional practices have emerged which rely partly or wholly on organic fertilizers and products developed from plant, animal, or naturally-occurring geological sources. Though the alternative fertilizers and products were derived from natural products, the extents to which the alternatives may reduce environmental concerns are uncertain.

Regular maintenance practices that include routine mowing, fertilization, irrigation, and occasional pesticide applications are commonly employed to maintain high quality lawn. Nitrogen is nearly always the limiting nutrient for turfgrass and supplemental applications of N are often required to optimize turfgrass growth and stand health (Kussow et al., 2011). To ensure plant availability of N between applications and reduce N loss, synthetic fertilizers are purposely produced in a variety of compositions that vary in nutrient content and release characteristics to meet plant requirements. Some forms of fertilizer contain “quick release” i.e. water soluble nitrogen (WSN) sources that are available for plant uptake immediately after application. Slow release products can either contain water insoluble N (WIN) that becomes available for plants following mineralization by microorganisms or slowly available N (SAN) fertilizers. Slowly available N sources are composed of either synthetic N sources that need to undergo mineralization (e.g. methylene urea) or WSN that has been encapsulated with coatings that moderate the release of N. Synthetic turf fertilizers can contain all three forms of N and although SAN are usually synthetically produced, WIN and to a lesser extent WSN, also exist in organic fertilizers.

Fertilizers used for turfgrass management present risks of increasing N_2O emissions as N compounds are degraded in the environment (Kaye et al., 2004; Townsend-Small and Czimczik, 2010; Townsend-Small et al., 2011) and atmospheric N_2O is considered to be "the dominant ozone-depleting substance emitted in the 21st century" (Ravishankara et al., 2009). The application of fertilizers with WSN such as urea, ammonium sulfate ($(NH_4)_2SO_4$), ammonium nitrate (NH_4NO_3), and calcium nitrate ($Ca(NO_3)_2$) may significantly increase N_2O emissions compared to a non-fertilized turf (Maggiotto et al., 2000; Bergstrom et al., 2001). Nitrogen sources with similar solubility characteristics such as ($(NH_4)_2SO_4$), ($Ca(NO_3)_2$), and urea tend to result in similar N_2O emissions (Bergstrom et al., 2001; Bremer, 2006), while Maggiotto et al. (2000) found that sulfur coated urea resulted in less N_2O emissions from turf when compared with urea and NH_4NO_3. The application of high annual rates of N fertilizer (250 kg ha^{-1}) to perennial ryegrass (*Lolium perenne* L.) turf can also increase emissions of N_2O compared with lesser annual N rates (50 kg ha^{-1}) (Bremer, 2006). A majority of the studies on N_2O emissions from turf have made comparisons among synthetic N sources and scant information was available regarding emissions from organic N fertilizers in turfgrass lawn situations. Row crop production studies have compared N_2O emissions from manure and synthetic N sources, with sometimes contrasting results (Lemke et al., 1999; Mogge et al., 1999; Jarecki et al., 2009; Ding et al., 2012; Nagano et al., 2012). Unlike agricultural management though, a single application of synthetic N fertilizers to managed stands of turf is usually limited to 50 kg N ha^{-1} or less and turf is not subject to tillage practices like row crops.

Nitrogen fertilization can also pose environmental risk by enhancing NO_3 leaching. Nitrate is one of the most common groundwater contaminants in the United States (Spalding, 1993). The US Environmental Protection Agency (EPA) considers NO_3 in drinking water to be of concern to human health when maximum contaminant limit (MCL) concentrations exceed 10 mg NO_3-N L^{-1} (USEPA, 2007). Though NO_3-N concentrations in groundwater are usually less than EPA MCL in urban areas where turfgrasses are frequently used as groundcover (Petrovic 1990; Petrovic and Easton, 2005), increased NO_3 leaching may occur depending on stand age (Miltner et al., 1996; Erickson et al., 2001; Frank et al., 2006; Mangiafico and Guillard, 2006), root development (Geron et al., 1993; Bowman et al., 1998; Sullivan et al., 2000), climatic conditions (Geron et al., 1993; Duff et al., 1997; Mangiafico and Guillard, 2006), grass species (Liu et al., 1997; Bowman et al., 1998; Jiang et al., 2000), and N rate (Gross et al., 1990; Brauen and Stahnke, 1995; Frank et al, 2006). The use of mature sod in place of establishing a turf stand from seed can reduce NO_3 leaching during establishment (Geron et al., 1993). Mancino and Troll (1990) determined that fertilizer source did not affect NO_3 leaching when frequent applications of relatively low N amounts were used, 9.8 kg N ha^{-1} every 7 days or 19.5 kg N ha^{-1} every 14 days, although ($Ca(NO_3)_2$) and NH_4NO_3 fertilizers had higher leaching rates than urea, isobutylidende diurea, ($(NH_4)_2SO_4$), and urea formaldehyde when making a single application at 48.8 kg N ha^{-1}. Geron et al. (1993) found that N source did not influence NO_3 leaching when using resin-coated urea and urea fertilizers. Guillard and Koop (2004) reported more NO_3

leached when turf was fertilized with (NH_4NO_3) than with either a polymer-coated sulfur-coated urea fertilizer or organic poultry manure, with only the NH_4NO_3 differing from the non-fertilized control.

This study was designed to compare professional and do-it-yourself (DIY) approaches to home lawn care utilizing conventional or alternative products for fertilization and weed control. The first objective of the study was to compare NO_3 leaching from turf established with sod and by seeding, managed with no-fertilizer, synthetic N (conventional), or organic N (alternative) treatments. The second objective was to compare N_2O emissions among non-fertilized, conventional, and alternative N fertilized treatments. The final objective was to evaluate aesthetics and the need for pesticide applications among establishment methods and management programs. Our hypothesis was plots established by sod would reduce NO_3 leaching and would require fewer herbicide applications than the seeded treatments, alternative fertilizers would have similar NO_3 leaching and N_2O emissions to conventional fertilizers, and the conventional and alternative management programs would have similar aesthetic qualities.

MATERIALS AND METHODS

Stands of cool-season turfgrass were established by seeding or with mature sod, on 25 August 2010, at the O.J. Noer Turfgrass Research and Education Facility in Madison, WI. Soil type was a Batavia silt loam soil (fine-silty, mixed, superactive, mesic Fluvaquentic Endoaquoll) and soil test (Peters, 2013) results indicated a pH of 6.8, 33 g kg^{-1} organic matter, 58 mg P kg^{-1} (Bray P1) and 153 mg K kg^{-1} (Bray P1). Prior to establishment existing turf was killed by applying glyphosate two weeks before site preparation. In order to establish the grasses onto bare soil the dead turf was cut by a sod cutter, removed, and the soil was tilled to a depth of approximately 13 cm. Starter fertilizer (20-12-4) was applied to the bare soil prior to seeding at a rate of 245 kg ha^{-1}. The mature sod was composed of *Poa pratensis* L. and grown on a mineral soil prior to harvest. The seed mixture (Earthcarpet Madison Parks Turfgrass Mixture, Winfield Solutions LLC., P.O. Box 54589, St. Paul, MN 55164) consisted of *P. pratensis* cultivars Arrowhead, Odyssey, Orfeo, SR 2100, and Wildhorse, *Lolium perenne* L. cultivars SR 4600 and Quebec, and *Festuca rubra* L. ssp *rubra* SR 5250 was broadcast-seeded to the appropriate plots at the recommended pure live seed rate of 195 kg ha^{-1}. Seeded plots were mulched with wheat straw (*Triticum aestivum* L.; 0.3 kg m^{-2}) to aid moisture retention during establishment. Seeded and sodded plots were irrigated twice daily (0.7 cm d^{-1}) during the first 14 days after planting and once daily (0.4 cm d^{-1}) for an additional 10 days. Following the establishment period, supplemental irrigation (0.3 cm) events were used only to incorporate fertilizer treatments following each fertilization event or prior to application for treatments that required a granular postemergent herbicide. Mowing began 12 Sept. 2011 with a walk behind mulching-mower (JX 75, Deere & Co. Moline, IL) and continued once weekly at a height of 7 cm during the growing season throughout the project.

Fertilizer and Pesticide Treatments

Experimental units (1.8 m x 2.7 m) were arranged in a randomized complete block design with four replications. The treatments consisted of a five x two factorial arrangement, representing five management programs and two grass establishment methods. The management programs consisted of a control (Cont) with no inputs of fertilizers and pesticides and four fertilized treatments designed to represent the professional (pesticide formulations or products labeled for certified pesticide applicators) and DIY options that a typical homeowner would have available to care for their lawn. The conventional (Conv) and alternative (Alt) fertilizers provided a range of slow and quick release N characteristics among the programs. Fertilizer was applied at a rate of 48.9 kg N ha^{-1} per application in August (before seeding) and mid-October of 2010, and mid-May, late-August, and mid-October of 2011 and 2012 based on recommendations for the turf species and geographic location (Stier, 2000; Stier, 2001). Herbicides applications for the Pro and DIY programs were based on regular scouting and action thresholds. Herbicides were applied in May and October only when plot weed cover exceeded 5% within a treatment (estimated visually), in which case herbicide was applied to all replications within the treatment. The implementation of a 5% weed abundance treatment threshold was intended to maintain a high quality stand of turfgrass. The method of herbicide application depended on the formulation of the product. Herbicide products that required mixing with water were applied with a CO_2 backpack sprayer using Turbo FloodJet VS5 nozzles (TeeJet Technologies, Springfield, IL) at a spray volume rate of 407 L ha^{-1}. Ready for use herbicides (DIY programs) were applied utilizing the manufacturer spray bottle packaging. Granular herbicides were incorporated with N fertilizer by the manufacturer and applied with a hand-held shaker jar.

Professional program treatments

One of the programs, Pro-Conv, was developed by consulting with a lawn care provider and designed to represent a professional lawn care service program that utilized synthetic fertilizers and herbicide formulations available for use only by professional applicators. The Pro-Conv treatment utilized a granular synthetic sulfur coated urea (SCU) fertilizer (24-0-5, 10% WSN, 14% SAN) (Lesco Inc., Cleveland, OH 44114-1849) and a liquid herbicide formulation (Momentum FX^2, Lesco Inc., Cleveland, OH 44114-1849) of 44.2% a.i. 2,4-Dichlorophenoxyacetic acid (2,4-D), 3.86% a.i. triclopyr (3,5,6-Trichloro-2-pyridinyloxyacetic acid), and 4.20% a.i. fluroxypyr (1-methylheptyl ((4-amino-3,5-dichloro-6-fluoro-2-

pyridinyl)oxy) acetate) to maintain turf quality and control weeds. The seeded treatments required herbicide applications in May and Oct of 2011. Herbicide applications were not necessary for the sodded treatments. The herbicide was applied at 7.76 kg a.i. ha^{-1}, as directed by the label.

A second professional program, Pro-Alt, employed a fertilizer composed of synthetic and organic N sources in conjunction with an EPA-recognized reduced risk herbicide. The program included synthetic N, N-derived from biosolids, and N-derived from poultry manure (16-1-2.5; 13.6% WSN, 2.4% WIN) (Nutrients Plus, Virginia Beach, VA 23454) and mesotrione (2-[4-(methylsulfonyl)-2-nitrobenzoyl]-1, 3-cyclohexanedione) (Tenacity, Syngenta Crop Protection Inc., Greensboro, NC 27709) herbicide to maintain turf quality and control weeds. At the time of the initial herbicide application mesotrione was labeled for use on home lawns, although prior to the conclusion of the study the product was no longer available for use on residential turf. The seeded treatments received herbicide applications (0.2 kg a.i. ha^{-1}) during May 2011 and Oct. of 2011 and 2012. The sodded treatments received a single herbicide application (0.2 kg a.i. ha^{-1}) in May 2011.

Do-it-Yourself treatments

A DIY synthetic fertilizer (DIY-Conv) program with granular herbicide represented an option readily available at home lawn care centers. The program used a different fertilizer mixture for each season of application and included granular herbicide when weed control was needed (Table 1). This treatment utilized synthetic fertilizers composed of urea, methylenediurea, and dimethylenetriurea N sources (EC Grow Inc., Eau Claire, WI 54703). The marketed fertilizer products for this program had options for weed control with granular formulations of Trimec herbicide (PBI Gordon Corp. Kansas City, MO), composed of 2,4-D, MCPP-p (2-(4-chloro-2-methylphenoxy) propanoic acid), and dicamba (3,6-dichloro-2-methoxybenzoic acid) when weed control was necessary. One of two combinations of fertilizer and herbicide were used when weed control was necessary, a 28-0-2.5 (20.7% WSN, 3.0% WIN, 4.3% SAN) fertilizer with (0.08 g a.i. g^{-1} N) was the option for May applications and a 23-0-8 (17.0% WSN, 2.5% WIN, 3.5% SAN) fertilizer with (0.06 g a.i. g^{-1} N) was the option for October (Table 1). There were also two fertilizer formulations when weed control was not necessary. A 30-0-2.5 (22.2% WSN, 3.2% WIN, 4.6% SAN) fertilizer was the option for May and August, while a 22-0-8 (16.2% WSN, 2.4% WIN, 3.4% SAN) fertilizer was the option for October applications (Table 1).

A DIY program (DIY-Alt) employing poultry manure fertilizer and an herbicide classified by the EPA as a biopesticide was used to represent an alternative program. The program utilized a pelletized poultry fertilizer (5-1-3, 2.0% WSN, 3.0% WIN)(Chickity Doo Doo, R&J Partnership, LLC, Lake Mills, WI) listed as organic by the Organic Material Review Institute (OMRI) and 1.5% a.i. ferric hydroxyethylenediaminetriacetic acid (FeHEDTA) (Elementals lawn weed killer, The Ortho group, Marysville, OH 43040) herbicide treatment to maintain turf quality and control weeds. FeHEDTA was chosen because the active ingredient is a biopesticide and has been determined by the EPA to have low toxicity and with little danger to non-target organisms. In addition, FeHEDTA has also been approved for use on turf in Ontario, Canada when other synthetic pesticides were banned (Ministry of the Environment, 2011). The sodded treatments did not require weed control, though herbicide applications were necessary for the seeded plots during May and August of 2011 and 2012. Weed control with FeHEDTA required a subsequent application two to three weeks following the initial treatment. The

FeHEDTA was purchased ready-to-use in a hand-held spray bottle, and applications were made to cover the leaf surface of the desired weed with solution as directed by label instructions.

Data Collection

Nitrate leaching

Leachate was collected from September 2010 to September 2012, at a depth of 36 cm using low tension wick lysimeters modeled after a design used by Holder et al. (1991). Lysimeter columns were 20 cm diameter and placed in the ground prior to turfgrass establishment. Fiberglass wicking material (1.3 cm diam. x 60 cm length; Pepperell Braiding Co. Pepperell, MA. 01463) was used as a source of hydraulic tension to transport leachate from the bottom of the lysimeter to the collection bottle. The site of each lysimeters was excavated with a 31 cm post soil-auger to a depth of 100 cm. The lysimeter was set in the hole, leveled with the soil surface, and the hole was backfilled with the same soil. Leachate was collected within 24 h following rainfall events. Leachate volumes were recorded and a 50 ml sample was collected and immediately frozen until analyzed. Leachate during the winter months was collected when snowmelt was sufficient to allow access to the collection chambers. A 5 to 8 ml aliquot of leachate was analyzed for NO_3 using automated colorimetry (USEPA, 1993) at the University of Wisconsin-Madison Soil and Plant Analysis Laboratory Madison, WI. Collections of leachate were made during September 2010, February, March, April, May, June, November and December 2011, as well as March, April, May, and August of 2012. All of the experimental units of each treatment required leachate collection on each date during the first year of the study. However, during the second year of the study the leaching was more variable and on every date some experimental units of each treatment did not result in leachate for collection.

Nitrous oxide emissions

Nitrous oxide data collection began 9 May 2011 and ended 13 May 2013. Due to time considerations, the N_2O measurements were conducted only on the Cont, the Pro-Conv, and the DIY-Alt treatments established by seed. Unless snow cover was greater than approximately 4 cm which prevented access to the permanent chamber mounting brackets, measurements were conducted in the field weekly and also every 24 h for a period of five consecutive days following fertilization events. A photoacoustic gas-monitor (1412 photoacoustic field gas-monitor, INNOVA Air Tech Instruments, Denmark) was used to analyze the air within a vented static collection chamber (Hutchinson and Moiser, 1981; Bremer, 2006). The chamber was constructed from a 20 cm SDR 35 PVC STI gasketed sewer end cap (Plastic Trends, Inc. Shelby Twp., MI 48316). Two holes were drilled in the top of the chamber to accommodate a 64 mm vent and a 4 mm sampling port. To maintain a constant temperature and reduce solar heating within the chamber, the exterior of the chamber was covered with an adhesive aluminum foil (Intertape Polymer Group, Danville, VA). During collection events, the chamber was placed on a permanently established 20 cm polyvinyl chloride (PVC) bases (Bremer, 2006; Rochette et al., 2008). A 0.5 m section of Teflon tubing 4.0 mm outside diameter was used to connect the gas analyzer to the collection chamber for sampling. The gas analyzer was programmed to automatically collect measurements at 3 min intervals for 15 min (0, 3, 6, 9, 12, and 15 min), using a 5 s flush period and 5 s sample integration period for each measurement. Ambient air and soil temperature to a 7 cm depth were recorded prior to each N_2O measurement with a thermocouple thermometer (DuaLogR model 600-1050, Barnant Co. Barrington, IL 60010-1050). Soil moisture was measured in the upper 3.8 cm of the soil using time domain reflectometry (Fieldscout TDR 300 Soil Moisture meter, Spectrum Technologies Inc. Plainfield, IL 60585) and by averaging the values at three locations directly adjacent to the chamber.

Nitrogen was applied on three dates each year and N_2O measurements were conducted 5 min after the application of 0.3 cm of irrigation water used to incorporate the N fertilizer. The N_2O fluctuations were calculated with the Hutchinson Mosier Regression (HMR) package developed by Pedersen et al. (2010) within the statistical program R (R Development Core Team, 2008).

Turf Aesthetics

Turf quality data were visually assessed monthly, April to October, using a 1 to 9 scale with 1, 6, and 9 representing dead or dormant turf, acceptable turf, and exceptional turf respectively. The percent cover (0 to 100) of weeds was assessed once each spring, summer, and fall during 2011 and 2012 using visual estimation.

Data Analyses

Nitrate, weed cover, and turf quality data were analyzed using mixed model analysis (SAS Institute, 2012) to test significance of effects ($P \leq 0.05$). Fixed effects included management program, establishment method, and date of observation with block as the random effect. Multiple correlation structures were tested using repeated measures analysis. The correlation structures used for the analyses were compound symmetry (for weed cover), unstructured (NO_3), and spatial power (turf quality). Annual losses of total NO_3-N (kg ha^{-1}) and NO_3-N concentration (mg L^{-1}) from events were used in the data analysis. The volume of leachate and NO_3-N concentration were used to calculate total loss of NO_3-N of an experimental unit for each event. Annual total losses of NO_3-N were the sum of individual leaching events. The control with no N inputs (Cont) was used as a benchmark of background NO_3 losses without N fertilization. For each program, the NO_3 leaching above or below background loss was used to calculate the proportion of NO_3-N in the leachate that may have resulted from annual N applications (147 kg N ha^{-1} yr^{-1}). The number of observation dates and the size of the data set for the measurements of N_2O exceeded the capacity of the computer processor to test correlation structures for repeated measures analysis. Instead each date of observation was analyzed separately to test significance of effects ($P \leq 0.05$). On the dates when differences among treatments occurred, treatment means were separated using Tukey's HSD (Saxton, 1998; SAS Institute, 2012). Cumulative N_2O emissions among the treatments were calculated by summing the N_2O flux measurements observed during the entire course of the study. To investigate if environmental conditions impacted N_2O emissions, flux rates were compared with data from an on-site weather station (Model CR-21X data logger, Campbell Scientific, Inc., Logan, UT) for air temperature, soil temperature, and soil moisture on days of observation, as well as precipitation and evapotranspiration two and seven days prior to flux measurements. Correlations between N_2O flux measurements and each of the environmental conditions were tested using Pearson's correlation coefficients. Whenever appropriate, the significant differences ($P \leq 0.05$) among means were separated using Tukey's HSD (Saxton, 1998; SAS Institute, 2012).

RESULTS

Nitrate leaching

There was a significant difference ($P \leq 0.05$) in total NO_3-N leaching between years but not among management programs, establishment methods, or any treatment combinations (Table 2). The annual total NO_3-N leaching of the treatments for the first year and second years of the study were 7.0 and 0.8 kg NO_3-N ha^{-1} respectively. The annual total NO_3-N leaching of the Cont was not different ($P > 0.91$) than the Pro-Conv, Pro-Alt, DIY-Conv, and DIY-Alt. The background (Cont) leaching was 6.4 kg N ha^{-1} during the first year and 0.5 kg N ha^{-1} the second year (Table 3). Leaching of NO_3-N from the management programs during the first year was 5.7 to 8.9 kg N ha^{-1} and 0.4 to 1.3 kg N ha the second year. The leaching from the N applications to the Pro-Conv and DIY-Alt programs were less than the background level during the first year, though the DIY-Conv and Pro-Alt may have leached up to 1.6% of applied N (Table 3). During the second year the management programs were estimated to have lost from 0.1 to 0.5% of the N applied to the treatments (Table 3).

The majority of the variation of mean NO_3-N concentration was an effect of year, although leaching was also affected by combinations of year by management program and year by establishment method (Table 2). The mean NO_3-N concentration for all the treatments the first year was 7.0 mg L^{-1} and decreased to 4.0 mg L^{-1} during year two. The mean NO_3-N concentration of management programs of during the first year ranged from 4.6 to 9.3 mg NO_3-N L^{-1} and 2.1 to 6.4 mg NO_3-N L^{-1} in the second year. The leaching of the Pro-Alt treatment had significantly greater ($P \leq 0.05$) NO_3-N concentration during the first year (9.3 mg L^{-1}) than during year two (2.1 mg L^{-1}), although leaching from the other programs were similar to the Pro-Alt during both years. The NO_3-N concentration of leaching events from the sodded treatments during the first year (7.7 mg L^{-1}) was greater ($P \leq 0.05$) than during the second year (2.6 mg L^{-1}), although nitrate concentrations from the seeded treatments (6.3 and 5.4 mg L^{-1}, respectively) were not significantly different from the sod. Leachate collection occurred on seven dates each year of the study. Ten of the leaching events over the two years were due to rainfall (> 1.8 cm) and four events were a result of snowmelt. Eleven of the leaching events occurred between the months of November to May of 2011 and 2012. The remaining leaching events resulted from rainfall events, including less than one month after seeding in September 2010 (1.8 cm precipitation), in early June 2011 (3.6 cm precipitation), and while the turf was recovering from a period of dormancy in August 2012 (2.5 cm precipitation).

Nitrous oxide emissions

The mean N_2O fluctuations of treatments ranged from 10 to 1800 μg N_2O-N m^{-2} h^{-1} during the two-year period of measurements (Fig. 1). The measurements immediately following two of the six fertilizer events (49 kg N ha^{-1}) resulted in greater ($P \leq 0.05$) N_2O emissions from either the Pro-Conv or DIY-Alt treatments than the Cont (Table 4). A majority of the N_2O observations immediately after fertilizer applications were greater than the collections on preceding dates and tended to decline within two days after fertilization events (Fig. 1). The exception occurred following the Oct. 2011 fertilization event, when the typical decrease was observed on the second day application and then emissions increased again over the next few days (Fig. 1). Though N_2O flux was often lower during the winter months, Pearson's correlation tests indicated weak correlations between N_2O flux and soil moisture (0.132), soil temperature (0.422), air temperature (0.378), on days of measurements or the precipitation (0.127), or evapotranspiration (0.373) prior to measurements.

Nitrous oxide flux differences ($P \leq 0.05$) among the Pro-Conv, DIY-Alt, and Cont treatments occurred on 25 of the 89 observation dates (Fig. 1). On one of the dates that differences in N_2O fluxes among the treatments occurred, the N_2O from the Cont was greater than the DIY-Alt (Table 4). On 8 of the 25 dates when significant flux differences were observed, the Cont resulted in fewer emissions than both fertilized treatments (Table 4). The N_2O flux from the DIY-Alt and Pro-Conv treatments differed on three dates: August and November 2011, and March 2012 (Table 4). The cumulative totals for 89 h of N_2O flux observations during the entire study from the Pro-Conv (47.9 mg N_2O-N m^{-2}), DIY-Alt (49.7 mg N_2O-N m^{-2}), and Cont (41.2 mg N_2O-N m^{-2}) treatments were similar ($P > 0.05$).

Aesthetics

Weed cover was affected ($P \leq 0.05$) by a combination of date, management program, and establishment method (Table 5). The Cont established by seeding had a mean weed cover of over 58%, two years following establishment (Table 6). During 2011 and 2012, the Pro-Conv, Pro-Alt, DIY-Conv, and DIY-Alt managed seeded plots required repeated applications of herbicide to suppress weed populations (Table 6). The seeded plots managed with the Pro-Conv program required two applications of herbicide and reduced weed cover from 17.5 to 2.0% during the two years following establishment. The Pro-Alt and DIY-Conv treatments established by seed required herbicide applications in the spring 2011, fall 2011, and fall 2012. The formulation of the weed control product for the seeded DIY-Alt required two treatments each spring and fall, therefore eight total herbicide applications were made during 2011 and 2012 (Table 6). The establishment of plots with sod suppressed weed encroachment and the mean weed cover of the Cont program was approximately 10%, two years following establishment (Table 6). The weed cover of the Pro-Conv, Pro-Alt, DIY-Conv, and DIY-Alt established with sod had a mean cover of less than 4% throughout the course of the study (Table 6). A single replication of sodded turf in the DIY-Alt and DIY-Conv programs exceeded the weed encroachment threshold and as a result herbicides were applied to those treatments in the spring of 2011. The weed species most frequently observed in the treatments were dandelion (*Taraxacum officinale* F.H. Wigg.), common plantain (*Plantago major* L.), curly dock (*Rumex crispus* L.), Canada thistle (*Cirsium arvense* (L.) Scop.), and white clover (*Trifolium repens* L.).

The quality of the turf was affected ($P \leq 0.05$) by a combination of date, management program, and establishment method (Table 5). The quality of the treatments established by sod were of greater quality for the Pro-Conv, Pro-Alt, DIY-Conv, and DIY-Alt programs during 2011, although by May 2012 the seeded stands that received fertilizer became of similar quality to the sodded treatments (Table 7). While differences in quality between the establishment methods diminished as the stand matured, the distinction between the treatments managed with and without inputs (i.e. fertilizers, herbicides) had increased. The mean quality for the seeded Cont (1.0 to 4.8) was significantly less than all other treatments (1.0 to 9.0) on 85% of the measurement dates (Table 7). The quality of treatments that received inputs was similar on all dates within each establishment method (Table 7). Turf quality for the sodded Cont program was lower (ratings of 2.5 to 8.0) than the sodded treatments that received fertilizer applications (3.0 to 9.0) on 9 of the 15 measurement dates and were more likely to be different as the study progressed. The quality for all management programs and establishment methods was lower during periods of summer dormancy (Table 7).

DISCUSSION

Nitrate leaching

The loss of N from managed landscapes was an environmental concern prior to the development of synthetic N fertilizers (Davis, 1926). The fertilization of turf has often been regulated to minimize N loss (WDNR, 2004; State of Minnesota, 2006; State of Michigan, 2010) although turf fertilization with either the synthetic or alternative N sources in my study leached similar amounts of NO_3 when compared with the Cont. My data were consistent with others that reported organic N did not reduce NO_3 leaching when compared with other N sources (Guillard and Koop, 2004; Kussow 2008; Fetter et al, 2012). Miltner et al. (1996) used ^{15}N fertilizer to investigate NO_3 leaching and determined the majority N lost during leaching events was not from fertilization but rather from existing soil N. The leaching of NO_3-N during the current study was potentially less than 1.5% of N applied during the first year following establishment and less than 1% during the second.

The NO_3 concentration of all treatments during both years of the current study was below the EPA MCL, although some experimental units occasionally had leaching events that exceeded 10 mg NO_3-N L^{-1}. The greater concentrations of NO_3-N leached during the first year of the study compared with the second year was consistent with other research that observed immature turfgrass stands and concomitant lack of root development allowed greater potential for NO_3 leaching than mature swards (Geron et al., 1993; Miltner et al., 1996; Erickson et al, 2001). The soil disturbance that occurred during establishment of the lysimeters could have promoted mineralization and existing soil N has been reported by others to contribute a large portion of the NO_3-N in leachate (Miltner et al., 1996; Guillard and Koop, 2004). Our observation that total NO_3 leaching was not significant between establishment methods during either year of the experiment was in contrast to Geron et al. (1993). The differences could be due to the higher N fertilization rate (218 kg N ha^{-1} yr^{-1}) employed by Geron et al. (1993) that reported sod reduced NO_3 leaching during the first year but the seeded treatments had less NO_3 losses during the second year. The majority of leaching events in the current study occurred from November through March when reduced turf growth limited uptake of water and N. The periods of greater leaching were consistent with other studies that reported the largest losses of NO_3 occurred during the colder months of the year, apparently because water and N uptake by the turf was reduced (Geron et al., 1993; Mangiafico and Guillard, 2006; Kussow, 2008; Lloyd et al., 2011).

The similarity of NO_3 leaching from the Cont and fertilized treatments during this study suggest that during the two years following establishment, N leaching was not due to fertilization and limiting N loss during the colder months of the year would likely be difficult.

Nitrous oxide emissions

The rise of anthropogenic N_2O emissions is due in part to the use of synthetic and organic N fertilizers that can stimulate microbial production of N_2O (Davidson, 2009). The Pro-Conv treatment of this study rarely resulted in significantly greater N_2O emissions than the Cont and was never greater than the DIY-Alt. The greatest daily mean of N_2O-N flux observed during this study (1800 μg N_2O-N m^{-2} h^{-1}) was higher than those reported from maintained turfgrasses in Kansas, USA (400 μg N_2O-N m^{-2} h^{-1}) (Bremer, 2006) and lower than Ontario, Canada (10800 μg N_2O-N m^{-2} h^{-1}) (Maggiotto et al., 2000). The mineralization of N from soil organic matter in the current study likely contributed a majority of the emissions observed on the measurement dates, as the Cont rarely had lower N_2O flux than both the treatments receiving N fertilizer. The lack of difference among fertilized and non-fertilized treatments in our study may have been due to the relatively low rates of N application (49 kg N ha^{-1}). Other research reported N_2O emissions were of greatest concern when using high N rates (> 50 kg N ha^{-1}) and non-linear increases of N_2O emissions were often observed when an application event exceeded 150 kg N ha^{-1} (Bremer, 2006; Jarecki et al., 2009; Hobben et al., 2011). Lower N_2O emissions have been observed from high maintenance ornamental turf than lower maintenance turf of less quality in urban ecosystems of Baltimore, MD (Groffman et al., 2009). The decrease in N_2O emissions was likely due to greater root growth for the highly maintained ornamental turf which resulted in the more efficient use of N and diminished the chances of N_2O production compared to the less intensive methods.

Though we found a lack of correlation between N_2O flux and environmental variables (i.e. soil conditions, precipitation, air temperature, and evapotranspiration), the N_2O fluxes were often greatest when irrigation was applied immediately prior to measurements on fertilization dates. The increase in N_2O emissions that occurred on those dates from the Cont indicated that the impact of irrigation was more influential on emissions than N applications. Since irrigation was not routinely applied, increase in N_2O that followed the irrigation to incorporate fertilizers may have resulted from several factors. For example, soil moisture was likely a limiting factor for the microbial production of N_2O and even the minor increase in anaerobic soil conditions that occurred after irrigation events may have stimulated biological production of N_2O (Horgan et al., 2002; Kim et al., 2012). Additional information is needed for confirmation though, as Raciti et al. (2011) reported saturation of turf soil cores increased N_2O emissions in the laboratory, but not in a field setting. Another possible source of the greater flux following some fertilization events could be N_2O dissolved in the irrigation water (Ueda et al., 1993). Groundwater was used for irrigation and there exists the possibility that N_2O dissolved in the water became gaseous under the reduced pressure at the ground surface. The applications of irrigation may have filled the soil pore spaces and displaced gases into the atmosphere (Denmead et al., 1979; Smith et al., 1998). Environmental conditions have been correlated with N_2O emissions by others (Clayton et al., 1997; Raciti et al., 2011), although similar to the current study, others (Bremer, 2006; Townsend-Small and Czimczik, 2010) have determined that those conditions were not always strongly correlated with N_2O emissions.

Aesthetics

The relatively high mowing height was chosen to deter weed encroachment, although herbicide treatments were expected to eventually be necessary to maintain turfgrass quality

(Busey, 2003). Also as anticipated, all the N and herbicide programs suppressed weed populations when compared with the Cont. The two applications of herbicides utilized in the seeded Pro-Conv treatment during the first year were effective enough to maintain weed abundance below the treatment threshold for the duration of the study, while other management programs required three or more herbicide applications. The mesotrione herbicide used in the Pro-Alt program, did cause some of the turf leaf blades to turn white (photo-bleaching) and was likely one of the reasons that home lawn applications were removed from the product label. The photo-bleaching of the turf was visually noticeable only between dates of aesthetic measurements, did not impact quality ratings, and was present approximately seven days or until the foliage was removed during mowing. Reduced risk pesticides are reported to have low toxicity to non-target organisms and very few reduced risk herbicides are permitted for home lawn care applications. For those reasons, the loss of mesotrione herbicide for lawn turf was unfortunate. Though the FeHEDTA used in the DIY-Alt treatment was not considered organic, it had a low toxicity and unlike commercially available organic pesticides, it was selective for broadleaf weeds without injuring or killing the turf. Overall, the FeHEDTA herbicide was effective although it required applications on eight occasions during the study and can also be more expensive than other herbicides (e.g. 2,4-D, fluroxypyr, dicamba). During the course of our study the treatments established with a mature sod provided an effective method of controlling weed encroachment even in the absence of a fertilization programs or herbicide applications. The capacity of the sod to resist weed encroachment without supplemental N may diminish over time. For the first two years of establishment the sodded Cont resisted weed encroachment equal to or better than the seeded method, regardless of fertilizer and herbicide treatments. Further research is necessary to determine if establishing a site with mature sod would provide homeowners an option to reduce pesticide applications over a period of time longer than the two years of the current study.

The visual appearance of turfgrass stands and weed abundance are typically used to assess aesthetic qualities and effectiveness of management programs. As expected, establishment using mature sod provided better initial quality than seeding. The 18 months required for the seeded plots to develop quality comparable with the sodded treatments was approximately the duration of time for sod to mature before harvest in Wisconsin. The turf quality of the Pro and DIY management programs were comparable at the end of the current study and these results were consistent with Caceres et al. (2010). Though irrigation was not used in my study, Caceres et al. (2010) also reported acceptable quality ratings during a majority of measurements for organic and synthetic turf management programs which included irrigation. In the absence of a supplemental irrigation program, the quality of the treatments in the current study was expected to vary by date because of heat and drought stresses. During both years of the study the turfgrass entered a period of summer dormancy leading to declines in quality. Except during periods of dormancy, the quality was greater for the treatments that received fertilizer and chemical inputs than the Cont.

The use of professional and DIY programs that utilized conventional and alternative products had similar NO_3 leaching, N_2O emissions, and aesthetic qualities. The risk of NO_3 leaching was greatest during the first year following establishment and between November and May. The frequency of NO_3 leaching decreased as water demand of the turf increased and also when the stand matured following establishment. Key observations were made during the course of the N_2O data collection. First, the use of a conventional N fertilizer did not reduce or increase emissions of N_2O when compared with the alternative N fertilizer. Second, on four of the six application dates, applying 48.9 kg N ha^{-1} did not increase N_2O emissions when compared with the Cont. Third, the Cont resulted in lower N_2O emissions on only 8 of the 89 measurement dates compared to the Pro-Conv and DIY-Alt treatments, and annual total emissions were similar from both fertilized and non-fertilized turf. Finally, the use of irrigation immediately prior to N_2O measurements on fertilizer application dates appears to have stimulated N_2O emissions equally among the fertilized and non-fertilized treatments.

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Table 1. Month of application (●) and product formulations with and without herbicides used for the do-it-yourself synthetic product (DIY-Conv) treatment to manage cool-season turfgrasses established by seeding or mature sod during 2011 and 2012 in Madison, WI.

		Seeded						Sodded					
		2011			2012			2011			2012		
N-P-K	Herbicide	May	Aug.	Oct.	May	Aug.	Oct.	May	Aug.	Oct.	May	Aug.	Oct.
28-0-2.5	Yes	●						●					
30-0-2.5	No		●		●	●			●		●	●	
23-0-8	Yes			●			●						
22-0-8	No									●			●

Table 2. Fixed effects of management program, establishment method, and year on the leaching of NO_3-N from stands of cool-season turfgrasses established using mature sod or seed. Nitrate data were analyzed annually by for total NO_3-N loss (kg ha^{-1}) and also for by mean concentration of leachate (g L^{-1}).

		Total NO_3-N loss		Mean concentration NO_3-N	
Source of variation	DF[†]	F value	Pr > F	F value	Pr > F
Management program (M)	4	0.23	0.9186	0.60	0.6645
Establishment method (E)	1	0.62	0.4360	0.46	0.5007
M x E	4	0.46	0.7779	0.90	0.4781
Year (Y)	1	29.80	<0.0001	11.57	0.0019
Y x M	4	0.47	0.7559	2.98	0.0347
Y x E	1	0.16	0.6922	6.04	0.0200
Y x M x E	4	0.50	0.7378	0.39	0.8163

[†] Degrees of freedom

Table 3. Nitrogen availability and NO_3-N leaching from conventional (Conv) and alternative (Alt) fertilizer products used as part of professional (Pro) or do-it-yourself (DIY) home lawn care programs in Madison, WI. N fertilizers are presented in forms of water soluble N (WSN), water insoluble N (WIN), and slowly available N (SAN). The control with no N inputs (Cont) was used as a benchmark of background N losses without N fertilization. For each program, the NO_3 leaching above or below background loss was used to calculate the proportion of NO_3-N in the leachate that was potentially leached from N applications (147 kg N ha^{-1} yr^{-1}). Means of total NO_3-N leaching of the management programs and the control treatment were similar ($P > 0.91$).

				NO_3-N leaching			
	N availability (% of total N)			Total N loss (kg N ha^{-1})		Potential N leached from fertilizer (%)	
Management program	WSN	WIN	SAN	Year 1	Year 2	Year 1	Year 2
Pro-Conv	41.7	0.0	58.3	5.8	1.0	-0.4[‡]	0.3
DIY-Conv[†]	74.0	10.5	15.5	8.1	0.7	1.1	0.1
Pro-Alt	39.4	15	45.6	8.9	0.4	1.6	0.1
DIY-Alt	40.0	60.0	0.0	5.7	1.3	-0.4	0.5
Cont	na[§]	na	na	6.4	0.5	na	na

† Mean values were used in the table for the DIY-Conv. management program. The WSN, WIN, and SAN varied as a result of utilizing four different fertilizer formulations with and without herbicides.

‡ Negative value indicates NO_3 loss from management program was less than the background losses.

§ na, Not applied. N was not applied to the Cont.

Table 4. Dates of measurements that N_2O fluctuations were significantly different ($P \leq 0.05$) for stands of cool-season turfgrasses managed with a non-fertilized control, organic (alternative) N fertilizer, and a synthetic (conventional) N fertilizer. Nitrogen fertilizer applications resulted in differences among treatments on 24 Aug. and 22 Oct. 2011. Observations were made May 2011 to May 2013.

	N_2O flux (μg N_2O-N m^{-2} hr^{-1})		
Date	Non fertilized	Alternative N	Conventional N
24 Aug. 2011	570 a†	920 b	705 ab
26 Aug. 2011	374 a	548 b	445 ab
28 Aug. 2011	333 a	502 b	411 ab
22 Oct. 2011	290 a	535 b	635 b
24 Oct. 2011	125 a	257 b	199 ab
1 Nov. 2011	476 ab	733 b	361 a
21 Nov. 2011	37 a	89 ab	98 b
5 Dec. 2011	83 a	150 b	119 ab
25 Mar. 2012	400 a	579 b	476 a
1 Apr. 2012	163 a	250 b	230 b
23 Apr. 2012	90 a	198 b	122 ab
30 Apr. 2012	142 a	258 b	208 ab
14 May 2012	346 a	536 b	468 ab
15 May 2012	332 a	459 b	467 b
16 May 2012	214 a	429 b	460 b
4 June 2012	427 a	606 b	635 b
13 Aug. 2012	541 a	868 b	817 b
14 Aug. 2012	328 a	826 b	522 ab
15 Aug. 2012	395 a	646 b	568 ab
19 Aug. 2012	389 a	690 b	486 a
23 Sept. 2012	94 a	164 ab	248 b
30 Sept. 2012	154 b	23 a	81ab
10 Oct. 2012	100 a	176 ab	206 b
28 Apr. 2013	91 a	323 b	240 b
4 May 2013	224 a	445 b	437 b

†Values followed by letters indicate significant differences ($P \leq 0.05$) among treatments within rows.

Table 5. Fixed effects of management program, establishment method, and date for weed cover and turf quality for stands of cool-season turfgrasses during the first two years following establishment by seed or mature sod.

		Weed Cover			Turf Quality	
Source of variation	DF[†]	F value	Pr > F	DF	F value	Pr > F
Management program (M)	4	839.96	<0.0001	4	353.52	<0.0001
Establishment method (E)	1	297.85	<0.0001	1	2779.65	<0.0001
M x E	4	37.79	<0.0001	4	82.07	<0.0001
Date (D)	3	10.42	<0.0001	13	945.40	<0.0001
D x M	12	9.50	<0.0001	52	15.89	<0.0001
D x E	3	4.73	0.0052	13	181.42	<0.0001
D x M x E	12	6.72	<0.0001	52	6.46	<0.0001

[†] Degrees of freedom

Table 6. Percentage of broadleaf and grassy weed cover within stands of cool-season turfgrass. Turfgrasses were established by seeding or with mature sod and managed with four management programs and a control treatment without inputs of fertilizer or herbicide. Differences among means were determined by Tukey's HSD (Saxton, 1998) at the 0.05 probability level.

	% Weed cover							
	Established with seed				Established with sod			
Management program	May 2011	Sept 2011	May 2012	Sept 2012	May 2011	Sept 2011	May 2012	Sept 2012
Professional (Conventional)	17.5† C‡	6.8† ABC	0.3 A	2.0 AB	0.5 A	1.5 A	0.0 A	0.5 A
Professional (Alternative)	17.5† C	11.8† ABC	2.5 AB	11.3† ABC	2.8† A	0.5 A	0.5 A	0.8 A
Do-it-Yourself (Conventional)	17.5† C	15.0† BC	1.8 A	5.5† ABC	3.3† A	1.3 A	0.0 A	1.0 A
Do-it-Yourself (Alternative)	12.5† ABC	10.0† ABC	5.8† ABC	11.3† ABC	0.3 AB	2.3 AB	0.5 A	1.8 A
No input control	17.5 C	33.8 D	38.8 D	58.8 E	5.0 ABC	6.5 ABC	5.0 ABC	10.0 ABC

† Indicates weed treatment threshold was exceeded on at least one replication within the treatment and an herbicide application was made on that date to all replications within the treatment following the assessment of weed cover.

‡ Values followed by letters indicate significant differences among treatments within columns and rows.

Table 7. Visual estimates of turfgrass quality from Apr. 2011 to May 2012 for turfgrasses established by seeding (Seed) or with mature sod (Sod). Turf was managed with programs designed to represent professional (Pro) and do-it-yourself (DIY) approaches to home lawn care. Turf fertilization included conventional N (Conv) fertilizer, alternative N (Alt) fertilizer, and no fertilizer or herbicide input control (Cont). Least significant difference ($P \leq 0.05$) among the treatments, within rows was 0.5.

	Seed					Sod				
Date	Cont	Pro-Conv	Pro-Alt	DIY-Conv	DIY-Alt	Cont	Pro-Conv	Pro-Alt	DIY-Conv	DIY-Alt
Apr 2011	3.0	3.0	3.0	3.0	3.0	7.0	7.0	7.0	7.0	7.0
May 2011	3.0	5.0	5.0	5.0	5.0	6.5	7.3	7.0	7.3	7.3
June 2011	4.0	5.0	5.0	5.0	5.0	8.0	8.0	8.0	8.0	8.0
July 2011	3.0	5.0	5.0	5.0	5.0	6.0	7.0	7.0	7.0	7.0
Aug. 2011	1.0	1.0	1.0	1.0	1.0	5.0	6.0	6.0	6.0	6.0
Sept. 2011	3.0	6.3	6.5	6.0	5.8	6.0	6.8	7.0	7.0	7.0
Oct. 2011	3.0	6.3	6.5	6.0	5.8	5.5	6.5	7.0	7.0	6.8
Apr. 2012	4.8	8.0	8.3	8.3	7.5	7.5	9.0	9.0	9.0	9.0
May 2012	3.0	8.3	8.3	7.5	7.0	7.0	8.3	8.0	8.3	8.3
June 2012	3.0	8.3	8.3	7.5	7.0	7.0	8.3	8.0	8.3	8.3
July 2012	1.0	3.0	3.0	2.8	3.0	2.5	3.0	3.0	3.0	3.0
Aug. 2012	3.0	5.8	6.0	6.0	6.0	5.3	6.0	6.0	6.0	6.0
Sept. 2012	3.0	7.0	7.0	7.0	6.8	5.0	7.0	7.0	7.0	7.0
Oct. 2012	3.0	5.0	5.0	5.0	5.0	4.3	5.0	5.0	5.0	5.0

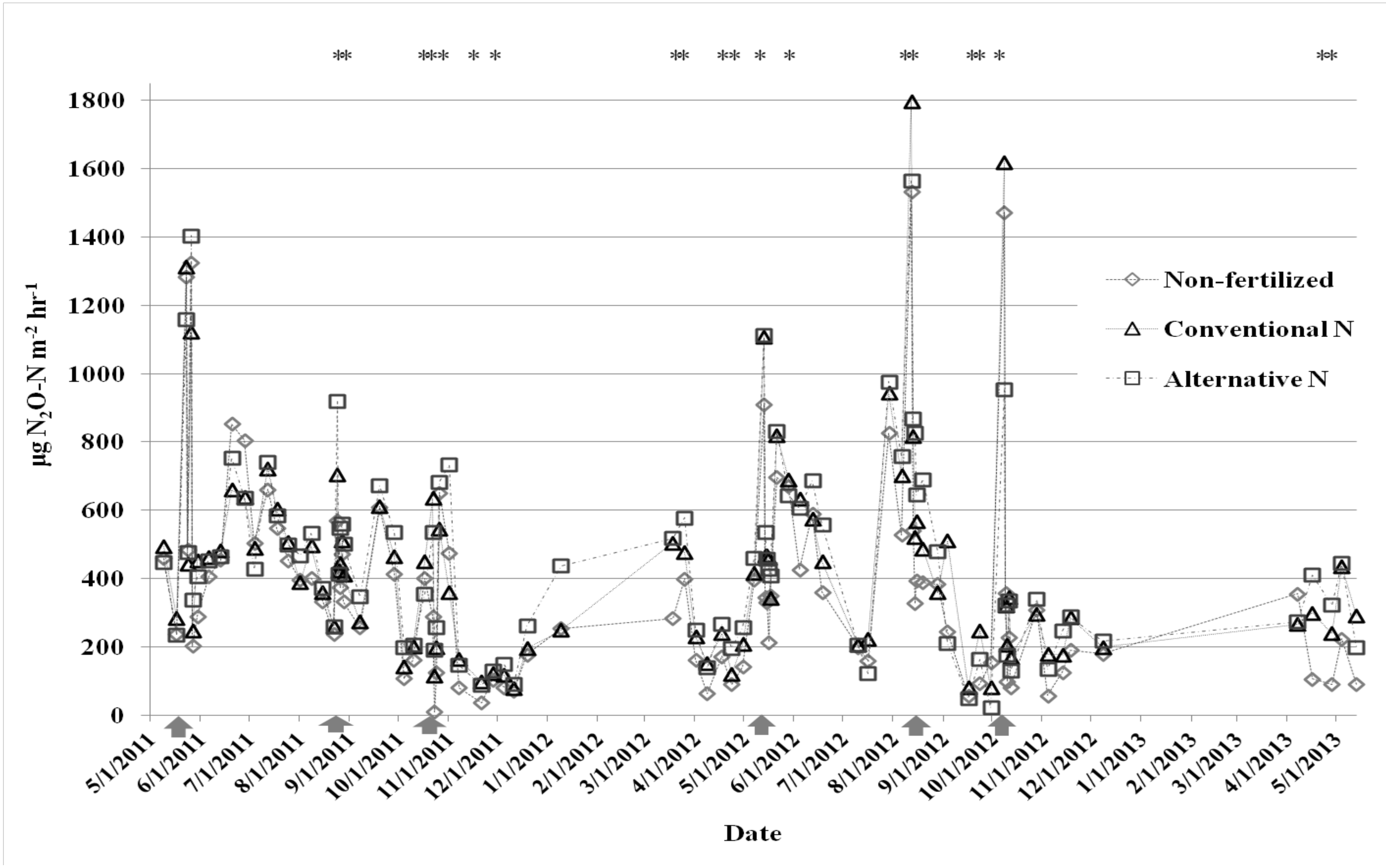

Fig. 1. Nitrous oxide fluctuations from cool-season turfgrasses managed with a non-fertilized control, a conventional N fertilizer, and an alternative N fertilizer. Measurements were collected once weekly and for five consecutive days following a fertilization event over a two year period. Significant differences ($P \leq 0.05$) among treatments are indicated by an asterisk along the top of the figure. Grey arrows indicate date of fertilizer application.

CHAPTER TWO

Change in Soil Carbon and Nitrogen

Following Establishment of Turfgrass in Topsoil and Subsoil

ABSTRACT

Turfgrasses often replace production agriculture in highly populated areas as a result of the suburban expansion in the United States. During new home construction, the topsoil has often been removed and there is an option of establishing the turfgrass on the subsoil. Establishing turfgrasses on subsoil has often been viewed as undesirable; however the subsoil lacks soil organic carbon (SOC) and may be capable of storing more atmospheric carbon than topsoil. The primary objective of the study was to compare the change in SOC and soil organic nitrogen (SON) following the establishment of turfgrass on topsoil and subsoil. The second objective was to compare N applications and irrigation on SOC and SON change in the two soil types. Treatments were arranged in a split plot design with four replications. Whole plot factor was irrigation, with fertilizer rate and soil type as subplot factors. Fixed effects of N fertilizer rate, soil type, date, and a soil type x date interaction were significant ($P \leq 0.05$) for SOC and SON. The amount of SOC and SON in the topsoil declined during the first three years following establishment. The SOC and SON of the subsoil increased during the same period. The amount of SOC and SON increased ($P \leq 0.05$) when fertilizer was applied to supply 147 kg $N \cdot ha^{-1} \cdot yr^{-1}$ when compared with no fertilizer, while irrigation did not affect SOC or SON ($P > 0.05$). The topsoil SOC decline was the first turfgrass field study to report losses of SOC following establishment.

In an effort to alleviate the rising atmospheric CO_2 levels, the planting of perennial grasses has emerged as a possible means of mitigating global warming while also improving soil quality (Bandaranayake et al., 2003; Gebhart et al., 1994; Qian et al. 2003; Schlesinger, 2000; Selhorst and Lal, 2013; Zirkle et al., 2011). Research studies have investigated the carbon sequestration potential of urban areas in which the primary vegetation was perennial lawn grasses. Results of these studies indicated that soil organic carbon (SOC) of arable lands formerly used for production agriculture can be replenished by the establishment and maintaining home lawns (Selhorst and Lal, 2013; Qian et al., 2003; Zirkle et al., 2011). The perennial nature of lawns may help replace the soil carbon that was lost due to the mechanical tillage and other practices when the sites were managed for agricultural production (Halvorson et al., 2002; Six et al., 2000).

The maintenance of turfgrass with N fertilizer facilitates the production of photosynthetic products that can accumulate in the soil as organic matter (Pouyat et al., 2009; Qian et al., 2003; Qian et al., 2010; Zirkle et al, 2011). Net ecosystem exchange (NEE) and dark respiration (Rd) have been used to estimate exchange rates of CO_2 between ecosystems and the atmosphere (Huxman et al., 2004; Bremer and Ham, 2005; Ignace et al., 2007). The rate at which turfgrass ecosystems accumulate carbon is dependent on multiple factors including disturbance events, carbon content of soil, pH, climate, nutrients, and water availability (Pouyat et. al., 2009; Qian and Follett, 2002; Qian et al., 2003, 2010; Zirkle et al., 2011). The addition of N fertilizer can increase SOC accumulation rate by 0.8 to 1.0 $Mg \cdot C \cdot ha^{-1} \cdot yr^{-1}$ (Zirkle et al., 2011). Qian et al (2003) simulated turfgrass scenarios using the CENTURY model and determined that N fertilization increases SOC and decreases the time required before the soil carbon accumulation rate reaches an asymptote known as a “steady state”. The CENTURY model also indicated that returning the grass clippings to the turf following mowing events during a period of 10 to 50 years could increase soil organic carbon between 11 and 59% compared with removing the clippings. The rate of increase observed in this scenario existed when the turf was managed with fertilizer rates of 75 and 150 $kg{\cdot}N{\cdot}ha^{-1}{\cdot}yr^{-1}$ although SOC remained relatively unchanged when clippings were removed from the 75 $kg{\cdot}N{\cdot}ha^{-1}{\cdot}yr^{-1}$ treatments. Adequate soil moisture is required for plant growth and to maximize carbon sequestration rates. The application of supplemental irrigation has been predicted to increase SOC by approximately 0.05 to 0.15 $Mg{\cdot}C{\cdot}ha^{-1}{\cdot}yr^{-1}$ (Lal, 2004).

A comparison of home lawns and native ecosystems around Denver, CO found that due to fertilization and irrigation practices turfgrass lawns greater than 25 years of age had twice as much SOC than native steppe grass soils of the surrounding area (Pouyat et al., 2009). The same study also investigated home lawns in the Baltimore, Maryland area and found SOC was equal to urban forest remnants and twice as much as rural forests. Selhorst and Lal (2013) evaluated turf sites across the United States with similar maintenance practices and reported that home lawns consisting of cool-season or warm-season grasses were increasing the SOC at a rate ranging from 0.9 to 5.4 $Mg{\cdot}C{\cdot}ha^{-1}{\cdot}yr^{-1}$. The SOC accumulation rates of the Selhorst and Lal (2013) study did not deduct the carbon equivalency (Ce) of emissions from maintenance practices but predicted the emissions, not including irrigation, to be approximately 250 $kg{\cdot}C{\cdot}ha^{-1}{\cdot}yr^{-1}$. A modeling study for home lawns with various management intensities predicted that stands of turfgrasses have a net sequestration potential of 2.5 to 20.4 $Mg{\cdot}C{\cdot}ha^{-1}{\cdot}yr^{-1}$, even after the Ce of maintenance emissions were included (Zirkle et al., 2011). The greater SOC rates accumulation rates reported with the model were for “best management practices” that received multiple fertilizer applications, regular irrigation, pest control and were mowed once each week.

Carbon accumulation rates observed in a home lawn environment are not expected to continue indefinitely and eventually plateau or achieve a “steady state” (Bandaranayake et al., 2003; Qian and Follett, 2002; Selhorst and Lal, 2013). The length of time it takes a stand of turf to reach the steady state depends on environmental factors, silt plus clay content, management inputs, and initial carbon capacity. The carbon storage capacity of turfgrass systems are expected to reach a “steady state” at approximately 40 to 80 $Mg \cdot C \cdot ha^{-1}$ at which point additions of new organic materials become equal to the losses (Bandaranayake et al., 2003; Golubiewski, 2006; Qian and Follett, 2002; Yao et al., 2009). Considering that the benefits of mitigating climate change are limited to the time period prior to turf reaching a steady state of C storage, a strong consideration should be put on the initial carbon content of soils in order to maximize the carbon sink potential of newly established home lawns. One study has shown significant increases in soil C during the first five years following turf establishment when soils had initially low C content (Yao et al., 2009).

During the construction of a new home, roadsides or other similar sites, the topsoil layer may become buried during the excavation of the foundation or relocated offsite. When the topsoil layer is no longer available for lawn establishment, a decision is required to either replace the topsoil at an additional expense or establish the turfgrass on the less desirable subsoil. Though the subsoil layers typically have less nutrients and organic matter than topsoil, the carbon deficiency may offer a greater C storage potential than topsoil. Consequently, two objectives were identified for the study. The first objective was to compare the change in SOC and SON within topsoil and subsoil of recently established stands of turfgrasses over a two year period. The second objective was to compare the effects that N applications and irrigation on SOC and SON change of the two soil types over a two year period. The null hypothesis was that

SOC and SON concentration would increase for both soils, while inputs of irrigation and N would amplify accumulation of SOC and SON.

MATERIALS AND METHODS

Establishment

A pit (approximately 0.4 m deep x 3.0 m wide x 20 m long) was excavated with a backhoe to hold constructed root zones for the study on 2 Aug. 2010, at the O.J. Noer Turfgrass Research and Education Facility in Madison, WI. Adjacent to the pit, a trench was excavated to separately stock pile the darker colored topsoil (A and E soil horizons) and the lighter colored subsoil (B soil horizons) that were used for the study. Oriented strand board partitions (1.2 m x 1.2 m wide x 0.4 m depth) were constructed within the pit to serve as experimental units, contain the soil profile, and keep the root zones between the treatments separated. The partitions were filled with either the topsoil or subsoil and four precipitation events, approximately 23 cm total, facilitated settling of the soil within the partitions. Following the rainfall events, additional soil was added to the partitions when necessary followed by repeated foot traffic to further compact the soil level with the top of the partition. The soil texture analysis (Bouyoucos, 1962) indicated that the topsoil was a loam texture (composed of 35% sand, 49% silt, and 16% clay) and the subsoil was a silt loam (19% sand, 55% silt and 26% clay). The topsoil had a pH of 7.2 at the time of establishment and consisted of 38.6 $g \cdot C \cdot kg^{-1}$ (dynamic flash combustion method; Campbell, 1992), 4.0 $g \cdot N \cdot kg^{-1}$ (dynamic flash combustion), 145 $mg \cdot P \cdot kg^{-1}$ (Bray P1), and 337 $mg \cdot K \cdot kg^{-1}$ (Bray P1). The pH of the subsoil was 5.9 at the time of establishment and consisted of 4.0 $g \cdot C \cdot kg^{-1}$, 0.5 $g \cdot N \cdot kg^{-1}$, 62 $mg \cdot P \cdot kg^{-1}$, and 129 $mg \cdot K \cdot kg^{-1}$.

Immediately prior to seeding, a starter fertilizer (20-12-4) was applied to the bare soil at a rate of 245 kg·ha^{-1}. On 27 Aug. 2010, a turfgrass seed mixture (Earthcarpet Madison Parks Turfgrass Mixture; Winfield Solutions LLC., Saint Paul, MN, USA) was applied to bare soil at a rate of 195 kg·ha^{-1} of pure live seed. The seed mixture had a germination rate of 85% and was composed of 49.2% Kentucky bluegrasses (*Poa pratensis* L. cultivars Arrowhead, Odyssey, Orfeo, SR 2100, and Wildhorse), 24.5% perennial ryegrass (*Lolium perenne* L. cultivars SR 4600 and Quebec), and 24.6% creeping red fescue (*Festuca rubra* L. ssp *rubra* SR 5250). To aid in moisture retention during establishment, a wheat straw (*Triticum aestivum* L.) mulch (0.3 kg·m^{-2}) was applied following seeding. Plots were irrigated twice daily (0.7 cm·d^{-1}) during the first 14 d of establishment (Stier, 2000) and once daily (0.4 cm·d^{-1}) for an additional 10 d. Irrigation applications after the first 24 d of establishment were conducted only for the irrigated treatments. Mowing began 16 Sept. 2011 and the plots continued to be mown with a walk behind mulching-mower (JX 75; Deere & Co. Moline, IL, USA) once a week at a height of 7 cm during the growing season of each year.

Treatments

The treatments were arranged as a two x three x two factorial arranged in a split plot, randomized block design with four replications. The whole plot factor was irrigation, with fertilizer rate and soil type as subplot factors. An onsite weather station (Model CR-21X data logger; Campbell Scientific, Inc., Logan, UT, USA) was used to measure daily precipitation and estimate evapotranspiration for determining irrigation needs. A deficit irrigation program (Fu et al., 2004) was used to investigate the impact of irrigation on carbon sequestration when compared with the non-irrigated treatment. Once weekly, 80% of the week's estimated evapotranspiration (FAO56 Penman-Monteith) minus the precipitation was returned to the irrigated treatments. Plots were irrigated using a hose and nozzle assembly regulated by an electronic water meter (TM-075; Great Plains Industries, Inc., Wichita, KS, USA) from 19 May to 30 Sept. 2011 and from 11 May to 30 Sept. 2012. Fertilizer treatments were implemented to investigate N additions effect on carbon sequestration. Three fertilizer treatments consisted of a control with no fertilizer inputs, 74 $kg \cdot N \cdot ha^{-1} \cdot yr^{-1}$, and 147 $kg \cdot N \cdot ha^{-1} \cdot yr^{-1}$. Fertilizer applications were made in Aug. (before seeding) and Oct. of 2010 and in May, Sept., and Oct. of 2011 and 2012 with a sulfur coated urea (SCU) fertilizer (24-0-6, 10% soluble urea-N, 14% slowly available urea-N) (Lesco Inc., Cleveland, OH, USA). Soil treatments were used to investigate carbon accumulation in soil types varying in initial carbon content. With the exception of the starter fertilizer prior to seeding, no additional P fertilizer was applied for the duration of the study.

Data Collection

Net ecosystem exchange (NEE) and dark respiration (Rd) of CO_2 were measured monthly Apr. through Nov. of 2011 and 2012 using an LI-6400 XT portable photosynthesis system (Licor, Lincoln, NE, USA). Two weeks prior to the first data collection sections of polyvinyl chloride (PVC) pipe (10 cm diam. x 10 cm height) were pressed into each plot leaving approximately 1.3 cm exposed above the soil surface. The PVC collars remained in the soil for the duration of the study in order to reduce soil disturbance preceding data collections and provide a permanent base for the chambers used for measurements. The measurements of NEE and Rd were collected during peak daylight hours (9:00 AM to 3:00 PM) on days with sunny to partly cloudy weather conditions. Once a month from April through November, a clear acrylic chamber attachment (10 cm diam. x 15 cm height) (Licor 6400-19 kit; Licor, Lincoln, NE, USA) was placed on the PVC bases to temporarily (4 min) enclose the turf canopy above the soil surface. Enclosing the turf canopy allowed the LI-6400XT to measure the rate of photosynthetic CO_2 fixation to calculate NEE. The Li-6400XT was programmed using the open system and settings included; a flow rate of 500 $\mu mol \cdot m^{-2} \cdot s^{-1}$, reference CO_2 at ambient level, soda lime and desiccant chambers set to scrub. Reference CO_2 values were determined during the calibration cycle by opening the clear acrylic chamber to the atmosphere and observing ambient air concentrations. To measure Rd of the canopy and soil, an opaque chamber was used in conjunction with the LI-6400XT. The soil chamber program settings included; reference CO_2 ambient level, three cycles of chamber draw down measurements, soda lime and desiccant chambers set to scrub. Because of the number of observations, two days were required to complete the data collection on all treatments. The measurements were conducted on consecutive days unless weather prohibited in which case data collection resumed when weather permitted.

Soil samples were collected prior to establishment in Aug. 2010, then twice annually (May and Oct.) during 2011 and 2012, and Aug. 2013. Soil samples collected prior to establishment were bulked by soil type. At all other sampling periods, three soil samples were randomly collected from within each experimental unit with a soil probe (2.5 cm inside diam.) to a depth of 13 cm. The samples from each experimental unit were placed in a plastic bag and stored individually in a freezer at -10°C until processing and analysis. To prepare the soil for analysis the samples were allowed to thaw at room temperature, air dried for 48 h, ground with a ceramic mortar and pestle until all particles could pass through a 2 mm sieve, and then oven-dried for 24 h at 105°C. The oven dried samples were placed in 2 mL microcentrifuge tubes along with two stainless steel ball bearings and shaken vigorously using a modified paint can shaker (Fleming Gray Ltd., Saint George, Ontario, Canada) as an agitating device for 15 min to further pulverize the soil. A representative soil sample (8 to 10 mg) was weighed, rolled in a tin sample container (CE Elantech, Lakewood, NJ, USA), and stored in a desiccation chamber until they could be analyzed. Total C and N were determined by combustion method using a Carlo Erba 1500 Series 2 Elemental Analyzer (CE Elantech, Lakewood, NJ, USA).

Bulk density was measured in May 2011, using a single soil core (7.5 cm·diam. x 7.5 cm·depth) from each experimental unit. The soil cores were oven dried at 105° C for 48 h then weighed to determine bulk density. The bulk densities of the topsoil and subsoil were 1.38 g cm^{-3} and 1.45 g cm^{-3} respectively. An additional soil core (5 cm·diam. x 13 cm·depth) was also collected in Aug. 2013 from the non-fertilized and 73.5 kg·N treatments of each soil type and irrigation strategy to compare root masses, although due to time constraints the 147 kg·N treatments were not collected. To aid in the separation of roots from soil, each core was submersed in a container filled with a sodium hexametaphosphate solution (50 g·L^{-1}) for six days. The core was then placed in a 1 mm soil sieve under running tap water to until the soil was separated from the roots. The roots that were retained in the sieve were collected, oven dried at 50˚ C for 48 h, and weighed to determine root mass. The application of fertilizer and irrigation did not significantly ($P > 0.1$) affect root masses of the soil treatments and data were not included in the results.

Soil conditions were monitored throughout the study using Campbell Scientific model 107-L temperature sensors and model CS616-L (Campbell Scientific, Inc., Logan, UT, USA) water content (time-domain) reflectometers that were buried horizontally at a 10 cm depth prior to turf establishment. Because of the cost of the equipment, sensors were not employed to observe soil conditions in all treatments. We used four sensors of each type within a replication to monitor soil type and irrigation regime (irrigated subsoil, irrigated topsoil, non-irrigated subsoil, and non-irrigated topsoil). The sensor wires were buried underground, run to a central location, and connected to a model CR-10 measurement and control data logger (Campbell Scientific, Inc., Logan, UT, USA). The control unit was programmed to collect measurements at two hour intervals and data were downloaded monthly. Power supply failures from 29 June 2011 to 18 July 2011 and 25 Aug. 2012 to 2 Oct. 2012, resulted in two periods of time that soil temperature and moisture data were unavailable.

Turf quality was assessed monthly, April through October, starting in Aug. 2010 and ending Oct. 2012. Quality was rated visually using a 1 to 9 scale with 1, 6, and 9 representing dead or dormant turf, acceptable turf, and exceptional turf respectively.

Data Analysis

Measurements of SOC, SON, NEE, Rd, soil temperature, soil moisture, and turf quality were analyzed using mixed model analysis of SAS to test significance of treatment effects. Fixed effects included fertilizer rate, soil type, irrigation, date of observation, and date x irrigation; the random effect was block. Multiple correlation structures were tested using repeated measures analysis to determine the best fitting among the temporal correlations. Akaike information criterion (AIC) was used to determine which model was selected for the repeated measures analysis. The correlation structures used for the analyses were compound symmetry (monthly soil moisture and monthly soil temperature), heterogeneous compound symmetry (SOC), unstructured (SON), and spatial power (NEE, Rd, and turf quality). Linear regression was conducted on SOC and SON to model the rate of change for increasing N rates and time after establishment (R Development Core Team, 2008). Significant differences among means were separated using Tukey's HSD at the 0.05 probability level (SAS Institute, 2012).

RESULTS

Fixed effects of N fertilizer rate, soil type, date, and a date x soil type interaction were significant ($P \leq 0.05$) for the accumulation of SOC and SON (Table 1). Most of the variation in SOC among the model effects was caused by soil type (Table 1). The mean SOC content across all dates of observation for topsoil was 29.5 $mg \cdot C \cdot g^{-1}$ and 6.80 $mg \cdot C \cdot g^{-1}$ for the subsoil. The SOC content of the topsoil decreased approximately 40% from the time of establishment (38.6 $mg \cdot C \cdot g^{-1}$) to the 35th month (22.7 $mg \cdot C \cdot g^{-1}$) measurements (Fig. 1). The SOC content of the subsoil increased between the time of establishment (3.99 $mg \cdot C \cdot g^{-1}$) and the 35th month (5.79 $mg \cdot C \cdot g^{-1}$) sampling date (Fig. 1). Soil organic carbon increased linearly ($y = 16.5x + 0.788$; $P \leq 0.001$) with N rate. The 0.0, 73.5, and 147 $kg \cdot N \cdot ha^{-1} \cdot yr^{-1}$ treatments had mean SOC across all sampling dates of 17.3, 18.0, and 19.0 $mg \cdot C \cdot g^{-1}$ respectively.

The majority of the variation in effects for SON was also due to soil type (Table 1). Mean SON content across all sampling dates was 2.91 $mg \cdot N \cdot g^{-1}$ in the topsoil and 0.68 $mg \cdot N \cdot g^{-1}$ in the subsoil. The SON content of the topsoil decreased ($P \leq 0.05$) by 40% from the time of establishment (4.02 $mg \cdot N \cdot g^{-1}$) to the 35th month (2.26 $mg \cdot N \cdot g^{-1}$) (Fig. 2). The SON content of the subsoil increased ($P \leq 0.05$) 50% between the time of establishment (0.475 $mg \cdot N \cdot g^{-1}$) and 35th month (0.73 $mg \cdot N \cdot g^{-1}$) sampling date (Fig. 2). Soil nitrogen increased linearly ($y = 1.59x + 0.0996$; $P \leq 0.001$) with N rate. The 0.0, 73.5, and 147 $kg \cdot N \cdot ha^{-1} \cdot yr^{-1}$ treatments had mean SON across all sampling dates of 1.71, 1.76, and 1.91 $mg \cdot N \cdot g^{-1}$ respectively.

Net ecosystem exchange of CO_2 was affected by a combination of date, soil type, and irrigation, though a majority of the variation was due to soil type (Table 1). Net ecosystem exchange of the treatments varied, with losses of carbon from the stands of turf as much as -0.8

$\mu mol \cdot CO_2 \cdot m^{-2} \cdot s^{-1}$ and carbon assimilation rates up to 9.7 $\mu mol \cdot CO_2 \cdot m^{-2} \cdot s^{-1}$ (Table 2). Net ecosystem exchange was less for the non-irrigated topsoil when compared with the irrigated topsoil during Aug. 2011, July, Aug., Oct., and Nov. of 2012. Irrigation also increased NEE for the irrigated subsoil compared with the non-irrigated subsoil during June, July, Aug., and Oct. of 2012 (Table 2). The irrigated topsoil treatment had a greater NEE than the irrigated subsoil during 12 of the 16 months of measurement (Table 2). The non-irrigated topsoil had a greater NEE than the non-irrigated subsoil during 11 months of the study. The minimum, maximum, and average photosynthetically active radiation (PAR) observed on measurement dates during 2011 were 143.9, 1962, and 1176 $\mu mol \cdot m^{-2} \cdot s^{-1}$ and 154.5, 2139, and 1095 $\mu mol \cdot m^{-2} \cdot s^{-1}$ in 2012. The lower PAR values observed during measurements of NEE occurred intermittently during periods cloud cover. Atmospheric CO_2 levels observed during measurements ranged from 370 to 400 $mg \cdot L^{-1}$. Low and high daily temperatures were lower in 2011 (5.2 and 29˚C) than during 2012 (8.7 to 36˚C).

Dark respiration of CO_2 was affected by a combination of date, soil type, and irrigation, although a majority of the variation was due to soil type (Table 1). The greatest losses of CO_2 from the soil from Rd was -18 $\mu mol \cdot CO_2 \cdot m^{-2} \cdot s^{-1}$ during May 2012 from the irrigated topsoil (Table 3). The least rate of Rd carbon loss from the soil was -1.5 $\mu mol \cdot CO_2 \cdot m^{-2} \cdot s^{-1}$ from the non-irrigated subsoil in July 2012 (Table 3). The loss of soil carbon during Rd was greater from the irrigated topsoil than the non-irrigated topsoil treatments during five months of the summer and fall of 2011 and 2012 (Table 3). Dark respiration of the irrigated topsoil was equal to the non-irrigated topsoil during every month except Aug. 2011. Dark respiration of the irrigated subsoil treatments resulted in greater losses of CO_2 than non-irrigated subsoil during Aug. 2011 and July through Oct. 2012. During Nov. 2012, the non-irrigated subsoil had greater Rd than the irrigated subsoil treatments (Table 3).

Soil type, date, and fertilizer rate accounted for a majority of the variation of effects for turf quality (Table 4). The turf quality was also affected by combinations of date, fertilizer rate, and soil type. The lowest turf quality measurements for each of the treatments on both soil types occurred during Aug.2011 and June to Oct. 2012 (Table 5). The non-fertilized subsoil had poor turf quality on all dates of the study, though the 147 $kg\cdot N\cdot ha^{-1}$ subsoil treatments were sometimes comparable to the topsoil treatments with or without N applications. When comparing the 147 $kg\cdot N\cdot ha^{-1}$ treatments, turf in the subsoil had lower quality than the topsoil in Oct. 2012 after irrigation had stopped for the year and during periods when high temperatures affected quality of all treatments (Table 5). The 0.0 $kg\cdot N\cdot ha^{-1}$ topsoil treatments had lower quality than the 147 $kg\cdot N\cdot ha^{-1}$ topsoil during 2011 but the treatments were equal on all dates during 2012 (Table 5).

The deficit irrigation program used to replace 80% ET totaled 27.1 cm H_2O during 2011 and 49.0 cm during 2012. The total volume of irrigation applied during 2011 and 2012 was 2504 $kL\cdot ha^{-1}$ and 4702 $kL\cdot ha^{-1}$ respectively. Soil moisture was affected by a combination of date, irrigation, and soil type, although the date of measurement and soil type each had a large effect on the variation in soil moisture (Table 6). The monthly mean soil moisture of the silt loam textured subsoil varied from 0.188 to 0.409 $cm^3\cdot cm^{-3}$ and was greater than the topsoil soil moisture (0.127 to 0.318 $cm^3\cdot cm^{-3}$) during all months of the study (Table 7). The irrigated subsoil had greater mean monthly water content than the non-irrigated subsoil during all months of the study (Table 7). The non-irrigated subsoil had soil water content equal to the irrigated topsoil on two dates during 2011 (Sept. and Oct.) and two dates in 2012 (June and Oct.) (Table

7). The irrigated topsoil had greater water content than the non-irrigated subsoil during July 2011, Aug. 2011, and July 2012. The non-irrigated topsoil had the lowest soil water content on four dates in 2011 and six dates in 2012 (Table 7). Data were not collected to determine the wilting point of the turf.

The temperature of the soil was affected by a combination of date, irrigation, and soil type, although a majority of the variation among effects was due to date (Table 6). The mean monthly soil temperatures of the treatments varied from -0.1°C in Jan. 2012 to 27°C in July of 2011 and 2012 (Table 8). There were differences in soil temperature among the soil treatments during Aug. 2010, July, Nov. and Dec. 2011, and Feb., June, July and Nov.2012 (Table 8).

DISCUSSION

Soil carbon and nitrogen

The subsoil and topsoil were chosen for the study primarily for the difference in SOC. Soil organic carbon and SON accumulated in the subsoil following the establishment as expected, although SOC and SON in the topsoil declined 40% within three years of establishment, not supporting the null hypothesis. The increases in SOC in the subsoil were consistent with the accumulation of SOC reported for mature turfgrass that were not recently established (Pouyat et al., 2009; Qian and Follett, 2002; Qian et al., 2010; Zirkle et al., 2013). The losses of SOC and SON in the topsoil were likely a result of the mechanical manipulation of the soil that occurred multiple times during the site construction. Losses of SOC following disturbance events are supported by Six et al. (2000) that observed lower SOC in agriculture fields managed with conventional tillage than those of no-tillage methods. The losses of SOC and SON of the topsoil could be explained by the greater Rd measurements when compared with the subsoil. Though the NEE of the turf grown on both soils was positive during most of the measurements, the greater Rd from the topsoil could have resulted in significant losses of C during the periods when low light conditions limited photosynthesis and reduced NEE. The increased Rd of the topsoil was likely facilitated by the disruption of soil structure during the construction and establishment processes. The soil disturbance exposed soil aggregates that previously protected SOC and SON from mineralization, increased gas exchange within the soil, and stimulated microbial activity (Qian et al., 2003; Six et al., 2000). The microbial mineralization of C is dependent on N and likely contributed to the decrease in N content as well. Though no field studies for turf have reported declines in SOC and SON following establishment, the CENTURY model suggested losses of SOC can persist for up over five years for stands of turf due to soil disturbances (Qian et al., 2003). The extensive manipulation of the soils during my study was analogous to the establishment processes used at many construction sites, including new homes, roadsides, and golf courses. The losses of SOC and SON from the topsoil in my study would likely have resulted in some NO_3 leaching, N_2O emissions, and contributions to atmospheric CO_2. In the short term, the losses of C and N from the topsoil following establishment were counterproductive to maximizing the environmental benefits of turfgrasses.

Nitrogen fertilization

The increase of SOC and SON that occurred in my study when N was applied to turf, supported the null hypothesis. The increase of SON from N applications was also reported by others (Frank et al., 2006; Miltner et al., 1996). Nitrogen fertilization contributed to an increase of SOC in my study and the results were supported by the modeling of SOC accumulation in golf course soils of Colorado (Qian et al., 2003) and home lawns across the United States (Zirkle et al., 2011). In conjunction with other routine maintenance, N fertilization has been demonstrated to increase soil organic matter (i.e. SOC and SON) accumulation in turfgrass systems (Pouyat et al., 2009; Qian et al., 2003; Qian et al., 2010; Zirkle et al., 2011). Nitrogen fertilization promotes production of plant biomass, which can eventually become soil organic matter (Qian et al., 2010). Accumulation of soil organic matter increases cation exchange capacity and the capacity of the soil to accommodate N.

The topsoil had greater quality ratings than the subsoil on many dates, even in the absence of N applications, likely because the organic matter served as a reserve of nutrients that were mineralized during the study. The lower quality of the subsoil turf detracted from the appeal as a medium for C storage. The mediocre aesthetics may have been caused by N deficiency that was not limiting for the turf grown in the topsoil. Additional N applications would have likely improved the quality of turf grown on the subsoil. The lower quality may also have been a result of the lower pH of the subsoil. The pH of the subsoil was slightly acidic, though at the lower range for growing turf (Stier, 2000). The low pH may have contributed to the lower turf quality. Soil P was determined to be satisfactory for turfgrasses prior to establishment (WDNR, 2006); however, no further soil testing was conducted in subsequent years and no additional P was applied during fertilization. The turf quality of the non-fertilized topsoil was greater than expected and better than the subsoil treatments during a majority of the first 18 months following establishment. The greater turf quality of the non-fertilized topsoil was likely due to increased N availability from mineralization of soil N and resulted in a reduction of N possibly lost into the environment.

Irrigation

Irrigation was predicted to increase SOC and SON and the lack of significance was unexpected. The results in the current study were contrary to others that reported irrigation increased the rate of SOC accumulation (Qian et al., 2010; Zirkle et al., 2011), although the temperate climate at the location of my study could have made irrigation less essential than drier or warmer sites. Modeling of home lawn carbon sequestration by Zirkle et al. (2011), using data from numerous locations in the United States, suggested irrigation would promote SOC accumulation in turf by up to 10 $g \cdot C \cdot m^{-2} \cdot yr^{-1}$. Qian et al. (2010) reported gains in SOC occurred during the four years following establishment for all turfgrasses and the greatest rates of change occurred when irrigation was applied. Increased rates of NEE and Rd for the irrigated treatments in my study resulted in net changes of SOC and SON that were similar to the non-irrigated treatments. In contrast to Qian et al. (2010), the turf established on the topsoil in my study lost C following establishment. The differences in SOC change between the results this study and Qian et al. (2010) may be due to the establishment methods and soil type. Qian et al. (2010) did not describe the intensity of the establishment methods that could be compared with my study in Wisconsin, although the more southern location of Nebraska had a longer growing season to accumulate C and the higher clay content of the silty clay loam soil could have helped to stabilize C, reducing mineralization (Six et al., 2002). The effect of irrigation in my study may also be masked by the losses SOC and SON of the topsoil caused by the soil disturbance of establishment. Irrigation may have a greater effect on SOC and SON as the turf stand matures or during years without adequate rainfall, although SOC accumulation would likely occur even in the absence of irrigation (Selhorst and Lal, 2013).

The SOC content of the soil prior to establishment can determine the C storage potential during subsequent years. There are a number of examples in the literature that SOC and SON increases during a period of many years after turf establishment, though the topsoil of my study was an atypical example of loses of SOC and SON during the short term. The amount of SOC and SON lost from the topsoil after establishment may take decades or longer to be replaced by the turf. The changes observed in the topsoil could present environmental concerns depending on the extent of NO_3 leaching, N_2O, and CO_2 emissions. The subsoil did have aesthetic quality deficiencies that would need to be addressed in order to be considered high quality turf, but my study suggests that the use of C deficient soils could reduce C loss following establishment. In addition, the lack of C in the subsoil suggests that the C deficient soils may provide an environmental benefit of sequestering more atmospheric carbon prior to reaching the “steady state”, than soils with greater initial SOC content.

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Table 1. Fixed effects of fertilizer rate, soil type, irrigation, and date on soil organic carbon (SOC) and nitrogen (SON), as well as net ecosystem exchange (NEE) of CO_2, and dark respiration (Rd) of CO_2 from cool-season turfgrass established on two soils types and managed with three N fertilizer rates in Madison, WI.

		SOC		SON			NEE		Rd	
Source of variation	DF[z]	F value	Pr > F	F value	Pr > F	DF	F value	Pr > F	F value	Pr > F
Fertilizer rate (F)	2	3.40	0.0356	6.23	0.1430	2	27.05	<0.0001	29.21	<0.0001
Soil type (S)	1	2048	<0.0001	1982	<0.0001	1	70.36	<0.0001	286.0	<0.0001
F x S	2	0.24	0.7862	1.46	0.7535	2	12.81	<0.0001	6.86	0.0014
Irrigation (I)	1	0.04	0.8429	0.01	0.4830	1	19.00	0.0001	17.84	0.0242
F x I	2	1.73	0.1802	1.89	0.0798	2	0.14	0.8731	6.54	0.0019
S x I	1	0.37	0.5421	0.01	0.5239	1	0.86	0.3609	5.26	0.0231
F x S x I	2	0.55	0.5758	0.23	0.2535	2	0.53	0.5952	0.13	0.8752
Date (D)	4	17.10	<0.0001	8.79	<0.0001	15	80.94	<0.0001	42.38	<0.0001
D x F	8	1.50	0.1740	1.09	0.2518	30	1.91	0.0132	1.74	0.0119
D x S	4	7.80	<0.0001	16.31	<0.0001	15	4.68	<0.0001	3.52	<0.0001
D x F x S	8	0.64	0.7451	0.56	0.6843	30	1.17	0.2877	1.31	0.1374
D x I	4	0.22	0.9256	0.22	0.8832	15	7.25	<0.0001	5.84	<0.0001
D x F x I	8	1.12	0.3604	0.85	0.6623	30	1.05	0.4171	0.93	0.5773
D x S x I	4	0.43	0.7879	0.75	0.5523	15	2.69	0.0026	3.26	<0.0001
D x F x S x I	8	0.10	0.9990	0.19	0.8838	30	0.88	0.6498	0.48	0.9907

[z] Degrees of freedom

Table 2. Net ecosystem exchange of CO_2 from stands of cool-season turfgrass established on two soil types and managed with irrigation or without irrigation in Madison, WI. Irrigation was applied from May through Sept. of 2011 and 2012. Positive NEE values indicate CO_2 was assimilated by the turf and negative values indicate respiration of CO_2.

	Net ecosystem exchange (μmol CO_2 m^{-2} s^{-1})			
	Topsoil		Subsoil	
Date	Irrigated	Non-irrigated	Irrigated	Non-irrigated
2011				
Apr.	7.7 a[z]	6.9 a	4.8 b	5.0 b
May	6.5 ab	7.5 a	5.3 b	5.3 b
June	5.3 a	4.7 a	3.9 a	4.0 a
July	2.6 a	2.7 a	2.3 ab	0.6 b
Aug.	1.6 a	-1.1 b	0.9 a	0.2 ab
Sept.	4.6 a	5.0 a	2.7 b	2.6 b
Oct.	7.1 a	6.9 a	5.7 ab	5.1 b
Nov.	5.4 a	5.8 a	3.7 b	4.1 b
2012				
Apr.	8.1 a	8.0 a	6.2 b	5.8 b
May	9.7 a	9.4 a	5.8 b	6.8 b
June	5.0 a	3.6 ab	2.6 bc	1.6 c
July	1.8 a	-0.8 b	1.4 a	-0.4 b
Aug.	6.6 a	4.7 b	4.3 b	0.0 c
Sept.	5.7 a	5.7 a	2.5 b	2.6 b
Oct.	5.5 a	0.7 c	3.5 b	0.3 c
Nov.	8.4 a	3.3 b	2.9 b	3.2 b

[z] Lower case letters indicate separation of means in rows ($P \leq 0.05$).

Table 3. Dark respiration of CO_2 from stands of cool-season turfgrass established on two soil types and managed with irrigation or without irrigation in Madison, WI. Irrigation was applied from May through Sept. of 2011 and 2012. Negative values indicate the respiration of CO_2 into the atmosphere when turf was subject to dark conditions under an opaque chamber.

	Dark respiration (μmol CO_2 m^{-2} s^{-1})			
	Topsoil		Subsoil	
Date	Irrigated	Non-irrigated	Irrigated	Non-irrigated
2011				
Apr.	-12 a[z]	-12 a	-3.0 bc	-6.5 b
May	-9.9 a	-9.6 a	-12 b	-6.1 b
June	-14 a	-14 a	-11 ab	-9.8 b
July	-14 a	-13 ab	-11 b	-11 b
Aug.	-18 a	-9.7 bc	-9.8 b	-6.8 c
Sept.	-16 a	-15 a	-6.8 b	-6.4 b
Oct.	-10 a	-10 a	-6.4 b	-7.0 b
Nov.	-6.4 a	-5.5 ab	-3.4 b	-3.2 b
2012				
Apr.	-11 a	-12 a	-7.8 b	-7.1 b
May	-18 a	-16 ab	-14 bc	-13 c
June	-12 a	-10 ab	-7.3 b	-6.8 bc
July	-13 a	-2.6 c	-7.1 b	-1.5 c
Aug.	-16 a	-16 a	-15 a	-8.9 b
Sept.	-14 a	-10 b	-8.3 b	-5.0 c
Oct.	-7.4 a	-3.0 bc	-4.6 b	-1.6 c
Nov.	-18 a	-12 b	-6.2 c	-9.9 b

[z] Lower case letters indicate separation of means in rows ($P \leq 0.05$).

Table 4. Fixed effects of fertilizer rate, soil type, irrigation, and date on the visual quality of cool-season turfgrass established on two soil types, with three N fertilizer rates, and with or without irrigation in Madison, WI.

		Quality	
Source of variation	DF[z]	F value	Pr > F
Fertilizer rate (F)	2	101.99	<0.0001
Soil type (S)	1	331.20	<0.0001
F x S	2	56.75	<0.0001
Irrigation (I)	1	18.37	0.0219
F x I	2	1.15	0.3234
S x I	1	3.54	0.0645
F x S x I	2	1.11	0.3362
Date (D)	12	257.39	<0.0001
D x F	24	5.88	<0.0001
D x S	12	27.08	<0.0001
D x F x S	24	3.67	<0.0001
D x I	12	17.19	<0.0001
D x F x I	24	0.74	0.8059
D x S x I	12	1.26	0.2420
D x F x S x I	24	0.64	0.9049

[z] Degrees of freedom

Table 5. Means of turf quality ratings (1.0-9.0) for stands of cool-season turfgrasses established on two soil types and managed with 0.0 kg N ha^{-1} yr^{-1} fertilizer (No N), 73.5 kg N ha^{-1} yr^{-1} (Low N), and 147 kg N ha^{-1} yr^{-1} (High N) in Madison, WI. Ratings of 1, 6, and 9 indicate dead or dormant turf, acceptable turf, and exceptional turf quality respectively.

	Turf quality					
	Topsoil			Subsoil		
Date	No N	Low N	High N	No N	Low N	High N
2011						
May	4.6 bc[z]	5.1 ab	5.5 a	2.3 d	4.0 c	4.6 bc
June	6.5 b	7.1 ab	7.5 a	2.0 d	4.8 c	5.4 c
July	6.5 b	7.1 ab	7.5 a	2.0 d	4.8 c	5.4 c
Aug.	3.8 b	4.3 abc	5.0 a	1.4 d	3.4 c	3.5 bc
Sept.	6.5 b	7.1 ab	7.5 a	2.0 d	4.8 c	5.4 c
Oct.	6.5 b	7.1 ab	7.5 a	2.0 d	4.8 c	5.4 c
2012						
Apr.	8.7 a	9.0 a	9.0 a	2.5 d	6.5 c	7.5 b
May	8.3 ab	8.6 a	9.0 a	2.1 c	7.1 b	8.4 a
June	3.5 a	3.5 a	3.5 a	1.1 b	3.5 a	3.5 a
July	3.5 a	3.5 a	3.5 a	1.1 b	3.5 a	3.5 a
Aug.	4.0 a	3.8 a	4.1 a	1.5 b	3.6 a	3.8 a
Sept.	4.8 a	4.4 a	4.9 ab	2.1 c	4.0 b	4.4 ab
Oct.	4.8a	4.4a	4.9ab	2.1 c	4.0b	4.4ab

[z] Lower case letters indicate separation of means in rows ($P \leq 0.05$).

Table 6. Fixed effects of soil type, irrigation, and date on soil moisture and soil temperature from plots established with cool-season turfgrass on two soils types in Madison, WI.

		Soil moisture		Soil temperature	
Source of variation	DF[z]	F value	Pr > F	F value	Pr > F
Soil (S)	1	41.77	0.0007	12.06	0.0133
Irrigation (I)	1	14.42	0.0090	0.18	0.6845
I*S	1	1.16	0.3236	6.30	0.0459
Date (D)	21	73.16	< 0.0001	3733.42	< 0.0001
D*S	21	6.34	< 0.0001	1.18	0.2682
D*I	21	22.30	< 0.0001	1.61	0.0484
D*I*S	21	2.24	0.0018	2.25	0.0018

[z] Degrees of freedom

Table 7. Mean monthly soil moisture content (7 cm depth) of loam topsoil and silt loam subsoil established with cool-season turfgrass that were managed with irrigation and without irrigation.

	Soil volumetric water content ($cm^3 \cdot cm^{-3}$)			
	Topsoil		Subsoil	
2010	Irrigated	Non-irrigated	Irrigated	Non-irrigated
Aug.	0.269 c[z]	0.269 c	0.420 a	0.342 b
Sept.	0.303 c	0.313 c	0.440 a	0.371 b
Oct.	0.206 c	0.208 c	0.334 a	0.274 b
2011				
Mar.	0.295 c	0.313 c	0.379 a	0.353 b
Apr.	0.304 c	0.316 c	0.409 a	0.371 b
May	0.230 c	0.236 c	0.345 a	0.306 b
June	0.267 c	0.265 c	0.360 a	0.328 b
July	0.271 b	0.142 d	0.352 a	0.221 c
Aug.	0.293 b	0.155 d	0.398 a	0.232 c
Sept.	0.302 b	0.210 c	0.407 a	0.291 b
Oct.	0.286 b	0.230 c	0.379 a	0.305 b
Nov.	0.318 c	0.281 d	0.392 a	0.345 b
Dec.	0.329 b	0.293 c	0.374 a	0.339 b
2012				
Jan.	0.314 a	0.277 b	0.320 a	0.301 ab
Feb.	0.318 c	0.288 d	0.407 a	0.362 b
Mar.	0.318 c	0.288 d	0.407 a	0.362 b
Apr.	0.281 c	0.257 c	0.391 a	0.334 b
May	0.262 c	0.231 d	0.358 a	0.301 b
June	0.217 b	0.127 c	0.284 a	0.188 b
July	0.291 b	0.189 d	0.323 a	0.265 c
Aug.	0.261 c	0.202 a	0.383 a	0.294 b
Sept.	na[y]	na	na	na
Oct.	0.286 b	0.218 c	0.367 a	0.288 b
Nov.	0.294 b	0.247 c	0.366 a	0.316 b

[z] Lower case letters indicate separation of means in rows ($P \leq 0.05$).
[y] na, Not available, data were missing for Sept. 2012.

Table 8. Mean monthly soil temperature (7 cm depth) of loam topsoil and silt loam subsoil established with turfgrass. Turfgrasses were managed with irrigation and without irrigation. Irrigation was applied from May through Sept. of 2011 and 2012.

	Soil temperature (°C)			
	Topsoil		Subsoil	
2010	Irrigated	Non-irrigated	Irrigated	Non-irrigated
Aug.	24 b[z]	24 b	24 b	26 a
Sept.	18 a	18 a	18 a	18 a
Oct.	13 a	13 a	13 a	13 a
2011				
Mar.	2.7 a	2.9 a	2.7 a	2.6 a
Apr.	7.5 a	7.5 a	7.7 a	7.8 a
May	14 a	14 a	14 a	14 a
June	20 a	20 a	20 a	20 a
July	26 b	26 b	26 b	27 a
Aug.	23 a	23 a	23 a	23 a
Sept.	18 a	18 a	18 a	18 a
Oct.	12 a	12 a	12 a	12 a
Nov.	5.6 a	5.8 a	5.8 a	5.2 b
Dec.	1.4 ab	1.7 a	1.6 ab	1.1 b
2012				
Jan.	0.1 a	0.4 a	0.4 a	-0.1 a
Feb.	8.2 ab	7.9 b	8.0 ab	8.5 a
Mar.	8.2 a	7.9 a	8.0 a	8.5 a
Apr.	11 a	11 a	11 a	11 a
May	17 a	17 a	17 a	17 a
June	21 b	21 b	21 b	22 a
July	26 b	26 b	26 b	27 a
Aug.	23 a	23 a	23 a	23 a
Sept.	na[y]	na	na	na
Oct.	11 a	11 a	11 a	11 a
Nov.	5.9 a	6.0 a	6.1 a	5.3 b

[z] Lower case letters indicate separation of means in rows ($P \leq 0.05$).
[y]Not available, data were missing for Sept. 2012.

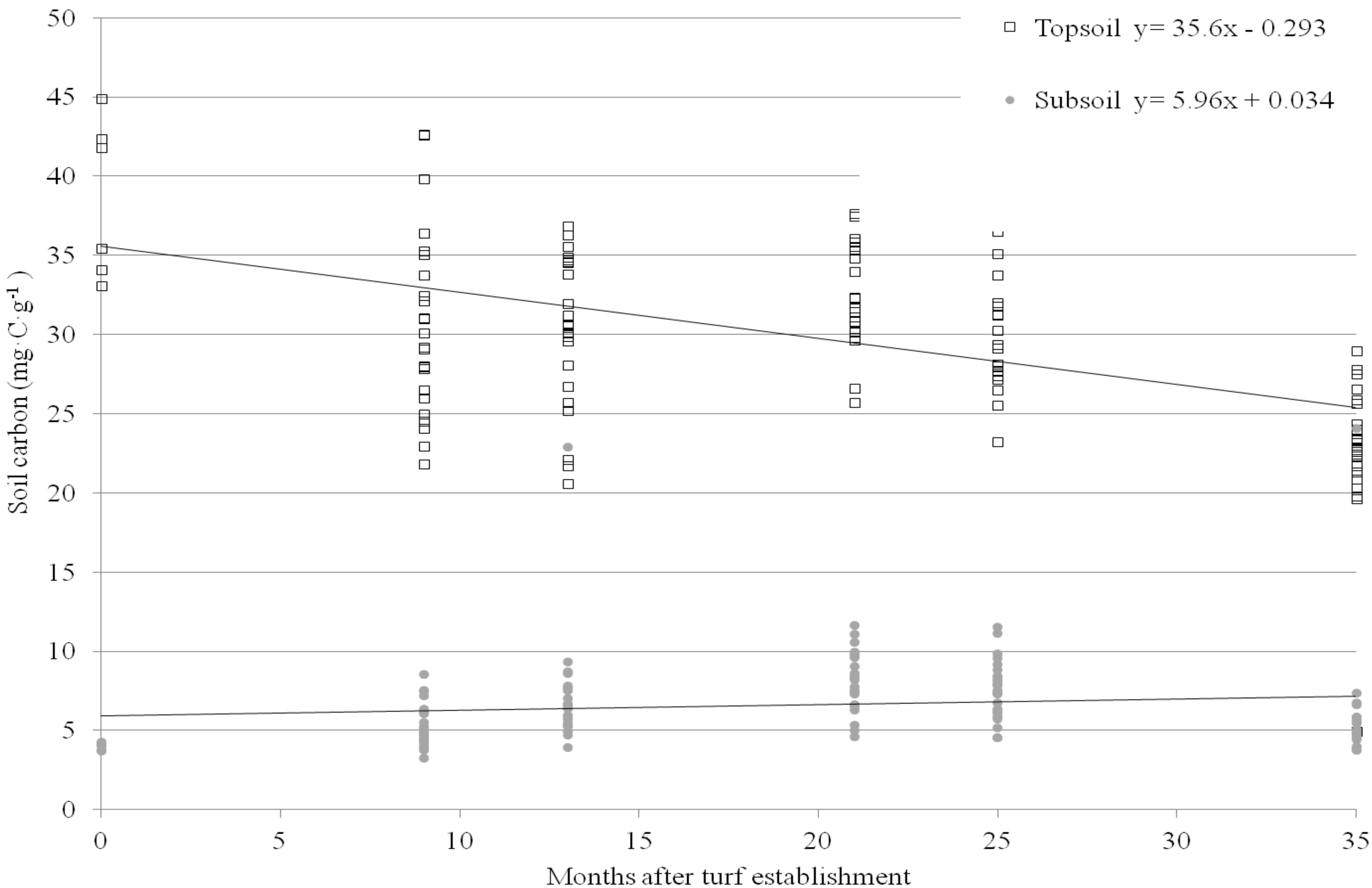

Fig. 1. Means of soil C content of topsoil and subsoil sampled periodically during the 35 month period following establishment. Soils were established with cool season turfgrasses during Aug. 2010 in Madison, WI. Linear regression of change in soil C content for both soil types was significant ($P \leq 0.05$).

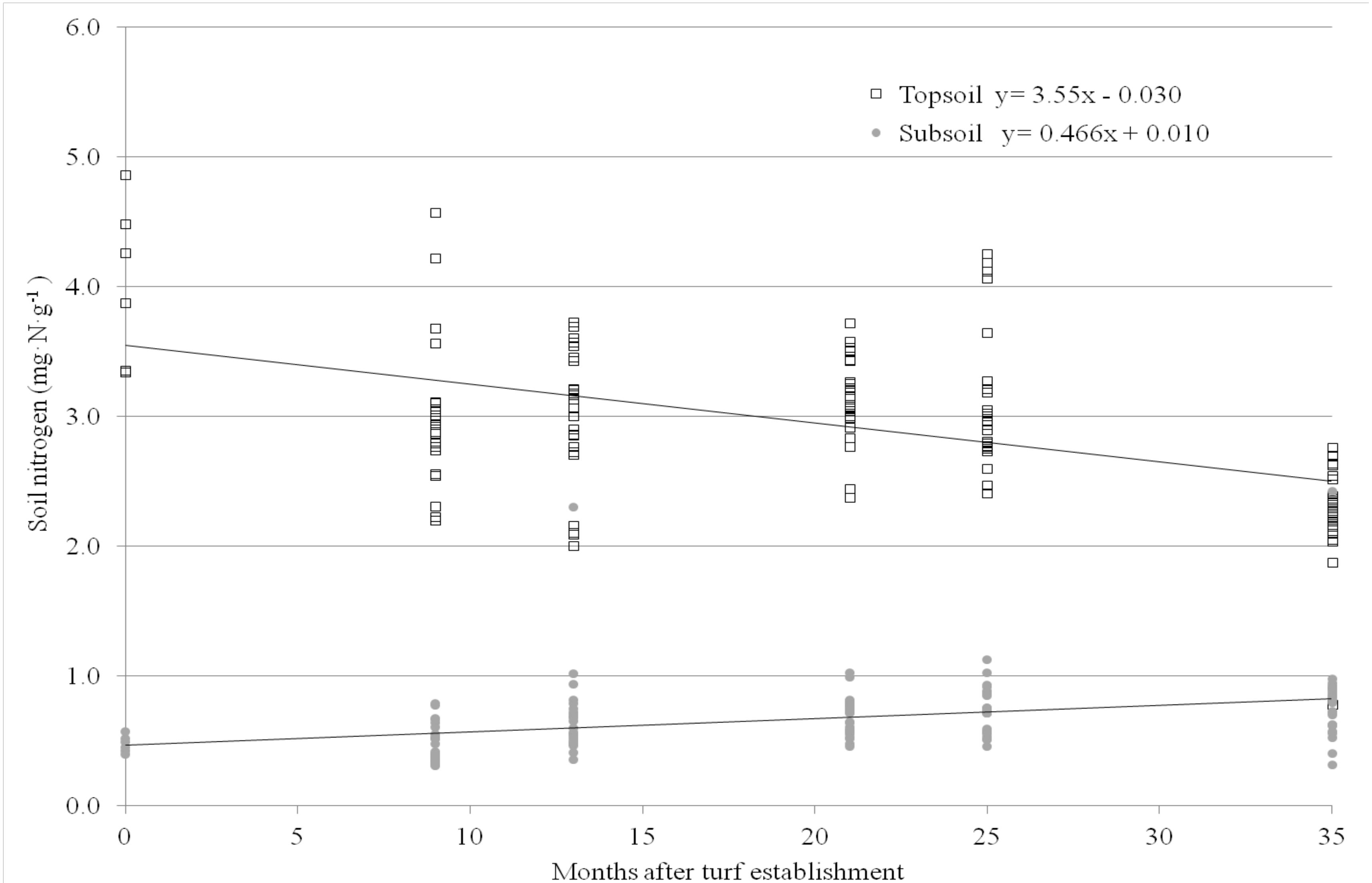

Fig. 2. Means of soil N content of topsoil and subsoil sampled periodically during the 35 month period following establishment. Cool season turfgrasses were established on the two soil types during Aug. 2010 in Madison, WI. Linear regression of change in soil N content for both soil types was significant ($P \leq 0.05$).

CHAPTER THREE

Carbon Emissions and Marginal Abatement Costs of Home Lawn Care Programs

ABSTRACT

Strategies are sought to mitigate the threat of global warming and reduce the emissions of CO_2 and other greenhouse gases. As a consequence, the routine maintenance practices employed for home lawn care have gained attention as a possible opportunity to reduce carbon emissions. The goals of this study were to construct data summaries for lawn care carbon emissions, compare the costs of lawn care strategies based on product consumption, and to demonstrate the disparity in emissions that often depends on water availability, energy generation technology, and climate of the geographic location. As a result of the geographical variances in availability of resources and emissions from power generation, lawn care strategies were also compared at two locations that emphasized these differences. To accomplish the comparisons, data summaries were constructed to conduct consumption-based estimations of the carbon emissions from conventional and alternative N fertilizer production, gasoline and electric lawn mowing, and irrigation. The cost effectiveness of the alternative practices to reduce carbon emissions was determined through assessments of both annual cost and the marginal abatement cost of carbon reduction. The cost effectiveness of the alternative strategy depended on the practice under consideration. The variation in emissions that existed between locations was as high as twenty fold. Irrigation was the largest cost and source of emissions at both locations. Though the marginal abatement cost of the alternative N fertilizer was an expensive option to make reductions, the electric lawn mower provided a cost effective method to reduce carbon emissions.

Global warming caused by the increase of greenhouse gases (GHG), such as CO_2, is a serious threat to the environment and strategies are sought that can mitigate sources of emissions (Cox et al. 2000). The prominence of the home lawn within the American landscape has brought attention to the carbon emissions from the routine of maintenance practices (e.g. mowing, fertilizers, and irrigation) required for turfgrasses (Townsend-Small and Czimczik 2010). Though studies have estimated the carbon footprint of lawn care on large scales (Townsend-Small and Czimczik 2010; Zirkle et al. 2011; Selhorst and Lal 2013), the application of the data to the home lawn scale remains difficult, due to variations in climate and consumer choices. In addition, previous data were not intended to demonstrate the extent to which alternative choices or strategies could provide a homeowner with an opportunity to reduce carbon emissions. Alternative lawn care strategies include less reliance on conventional (synthetic) N fertilizers and mowers powered by fossil fuels. With the growing interest in alternative lawn care strategies, data are needed to determine their economic utility and potential for reducing carbon emissions when compared with conventional strategies. The development of data summaries to assess carbon emissions from lawn care practices could provide a tool to reduce the carbon footprint of tens of millions US homes.

The disparity in home owner preferences regarding lawn appearance can result in great variation in the amount of inputs used for turf care (Zirkle et al. 2011). Approximately 50% of US homeowners employ a low-maintenance strategy that forgoes the application of irrigation or fertilizer entirely. High-maintenance strategies utilizing repeated applications of N and irrigation account for around 20% of home lawns, while approximately 30% of home lawns receive an intermediate level of fertilizer and irrigation inputs (Zirkle et al. 2011). Urban and suburban areas with higher-income residents were more likely than rural or lower-income residents to employ a rigorous high maintenance strategy to maintain a high quality lawn comparable to that of their neighbors (Robbins and Sharp 2003). The economic cost and carbon footprint often increase as a result of the higher intensity maintenance strategy (Zirkle et al. 2011). Regardless of the management intensity, accommodating for variation in maintenance practices by utilizing consumption-based calculations for carbon emissions may be a useful tool to make carbon reduction decisions (Jones and Kammen 2011). Jones and Kammen (2011) reported that consumption-based calculations can have many benefits including the facilitation of informed choices by the consumer and establishing carbon emission benchmarks to compare with alternative strategies. Alternative practices that reduce carbon emissions must be affordable to the consumer and the identification of economically viable alternative practices is essential. Economic studies that have investigated alternative practices for home lawn care are limited (Caceres et al. 2010; Yue et al. 2012) and failed to link the economics of alternative practices to carbon reduction. Caceres et al. (2010) compared the cost and aesthetic qualities of lawn care programs, and reported that although alternative (organic) N fertilizers provided comparable turf quality much of the time, the products were more expensive than synthetic N sources. Yue et al. (2012) conducted surveys that investigated the consumer's willingness-to-pay for lawn care and reported consumers were more likely to pay a premium for lawn care that required less fertilizers, irrigation, and work for the homeowner, than for native grass species or high input strategies. Marginal abatement costs (MAC) can be used as an economic assessment of carbon mitigation measures that result from the use of alternative practices and MAC has yet to be applied to evaluate turfgrass management. In the current study, MAC was calculated for alternative practices to assess the cost effectiveness of the practices to reduce carbon emissions when compared to conventional practices.

Alternative management strategies are believed to reduce environmental concerns and improve the sustainability of home lawn care (Belows 2003; Tukey 2007). These alternative strategies often utilize non-synthetic fertilizers and eliminate fossil fuel powered engines to reduce carbon emissions from home lawn care. The combustion of gasoline during mowing has been perceived as one of the largest sources of carbon emissions of lawn maintenance practices (Townsend-Small and Czimczik 2010; Zirkle et al. 2011; Selhorst and Lal 2013). There are approximately 6 million gasoline residential walk behind or push mowers sold in the United States annually (US Department of Commerce 2005). Although electric motor and non-motorized mowers are available alternatives, data regarding the extent of their use are not well known. In an effort to reduce emissions from gasoline mowers, several major cities in at least five states have sponsored annual events to exchange a used gas mower for a new electric mower at a discounted price (Perratore 2009). However, the production of the electricity to power electric lawn mowers often results in carbon emissions and the benefits of zero or low carbon emissions may be limited to the availability of clean energy sources such as solar or hydroelectric power. Unfortunately, electricity in the United States is often generated using fossil fuels, with coal (37%), natural gas (30%), and petroleum (1%), accounting for two-thirds of production (US Energy Information Administration 2013a). Regions of the United States that invested resources in developing a clean energy infrastructure can have significantly less carbon emissions than those that rely on fossil fuels for electricity production. For example, emissions from electricity use within the California power grid system resulted in less than half as much carbon emissions compared to other regions of the United States that rely on coal or natural gas (Table 1).

Nitrogen is often the most limiting nutrient for plants and the application of N fertilizer is often essential for maintaining turf quality (Kussow et al. 2011). Synthetic fertilizer manufacturing can be an energy intensive process that results in GHG emissions (Sawyer et al. 2010) and the use of non-synthetic or organic fertilizers has been proposed as an alternative (Blessington et al. 2013). The Haber-Bosch process is used to produce anhydrous ammonia for the creation of other forms of synthetic nitrogenous fertilizers (e.g. Urea, NH_4NO_3, and $((NH_4)_2SO_4)$). The carbon emissions for the manufacturing of synthetic fertilizers vary by formulation and are estimated to range from 0.1 to 2.0 kg of carbon equivalents (Ce) for each kg of fertilizer produced (Kongshaug and Jenssen 2003; Lal 2004). Alternative fertilizers also require inputs of fossil fuel energy for equipment used to manipulate the material and produce a finished product. Though the energy required to produce alternative fertilizers has received little attention, initial research has determined that composting and other material stabilization processes can also be energy intensive (Helikson 1991; Fadare et al. 2009). Alternative turfgrass fertilizer products composed of poultry manure, sewage sludge, fish emulsion, and seaweed extracts are commonly available N sources for home lawns and often considered to be organic. Poultry manure-based fertilizers are popular choices for turf maintenance because they contain roughly equal amounts of water soluble and water insoluble N to extend nutrient availability between applications. There are many processes that are used to produce poultry manure fertilizers and the energy required depends on the intensity of processes the material undergoes to produce a finished product.

Lawn irrigation is an often overlooked source of carbon emissions and as much as nine million hectares of urban turfgrasses in the United States may receive irrigation at some point during the year (Milesi et al. 2009). Lawn irrigation needs are the greatest in semi-arid or arid climates such as Denver, CO and Salt Lake City, UT where more than 60% of lawns likely receive supplemental irrigation (UDNR 2010). Under many circumstances the carbon emissions from irrigation can be difficult to quantify and consequently have not been assessed as thoroughly as mowing and chemical costs of lawn care (Zirkle et al. 2011; Selhorst and Lal 2013). Significant carbon emissions can result from the distribution of water for irrigation and the local climate likely dictates the intensity of irrigation practices (Sloggett 1992). In addition, the use of potable water has often utilized as an irrigation source within municipalities and the energy requirement from water treatments contributes to carbon emissions. Home lawn irrigation systems often do not require the supplemental pumping equipment that is necessary for large scale irrigation, instead relying on the hydraulic head pressure created for water distribution within the municipality. The lack of supplemental pumping equipment simplifies assessments of energy consumption and carbon emissions from home lawn irrigation. Estimates of carbon emissions from home lawn irrigation can be made using a combination of United States Environmental Protection Agency (EPA) emissions data for electrical power generation (USEPA 2012) and municipal water statistics that quantify energy required for distribution (Elliott et al. 2003; Taylor et al. 2008; Hoover 2009; Schroeder et al. 2012; UDWR 2012; Coleman et al. 2013). Providing home owners with carbon footprint estimates for irrigation may facilitate the consideration of implementing water saving strategies.

The goals of this study were to construct data summaries for lawn care carbon emissions, compare the costs of lawn care strategies based on product consumption, and demonstrate the disparity in emissions that often depends on water availability, energy generation technology, and climate of the location. To accomplish these goals data summaries were constructed to assess the emissions of lawn care practices. The data summaries were applied to conventional and alternative lawn care strategies at two locations that exhibited diversity in resources and climate. The use of two locations allowed a demonstration in the disparities in cost and emissions that can occur due to the availability to resources such as water or clean energy. This research presents a new approach for assessing carbon emissions from lawn care through the use of the data summaries to make comparisons of conventional and alternative methods based on product consumption. This research improves upon earlier studies of lawn care (Townsend-Small and Czimczik 2010; Zirkle et al. 2011; Selhorst and Lal 2013) by quantifying irrigation emissions from data for municipal water sources and also includes an economic evaluation of the programs to assess the cost effectiveness of the lawn care strategies to reduce carbon emissions.

MATERIALS AND METHODS

Program Descriptions

Scenarios representing home lawn care in south central Wisconsin and coastal southern California were used to compare the carbon emissions of two management strategies (conventional and alternative). Emissions were calculated using the data summaries and when appropriate using the data specific to the location. The area of the lawn used for the assessment was 740 m^2 and represented an average lawn size in the United States (Vinlove and Torla 1995; Osmond and Hardy 2004). The home lawns at both locations were composed of cool season turfgrasses that received regular maintenance and irrigation when necessary. The conventional lawn care program (CL) utilized synthetic fertilizers and a gasoline powered lawn mower. The alternative lawn care program (AL) utilized a composted poultry manure fertilizer and an electric powered lawn mower. An irrigation program was assessed for the CL and AL programs at both locations during the seasons of active turf growth when precipitation was insufficient for returning 80% of the weekly estimated evapotranspiration. This level of irrigation was based on the goal of maintaining a quality turf while minimizing the amount of water in an approach known as deficit irrigation (Fu et al. 2004).

Electrical power generation

The comparison of alternative turf care strategies sometimes required the quantification of emissions from electrical power generation used for water distribution and electric lawn mowers. To calculate emissions rates for a location, the current study used the EPA guidelines for developing an emission inventory for electrical consumption that utilized annual total output of GHG for the desired location (USEPA 2012). Location specific information was necessary for each site in the comparison because the technology used to generate power in the United States varies greatly by subregion, resulting in differences in carbon emissions (Table 1). To quantify the emissions of electricity production for my study, discharges of carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) were converted to units of carbon equivalents (Ce). The conversion to Ce changed CO_2 to elemental C, while CH_4 (25 times) and N_2O (298 times) were converted based upon their 100-year global warming potential when compared with CO_2 (Solomon et al. 2007). The Ce for south central Wisconsin (MRO East) electrical power generation was 0.435 kg Ce kWh^{-1} and 0.180 kg Ce kWh^{-1} for coastal southern California (WECC California) (Table 1). The differences in emissions between the two locations chosen for my study resulted from a greater use of clean power technologies for the generation of electrical power in the WECC California subregion. In addition to the two locations presented in the comparison, the data summary for electrical power generation contained data necessary for estimating emissions for sites across the continental United States (Table 1).

Fertilizer

This study focused on two fertilizer products that represent choices available to a homeowner. The CL option used a synthetic fertilizer (24-0-6, 21.6% N derived from urea, 2.4% N derived from NH_4NO_3, and 6% K_2O derived from KCl) and the AL option utilized a composted poultry manure (5-3-2.5, 3% water insoluble N and 2% water soluble N). Fertilizer rates were based upon applying 147 kg N ha^{-1} yr^{-1}. Greenhouse gas emission estimates for synthetic fertilizer production (Table 2) were adopted from the Kongshaug and Jenssen (2003). Emissions from organic fertilizer production were estimated from Helikson (1991) and Fadre et al. (2009). The assessment of Ce from fertilizer production included the consumption of raw materials and energy during the production of the nutrient sources. The Ce for mixed fertilizers with multiple nutrient sources can be determined with the information in data summary (Table 2) and using the method outlined by Kongshaug and Jenssen (2003) that accounted for the proportion of each nutrient source in the product. The sum of the Ce for each nutrient sources in the fertilizer equal the total carbon emissions from the fertilizer product. The equation for calculating the emissions for each nutrient source was;

$$\text{Fertilizer emissions} = N_C \times N_E.$$

Where N_C is the concentration (kg) of the nutrient in one kg of fertilizer product and N_E is the emissions from the production of one kg of the nutrient source (Table 2). For example, based on the N and K sources for the CL fertilizer the emissions were 0.282 kg Ce kg^{-1} fertilizer and were a result of the production of urea (0.259 kg Ce), NH_4NO_3 (0.020 kg Ce), and potassium chloride (KCl; 0.003 kg Ce). The data summary (Table 2) also listed carbon emissions for many fertilizers occasionally used for turf, although not all of the fertilizers were included in the CL and AL program comparison.

For the organic manure N fertilizers, indoor egg layer chicken operations were assessed as they provide a means of manure collection that is more efficient than free range practices (Xin et al. 2011). The processing of poultry manures into an agricultural fertilizer has been accomplished by numerous methods. Composting and non-composting methods were assessed for the fertilizer emissions data summary (Table 2). To calculate the carbon emissions from the organic manure sources, data from Helikson (1991) and Fadre et al. (2009) were used as the basis for energy requirements of each processing component. The data presented by Helikson (1991) and Fadre et al. (2009) required units conversion to Ce for diesel (35.14 MBtu L^{-1}, 0.94 kg Ce kg^{-1} fuel, 0.85 kg^{-1} L^{-1}, 0.803 kg Ce L^{-1}) and electricity (1 MJ = 0.278 kWh, 0.435 kg Ce kWh^{-1}) (Lal 2004; USEPA 2012; US Department of Energy 2013).

The non composting process used proprietary innovations that utilized the heat created by the birds within their cages to dry the waste material for processing (E. Hansen, personal communication 2013) in lieu of other drying methods (e.g. fans, ovens, air-drying). Using these techniques the facility eliminated many energy inputs for processing and only the mechanical equipment used to convert the manure into a fertilizer pellet was necessary to calculate Ce. The energy consumption of the pellet equipment presented by Fadare et al. (2009) required 0.665 MJ kg^{-1} product (0.02 kWh kg^{-1} product) and resulted in 7.83 kg Ce Mg^{-1} fertilizer. The non-composted process did not eliminate pathogens and was still considered to be fresh manure. Though fresh manure can be used as a lawn fertilizer, the non composted fertilizer was included in the data summary but not in the comparison.

A second assessment of poultry manure fertilizer techniques was conducted for a process that utilized a windrowing method to compost manure into a finished product. Helikson reported composting methods would require energy inputs (1.455×10^5 BTU derived from diesel) for the movement of the manure to create windrows (1.22 kg Ce Mg^{-1} fertilizer), turning and moisture control to facilitate the composting (0.87 kg Ce Mg^{-1} fertilizer), moving and loading of the windrow material (1.22 kg Ce Mg^{-1} fertilizer), and machinery to form the pellet (7.83 kg Ce Mg^{-1} fertilizer) (Helikson 1991; Fadare et al. 2009). The composted poultry manure finished product resulted in 0.22 kg Ce kg^{-1} N (Table 2). The method used to determine the emissions of composted fertilizer did not include energy to transport the material to or from the composting facility and any relocation of the manure not accounted for in other components of process was assumed to be minimal.

Mowing

The calculations for the mowing events were made assuming a walk behind push mower was used at each location. The estimated time required to complete a mowing event of 740 m^2 was 45 min, assuming a 1.8 km h^{-1} mowing pace with a 53 cm cutting width mulching mower. The 1.8 km h^{-1} speed accounted for turning of the mower and was slower than a typical walking pace. The estimates of emissions from mowing were calculated by determining the rate of fuel or electrical power consumption and the emissions from the source of power. Depending on the mower and lawn conditions, gasoline powered mowers consume between 8.0 ml to 16.0 mL min^{-1} with an average of 12.7 mL min^{-1} (Priest et. al. 2000; Christensen and Westerholm 2001). The mowers for the comparison were chosen because they were of similar construction and built by the same manufacturer. The CL program used a gasoline motor powered push mower and assumed the fuel consumption rate to be 12.7 mL min^{-1} (Priest et. al. 2000; Christensen and Westerholm 2001). The lawn care emissions (LCE) from the gasoline powered mower were determined from the volume of fuel consumed. Gasoline has a mass of 0.76 kg L^{-1} and was assumed to contain 0.85 kg Ce kg^{-1} (Lal 2004; Selhorst and Lal 2013). The AL program employed an electric motor driven push mower with an energy consumption rate of 0.43 kWh (Toro 2013). Two primary factors impact the quantity of emissions from the electric mower and the emissions were calculated by multiplying the power consumed by the mower power with the electrical power generation emissions (Table 3). The energy consumption rates of electric lawn mowers vary but generally range between 0.02 kWh to 3.0 kWh. To determine the LCE of the electric mower at each location, emissions from electrical power were estimated to be 0.435 kg Ce kWh^{-1} for the south central Wisconsin location and 0.180 kg Ce kWh^{-1} for coastal southern California (USEPA 2012). Lawn mowing events are required every 7-10 days during the growing season (Stier 2001). Assumptions made from climate data (Table 4) regarding the months of active turf growth for south central Wisconsin (April to October) and coastal southern California (January to December), suggested 28 and 40 mowing events a year respectively (Zirkle et al. 2011; Selhorst and Lal 2013). The number of events was adjusted to accommodate less frequent mowing during periods when heat stress (> 29°C) or cold temperatures (< 13°C) were expected to reduce turf growth. The equation for calculating the emissions for each mower type was;

$$\text{Gasoline mower emissions} = M_G \times E_G.$$

$$\text{Electric mower emissions} = M_E \times E_E.$$

Where M_G is gasoline consumed by mower (L), E_G is emissions from gasoline combustion (kg Ce L^{-1}), M_E is electricity consumed by mower (kWh), E_E is emissions from local electrical power generation (kg Ce kWh^{-1}; Table 1). The push mowers in the current study were not practical for lawns much larger than 950 m^2 and therefore scaling of the numbers to a larger area was not conducted.

Irrigation

The social pressure of maintaining a well kept lawn is greater in urban and suburban communities than for rural locations and only municipal water sources of irrigation were considered for this study (Robbins and Birkenholtz 2003; Robbins and Sharp 2003; Milesi et.al. 2009). Home lawn irrigation systems in urban areas typically utilize the hydraulic force of municipal water systems created by water towers to pressurize the sprinkler system. The use of existing water pressure eliminated the need for supplemental pumps and energy inputs for the home lawn irrigation system. Available data indicates the energy for municipalities in the United States to provide potable water to the community varied from 0.187 to 2.7 kWh kL^{-1} (Elliott et al. 2003; Taylor et al. 2008; Hoover 2009; Schroeder et al. 2012; UDWR 2012; Waller and Talbot 2012; Coleman et al. 2013) and was dependent on the location of the water source, technology used for treatment purposes, and energy for distribution (Table 5). In Wisconsin, the energy required to provide 1.0 kL of treated potable water to households from ground and surface water sources was 0.476 and 0.423 kWh (Table 5) respectively (Elliott et al. 2003). The delivery of 1.0 kL of potable water in coastal southern California required 2.70 kWh (Table 5) and was much greater than in Wisconsin due to additional energy needed to pump the water over long distances through the Colorado River aqueduct system (Schroeder et al. 2012). Emissions from water used for irrigation can be calculated from combinations of emissions from electrical production and energy requirements for supplying municipal water to customers as demonstrated in Table 5. The equation for calculating the emissions for irrigation was;

$$\text{Irrigation emissions} = W_V \times W_E \times E_E.$$

Where W_V is water volume used for irrigation (kL), W_E is energy consumed to supply potable municipal water (kg Ce kL^{-1}; Table 5), E_E is the emissions from local electrical power generation (kg Ce kWh^{-1}; Table 1). As with the other summaries, additional data was included to facilitate estimates of carbon emissions at locations other than those presented in the comparison.

Thirty year climate data for each location (International Water Management Institute 2014) were used to determine the irrigation necessary during a typical weather year (Table 4). At 80% replacement of estimated ET for a 740 m^2 lawn, the south central Wisconsin location required 92 mm of irrigation annually (68.1 kL yr^{-1}) with applications made May through the August (Table 4). The warmer and drier climate of the coastal southern California site required applications of irrigation year round, totaling 866 mm (641 kL yr^{-1}; Table 4).

Economics

Costs of each lawn management program (AL and CL) were calculated to compare the annual investments necessary for each method. Marginal abatement costs of carbon from the use of the alternative program were also determined to compare the relationship between investment cost and carbon reduction with those of the conventional method. Marginal abatement costs have been used as a cost effectiveness measurement of potential carbon reduction when considering alternatives to business as usual practices (Bättig and Ziegler 2009; Lutsey and Sperling 2009). The equation for calculating the MAC of fertilizer, mowing, and annual cost at each location were calculated as;

$$MAC = (C_{AL} – C_{CL})/(E_{AL} – E_{CL}).$$

Where C_{AL} is the cost of the alternative practice under consideration, C_{CL} is the cost of the conventional practice under consideration, E_{AL} is the carbon emissions from the alternative lawn care practice, and E_{CL} is the carbon emissions from the conventional lawn care practice. When negative MAC values were the result, the AL program was a more expensive option to reduce carbon emissions and the AL program was the less expensive choice when positive values resulted. Whenever possible the suggested manufacture's retail prices or published utility rates were used for the comparison. The original purchase prices of the electric (e-Cycler, model 20360, The Toro Co. Bloomington, MN. 55420) and gasoline (Recycler, model 20323, The Toro Co. Bloomington, MN. 55420) mowers were US$419 and $279 respectively. The expected useful life of the gasoline mower was 10 years or 250 hrs (Sivaraman and Lindner 2004). Costs were amortized over a ten year period for the electric mower as well, though their life expectancy was less certain than the commonly used gasoline mowers. Annual maintenance of the gasoline mower was assumed to be \$66 yr^{-1} and included oil change, new spark plug, and blade sharpening (Middleton Power Center, Middleton WI). Blade sharpening for the electric mower was $5 annually. The cost of gasoline used for the comparison was determined using current market rates on 10 June 2013 (AAA 2013) and was\$1.06 L^{-1} for south central Wisconsin and \$1.05 L^{-1} in coastal southern California. Fertilizer prices were \$8.43 kg^{-1} N for the synthetic fertilizer used in the CL program and \$22.90 kg^{-1} N for the composted poultry manure of the AL program. To avoid the variations in cost between on-peak and off-peak electrical use, average retail prices for household electricity were used to calculate battery charging costs of the electric mowers and irrigation. The average costs of household energy use were $0.13 kWh for south central Wisconsin and $0.15 kWh coastal southern California (US Energy Information Administration 2013b). Annual maintenance of the electric mower was limited to blade sharpening ($5, Ace Hardware Verona, WI) and battery replacement ($135) was anticipated to be necessary every 4.5 yr (Sivaraman and Lindner 2004). Current prices for local utilities were used to determine the cost of irrigation for south central Wisconsin (\$0.74 kL^{-1}) and coastal southern California (\$0.84 kL^{-1} H_2O) (City of Madison 2013; City of San Diego 2013).

RESULTS

South Central Wisconsin

The LCE for maintaining a 740 m^2 lawn using the CL program in south central Wisconsin were 39.0 kg Ce and the annual cost was $253.27 (Table 6). The use of conventional fertilizer resulted in 12.7 kg Ce annually, accounting for approximately 33% of the LCE and 36% of the monetary cost (Table 6). The annual cost of the mower, including gasoline and maintenance, with the purchase price amortized over a ten year period was $110.82 yr^{-1}. The mower was 31% of the LCE and 44% of the total cost of the CL program (Table 5). Irrigation was the largest source of LCE for the CL program, accounting for 36% of Ce and 20% of the annual cost (Table 6).

The LCE from the AL program in south central Wisconsin were 20.4 kg Ce and the annual cost was $374.88 (Table 6). The MAC of the AL program in Wisconsin increased cost $6.50 kg^{-1} Ce emitted. The alternative fertilizer used for the AL program reduced LCE by approximately 10 kg Ce yr^{-1} and increased the annual monetary cost $157, when compared with the CL program. The AL program fertilizer increased MAC in Wisconsin $15.27 kg^{-1} Ce (Table 6). The alternative N fertilizer was the largest single cost for the AL strategy, accounting for 66% of the total cost, although only 12% of LCE (Table 6). The annual cost of the mower, including electricity and maintenance, with the purchase price amortized over a ten year period was $75.02 yr^{-1}. The mower accounted for 19% of Ce and 20% of the annual cost. The electric powered mower reduced Ce by 68% and mowing costs by 30% when compared with the gasoline mower. The electric mower in Wisconsin decreased MAC of the program by $4.33 kg^{-1} Ce (Table 6). Irrigation was the largest source of Ce for AL program and accounted for approximately 69% of Ce, although only 14% of the annual cost.

Coastal Southern California

The LCE for maintaining a 740 m^2 home lawn using the CL program in coastal southern California was 342.2 kg Ce yr^{-1} with an annual cost of $747.03 (Table 6). The use of conventional fertilizer in the CL program accounted for 12.7 kg Ce, 4% of the LCE and 12% of the annual cost (Table 6). The annual cost of the mower, including electricity and maintenance, with the purchase price amortized over a ten year period, was $118.09 yr^{-1} (Table 6). The gasoline mower accounted for 5% of the CL program's LCE and 16% of the annual cost. The irrigation for coastal southern California accounted for 91% of Ce and 72% of the annual cost of the CL program (Table 6).

The LCE from the AL program in coastal southern California were 316.7 kg Ce yr^{-1} and the annual cost was $862.19. The MAC of the AL program increased cost $4.53 kg^{-1} Ce emitted (Table 6). The AL program reduced LCE by approximately 7% (25.4 kg Ce yr^{-1}) when compared with the CL program. The alternative N fertilizer accounted for less than 1% of the total LCE and 29% of the annual cost (Table 6). The AL program alternative N fertilizer increased MAC in California, $15.27 kg^{-1} Ce (Table 6) when compared with the CL program. Similar to Wisconsin, the use of the alternative fertilizer resulted in fewer emissions than the conventional N and the MAC indicated fertilizer was the most expensive choice to make carbon reductions. The annual cost of the electric mower, including electricity and maintenance, with the purchase price amortized over a ten year period was $75.84 yr^{-1} (Table 6). The electric powered mower also reduced financial cost by 36% and LCE by 87%, when compared with the gasoline mower. The electric mower in California decreased MAC of the program by $2.80 kg^{-1} Ce (Table 6). As with the CL program, the carbon emissions for irrigation accounted for nearly 99% of LCE and 62% of the annual cost (Table 6).

Comparison of Wisconsin and California

The total LCE and annual costs of the programs were less in the cooler climate, more humid climate of south central Wisconsin than coastal southern California, which translated into fewer mowing and irrigation requirements (Table 6). When compared with the Wisconsin site, the California location received 12 extra mowing events (Table 6) and an additional 77 cm (Table 4) of irrigation annually. The LCE and annual costs of conventional and alternative N fertilization were identical between the locations. Increasing the mowing events had little impact on annual costs between the two locations, although LCE were greater in Wisconsin and MAC were less expensive in California (Table 6). Emissions from gasoline use increased from 12.2 kg Ce yr^{-1} in south central Wisconsin to 17.4 kg Ce yr^{-1} for coastal southern California, although due to the clean power generation technology in California the emissions from the electric mower decreased from 3.93 to 2.32 kg Ce yr^{-1} (Table 6). The irrigation volume was more than nine times greater for the California location than Wisconsin and the LCE from irrigation were 20 times greater between the locations (Table 6). The increased emissions resulted from a greater water quantity due to drier climate, the longer growing season, and the higher energy demand from water delivery. The annual cost for irrigation in coastal southern California was ten times greater than for south central Wisconsin (Table 6).

DISCUSSION

Economics

Cost effective reductions in carbon emissions were assessed in the current study using MAC and annual cost of the alternative lawn care program. From the home owner's perspective, the adoption of carbon reduction practices that reduce annual cost would likely be the most appealing. The greater MAC and annual cost of the entire AL program was expected; however the cost effectiveness to reduce carbon emissions demonstrated by the use of an electric mower was a greater benefit than anticipated. The reduction of MAC for the electric mower was not unexpected however, as the decrease of MAC achieved from improved engine technology has been reported to be a cost effective means of carbon abatement (Bättig and Ziegler 2009; Lutsey and Sperling 2009). Irrigation was a significant portion of the annual cost, especially for the coastal southern California location. Reducing irrigation by 25% at the Wisconsin or California locations would have reduced carbon emissions by 3.5 and 78 kg Ce yr^{-1} respectively. An alternative irrigation strategy was not included for the comparison and therefore the MAC of irrigation could not be calculated. The higher cost of the N source for the AL program was anticipated and would likely decrease the appeal of the fertilizer as a carbon reduction option. The AL program N fertilizer resulted in a far greater cost to abatement carbon emissions than other opportunities outside of lawn care. Investments to decrease carbon emissions by improving the energy efficiency of residential buildings and improving power plant technologies require investments of US$70 Mg CO_2 or less (Bättig and Ziegler 2009). The trading of carbon permits on European financial markets (US$12 Mg C; Querejazu 2012) also resulted in less expensive options to decrease carbon emissions, when compared with alternative N fertilizer for lawn care.

In contrast to carbon emissions, investments of time and resources in lawn turf may benefit environmental quality through photosynthetic fixation of atmospheric CO_2 (Qian and Follett 2002; Qian et al. 2010; Selhorst and Lal 2013). From an environmental prospective, the cost of the conventional or alternative lawn care scenarios could be justified as an investment in improving environmental quality if carbon emissions from maintenance practices were less than the quantity of carbon sequestered in the soil. The sequestration rates reported for turf vary and have been estimated to be as high as 5.4 Mg C ha^{-1} yr^{-1} (Selhorst and Lal 2013). If an annual sequestration rate of 1.0 Mg C ha^{-1} (Qian and Follett 2002) was assumed to have occurred at both locations of the current study, approximately 74 kg C yr^{-1} could of be accumulated in the soil. The difference in emissions from maintenance practices and carbon sequestered in the soil would have resulted in net reductions of 35 to 54 kg C yr^{-1} for Wisconsin. However, due largely to the emissions from irrigation, California would result in a net loss of 243 to 268 kg C yr^{-1} if the annual sequestration rate was 1.0 Mg C ha^{-1}. As an investment in environmental quality, the expenditures for Wisconsin lawn care strategies were \$6.85 to \$6.94 kg^{-1} C sequestered and costs were comparable to the benefit of urban tree maintenance that has been estimated at \$3.13 to \$8.89 kg^{-1} C (Kovacs et al. 2013). Investments in Wisconsin lawn care were a less cost effective means of reducing carbon emissions than adapting home solar water heater technology at costs from \$0.71 to \$1.71 kg C yr^{-1} (North Carolina State University 2014). The economic components of the current study were based on assumptions made at the time of the comparison and market fluctuations could affect the cost effectiveness of the assessments.

Carbon Emissions

The use of irrigation in my study was the largest source of LCE among the maintenance practices used for lawn care at both sites. The large difference in LCE between locations was due to variations in climate as well as the energy requirements for the treatment and distribution of municipal water (Elliot et al. 2003; Schroeder et al. 2012). The higher energy requirement to supply potable water over long distances through an aqueduct system for coastal southern California increased the carbon emissions. Regardless of location, the identification of alternative strategies to reduce the intensity of irrigation practices could offer significant decreases in the carbon emissions from lawn care. In Wisconsin, an application of 2.5 cm water released a comparable amount of LCE (5.2 g Ce m^{-2}) to an application of 49 kg N ha^{-1} of conventional N fertilizer (5.7 g Ce m^{-2}). Fertilization would typically be conducted one to four times annually, while irrigation might be done much more frequently depending on the annual precipitation or local climate. In coastal southern California, an application of 2.5 cm of water released about twice the amount of LCE (12.2 g Ce m^{-2}) than would result from an application of synthetic urea fertilizer (5.7 g Ce m^{-2}). Comparing irrigation emissions from our study with others (Townsend-Small and Czimczik 2010; Zirkle et al. 2011; Selhorst and Lal 2013) was difficult. Zirkle et al. (2011) used an approach perhaps better suited for assessing LCE for large areas than individual home lawns, by using the proportions of both irrigated and non-irrigated lawns in the LCE calculations. Townsend-Small and Czimczik (2010) estimated LCE from irrigation in Irvine, CA to be 530 kg C ha^{-1}, although the authors were unclear of the quantity of irrigation that was used in the calculations. Selhorst and Lal (2013) conducted LCE assessments for lawns across the United States that did not use irrigation. Alternatives to municipal water, such as untreated or reclaimed water sources could likely decrease the emissions that result from irrigation. For example, in the Jordan Valley region of Utah, the untreated irrigation water supplied to residents resulted in 0.042 kg Ce kL^{-1} and was 0.165 kg Ce kL^{-1} less than municipal water in south central Wisconsin (Table 5). The deficit irrigation strategy of the current study was employed as a scientific approach to water management (Fu et al. 2004), although less precise methods are likely used by many homeowners. Basic watering strategies apply 2.5 cm wk^{-1} during the irrigation season (Stier 2001) and homeowners often do not adjust to accommodate weather conditions. These basic watering strategies that do not account for variations in climate would result in a much greater quantities of water and carbon emissions than presented in the current study. The data available for the energy requirements of municipal water distribution could benefit from data at more locations and the lack of data restricted precise predictions of carbon emissions to a limited number of places in the United States. To determine irrigation LCE for the locations not available in the water distribution summary (Table 5), the data in the summary could be used for estimation purposes or to determine the range of emissions that likely would occur.

The production of many synthetic N sources can be energy intensive and the variation in carbon emissions that resulted from the production of the organic and synthetic fertilizers was expected. Due to the variation in emissions between the N sources and also the quantity of N applied, annual LCE from programs were different than those reported by others (Townsend-Small and Czimczik 2010; Zirkle et al. 2011; Selhorst and Lal 2013). The current study calculated fertilizer applications totaling 147 kg N ha^{-1} yr^{-1} would result in LCE of 32.3 to 172 kg Ce ha^{-1} yr^{-1}. Townsend-Small and Czimczik (2010) calculated annual LCE for fertilizers with application rates of 100 and 750 kg N ha^{-1} yr^{-1} derived from of a variety of synthetic and organic N sources would resulted in 122 and 925 kg Ce ha^{-1} respectively. The application of 750 kg N ha^{-1} yr^{-1} presented by Townsend-Small and Czimczik were excessive for a majority of turf applications, especially lawn care. Zirkle et al. (2011) estimated that applying 147 and 250 kg N ha^{-1} yr^{-1} had resulted in 155 to 495 kg Ce ha^{-1} yr^{-1}. Selhorst and Lal 2013 calculated annual emissions from a 49 kg N ha^{-1} application resulted in 64.5 kg Ce ha yr^{-1}. The variation of carbon emissions that result among N sources has not been addressed in previous lawn care studies (Townsend-Small and Czimczik 2010; Zirkle et al. 2011; Selhorst and Lal 2013) and N sources other than urea may often result in fewer carbon emissions (Table 2). These variations in carbon emissions among the N sources, illustrated the importance of the approach taken in the current study to calculate the carbon footprint of fertilizer based on quantity and source of N applications. Not accounting for these variations can over or under estimate carbon emissions of fertilizer production. The data summary (Table 2) for this study provided a baseline for many common types of conventional and alternative N lawn fertilizers, though additional data that defines the Ce for other types of lawn fertilizers (e.g. sulfur or polymer coated urea, methylene urea) would be helpful for more complete assessments. The emissions from fertilizer production are relatively consistent for best available technologies, however, the transport of fertilizers from the manufacturer to the consumer can be highly variable and all fertilizers can result in significant carbon emissions (55 kg Ce $Mg\text{-}km^{-1}$) (USEPA 2008). The carbon emissions from transporting the fertilizer may exceed those of fertilizer production, even when the products were only hauled short distances (< 10 km), and would likely negate the carbon reduction achieved by utilizing the alternative AL fertilizer in the current study.

The current study was focused on the emissions from a single home lawn using residential-sized mowing equipment. Initially, due to economies of scale it was anticipated that the LCE estimates might differ from others which calculated data based on large land areas (Zirkle et al. 2011; Selhorst and Lal 2013), although that was not the case with the gasoline mower. The estimates of emissions for the gasoline mower (12 to 17 kg Ce) in this study were similar to the 9.6 to 15 kg Ce predicted by Zirkle et al. (2011) and Selhorst and Lal (2013) predicted 14 kg Ce. Our LCE results were less than the 24.6 kg Ce estimates reported by Townsend-Small and Czimczik (2010) that calculated emissions for maintaining multiple sites and included fuel use to transport lawn care equipment to and from the sites. Other studies have not included assessments of emissions from electric mowers, so economies of scale could not be compared for alternative mowers. In the current study, the electric mower appeared to be the better choice at both locations as a result of the reduced carbon emissions, MAC, and annual costs when compared with gasoline power. Although the electric mower had a higher purchase price than the gasoline mower, the amortized costs over a period of 10 yr were less expensive and reduced cost approximately 30%. Though solar powered electric and manually propelled rotary mowers were not evaluated in the current study, their use could offer an additional reduction of carbon emissions over the electric mower of the comparison. Further research will be needed to compare the quality and performance of electric mowers with gasoline models.

The use of the data summaries in the current study demonstrated that differences in carbon emissions between locations existed and these variations were greater than were expected. The LCE for the coastal southern California location were 10 times greater than for south central Wisconsin (Table 6) due to climate and availability to resources. The amount of variation in carbon emission between locations was also much greater than existed between the management strategies. The LCE summaries were used as a benchmark for comparison of carbon emissions from conventional lawn care practices with alternative methods. The adoption of the alternative practices in the current study resulted in slightly lower carbon emissions, although the reductions in the contribution to annual household emissions that occurred were a fraction of one percent. The data summaries were created using current information and the duration the data will remain viable are unknown. The data summaries will remain subject to change as new technologies, availability in resources, or other circumstances alter the emissions from the practices. For instance, if coal fueled power plants are replaced with clean power technologies, the emissions of electrical power generation will decrease. The emissions from irrigation may also change if the availability of fresh water for irrigation decreases or reclaimed water sources become available for homeowner irrigation.

The scenario presented here was intended to illustrate the cost and the capacity of utilizing alternative products to reduce carbon emissions. The use of a high input strategy in this study represents less than 20% of the home lawn care strategies employed in the United States, while a majority of home lawns are managed less intensively and receive fewer inputs (Zirkle et al. 2011). Though using a high input strategy could result in greater carbon emissions than other lower input methods (Zirkle et al. 2011), the results of the current study indicated that even intensive lawn care methods were comparable with other common household items. To put the LCE from the programs of this study into perspective, the power consumption of a refrigerator, personal computer, television, and coffee maker (Cohen 2007) can result in a carbon footprint comparable to lawn care. For example, in Wisconsin LCE (20 to 39 kg Ce yr^{-1}) were less than emissions from a refrigerator (539 kg Ce yr^{-1}), personal computer (131 kg Ce yr^{-1}), television (59.6 kg Ce yr^{-1}), and coffee maker (50.5 kg Ce yr^{-1}). Though the California site had much greater LCE (317 to 342 kg Ce yr^{-1}), those emissions remained less than powering a refrigerator in Wisconsin. However, as a result of lower electrical power emissions, the LCE from the coastal southern California site were as much as all of the examples of household items combined (323 kg Ce yr^{-1}). Annual single family household emissions in the United States are approximately 13 Mg C (Jones and Kammen 2011). Lawn care emissions for the Wisconsin and California locations represented 0.2 and 2.6% of annual household emissions respectively.

Though lawn care emissions appeared to be a minor portion of the total household carbon footprint, the identification of economically efficient methods to reduce carbon emissions will likely remain important as the research further defines the risks of global warming. The data summaries presented in this study were constructed with the hope that estimations of carbon emissions from a variety of lawn care scenarios could be made in order to help facilitate such change. The alternative lawn care choices investigated in this study provided minor abatements of carbon although were sometimes much more expensive. The use of lawn mowers powered with electricity was an interesting option for lawn care to reduce carbon emissions and warrants further investigation. Additional research into the consumer's willingness-to-pay for alternative lawn care practices to reduce carbon emissions could provide important data for the acceptance of some of the more costly strategies.

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Table 1. Carbon equivalents (Ce) of greenhouse gas emissions from the consumption of electrical power in the continental United States. Emission values are presented within Environmental Protection Agency classified subregions (USEPA, 2012). Map of subregions available at http://www.epa.gov/cleanenergy/energy-resources/egrid/index.html

	Greenhouse Gas Emissions Annual Output
Subregion name	kg Ce kWh^{-1}
Western Electricity Coordinating Council (WECC) Southwest	0.326
WECC California	0.180
WECC Rockies	0.499
WECC Northwest	0.224
Electric Reliability Council of Texas	0.323
Florida Reliability Coordinating Council	0.322
Midwest Reliability Organization (MRO) East	0.435
MRO West	0.445
Northeast Power Coordinating Council (NPCC) New England	0.199
NPCC NYC/Westchester	0.167
NPCC Long Island	0.368
NPCC Upstate NY	0.136
Reliability First Corporation (RFC) East	0.259
RFC Michigan	0.454
RFC West	0.416
Southwest Power Pool (SPP) North	0.496
SPP South	0.437
SERC Reliability Corporation (SERC) Mississippi Valley	0.274
SERC Midwest	0.478
SERC South	0.362
SERC Tennessee Valley	0.371
SERC Virginia/Carolina	0.283

Table 2. Emissions of carbon equivalents (Ce) released during the production of fertilizer sources. Table adapted from Kongsaug and Jensen (2003).

Building block (Nutrient source)	Fertilizer analysis ($N-P_2O_5-K_2O$)	kg Ce emissions kg^{-1} Nutrient[z]
Urea	46-0-0	1.20 [y]
Ammonium nitrate	35-0-0	0.82
Ammonium sulfate	21-0-0	0.18
Calcium nitrate	15.5-0-0	1.14
Potassium nitrate	14-0-44	1.85
Mono Ammonium Phosphate	11-52-0	-0.67[x]
Di Ammonium Phosphate	18-46-0	-0.11
Poultry manure (Composted)[w]	5-3-2.5	0.22
Poultry manure (Dried)[v]	4-3-2	0.20
Triple Superphosphate	0-48-0	-0.10
Single Superphosphate	0-21-0	-0.50
Potassium Chloride	0-0-60	0.05
Potassium Sulfate	0-0-50	-0.02

[z] Nutrients include N, P_2O_5, and K_2O

[y] No carbon reduction credit (0.44 kg Ce kg^{-1} urea) was applied to urea production.

[x] Negative emissions result from production methods that utilize energy created by exothermic chemical reactions.

[w] Emissions adapted from Helikson (1991) and Fadare et al. (2009).

[v] Emissions adapted from Fadare et al. (2009).

Table 3. Estimates of carbon emissions from the electric motor powered push lawn mowers with a range of power consumption rates. The emissions from electrical power generation vary by location. For comparative purposes the emissions of a gasoline powered mower were included in the table. Emissions data are in units of carbon equivalents (Ce).

		Lawn mowing emissions (kg Ce hr^{-1})	
Mower type	Power consumption rate	South Central Wisconsin[z]	Coastal Southern California[y]
Electric	0.43 kWh	0.187[w]	0.077
Electric	1.0 kWh	0.435	0.180
Electric	1.2 kWh	0.522	0.216
Electric	1.8 kWh	0.820	0.324
Electric	2.4 kWh	1.04	0.432
Gasoline[v]	12.7 ml min^{-1}	0.488	0.488

[z] Electrical power generation in Wisconsin results in 0.435 kg Ce kWh^{-1}.
[y] Electrical power generation in California results in 0.180 kg Ce kWh^{-1}.
[w] Electric lawn mower emissions (kg Ce hr^{-1}) = power consumption rate hr^{-1} x electric power generation emissions.
[v] Gasoline contains 0.85 kg Ce kg^{-1} fuel and has a mass of 0.76 kg L^{-1}.

Table 4. 30 year historic averages (IWMI, 2014) of mean temperature (Temp), precipitation, evapotranspiration (ET), were used to calculate irrigation requirements for cool-season turfgrasses in south central Wisconsin and coastal southern California. Irrigation was calculated as the amount of water necessary to replenish soil moisture during months when precipitation was insufficient to replace 80% ET.

	South Central Wisconsin				Coastal Southern California			
Month	Temp (˚C)	Precipitation (mm)	80% ET (mm)	Irrigation (mm)	Temp (˚C)	Precipitation (mm)	80% ET (mm)	Irrigation (mm)
Jan	-9.1	25.0	13.1	0.0	13	31.5	56.8	25.3
Feb	-6.6	23.6	17.7	0.0	14	24.6	61.8	37.2
Mar	0.2	49.4	37.0	0.0	14	31.4	81.1	49.7
Apr	7.5	75.4	67.0	0.0	16	11.5	96.0	84.5
May	14	75.2	100	24.5	17	1.59	102	100
June	19	90.5	119	28.3	19	0.20	101	101
July	22	94.5	124	30.1	21	0.05	117	117
Aug	20	91.8	101	9.35	22	0.12	113	113
Sept	16	73.9	66.2	0.0	21	0.83	94.8	94.0
Oct	9.4	53.2	47.4	0.0	19	3.75	80.1	76.4
Nov	1.8	48.9	23.3	0.0	16	23.2	62.2	39.0
Dec	-5.9	36.8	13.1	0.0	14	25.0	54.1	29.1
Total		738	729	92.2		154	1020	866

Table 5. Energy consumed and carbon equivalents (Ce) of emissions from the treatment and distribution of municipal water in cities and regions of the United States.

State	City or Region	Emissions from power generation (kg Ce kWh^{-1})	Water energy requirements (kWh kL^{-1} H_2O)	Emissions from distribution (kg Ce kL^{-1} H_2O)	Source
Arizona	Phoenix	0.326	0.861	0.280[z]	Hoover 2009
	Tucson	0.326	1.49	0.483	
California	Coastal southern	0.180	2.70	0.486	Schroeder et al. 2012
	Coastal northern	0.180	0.396	0.071	
Florida	Tampa	0.366	0.423	0.155	Taylor et al. 2008
Indiana	Bloomington	0.416	0.581	0.242	Waller and Talbot 2012
	Mishawaka City	0.416	0.436	0.181	
	Valparaiso	0.416	0.523	0.180	
Illinois	State - low	0.478	0.433	0.207	Waller and Talbot 2012
	State - high	0.478	0.769	0.367	
South Dakota	cities <10k persons	0.445	0.851	0.378	Coleman et al. 2013
Utah[y]	Jordan Valley	0.224	0.187	0.042	UDWR 2012
Wisconsin	South central	0.435	0.476	0.207	Elliot et al. 2003
	Milwaukee	0.416	0.370	0.154	

[z] $\text{Emissions from distribution} = \text{Emissions from power generation} \times \text{water energy requirements}$

[y] Municipal water delivery in Utah provides untreated freshwater for irrigation purposes.

Table 6. Estimates of carbon emissions, annual cost, and marginal abatement cost (MAC) of carbon reduction from turf maintenance practices for home lawns (740 m^2) in south central Wisconsin and coastal southern California. The alternative maintenance strategy used composted manure N fertilizer and electric powered mower. The conventional strategy used synthetic N fertilizer and gasoline powered mower. Irrigation was equal for both programs and applied to return the portion of 80% evapotranspiration that was not replaced by precipitation. Negative MAC values indicate the alternative program was a more expensive option to reduce carbon emissions than the conventional program and positive values indicate the alternative was less expensive. Emissions are expressed in carbon equivalents (Ce).

	South Central Wisconsin				Coastal Southern California			
Strategy	Fertilizer	Mowing	Irrigation	Total	Fertilizer	Mowing	Irrigation	Total
	Carbon emissions (kg Ce yr^{-1})							
Conventional	12.7	12.2	14.1[z]	39.0	12.7	17.4	312[y]	342.1
Alternative	2.39	3.93	14.1	20.4	2.39	2.32	312	316.7
	Annual cost (\$ yr^{-1})							
Conventional	\$91.70	\$110.82[w]	\$50.75	\$253.27	\$91.70	\$118.09	\$537.24	\$747.03
Alternative	\$249.11	\$75.02	\$50.75	\$374.88	\$249.11	\$75.84	\$537.24	\$862.19
	MAC (\$ kg^{-1} Ce)[v]							
Alternative	-\$15.27	\$4.33	na[u]	-\$6.50	-\$15.27	\$2.80	na	-\$4.53

[z] Irrigation was applied May to August and totaled 9.2 cm.
[y] Irrigation was applied year round and totaled 87 cm.
[w] Annual cost of mowing includes purchase of mower, fuel or electricity, and maintenance amortized over a 10 years.
[v] MAC = (annual cost Alternative – annual cost Conventional)/(emissions Alternative – emissions Conventional).
[u] na = not applicable, irrigation programs were the same for both programs.