CARBON AND NITROGEN CYCLING IN TURFGRASS SOILS OF WISCONSIN

By

Sabrina Ruis

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The Dissertation is Approved by the Following Members of the Final Oral Committee:
Doug Soldat, Associate Professor, Advisor
Alfred Hartemink, Professor
Chris Kucharik, Associate Professor
Erika Marin-Spiotta, Associate Professor
Matt Ruark, Associate Professor

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Table of Contents

List of Tables

CHAPTER 1

List of Figures

ABSTRACT

Current University of Wisconsin extension guidelines recommend fertilizing new lawns with 150 kg N ha^{-1} annually until the site is 10 to 12 yrs old when the N-inputs should be reduced by half. This recommendation is based on a conceptual understanding of organic nitrogen accumulation in soil but is customizable based on soil testing information that can account for site to site variability. Turfgrass soils encompass a large area of the United States and are capable of storing large quantities of C and N. A method to make more refined fertilizer recommendations would reduce the potential for N-leaching, which has environmental and human health effects. However, we need background data on the quantities of soil organic C and N, how DAYCENT may best be used to simulate soil organic N (which could be used as a tool to aide in making those fertilizer recommendations), and an understanding of the mechanisms for C and N storage in turfgrass soils.

Chapter 2 quantified the amount of soil organic C, soil organic N, and mineralization across a variety of ages, soil types and management of turfgrass areas. Soil organic C and N increased linearly at rates of 0.27 and 0.019 Mg ha^{-1} yr^{-1}, respectively, and was not influenced by management and climate. Land use history and soil series were important components that differed in their rates of accumulation in soil organic C and N. The increase of soil organic C with age was markedly lower relative to other studies, but at individual soil series, these rates of soil organic C may match the literature values for soil organic C accumulation. This suggests that making broad-scale estimates of C accumulation from turfgrass based on studies using a single value of a few soil series may resulting in overestimation of potential C storage.

In Chapter 3, the source of soil property data for running DAYCENT was analyzed and the question of what level of detail would be required for land use history was explored. Web Soil Survey based soil properties could be used in place of or in combination with measured soil properties without sacrificing model accuracy. Specific land use history data generally improved estimates of soil organic C and N compared to using a simple, universal land use history like forest in the sensitivity analysis stage. Nitrogen leaching was simulated in DAYCENT for the chemical fertilized cropped sites, with soil organic N content at threshold leaching of 5.0 Mg ha^{-1} and time to threshold leaching averaged ~7 yrs. This implies that these sites should have reductions in fertilizer much earlier than current UW-Extension guidelines recommend.

Fractionation of soil organic C and N into various pools allows for determination of soil C and N saturation, further detail on soil organic C and N accumulation, and allows for comparison of soil organic C and N fractions with existing conceptual models from the literature. Our data indicated that our sites were likely at an effective stabilization capacity. Our C and N saturation ratios showed predominantly undersaturated clay minerals, aggregate stability did not change with time, and while our particulate organic matter C and N increased with age (and whole soil C and N), the mineral pools did not, all of which support the hypothesis of an effective stabilization capacity in our turfgrass soils. Turfgrass soils of Wisconsin that we studied do accumulate soil organic C and N with the system reaching stabilization quickly, which could mean losses of N; however, DAYCENT represents a potential tool to reduce that leaching through making better fertilizer recommendations.

CHAPTER 1

INTRODUCTION

Human activities have altered the N-balance in terrestrial ecosystems. With increasing human population, production of fertilizers, land use change and combustion of fossil fuels reactive N has increased, with important negative consequences to the environment and human health (Galloway et al., 2004). Terrestrial N-deposition had more than tripled between 1860 and 1990 and is projected to nearly double from todayĝs levels by 2050 (Galloway et al., 2004). Increased N-deposition has the potential to alter N status in ecosystems and may saturate some ecosystems with N (Aber et al., 1989; Lovett and Goodale, 2011).

Carbon dioxide levels have risen rapidly since the Industrial Revolution, to levels that are changing the climate. The atmosphere is not the only pool for C, others include: plants, the oceans, fossil fuels, the earthĝs crust, and soils. Soils represent the largest terrestrial sink for storing excess C. Soils may store the C that has been taken out of the atmosphere through photosynthesis and plant death for a period of time by various mechanisms before being oxidized to CO_2 by microbial activity or lost through erosion or leaching.

Research in C and N dynamics has encompassed a variety of ecosystems or land uses from prairie to forest to croplands. Carbon sequestration in agricultural systems has been the focus of numerous scientific research endeavors. Natural and semi-natural ecosystems have served as comparison land use with croplands, but in recent research have also served as a comparison land use for urban systems. Urban or turf systems differ from other systems evaluated for C and N dynamics in productivity, soil disturbance regime (which is very infrequent in turfgrass), inputs (fertilizer, irrigation), and management (mowing, removal of vegetation, fire), which in turn influence organic C and N accumulation. No matter what system was studied for C or N dynamics, the aims were generally to understand how or where C is stored, what the processed organic materials were like, and how to best manage the soil to store C.

Turfgrass C and N

Turfgrass accounts for over 16 million ha of land cover in the United States (Milesi et al., 2005) ï nearly equivalent to the area of Wisconsin. Turfgrass areas in the United States have been growing tremendously since the post-World War II era, when the lawn became a staple of the American landscape and an outdoor living space for recreation, not just food production as it had previously. The United States contains over 58 million home lawns, over 16,000 golf courses, and 700,000 athletic fields (Steinberg, 2006) where management and soil properties are as diverse as the people who own the land. Over this time since WWII, C and N have been accumulating in these grassy areas and accumulation appears to be driven by inputs of N-fertilizer and irrigation (Bandaranayake et al., 2003; Qian et al., 2003).

Soil organic C and N are influenced by soil texture, fertility, management, irrigation, and climate. Soils with greater quantities of silt and clay often accumulate greater quantities of soil organic C and N, where the rate of accumulation with fine soil particles is dependent upon the type of system ï turfgrass, crops, grasslands, or forest (Bandaranayake et al., 2003; Six et al., 2002; Hassink, 1997). Climates that are somewhat cooler (0.95 to 1.35°C between Denver and Fort Collins), showed greater contents of soil organic C (by ~5 Mg ha^{-1} to 20 cm after ~40 yrs) than warmer areas when soil textures are similar, perhaps due to differences in decomposition rates (Bandaranayake et al., 2003). Nitrogen inputs at high rates (150 kg N ha^{-1} annually for fairways) increased soil organic C by ~10 Mg ha^{-1} and soil organic N by ~1 Mg ha^{-1} compared to lower rates (75 kg N ha^{-1} annually for roughs) (Qian et al., 2003).

Surveys have shown that 56-78% of homeowners reported fertilizing annually or hiring a lawn company to fertilize their lawns (Law et al., 2004). The amount of fertilizer N applied ranged from 0-370 kg ha^{-1} yr^{-1} with an average of 107 kg N ha^{-1} annually (Law et al., 2004). The average annual application amount in the afore mentioned studies was below current University of Wisconsin Extension guidelines which recommend 150 kg N ha^{-1} yr^{-1}, split into three applications. Half rate N-fertilizer applications are recommended after ten to 12 yrs (Kussow et al., 2011). Decreased N-fertilizer after a decade is recommended because of the accumulation of soil organic C and N in the soil, which serves as an N-source. However, sites that are organic matter poor may have not achieved sufficient soil organic N to sustain turfgrass in ten years, while sites that are initially high in organic matter may not need additional N-fertilizer for the full decade after starting turfgrass. From research in Colorado using the CENTURY model (Qian et al., 2003), there appears to be a link between reaching the plateau in soil organic N and N-leaching. Knowing the soil organic N content that correlates with the plateau may allow for reduction in N-leaching through reduction in fertilizer inputs. The reductions in N-leaching would reduce the introduction of N into surface and ground waters, which have important environmental and health effects, such as, eutrophication.

Researchers have often found low levels of N loss to the environment relative to N application rates (Petrovic, 1990); however, N losses vary based on N source, N rate, timing, soil texture, and irrigation. Miltner et al. (1996) and Frank et al. (2006) demonstrated that after about 10 yrs in Michigan, leaching losses greatly increased from

<1 kg N ha^{-1} to over 35 kg N ha^{-1} in turfgrass stands fertilized at 250 kg N ha^{-1} yr^{-1}. Turfgrass stands on clay loam soils in Colorado were modeled with 150 kg N ha^{-1} yr^{-1} or 75 kg N ha^{-1} yr^{-1} and with or without clipping return (Qian et al., 2003). Soil organic C and N both increased rapidly over time before reaching a plateau in the high N rate with clippings returned while the low N rate and clippings removed remained relatively constant (Qian et al., 2003). Leaching increased rapidly over time, beginning about 30 years after establishment in the high N rate, clippings returned; while all other treatments leached little or no N over a 100 yr simulation as soil organic N continued to increase (Qian et al., 2003). Petrovic (1990) concluded that since total soil N accumulates quickly in the first 10 yrs the rate of N input should be similar to soil N storage, and that in older sites, since accumulation of total N is minimal, the N fertilizer rate should be similar to plant removal.

The CENTURY and DAYCENT models and Their Use in Turfgrass

The CENTURY model (monthly time scale) and its sister program DAYCENT (daily time scale) simulate C and nutrient cycling in the surface 20 cm of a variety of ecosystems with early use in grasslands (Bandaranayake et al., 2003; Parton et al., 1987). The model contains a number of submodels including soil organic matter/decomposition, water balance, N, P and S. In the soil organic matter/decomposition submodel, biomass is partitioned into aboveground and belowground biomass. The quantity of biomass is based on precipitation/irrigation, temperature, nutrients, and the growth factors of the plants. Upon death, each biomass component is separated into structural and metabolic pools based on the quantity of lignin in the tissue where an abundance of lignin goes to the structural pool. A portion of the structural pool automatically goes to the slow pool (20-

40 yr turnover) due to the inherent resistance to decay of the lignin polymer while the remainder is partitioned into the active pool (1-5 yr turnover). The partitioning of the structural pool was based on decay patterns during incubation studies. Once microbial processing occurs in the active pool a portion of that product goes to the slow pool based on sand content of the soil and another portion to the passive pool (100+ yr turnover) based on the clay content of the soil. Some processing occurs in the slow pool before the byproducts enter the passive pool, the quantity that enters from the slow pool to the passive pool is based on clay content (Metherell et al., 1993; Parton et al., 1987).

The N submodel works similar to the soil organic matter submodel because N cycling is microbially mediated and tightly linked to C cycling. Nitrogen losses (volatilization, denitrification, leaching) are also accounted for in the model (Metherell et al., 1993). Required input parameters are site environmental factors, plant C, N, and lignin, precipitation and irrigation, fertilization, and management (Bandaranayake et al., 2003; Parton et al., 1987).

Work with the CENTURY model has showed it to be applicable to turfgrass systems in Colorado. Banadaranayake et al. (2003) simulated soil organic C accumulation in golf course fairways and putting greens using the CENTURY model. Model estimates of soil C content for fairways (R^2=0.67) and putting greens (R^2=0.83) were well correlated with field measurements. Qian et al. (2003) used the CENTURY model to estimate aboveground biomass (clipping yield) of Kentucky bluegrass (*Poa pratensis* L.), soil organic C, soil organic N, and nitrate leaching under two fertilization regimes and clipping scenarios (returned or removed) on clay loam soils in Colorado. Compared to field data, CENTURY accurately simulated total clipping yield. Soil organic C and soil organic N were simulated for 100 yrs following turfgrass establishment. Both parameters showed rapid increases during the first 30 years followed by gradual slowing of soil organic C and N accumulation in all N fertilizer and clipping treatments except the low N, clippings returned management system. The low N, clippings returned management system remained steady for soil organic C, and increased slowly for soil organic N. Nitrate leaching in the high N, clippings returned treatment increased over time reaching ~60 kg ha^{-1} yr^{-1} (~40% of applied N) at 100 yrs since turf establishment, while the other treatments leached insignificant quantities (<2 kg NO_3^- ha^{-1} from 0 to 100 yrs). DAYCENT could potentially be used to predict soil organic N content over time, before the onset of N-leaching, which may then be used to give fertilizer recommendations and reduce N-leaching.

In order to operate DAYCENT, weather data from the site, soil texture, bulk density, pH, and management history (previous land use as well as fertility, irrigation, and cultivation regimes) are necessary. The previous information may be gathered from weather websites, Web Soil Survey, and land managers. I ran a DAYCENT simulation as a pilot study to determine how DAYCENT would simulate soil organic C and N in different locations, using several turfgrass chronosequences in the literature across the United States, plus three sites proposed for use in this study (Table 1.1 for sites and studies). These sites represent very different areas of the country with different climate regimes and soil properties (Table 1.1). The large discrepancy between the simulated and measured data in Table 1.1, especially in some of the oldest sites, indicates that there is work to be done in order to reduce the differences between the measured and simulated data.

General management history, given good records, is easy to obtain via a questionnaire or discussion with the land manager and weather data is easily retrieved from online sources. What is more difficult is the more detailed history and soils data from the site. Early land use history may be pieced together through agricultural statistics and land use surveys, but the question of type of tillage or crop or plant species in pastures prior to turf may be influential. Aside from management and land use considerations, soil properties may be obtained through sampling the site itself or through Web Soil Survey. Site sampling is time consuming and costly; however, it is possible that soil property data from Web Soil Survey may be utilized without sacrificing model accuracy, while keeping costs and labor to a minimum.

Conceptual Models of C and N in Soil

Conceptual models fractionate the soil C and N into various pools to better understand the cycling of C and N in the soil. Conceptual models provide additional detail on stabilization of soil organic C and N that we do not have with whole soil C or N. Conceptual models allow us to visualize and better understand soil organic C and N cycling within an ecosystem. Early conceptual models have been expanded and revised to current models, especially those on N limitation and saturation (filling of the soil with N).

There are a number of conceptual models that attempt to better understand N limitation and its impacts on cycling. One of the earliest of these conceptual models, and probably best known, was proposed by Vitousek and Reiners (1975). In their conceptual model as a vegetational system grows, it incorporates nutrients readily; however at some point the system reaches capacity and the nutrient decreases to the point where inputs and outputs are equal, or steady state (Vitousek and Reiners 1975).

Later conceptual models of N built upon the Vitousek and Reinerôs (1975) model by incorporating components of the N-cycle, and describing what would occur to those processes under increased N deposition (Aber et al., 1989). Lovett and Goodale (2011), tested Aber et al.ôs (1989) conceptual using a mixed-oak forest where plots were fertilized vs not and measuring foliage N, mineralization through incubations, and mineral N. Lovett and Goodale (2011) found that their forest esponded similarly to N additions as would be expected from the model. Lovett and Goodale (2011) proposed that N added through deposition may take any one of four pathways (into SOM, into plant biomass, lost as gaseous N compounds, NO_3^- leaching), but the largest refinement was completing the N-cycle by adding the soil organic matter pool.

Perhaps Lovett and Goodaleôs (2011) greatest contribution to our thinking about N saturation was proposing that there are two types of N-saturation which may be managed in different ways. The first is ñkinetic N saturationò where N input overwhelms the system ï the quantity input is greater than the rate at which it may be taken up and stored (Lovett and Goodale, 2011). The second is ñcapacity N saturationò in which further N cannot be retained or what is retained is negligible as all retention pathways are full (Lovett and Goodale, 2011). When kinetic saturation is concerned, prevention of losses from this type of saturation may be through changing form or frequency of fertilizer application; however if the system is capacity saturated N inputs must be reduced.

Many soil organic C models assume linear increases in soil organic C with increased C inputs and limitless storage ability; however there is discrepancy in how agricultural systems respond to additional organic matter (Stewart et al., 2007). Three conceptual models have been proposed to describe C-saturation and have been tested using long term agricultural data. The linear model showed that as C input increased, soil organic C also increased. The different pools for C have no limit so any new C can enter any pool under the linear model. The whole soil C-saturation model states that the soil has an upper limit to the amount of C inputs that may be incorporated. Once soil organic C reaches its maximum, the level of soil organic C plateaus no matter how much more C is added to the soil. If a soil is not at its maximum it is said to be at a saturation deficit, and the more depleted in C, the more the soil may stabilize. As a soil gets close to its maximum, the less C will be stabilized at any one time. The third model is a mixed model which is a combination of the linear and whole soil saturation models where one pool of C saturates and the other does not. Theoretically, the mixed model would have no maximum to the amount of C that may be stored, but the rate of accumulation would be slower than the linear model (Stewart et al., 2007).

Stewart et al. (2007), tested the three conceptual C models with soils from 14 long term agricultural experiments, and found that the whole soil saturation model best fit the data. Individual sites showed that C-saturation may not be that simple and further exploration would be required (Stewart et al., 2007). Stewart et al. (2008) fractionated soil organic C from long term agricultural systems in order to test the linear and whole soil saturation models. Pools of soil organic C associated with silt and clay tended to reach saturation while particulate organic matter tended to accumulate C linearly (Stewart et al., 2008). Particulate organic matter may be incorporated into soil aggregates; however, aggregates are continually forming and breaking down, resulting in a transient, dynamic pool of soil organic C. While it makes sense to explore what quantity of C is temporarily stored in aggregates, the key to long-term storage is likely through the mineral pool, which contains the oldest C and has long residence time (Six et al., 2000; Wattel- Koekkoek, 2003).

Stewart et al.ôs (2008 and 2009) results indicate that soil organic C saturates in the mineral pool of agricultural systems while particulate organic matter continues to increase; however, these studies are from tilled systems, not perennial systems and they typically do not include any information on N. From the forest systems studied by Vitousek and Reiners (1975), Aber et al. (1989), and Lovett and Goodale (2011), it appears that soil organic N saturation may occur in one or more pools as well. Whether or not N follows the same saturation pattern as C as demonstrated by Stewart et al. (2007, 2008) is important for our understanding of how perennial systems behave under N saturated conditions and how perennial systems may be managed to minimize N loss once saturation is reached.

A new conceptual model of N saturation linked to C saturation predicts that when mineral associated organic matter (MAOM) increases, N inputs to MAOM decrease, particulate organic matter C:N ratios decrease, net nitrification increases, and POM N increases (Castellano et al., 2012). This conceptual model of N saturation is based in part on Stewart et al.ôs (2007) work and was tested in a Maryland forest of two soil orders using ^{15}N additions during incubations and fractionating the soil into particulate organic matter and silt-clay fractions (Castellano et al., 2012). The data supported a number of model predictions: N inputs to particulate organic matter increased, C:N ratios of particulate organic matter decreased, net nitrification increased before reaching a plateau, and particulate organic matter N increased as MAOM increased. If the deficit in MOAM is calculated (see Stewart et al., 2008 or Six et al., 2002) it may be used as a way to predict how saturated a system is and how additional inputs may influence the system.

Nitrogen models have changed from simple input-output models to including all major components of the N cycle; however, perhaps the most important component of working with N in managed systems is whether or not the system exhibits kinetic or capacity saturation. Data from agricultural systems showed that C in the mineral pool saturated while the particulate organic matter pool continued to accumulate linearly, and the capacity and deficit of the mineral pool may be calculated to determine the quantity of C accumulated (Stewart et al., 2007; Stewart et al., 2008). Similarly, N accumulated linearly in the particulate organic matter pool and saturated in the mineral pool (Castellano et al., 2012). The possibility exists to calculate the degree of N saturation similar to Stewart et al. (2008) in order to predict the behavior of a system when inputs are added.

In turfgrass systems, we know C and N accumulate and that C accumulation is driven in part by N inputs (Qian et al., 2003). We have the ability to control kinetic N saturation through changing the form of the N inputs, but it would be useful to predict capacity N saturation as N losses have been observed to increase after prolonged, consistent management (Miltner et al., 1996; Frank et al., 2001). Turfgrass serves as a great model system to test these ecological theories as C and N saturation seem to occur quickly, especially given abundant N fertilization, which drives the C input to the soil. Saturation in soil organic N or achieving the plateau in soil organic N accumulation may correlate with N-leaching. Determining quantities of soil organic C and N in turfgrass soils of Wisconsin, using DAYCENT to predict N-leaching, and testing soil organic C and N conceptual models would set the background information for revising fertilizer recommendations.

Chapter 2 quantified the soil organic C and N content across an array of management (fairway, rough), climate (northern, southern WI), and soil types in turfgrass areas planted on Alfisols throughout Wisconsin. In addition to the soil organic C and N content, the mineralization of C and N from these sites was characterized. Chapter 3 evaluated DAYCENT predictions of soil organic C and N in Wisconsin turfgrass, through a six step calibration-validation procedure. DAYCENT was also used to predict N-leaching from Wisconsin turfgrass sites. Chapter 4 evaluated turfgrass soil organic C and N accumulation on a more detailed level through particle size fractionation, calculation of saturation ratio, and evaluation of conceptual models in turfgrass of Wisconsin.

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Table 1.1: Comparison of turfgrass sitesô soil organic C and soil organic N with measured data from the papers indicated to our DAYCENT simulations of their study sites.

Location	Age of Site	Depth of Sampling	Measured SOC	DAYCENT SOC ÿÿ	φSOC	Measured SON	DAYCENT SON	φSON	Bulk Density §§	pH	Sand	Silt	Clay
	(yrs)	(cm)	------------------------------(Mg ha^{-1})------------------------------						(g cm^{-3})		(%)	(%)	(%)
NCÀ	95	15	108.9	50.50	58.40	10.11	3.297	6.813	1.65	6.0	96	1.5	2.5
NC	6	15	39.59	47.97	-8.380	3.190	2.783	0.407	1.65	6.0	96	1.5	2.5
MDÿ	58	100	67.50	44.32	23.18	5.500	2.871	2.629	1.27	5.7	52	33	15
MD	58	100	75.00	28.32	46.68	7.250	1.805	5.445	1.27	5.7	52	33	15
MDÿ	4	100	67.50	47.43	20.07	5.500	2.847	2.653	1.27	5.7	52	33	15
MD§	4	100	40.00	15.07	24.93	3.750	1.111	2.639	1.27	5.7	52	33	15
CA¶	33	20	60.00	67.12	-7.120	1.500	4.999	-3.499	1.50	7.3	33	35	31
CA	2	20	10.00	19.75	-9.750	5.500	1.572	3.928	1.50	7.3	33	35	31
CO#	5	11.40	55.96	53.30	2.660	ND	ND	NA	1.33	7.0	45	19	35
CO	45	11.40	29.04	27.65	1.390	ND	ND	NA	1.33	7.0	45	19	35
WIÀÀ	119	20	55.70	62.99	-7.290	ND	NA	NA	1.45	7.0	30	56	13
WI	41	20	73.11	60.25	12.86	ND	NA	NA	1.38	6.5	21	54	23
WI	9	20	13.96	41.91	-27.95	ND	NA	NA	1.21	6.5	29	54	16
Mean	37	42.5	53.55	43.58	9.97	5.28	2.30	2.63	1.39	6.4	48.9	31.3	19.2

ÀMeasured data for North Carolina locations from Shi et al., 2006.

ÿMeasured data from Maryland locations from Raciti et al., 2011. Formerly forested. No significant trend reported for former forested sites over time in Raciti et al., 2011.

§Measured data from Maryland locations from Raciti et al., 2011. Formerly agriculture.

¶Measured data for California locations from Townsend-Small and Czimczik, 2010.

#Colorado measured data from regression equations in Bandaranayakke et al., 2003; Qian and Follet, 2002. CENTURY was used for their simulation values which are in the table for Colorado.

ÀÀThree of my Wisconsin study sites.

ÿÿDAYCENT simulated C and N to 20 cm depth.

§§Soil properties (bulk density, pH, and texture) are from WSS.

CHAPTER 2

CARBON AND NITROGEN STORAGE AND MINERALIZATION IN TURFGRASS SOILS OF WISCONSIN

Abstract

Turfgrass areas have rapidly increased to encompass over 16 million ha in the USA. Turfgrass areas are capable of storing large quantities of soil organic C and N, yet we know little about the variability of soil organic C and N in turfgrass sites across a region with different soil types, management, and turfgrass age. The objectives of this study were to: 1) use multiple sources to create a list of golf course to soil sample, and 2) use soil samples from those selected sites to quantify soil organic C and soil organic N contents and rates of mineralization across a range of Wisconsin turfgrass soils. Soil samples were from five fairway and rough pairs at 37 golf courses planted on Alfisols. Soil organic C and N, site age, soil texture, fertility regime, and previous land use history were determined for each site. Soil organic C and N increased linearly at rates of 0.27 and 0.019 Mg ha^{-1} yr^{-1}, respectively. Soil organic C and N had y-intercepts of 49 and 5.2 Mg ha^{-1}, respectively. Management (fairway or rough), and climate did not influence soil organic C and N. Sites that were formerly forested retained soil organic C, but lost soil organic N at a rate of 0.35 Mg ha^{-1} yr^{-1}. Formerly cropped sites increased in soil organic C and N by 0.12 and 0.03 Mg ha^{-1} yr^{-1}, respectively. Soil series contributed to the slow rate of increase overall through differences in soil organic C rates of increase, which ranged from 0.3 to 0.8 Mg C ha^{-1} yr^{-1}. This study demonstrated the influence of heterogeneity in soil texture, previous land use history, and age on soil organic C, N, and mineralization in turfgrass systems of Wisconsin. The heterogeneity inherent in the turfgrass system, shown in this study may make discerning differences between single variables influencing soil organic C and N difficult. The slower rate of accumulation in soil organic C we observed compared to literature values suggests that projections of soil organic C storage under turfgrass could be overestimated .

Human activities have altered the CO_2 levels in the atmosphere to levels that are capable of changing the climate. Plants are capable of transferring portions of the excess CO_2 to the soil, where it may be stored for a period of time. Similarly, N cycling has been altered, with effects of increased N in the environment including increased NO_x, N_2O (two greenhouse gases), and increased NO_3^- leaching, which in turn leads to eutrophication of surface waters and contamination of ground water resulting in human health effects.

Turfgrass is an important vegetative cover in urban areas. From 1945 to 2002, turfgrass area was estimated to have quadrupled (Lubowski et al., 2006). Milesi et al. (2005) estimated that 16 million ha of turf existed in the United States. There are over 16,000 (~4,400 ha) golf courses in the US; however, home lawns greatly outnumber golf courses at over 58 million (~15.7 million ha) (Steinberg, 2006; Milesi et al., 2005). The large quantity of home lawns under management similar to golf courses can have important environmental implications. Turfgrass areas are capable of storing large quantities of soil organic C (Qian et al., 2002; Pouyat et al., 2009). Because of this rapid expansion and the large area covered by turf, it is relevant to study soil organic C and soil organic N accumulation and cycling in turfgrass areas.

Management practices, including tillage, fertilizer inputs, and plant species influence the quantity of C stored in the soil (Purakyastha et al., 2008; Conant et al., 2001). Turfgrass is similar to prairies in plant type (perennial), minimal soil disturbance, biomass production, and duration of growing season. However, management of turfgrass is more like an agricultural system because both receive inputs of fertilizer and sometimes irrigation. Aboveground biomass production in agricultural crops, turf, and prairies are similar (Guzman and Al-Kaisi, 2010; Qian et al., 2003); however, belowground biomass of agricultural crops is less than half the belowground biomass of turfgrass and prairies (Guzman and Al-Kaisi, 2010; Qian et al., 2003).

Turfgrass receives regular applications of fertilizer and irrigation (Osmond and Hardy, 2004; Law et al., 2004), which may increase soil organic C and soil organic N (Zirkle et al., 2011; Qian et al., 2003). Turfgrass is regularly mowed and clippings returned, so additions of biomass occur from both above and below the soil surface. This is in contrast to agriculture which has smaller biomass inputs and prairie which has a strong belowground biomass input, and aboveground biomass that is not finely chopped. Soil organic C stocks in turfgrass are 50 to 60 Mg C ha^{-1} (20 cm depth) would be predicted to be greater than agriculture (~44 Mg C ha^{-1} to 15 cm) and similar to or slightly less than prairies (~56 Mg C ha^{-1} to 15 cm) (Bandaranayake et al., 2003; Guzman and Al-Kaisi, 2010).

Soil organic C and N storage under turfgrass is also influenced by climate. Comparisons between CENTURY-simulated soil organic C under turfgrass in Denver, CO and Fort Collins, CO showed that warmer Denver accumulated approximately 5 Mg ha^{-1} less soil organic C after 40 yrs than cooler Fort Collins (Bandaranayake et al., 2003). The difference in soil organic C in Colorado occurred with small differences in temperature (temperature differed by ~0.5ºC for annual average maximum and 0.1ºC for annual average minimum) and a difference of ~13 days of growing season length within a soil series. Small differences in climatic factors can have an influence on turfgrass soil organic C and N accumulation, and should be considered when analyzing data from multiple sites within a region.

Selhorst and Lal (2011) showed a significant difference in soil organic C between Ohio golf course roughs and fairways in a chronosequence, where soil organic C was 4.5% for roughs and 5.4% for fairways. The authors did not report fertility or irrigation practices of the fairways and roughs they studied. The difference between fairway and rough in Ohio could be due to differences in turfgrass species. Qian et al., (2010) showed that creeping bentgrass (*Agrostis* *palustris*, Huds.), which is commonly used on golf course fairways, and Kentucky bluegrass (*Poa pratensis*, L.), which is typically used on golf course roughs, were different in quantity of C stored; however, the C input and decomposition were similar between the two species. Qian et al., (2003) used two different N fertilizer application rates to simulate soil organic C and N with the CENTURY model and showed that fertilization at 75 kg ha^{-1} of N annually with clippings returned would store ~10 Mg ha^{-1} less of soil organic C than turfgrass fertilized at 150 kg ha^{-1} of N annually. A difference of only 1 Mg ha^{-1} was simulated for soil organic N under the same treatments. Simulated soil organic N was not verified with measured soil organic N contents from the sites in the Qian et al., (2003) study, but soil organic C was verified on similar soils by Bandaranayake et al., (2003).

When turfgrass is planted on a soil with low soil organic C, initial increases in soil organic C are linear, but can reach a plateau where accumulation of soil C slows dramatically. Soil organic C rates of increase during the linear phase vary greatly (0.7 to 1.5 Mg soil organic C ha^{-1} yr^{-1}, and up to 5 Mg soil organic C ha^{-1} yr^{-1}), and time to the plateau may vary by as much as 40 yrs or more (Table 2.1). Within a very specific region the time to the plateau and soil organic C at the plateau may vary by 20 years and nearly 10 Mg soil organic C ha^{-1}. When starting levels of soil organic C are relatively similar (Table 2.1, see CO and OH examples), the rate of increase was ~1 Mg soil organic C ha^{-1} yr^{-1} and time to the plateau ranges from 30-50 yrs. Differences in the rate of accumulation could be due to differences in previous land use before conversion to turfgrass, management (fertilizer input), and soil type. For example, the Colorado sites were shortgrass prairie and Ohio was originally agriculture. In Ohio, fairways and roughs were likely fertilized at different rates, leading to different times to reach the plateau.

Soil organic C and N generally increase with increasing clay or silt plus clay content, regardless of vegetation type ï grassland, cultivated, or forested (Hassink, 1997; Six et al., 2002; Castellano et al., 2012); however, the rates are often different for each land use. In grassland soils, the rate of increase in soil organic C and N with mineral particles <20µm were ~0.4 g silt+clay C kg soil^{-1} % silt+clay^{-1} (Hassink, 1997; Six et al., 2002) and ~0.04 g silt+clay N kg soil^{-1} % silt+clay^{-1} (Hassink, 1997) respectively. For soils under forest the rate of increase in soil organic C with mineral particles <20 µm increased by approximately one-third compared to grasslands, and in cultivated sites the rate decreased by a third compared to grasslands (Six et al., 2002). When soil particles <50 µm were considered, the rates changed to 0.20, 0.32, and 0.42 g silt+clay C kg soil^{-1} % silt+clay^{-1} for cultivated, grassland, and forested sites (Six et al., 2002). The influence of soil texture on soil organic C and N in turfgrass has not been quantified.

Turfgrass soils appear to be sinks for soil organic C, but broad characterization of the quantity of soil organic C and N under varying management and soil properties within a specific area is lacking. The objectives of this study were to: 1) Use Curtisô 1959 climatic delineation, the Wisconsin Golf Course Superintendents directory, Web Soil Survey, and golf course superintendent surveys (to determine N-fertilizer regimes, age of course, and previous land use history) to select sites for soil sampling, and 2) Use soil samples from the selected sites to quantify soil organic C content, soil organic N content, and mineralization in Wisconsin turfgrass soils across varying age, soil properties, climate, and management. With a better understanding of soil organic C and N dynamics, improved N-fertilizer recommendations may be possible.

Materials and Methods

Site Selection

Golf courses make up only ~0.1% of the turfgrass in the US (Milesi et al., 2005; Tilly, 2000), they use similar species as lawns, and often have records of fertilizer input and other management practices. Information on age of golf course and soil series were collected for the 202 golf courses listed in the directory of the 2012 Wisconsin Golf Course Superintendents Association. A spreadsheet was used to organize soils data from Web Soil Survey and calculate the percentage of each soil series occupied by the various courses, followed by grouping the soils by soil order.

Of the over 13,000 ha of golf course area tabulated from the directory, ~7200 ha were found to be on Alfisols, ~2500 ha on Mollisols, ~1200 ha on Entisols, and ~750 ha on Spodosols. Only Alfisols were considered for study selection. Remaining courses were split into northern and southern zones (Table 2.2 for major climate data for the two regions, updated data from Wisconsin State Climatology Office, tension zone excluded) based on Curtis (1959) categorizing the vegetation of Wisconsin. Courses north of the tension zone were classified as northern sites, and those south were classified as southern sites. The number of age classes was determined based on distribution of course ages. We selected 37 courses for study, with year of course opening ranging from 1894 to 2004 distributed across northern and southern Wisconsin (Table 2.3).

Soil sampling

Soil sampling of paired fairways and roughs from selected golf courses took place in spring 2013 after the soils were thawed and turf was actively growing (late April through first week of June). Soil samples were collected from roughs and fairways as these two areas represented different management practices and encompassed a breadth of turfgrass management practices, including fertilizer input, irrigation, species, and mowing height. Management for fairways and roughs and land use history at individual courses was determined through a survey of the superintendents (Appendix A).

Paired samples were collected from the center of the area of play (rough or fairway) to minimize edge effects from adjacent areas and in an area where the fairway and rough were both flat (<2% slope). Ten to 12 soil cores were collected with a 2.5 cm diameter sampling probe to a depth of 20 cm. Soils from roughs were collected in the same manner. One 7.5 cm diameter core, 20 cm deep, was collected to assess bulk density from each of the five randomly chosen fairways and five randomly chosen roughs for each course.

Soil analysis

The 2.5 cm diameter core samples were frozen until processing. Samples were thawed and field moist sieved (when possible or allowed to partly dry before sieving) to 4 mm and plant debris removed. Samples were then allowed to air dry in paper bags. Samples for bulk density were dried at 105°C to constant weight, weighed, and bulk density calculated (Blake and Hartge, 1986). Determination of pH was made in 1:1 soil water (Thomas, 1996). Soil texture was determined using the hydrometer method (Bouyoucos, 1962).

From the 2.5 cm diameter soil samples, total C and N content was determined with a Carlo Erba 1500 Series 2 Elemental Analyzer (CE Elantech, Lakewood, NJ). Preparation included homogenizing and finely grinding the soil to flour-like consistency using a paint shaker that held 2 mL microcentrifuge tubes containing the sample and two ball bearings, followed by measuring 8-10 mg of soil into tin capsules.

A subset of sites (81-91 yrs old, *n*=60 samples and less than 13 years old, *n*=50 samples, a middle age site was not included due to time constraints) from both North and South groups were used to determine C mineralization rates during laboratory incubation at ~22 °C. Specimen cups containing 50 g of soil were placed in canning jars (946 mL) with 30 mL of water in the bottom of the jar to maintain high humidity and reduce water loss. Soil was maintained at a consistent moisture level (60% water filled pore space) and jars were covered between measurement days with vented aluminum foil. During measurement periods, foil was removed and replaced with the jar lid fitted with two rubber septa and needles inserted through the septa, which were connected to the inlet and outlet ports on the gas analyzer, and measurements of CO_2 made at 0, 1, 40, 41, 80, and 81 minutes after closing the jars. Measurements of CO_2 flux were made on days 1, 3, 7, 21, and 28 days after rewetting using a photoacoustic gas analyzer (Innova 1412 field gas-monitor, INNOVA Air Tech Instruments, Denmark) (Parkina and Ventereab, 2010; Velthof et al., 2003).

N mineralization

To measure net nitrogen mineralization/immobilization, the same subset of soils used for C mineralization were used, plus the 45-55 year old sites (to give a gradient in age) (16 sites). Ten g (dry weight equivalent) of 4 mm sieved soil was placed in specimen cups. One ten g soil sample from each site was immediately extracted with 100 mL 2*M* KCl. A second sample from each site was covered with plastic wrap with a small hole to allow gas exchange and reduce water loss. Water was added to maintain moisture levels at 60% water filled pore space. After 28 days of incubation, the samples were extracted with 100 mL 2*M* KCl and shaken on an oscillating shaker for 1 hr before filtering through No. 1 Whatman filters (Hart et al., 1994). Net mineralization/immobilization of NH_4^+ and NO_3^- were then determined by difference between starting and ending samples.

The second method of determining potential mineralization involved an anaerobic incubation lasting 7 days at 40°C before extracting with 2.67*M* KCl and filtering the sample (Bundy and Meisinger, 1994; Drinkwater et al., 1996). Eight g of soil and 10 g of water in test tubes were placed in an incubator; after seven days samples were removed, 30 mL of 2.67*M* KCl added, and samples placed on an oscillating shaker for one hr before filtering through No. 1 Whatman filters. Potential mineralization of NH_4^+ was determined by difference between starting and ending NH_4^+.

Measurement of NH_4^+ from sample extracts was determined using a microplate colorimetric method adapted from Rhine et al. (1998). Fifty µL aliquots of soil extract were placed in wells (done in triplicate), followed by addition of citrate, 2-phenylphenol Na salt tetrahydrate/sodium nitroprusside, buffered hypochlorite for reagents and deionized water to bring the samples to the appropriate volume. Samples were then mixed for 30s on a vortex mixer and allowed to sit for 45 min for color development after which color was read on a spectrophotometer (BioTek Instruments, Inc, Winooski, VT) at 650 nm.

Measurement of NO_3^- from sample extracts was determined using a microplate colorimetric method adapted from Doane and Horwath (2003). Twenty µL of each sample were loaded into cluster tubes in triplicate, followed by addition of 1 mL of reagent containing VCl_3, sulfanilamide, N-(1-naphtyl)ethylenediamine dihydrochloride, and HCl. Samples were allowed to color for 14-24 hrs before transferring 275 µL to microplates and reading the color on the spectrophotometer at 540 nm.

Statistical Analysis

Categorical variable (climate, management-fairway or rough) effects on soil organic C, soil organic N, C-mineralization, and N-mineralization were analyzed with ANOVA. Continuous variables of annual fertility, soil pH, and soil texture effects on soil organic C, soil organic N, C-mineralization, and N-mineralization were analyzed through linear regression. The smaller data sets included in the discussion of land use history and soil series effects on soil organic C, soil organic N, C-mineralization, and N-mineralization were analyzed with ANOVA and linear regression, respectively.

Results

Annual fairway fertilizer N inputs as determined through the superintendent survey, ranged from 35 to 371 kg ha^{-1} with an average of 109 kg ha^{-1}. Annual rough fertilizer N inputs ranged from 0 to 98 kg ha^{-1} and averaged ~41 kg N ha^{-1}. Fairways were irrigated as needed at ~81% of the sites with an additional ~11% irrigating at ~80% of potential evapotranspiration or greater. Roughs were not irrigated at ~62% of the sites. The dominant previous land use was agricultural crops (15 sites), followed by forest and pasture-forest (6 sites each), pasture (5 sites), and pasture-crop (5 sites).

Soil organic C or soil organic N did not different between fairway or rough (p=0.63 for soil organic C and management; p=0.94 for soil organic N and management). Mean fairway soil organic C was 64.2 Mg ha^{-1} and rough soil organic C was 62.5 Mg ha^{-1}. Mean fairway soil organic N was 6.16 Mg ha^{-1} and rough soil organic N was 6.14 Mg ha^{-1}.

There were statistical differences between northern and southern sites for soil organic C (p<0.0001). Northern sites contained an average of 55.6 Mg ha^{-1} soil organic C while southern sites had soil organic C stocks of 69.9 Mg ha^{-1}. Soil organic N followed the same trend where the northern sites averaged 5.8 Mg ha^{-1} and southern sites 6.4 Mg ha^{-1} (p=0.0245). Management (fairway or rough) and climate (north or south) did not influence aerobic N-mineralization as ammonium and nitrate, maximum net potential N mineralization, or C-mineralization (Table 2.4).

Soil organic C and soil organic N increased linearly with site age (Figure 2.1 A and B) with no interaction between age and climate. Soil organic C began at ~49 Mg ha^{-1} (y-intercept) and increased by 0.27 Mg ha^{-1} yr^{-1}. Soil organic N increased by 0.019 Mg ha^{-1} yr^{-1}, beginning at ~5.2 Mg ha^{-1} (y-intercept). Soil organic C and soil organic N increased by about 50% during 100 yrs of turfgrass vegetative cover. Carbon to N ratio increased very slowly with site age (C:N ratio=0.0108*age+9.74, $p<0.0001$). Carbon mineralization (p=0.347), maximum net potential mineralization (p=0.590), and nitrate produced through the aerobic incubation (p=0.409) were not influenced by age of the course. Ammonium produced from the aerobic incubation increased very slowly with age of course (Figure 2.2 A).

Clay was not an important predictor for soil organic C (p=0.352) or soil organic N (p=0.0914). Clay content averaged about 16% and ranged from 2.5 to 44%. Silt plus clay were not related to soil organic C (p=0.0678) and soil organic N (p=0.113). Silt plus clay content averaged 53% and ranged from 8.5 to 85%. Carbon mineralization, maximum net potential N mineralization, and NO_3^- production from the aerobic incubation were not influenced by silt plus clay content (p=0.210 for C-mineralization, p=0.648 for maximum net potential mineralization, p=0.754 for NO_3^- production). Ammonium mineralization from the aerobic incubation decreased as silt and clay content increased (Figure 2.2 B). A number of analyses could be conducted to explore relationships that were either not an objective of the study or would have too few data points for strong statistical analysis. These analyses will be reported and described in the discussion.

Discussion

Fertilizer and Irrigation Inputs

The quantity of courses irrigating (92% on fairways and 38% on roughs) in this study was similar to what homeowners apply in North Carolina (54 to 89%) (Osmund and Hardy, 2004) and Maryland (32%) (Law et al., 2004). Annual N fertilizer rates were within the range of rates used by homeowners in North Carolina at 24 to 151 kg N ha^{-1} (Osmund and Hardy, 2004) and Maryland at 10.5 to 369 kg N ha^{-1} (Law et al., 2004). This suggests that golf course management is comparable to home lawn management and therefore may be useful to study the influence of turfgrass on soil organic C, soil organic N, and likely other influences of turfgrass on soil.

Influence of Climate

Southern sites had higher quantities of soil organic C than northern sites whereas CENTURY simulations found that warmer areas under the same soil series were likely to have lower soil organic C (Bandaranayake, et al., 2003). Northern and southern sites were similar in soil organic C contents until they were 55 yrs old, at that time the older northern sites had lower C than the southern sites (Figure 2.1A). The trend was similar for soil organic N (Figure 2.1B). An ANOVA comparing the effect of climate in the sites younger than 55 yrs old showed no significant difference (p=0.404 for C and p=0.479 for N). Sand content appeared to be driving the difference between these two groups of sites. To determine why these sites were different, ANOVA was used to compare the sand content of sites greater than 55 yrs old in the north and south. A second ANOVA was used to compare the sand content of sites less than 55 yrs in the north and south. The northern old sites (>55 yrs old) had an average sand content of ~60% and the old southern sites had an average sand content of ~42% (p<0.0001). There was no difference in sand content between the northern and southern sites for those sites younger than 55 yrs (p=0.354).

Management and the Cumulative N Effect

There was no effect of N fertilizer input on soil organic C and N when all sites were evaluated together. Evaluating site and management together showed that only two sites, Green Bay 3 (p=0.048) (52 yrs) and Sheboygan 3 (p=0.007) (39 yrs), showed differences in soil organic N between fairway and rough. Green Bay 3 fairways averaged 11.4 Mg ha^{-1} soil organic N and roughs averaged 8.9 Mg ha^{-1} soil organic N. Annual N fertilizer rates for Green Bay 3 were 148.5 kg ha^{-1} for fairways and 0 kg ha^{-1} for roughs. Sheboygan 3 averaged 7.9 and 4.5 Mg ha^{-1} soil organic N for fairway and rough, respectively. Annual N fertilizer rates for Sheboygan 3 were 99 kg ha^{-1} on fairways and 0 kg ha^{-1} on roughs.

Analysis of the effect of management and site on soil organic C showed two sites that were different for management: Sheboygan 3 (p=0.035) and Green Bay 5 (p=0.011). Sheboygan 3 had soil organic C contents of 77.2 Mg ha^{-1} in fairways and 43.2 Mg ha^{-1} in roughs. Green Bay 5 had higher soil organic C content in the rough at 105.7 Mg ha^{-1} than in the fairway at 68.6 Mg ha^{-1}. Green Bay 5 was fertilized at 186 kg ha^{-1} N annually on fairways and 37 Mg ha^{-1} N annually on roughs.

Green Bay 3 and Sheboygan 3 were two of six sites to fertilize their fairways at ~100 or 150 kg ha^{-1} N annually and their roughs at 0 kg ha^{-1} N annually. Green Bay 5 was one of 13 sites to fertilize fairways and not fertilize roughs. Analysis of clay content from the three sites showing management differences compared to all other sites showed differences in clay content.

Clay content averaged across the three sites that showed statistical differences between fairways and roughs was 20.9%, compared to an average clay content of 15.7% across all other sites ($p<0.0001$ for clay content comparison). This suggests that clay content may have had an important role in retention of C and N.

Selhorst and Lal (2011) showed statistical differences between fairways and roughs on an Ohio golf course chronosequence where fairways and roughs differed by 0.9% soil organic C. They did not include fertilizer rates for their fairways and roughs, but since there was a difference between management regimes at their sites, we also expected to observe differences. What may have masked the difference between fairway and rough was the wide range in soil fertility between the two management scenarios. One method to investigate this range in N fertility is to evaluate cumulative N applied over the history of the site (Figure 2.4 A and B). Calculation of N input assumes a constant N rate unless otherwise stated by the golf course superintendent. Linear regression of cumulative N explaining soil organic C and soil organic N shows a significant increase with increased cumulative N; however, cumulative N is also highly influenced by course age. When linear regression was run for annual N fertilizer rates, age and the interaction age explains ~ 8.5% of the variability. The majority of the variability was in the error term; however, annual N fertilizer rate and the interaction between age and annual N fertilizer rate explained less of the variability relative to age. This means that age was a more important factor for soil organic C and N accumulation than N fertilizer input on these courses.

Soil organic C and soil organic N increased with age (0.27 Mg C ha^{-1} yr^{-1}), but the increase did not occur as rapidly or plateau as in other studies that measured soil organic C across an age gradient (0.9 to 1.2 Mg ha^{-1} yr^{-1} in Colorado (Qian and Follett, 2002; Bandaranayake et al., 2003) and 1.5 Mg ha^{-1} yr^{-1} in Ohio (Selhorst and Lal, 2011)). Linear regression of soil organic C and total soil N with age show R^2 values ï 0.171 for soil organic C and 0.129 for total soil N. The variability is likely due to combination of soil series (over thirty soil series) and land use history (five land uses), which will be explored later.

The C:N ratio in the North Carolina (Shi et al., 2006) decreased from ~15 to ~11 during 94 yrs whereas our C:N ratio increased from ~9.7 to ~12.6 over 119 yrs. The difference may be partially explained by differences in prior land use, where our study was formerly dominated by deciduous trees and North Carolina with coniferous trees. Coniferous trees often have a much wider C:N ratio (C:N ratio of ~35 in deciduous trees, ~60 in coniferous trees versus ~9 to 20 in turfgrass) (McGroddy et al., 2004; Golubiewski, 2006), the C:N ratio needed to decrease to achieve a new C:N ratio based on turfgrass to balance the C and N levels within the soil.

Land Use History Influence on Data Variability

Land use history was an important parameter for determining trends in soil organic C and soil organic N contents, C:N ratio, and maximum net potential mineralization over time as well as y-intercepts (Table 2.5). Soil organic C and N increased with time with poor fit for both regression lines in the cropped previous land use history (Table 2.5). The poor fit for the cropped land use (n=150) likely resulted from the heterogeneity in management, where manure rates likely varied in addition to the agricultural crops. Formerly forested sites (n=60) increased in C:N ratio with age, had no change in C, and decreased in N while net N mineralization decreased (Table 2.5). Heterogeneity in deciduous tree and herbaceous vegetation probably lead to the low fit for soil organic C and N. Cropped sites had a lower y-intercept (52.7 Mg ha^{-1} for soil organic C and 5.83 Mg ha^{-1} for soil organic N) compared to forested sites (81.3 Mg ha^{-1} for soil organic C and 8.45 Mg ha^{-1} for soil organic N).

Land use history is often controlled for in other turfgrass research studies where land use history includes agriculture, former pines, or native prairie (Selhorst and Lal, 2011; Shi et al., 2006; Qian and Follett, 2002). Soil organic C, N, C:N ratio, and net N mineralization had different trends in formerly cropped, forested, and pastured which led to the poor regression when sites were evaluated in aggregate.

Variability in Soil Series

Another source of variation that likely contributed to the poor fit for the overall data on soil organic C and N was the large number of soil series in our study. There were three soil series found at multiple sites (Dodge-6 sites n=22 data points; McHenry-5 sites n=26-data points; Fox-4 sites n=8 data points), which allowed us to explore the relationship among age, soil series, and soil organic C or N. Individual soil seriesô effect on soil organic C and total soil N over time showed much stronger linear relationships (Figure 2.4A and B). All three soil series increased in soil organic C and N, but each soil series had a y-intercept at a different quantity of soil organic C or N and increased at different rates. For example, the Dodge soil series had a y-intercept at a higher level of C and N compared to McHenry and Fox, and increased at the slowest rate (Figure 2.4 A and B). The Fox soil series increased in soil organic C at a rate of 0.81 Mg C ha^{-1} yr^{-1}, which was similar to a Colorado study that showed rates of ~1.0 Mg C ha^{-1} yr^{-1} (Bandaranayake et al., 2003). Soil texture and previous land use history may explain why these soil series in our study were different. All three soils had similar quantities of clay ~17 to 18 g clay 100 g $soil^{-1}$. The Dodge and McHenry soils contained 55.7 and 50.4 g silt g $soil^{-1}$ with the remainder in sand. The Fox soil series contained greater quantities of sand than the other two soils at 48.2 g sand g $soil^{-1}$. The Dodge series was primarily a mix of slowly increasing soil organic C cropped (10 data points) and moderately increasing in soil organic C pasture-forested (10 data points) land use histories. The McHenry series also contains pasture-forest (10 data points), no increase in soil organic C pasture-cropped (10 data points) with a small portion of cropped (4 data points). Four data points were formerly pastured for the Fox soil series, another two were formerly cropped, and the remainder pasture-forest, both of the pastured options, being moderately increasing in soil organic C.

The Fox soil series increased at a rate of 0.81 Mg soil organic C ha^{-1} yr^{-1}, similar to Bandaranayake et al. (2003); however, with more heterogeneity (all data points), the rate of soil organic C increase dropped to 0.27 Mg ha^{-1} yr^{-1}. Other soil series were half or a quarter the rate of increase, which may mean that the overall rate of turfgrass C sequestration in turfgrass at a country-wide scale could be overestimated if the heterogeneity is not taken into account. In Table 2.1, most of these turfgrass areas were golf courses and may have not had topsoil removed for golf course construction, or if it was, it was returned. A turfgrass site established on subsoil would have the potential to sequester C at a faster rate due to the greater deficit between actual C content and capacity (Stewart et al., 2009). A site that is depleted in C due to previous land use history would also be more likely to sequester C at a faster rate; however, because of the rather limited number of data points within a soil series and within a land use history, we cannot test this hypothesis with our data set.

Silt and Clay

Silt and clay influence on soil organic C accumulation was not observed in our study as in other studies (Six et al., 2002; Hassink et al., 1997). A land use history and silt+clay interaction was important for explaining soil organic C and N content (p<0.0001 for both soil organic C and N).

Formerly cropped and crop-pastured sites decreased or did not change in soil organic C and N with increase silt plus clay (Table 2.6) Forested, pastured, and pasture-forested sites increased or did not change in soil organic C and N with increase in silt plus clay. Ammonium mineralization decreased with increased silt and clay, which suggests that there is an association with silt and clay preventing microbial access despite a lack of influence of silt plus clay on soil organic N.

Predictive Model

Through analysis of our data set, it appeared that age, land use history, and silt plus clay content were important factors for determining soil organic C and N contents in Wisconsin turfgrass soils. Soil series could be another component; however, the lack of sufficient replication of data points across all soil series prevented its inclusion in this model. An ANOVA that included age, silt plus clay content, land use history, and all interactions explained 23% of the variability in soil organic C and 20% of the variability in soil organic N. While ecological studies may consider this a sufficient percentage of the variability explained, it would likely not be sufficient for making fertilizer recommendations to reduce N-leaching.

Conclusions

This study demonstrated the heterogeneity of the turfgrass system across land use history, soil texture, age, and N fertilizer input. Overall increases in C and N were slow and linear over a 100 yr timeframe. Land use history influenced the rate of accumulation of soil organic C and N, with forested sites not increasing in C and decreasing in N while cropped sites increased in soil organic C and N. Soil series appeared to be an important factor as different soil series increased at different rates. When sites are analyzed in aggregate, both land use history and soil series variability seemed to contribute to the slow rate of increase in soil organic C and N. The extensive variability makes discerning differences and predictability in soil organic C and N in turfgrass difficult in large scale. Even predictive models such as those that included age, land use history, and silt plus clay content only accounted for ~20% of the variability in the data set.

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Table 2.1: Turfgrass soil organic C starting C content, times to plateau, and soil organic C at plateau, and rate of increase in soil organic C in the United States. MAT=mean annual temperature. MAP=mean annual precipitation

Location	MAT	MAP	Growing Season Length	Initial Soil Organic C	Approximate Time to Plateau	Soil Organic C at Plateau	Rate of increase in Soil Organic C
	°C	cm	d	Mg ha^{-1}	Yrs	Mg ha^{-1}	Mg ha^{-1} yr^{-1}
Fort Collins, COÀ	10.1	40.8	153	37.0	35	70	0.95
Denver, CO	10.1	43.5	157	37.0	40	65	0.7
Pinehurst, NC§	16.0	112	204	31.5	70	115	1.2
Las Vegas, NV	20.7	10.6	365	13.0	20	115	2.8
Ohio Fairwayÿ	11.6	143	173	35.2	30	80	1.5
Ohio Rough	11.6	143	173	35.4	40	80	1.1
CO/NE/WY¶	9.08	42.6	147	23.7	30	61	1.2

ÀFort Collins, CO and Denver, CO: Bandaranayake et al. (2003)
ÿSelhorst and Lal (2013)
§ Pinehurst, NC and Las Vegas, NV: Yao et al. (2009)
¶Qian et al. (2002)

Table 2.2: Climate data from the Wisconsin State Climate Office for Northern and Southern zones (based on Curtis, 1959, data updated from Wisconsin State Climatology Office and US Climate Data) in Wisconsin. Northern site data was averages from Wausau, Sheboygan, Eau Claire, and Green Bay. Southern site data was means from Madison, Racine, Lake Geneva, and Fond Du Lac.

	Southern Sites	Northern Sites
Jan mean temperature	-8.16°C	-9.83°C
Jul mean temperature	21.11°C	19.44°C
Annual mean temperature	8.33°C	6.83°C
Length of growing season	158 days	160 days
Annual precipitation	86.36 cm	78.23 cm
Annual total snowfall	114.3 cm	138.5 cm
Days with minimum temperature above 20°C	5.6 d	3 d
Days with maximum temperature above 32°C	13.9 d	5.9 d

Table 2.3: List of golf courses by code, ages, and climatic distribution (based on Curtis, 1959) for Wisconsin turfgrass C and N study. Land use history and age of course were determined through superintendent surveys.

Course	Climate	Year Started	Years Old	Land Use History
Wausau 1	N	1926	87	forested
Green Bay 1	N	1931	82	forested
Green Bay 2	N	1928	85	cropped
Sheboygan 1	N	1963	50	pasture-forested
Green Bay 3	N	1958	55	forested
Sheboygan 2	N	1962	51	pasture-cropped
Green Bay 4	N	1974	39	pasture-forested
Green Bay 5	N	1970	43	forested
Sheboygan 3	N	1974	39	cropped
Sheboygan 4	N	1988	25	pasture
Eau Claire 1	N	1984	29	cropped
Sheboygan 5	N	1981	32	pasture-cropped
Eau Claire 2	N	1994	19	cropped
Green Bay 6	N	1996	17	pasture
Wausau 2	N	1993	20	forested
Sheboygan 6	N	2002	11	cropped
Sheboygan 7	N	2003	10	cropped
Janesville 1	S	1894	119	pasture
Racine 1	S	1895	118	pasture
Madison 1	S	1900	113	pasture-forested
Madison 2	S	1915	98	cropped
Madison 3	S	1916	97	pasture-forested
Racine 2	S	1907	106	cropped
Janesville 2	S	1924	89	forested
Racine 3	S	1927	86	cropped
Madison 4	S	1922	91	cropped
Wisconsin Dells 1	S	1961	52	pasture-cropped
Racine 4	S	1972	41	pasture-cropped
Green Lake 1	S	1980	33	pasture
Janesville 3	S	1981	32	cropped
Green Lake 2	S	1983	30	pasture-cropped
Madison 5	S	1990	23	cropped
Madison 6	S	1994	19	cropped
Wisconsin Dells 2	S	1991	22	pasture-forested

Madison 7	S	2004	9	cropped
Madison 8	S	2001	12	cropped
Wisconsin Dells 3	S	2001	12	pasture-forested

Table 2.4. Mean C and N mineralization from Wisconsin golf course by management and climate. *p*-values from ANOVA comparing influence of management or climate on the mineralization parameters.

Management or Climate	Aerobic, NH_4^+À	Aerobic, NO_3^-	Anaerobic N-Mineralizationÿ	C-Mineralization§
	µg g $soil^{-1}$ d^{-1}	µg g $soil^{-1}$ d^{-1}	µg g $soil^{-1}$ d^{-1}	g C kg^{-1} d^{-1}
Fairway	0.0241	1.76	12.5	27.2
Rough	0.0290	1.52	13.5	24.0
p-value	0.881	0.165	0.285	0.103
North	0.0538	1.71	12.8	24.8
South	-0.000765	1.57	13.3	26.3
p-value	0.0901	0.402	0.553	0.441

ÀAerobic-NH_4^+ and NO_3^- produced during 28-d aerobic incubation.

ÿ Anaerobic N-Mineralization-Maximum net potential N mineralization (NH_4^+) produced during 7-d anaerobic incubation.

§ C-produced during 28-d incubation, measured as gaseous flux using photoacoustic gas analyzer.

Table 2.5. Linear regression of age and land use history for soil organic C (SOC), soil organic N (SON), maximum net potential N mineralization (MaxNMineralization or MaxN) and C:N ratio for Wisconsin golf courses.

Land Use	*n*	Parameter	Regression Equation	R^2	*p*-value
Cropped	150	SOC	SOC=52.7+0.122*age	0.35	<0.0001
	150	SON	SON=5.83+0.0302*age	0.18	0.0007
	70	MaxNMineralization	MaxN=9.74+0.0782*age	0.514	<0.0001
	150	C:N Ratio	C:N=10.1-0.00149*age	0.00063	0.761
Forested	60	SOC	SOC=81.3-0.188*age	0.016	0.331
	60	SON	SON=8.45-0.357*age	0.092	0.0182
	40	MaxNMineralization	MaxN=15.85-0.0824*age	0.179	0.0065
	60	C:N Ratio	C:N=8.40+0.0462*age	0.0806	0.0279
Pasture	50	SOC	SOC=43.3+0.446*age	0.353	<0.0001
	50	SON	SON=5.83+0.0302*age	0.217	<0.0001
	0	MaxNMineralization	N/AÀ		
	50	C:N Ratio	C:N=7.27+0.0264*age	0.335	<0.0001
Pasture-Forest	60	SOC	SOC=46.5+0.447*age	0.21	0.0002
	60	SON	SON=4.76+0.0299*age	0.18	0.0007
	20	MaxNMineralization	MaxN=31.4-0.380*age	0.951	<0.0001
	60	C:N Ratio	C:N=10.57+0.010*age	0.0186	0.299
Pasture-Crop	50	SOC	SOC=61.7-0.182*age	0.0069	0.57
	50	SON	SON=2.69+0.0666*age	0.095	0.029
	30	MaxNMineralization	MaxN=-6.33+0.359*age	0.144	0.0385
	50	C:N Ratio	C:N=16.23-0.144*age	0.259	0.0002

ÀN/A=not applicable. A subset of sites based on age were used to conduct the anaerobic incubation for maximum net potential N mineralization and no former pasture sites were included in that subset.

Table 2.6. Linear regression of land use history and silt plus clay content influence on soil organic C and N in Wisconsin golf courses.

Land Use History	Slope	R^2	*p*-value
Forested	SOC=0.429x+50.0	0.0300	0.152
	SON=0.0470x+4.09	0.0400	0.0112
Cropped	SOC=-0.366x+77.9	0.0400	0.0157
	SON=-0.0309x+7.39	0.0400	0.0112
Pastured	SOC=1.06x+11.8	0.280	<0.0001
	SON=0.0679x+3.91	0.150	0.0051
Pasture-Forest	SOC=0.687x+30.9	0.0900	0.0170
	SON=0.0431x+3.89	0.0700	0.0377
Crop-Pasture	SOC=-0.311x+68.5	0.0500	0.118
	SON=-0.0293x+6.82	0.0500	0.136

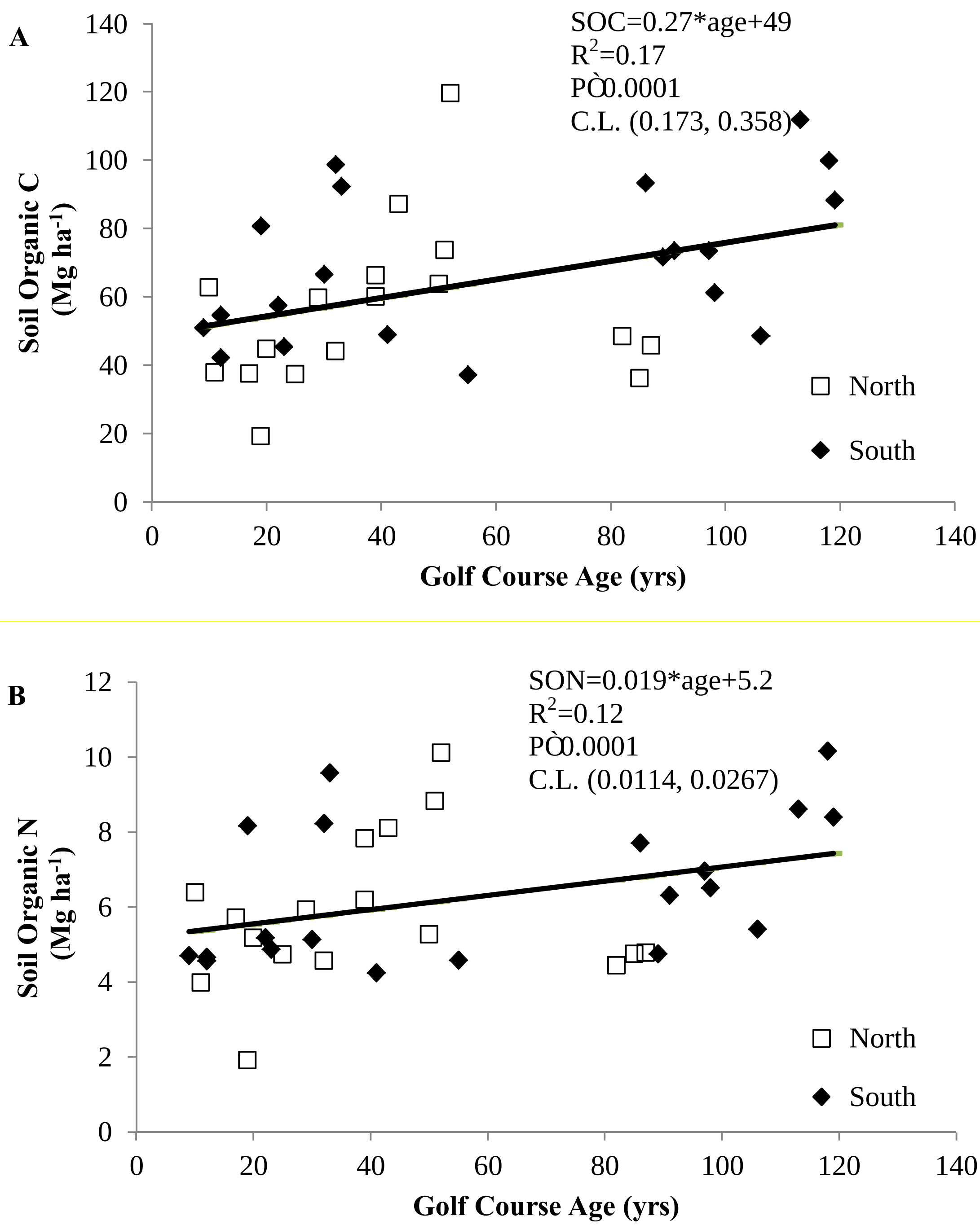
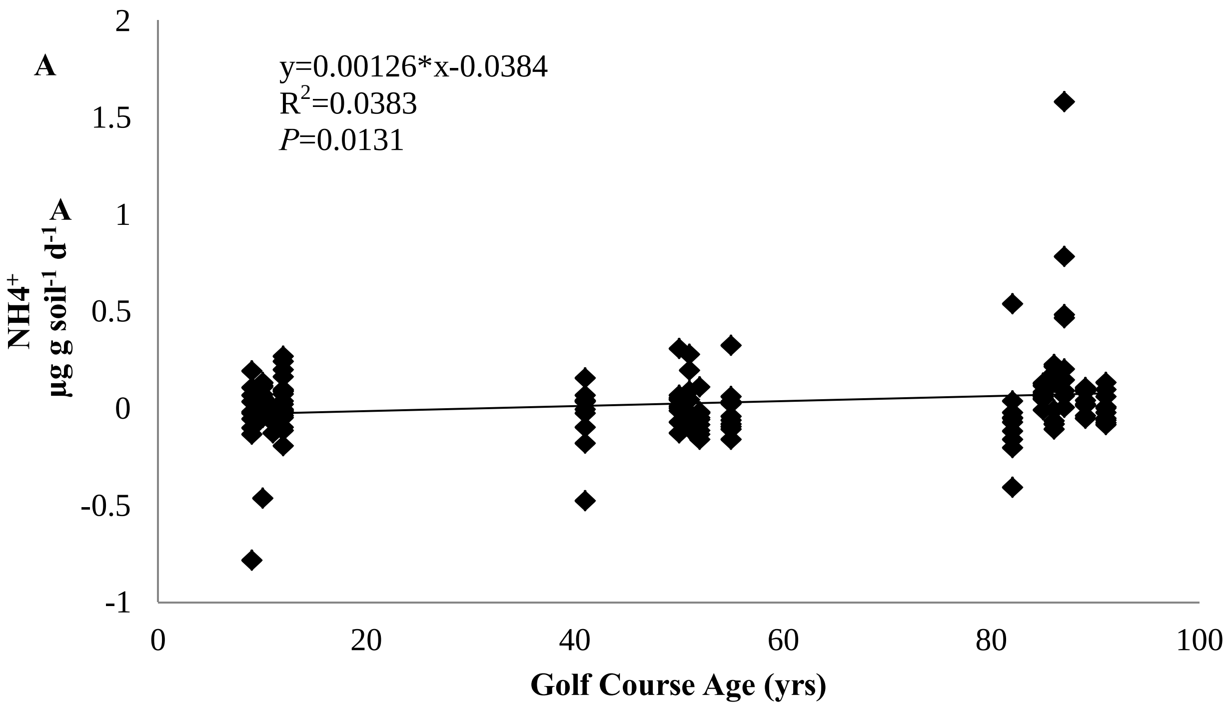


Figure 2.1: Soil organic C (A) and soil organic N (B) in turfgrass across gradient in age on Alfisols in Wisconsin. C.L.=confidence limit of the slope.



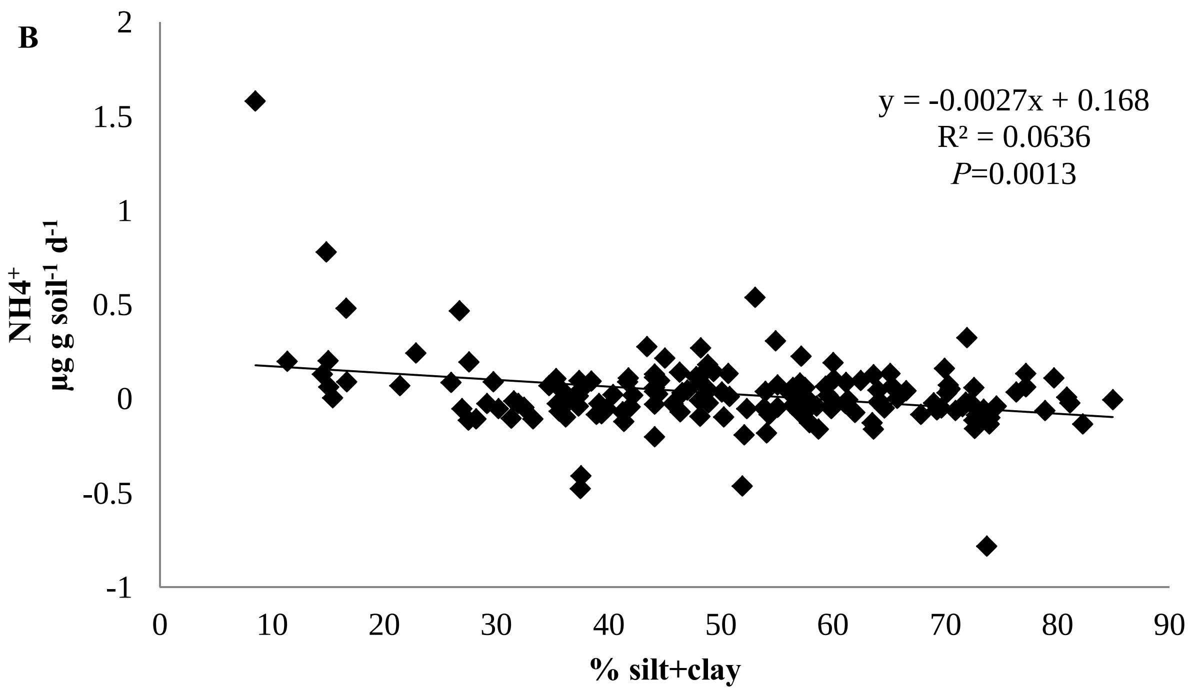


Figure 2.2: (A) Ammonium produced during aerobic incubation, across age of Wisconsin golf courses. (B) Ammonium produced as influenced by % silt and clay in Wisconsin golf courses.

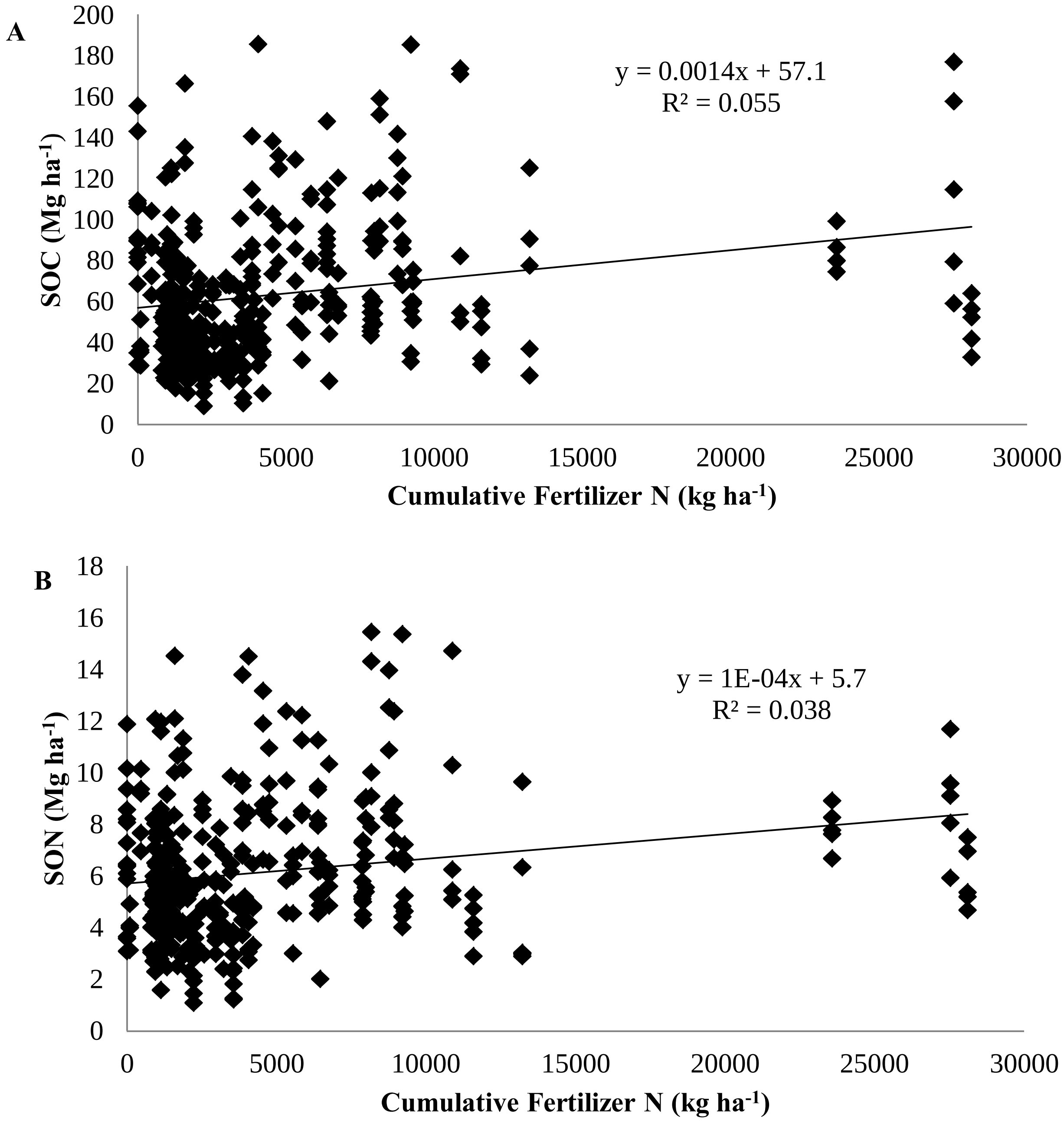


Figure 2.3. Soil organic C (A) and soil organic N (B) as influenced by cumulative fertilizer N applied throughout the lifetime of the golf course in Wisconsin. Cumulative fertilizer was calculated as years of golf course existence multiplied by fertilizer rate.

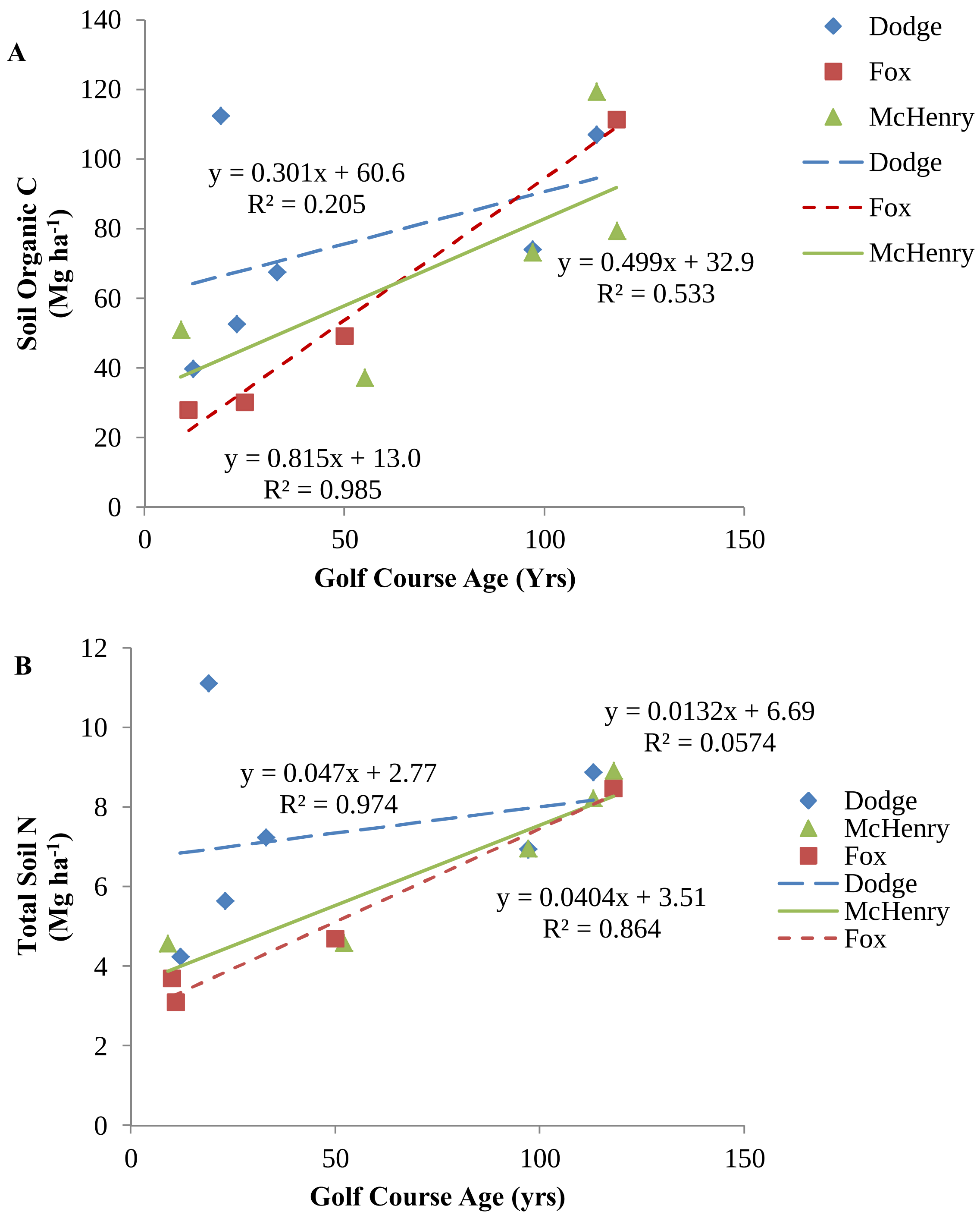


Figure 2.4: Soil organic C (A) and soil organic N (B) as influenced by age and soil series on Wisconsin golf courses.

CHAPTER 3

PARAMETERIZATION AND SENSITIVITY ANALYSIS OF DAYCENT FOR WISCONSIN TURFGRASS TO SIMULATE CARBON AND NITROGEN

Abstract

The amount of land under turfgrass has increased rapidly since the mid-1900s. Turfgrass can accumulate soil organic C and N with rates similar to that of prairies at 20 cm depth. Computer modeling of turfgrass areas has shown good relationships between measured and simulated values of soil organic C in Colorado. Use of DAYCENT would allow for analyzing relationships between soil organic C, N, and N-leaching and perhaps useful for making better fertilizer recommendations. The objectives of this study were to: 1) predict soil organic C and N in Wisconsin turfgrass by calibrating and validating DAYCENT with a six step process and 2) predict N-leaching using DAYCENT and explore relationships between soil organic N and N leaching. Soil samples (20 cm depth) from 26 Wisconsin golf course fairway-rough pairs were collected and analyzed for soil organic C, N, texture, pH, and bulk density. DAYCENT was used to simulate soil organic C and N with measured and Web Soil Survey derived soil properties. DAYCENT was then run under various land uses appropriate to the site and under a general land use history of forest. A combination of measured soil texture, pH and Web Soil Survey bulk density had simulated soil organic C and N closest to measured values (within ~13 Mg ha^{-1} for C and ~3 Mg ha^{-1} for N). Soil organic C was generally well predicted for all land use histories when simulated and measured data were compared; however, soil organic N was generally underpredicted. Use of a single, easy to program land use history was ineffective for soil organic C, but worked well for N. Soil organic N and time to N leaching above a threshold leaching level of 0.5 g m^{-2} was not influenced by management. The mean soil organic N content at threshold leaching was 5.0 Mg ha^{-1}, and time to threshold leaching was ~7 yrs. This means that if the DAYCENT estimated leaching is correct, that N-inputs should be reduced earlier than current extension guidelines recommend in Wisconsin. Before DAYCENT could be effectively used for

making turfgrass fertility the differences between measured and simulated soil organic N must be reconciled.

Turfgrass land cover has increased rapidly since 1945 (Lubowski et al., 2006), and now covers over 16 million ha of land in the United States (Milesi et al., 2005). Turfgrass is a perennial cover where N fertilizer applications and infrequent disturbance serve to increase soil organic C. Turfgrass soils can store large quantities of C, ranging from 40 to 177 Mg ha^{-1} to 20 cm after 50 yrs (Selhorst and Lal, 2011; Bandaranayake et al., 2003). Since turfgrass is a growing land cover capable of storing large quantities of soil organic C, it is important to understand how accumulation of C and N may influence need for and environmental safety of N fertilizer inputs.

Computer modeling is one way to assess how changes in site management affect soil organic C and N. The program allows the user to collect a set of measured data to parameterize the model, and obtain additional data through running the model. They also serve as tool to generate and test scientific research questions beyond what typical short term field experiments are able to do. The CENTURY model (a monthly timestep model) and its daily timestep model, DAYCENT, consist of C, N, P, S, forest, plant, and water-balance submodels that work together with inputs of weather, site history, and site soil data to simulate nutrient cycling (Parton et al., 1993; Parton et al., 1987). The CENTURY and DAYCENT models have been used successfully around the world in a variety of land uses from crops to grasslands to forests (Kelly et al., 1997; Parton et al., 1993; Carter et al., 1993) where good relationships or similar trends between measured and simulated values have been observed.

DAYCENT requires soil property inputs of soil texture, bulk density, and soil pH. Soil pH and bulk density are influenced by management (no-till, till, liming, and fertilization). Soil organic C is known to increase as silt and clay content of the soil increase (Six et al., 2002; Hassink 1997). Soil mineral associations are known to occur with organic matter and explain the textural influence on soil organic matter (Kleber et al., 2007). Bulk density can change seasonally, annually, and with management (Alletto and Coquet, 2009). Soil pH may also change with management, and many agricultural practices manipulate soil pH directly or indirectly. Due to the variability in soil management and soil heterogeneity, measured soil properties of texture, bulk density, and pH are important, but the time and equipment required to sample and analyze these properties may be too costly for homeowners or practitioners. The use of DAYCENT (or similar models) to make fertilizer recommendations would be easier if Web Soil Survey generated data are suitable in place of measured data.

The CENTURY model was used successfully in Colorado turfgrass areas to simulate soil organic C and N, aboveground biomass, and N-leaching from turfgrass (Bandaranayake et al., 2003; Qian et al., 2003; Zhang et al., 2013). Soil organic C from CENTURY simulations in Denver and Fort Collins, Colorado began at ~12 Mg ha^{-1} under shortgrass prairie and reached levels of 30-40 Mg C ha^{-1} after ~40 yrs of irrigated turfgrass management. Simulated versus measured soil organic C were well correlated with measured values with slope of 1.05 and fit of 0.67 for the R^2 value on the golf course fairways studied (Bandaranayke, 2003). CENTURY simulations of aboveground biomass in Colorado were within 21% of measured biomass levels (Qian et al., 2003). CENTURY simulations in Colorado showed soil organic C and N under turfgrass increased at faster rates and reached higher levels with a high N fertility regime (150 kg ha^{-1} yr^{-1} N) than low N fertility regime (75 kg ha^{-1} yr^{-1} N). The high N fertility regime reached soil organic C levels of ~65 Mg ha^{-1} 100 yrs after beginning turfgrass management and low fertility regime reached ~55 Mg ha^{-1} soil organic C after 100 yrs. Soil organic N also showed differences between N-fertility regimes, achieving contents of 5 Mg ha^{-1} and 4 Mg ha^{-1} after 100 yrs of turfgrass management for the high and low rates respectively (Qian et al., 2003).

Soil organic C and N from Wisconsin golf courses both increased over time at rates of 0.27 Mg C ha^{-1} yr^{-1} and 0.019 Mg N ha^{-1} yr^{-1}, respectively (Chapter 2). Other studies have shown soil organic C accumulation rates ranging from 0.7 to 1.5 Mg ha soil organic C across the country (Qian et al., 2002; Bandaranayake et al., 2003; Shi et al., 2006; Selhorst and Lal, 2011). Since both land use history and texture may influence the rate of accumulation in soil organic C and N, both may be relevant for making N fertilizer requirements. The research in Wisconsin (Chapter 2) suggests that land use history and fertility of a site may need to be accurate in order to achieve good DAYCENT simulation output.

Nitrogen leaching is problematic for drinking water quality and surface water eutrophication. Qian et al. (2003), demonstrated through CENTURY modeling in Colorado, that after ~30 yrs of N-fertility at a rate of 150 kg ha^{-1} annually, that N-leaching rapidly increased from < 1 kg ha^{-1} annually to levels of ~20 kg ha^{-1} annually (~13% of N applied annually) after 50 yrs of turfgrass management. The low rate of N at 75 kg ha^{-1} annually leached <1 kg ha^{-1} annually (~1% of N applied annually), even after 50 yrs of simulation. High rates of N-leaching from the 150 kg ha^{-1} simulation appeared to coincide with changes in the rate of soil organic N accumulation (N-leaching slowed as an asymptote in soil organic N content was reached) driven by fertilizer inputs. Zhang et al. (2013), showed that early in a turfgrass system (<11 yrs), N-leaching was minimal in both measured data and DAYCENT data in Colorado. Allowing the model to run an additional 40 yrs showed that N-leaching could increase to as much as 50 or 60 kg N ha^{-1} annually with 150 kg ha^{-1} of fertilizer input annually and frequent irrigation. The authors used DAYCENT to establish fertility levels through the lifetime of a site that would maintain established levels of quality based on annual net primary productivity, which resulted in gradual reductions in N-inputs from 150 kg ha^{-1}. As fertilizer inputs were reduced through time, N-leaching was reduced to <6 kg N ha^{-1} annually, a rate similar to atmospheric deposition. Measured data from Michigan showed similar trends in N-leaching as turfgrass systems aged, with leaching reported as minimal during the first 10 yrs after turfgrass establishment, and increasing to levels greater than the EPA threshold quantity of 10 mg L^{-1} when 245 kg ha^{-1} N fertilizer was annually (Miltner et al., Frank et al., 2006).

Nitrogen leaching in turfgrass of Wisconsin was shown to be similar to prairies at around ~5 kg N ha^{-1} yr^{-1} at ~ 3 yrs old (Stienke et al., 2009). Forested sites in the Northeastern United States were shown to be <2 kg N ha^{-1} yr^{-1} (Aber et al., 2003). If normal natural system leaching is ~2-5 kg N ha^{-1} yr^{-1}, then perhaps an acceptable threshold of N leached from turfgrass should be <5 kg ha^{-1} annually.

DAYCENT has potential to be used by soil testing labs to making more refined fertilizer recommendations that maintain turfgrass quality and minimize N-leaching. Researchers have used DAYCENT to accurately simulate soil organic C in turfgrass; however, N has not been evaluated to the same extent. Before DAYCENT could be used in soil testing labs for making turfgrass fertilizer recommendations, it must be calibrated and validated for Wisconsin turfgrass. We donꞌt know how substituting Web Soil Survey properties for measured soil properties will affect the accuracy of DAYCENT predictions of soil organic C and N in turfgrass. DAYCENT also requires land use history to simulate soil organic C and N, but the extent of how detailed the land use history inputs must be is unknown. The objectives of this study were to: 1) Calibrate and validate DAYCENT with a six step process so it may be used to predict soil organic C and N in Wisconsin turfgrass and 2) Use DAYCENT to predict N-leaching from turfgrass areas and explore relationships with soil organic N.

Materials and Methods:

Data Collection.

Soil samples were collected from 260 Wisconsin golf course fairways and roughs in spring 2013 (26 sites, 5 fairway-rough pairs from each). Ten to 12 soil cores were collected with a 2.5 cm diameter sampling probe to a depth of 20 cm to determine soil texture, soil pH, soil organic C and soil organic N content. Soils from roughs were collected in the same manner. One 7.5 cm diameter core, 20 cm deep, was collected to assess bulk density from each fairway and rough. Ten sites were formerly cropped sites that used chemical fertilizer after the 1950s (Chemical Fertilizer Cropped) (time in turfgrass ranged from 9 to 39 yrs) where cropping took place from 1850 until the time turfgrass was planted after 1950. An additional five cropped sites were manured and were cropped from 1850 through the start of turfgrass before 1950 (Manured Cropped) (time in turfgrass ranged from 85 to 106 yrs). Six sites were formerly forested (time in turfgrass to ranged from 20 to 89 yrs), and five were formerly pastured (time in turfgrass ranged from 17 to 119 yrs). Samples for bulk density were dried at 105°C to constant weight, weighed, and bulk density calculated (Blake and Hartge, 1986). Determination of pH was made in 1:1 soil water (Thomas, 1996). Soil texture was determined using the hydrometer method (Bouyoucos, 1962).

Total C and N content was determined with a Carlo Erba 1500 Series 2 Elemental Analyzer (CE Elantech, Lakewood, NJ). Preparation included homogenizing and finely grinding the soil to flour-like consistency using a paint shaker that held 2 mL microcentrifuge tubes containing the sample and two ball bearings, followed by measuring 8-10 mg of soil into tin capsules.

Soil texture, pH, and bulk density information were also collected from Web Soil Survey for each sampling location. Fertilizer inputs, irrigation, and previous land use history were determined from the golf course superintendent surveys in Appendix A. Weather data for each site were collected from the Daymet interpolated data (dates back to 1980).

Calibration and Validation Approach.

Approach. Calibration and validation of DAYCENT (Parton et al., 1987; Qian et al., 2003) for Wisconsin aboveground biomass, belowground biomass, N-leaching, soil organic C, and soil organic N occurred in several stages. The first stage was to set annual above-and below-ground productivity of turfgrass to levels within the ranges reported in the literature for turfgrass. The second stage adjusted N-leaching parameters so simulated N-leaching was similar in magnitude (determined through linear regression of observed and simulated N-leaching) for a long-term turfgrass N-leaching study in a Michigan Alfisol (Frank et al., 2006; Frank, personal communication). Stage three investigated which source of soil properties (measured or Web Soil Survey or a combination of the two) produced the C and N output closest to measured values. Stage four evaluated how land use history influences the soil C and N output, and how specific that land use history must be. Stage five was a fine-tuning process where individual site soil C and N were adjusted using adjustments to manuring or early turfgrass fertility. Stage six evaluated whether or not the original vegetation of forest, and also the simplest scenario to program could be used to achieve C and N output similar to the measured values.

DAYCENT blocking: The equilibrium block was run under the original vegetation of forest. The weather from Daymet, soil texture, bulk density, and pH were entered into the DAYCENT program and a simulation run for 4850 yrs to allow soil organic C to stabilize under the native vegetation. A kill block continued where the equilibrium block left off in time with soil organic C content and removed existing vegetation through clear-cutting. An agricultural block continued after the kill block using a generic cropping sequence that consisted of a wheat, corn, and alfalfa rotation through the 1950s and corn, soybean, and wheat rotation through present day.

Stage 1: Calibration-Validation:Biomass.

In order to set plant productivity values and allocation of C and N to above- and below-ground tissues, an ñidealò soil environment was created using loam texture, 6.5 pH, an average bulk density for loam textured soil and inputs of 100 kg ha^{-1} N annually in fairways and 75 kg ha^{-1} N annually in roughs. The biomass levels in the simulation were compared with literature values for creeping bentgrass (*Agrostis stolonifera* L.) (fairway) and Kentucky bluegrass (*Poa pratensis* L.) (rough) (Qian et al., 2010; Qian et al., 2003). Turfgass lignin content was set to 0.06 for shoots and 0.18 for roots (Bandaranayake et al., 2003; Shearman and Beard, 1975; Ledeboer and Skogley, 1967). Turfgrass C:N ratio was set to range from 20 to 40 (Bandaranayake et al., 2003). Weekly turfgrass mowing events were set to remove ~4% of the aboveground biomass and return it to the soil. Nitrogen fertilizer inputs were distributed in weekly fertilizing events to mimic the use of slow release fertilizers, or spoon-feeding of soluble fertilizer. The simulation was run and plant production values extracted from the model. Simulated plant production values were compared to literature values (Falk, 1976; Qian et al., 2003; Qian et al., 2010), the plant production factor (PRDX) was adjusted if plant biomass were outside of literature values. Once this was complete, calibration and validation proceeded with site-based data.

Five of the formerly cropped sites (50 data points) were simulated with DAYCENT with N fertilizer levels indicated by the superintendent. Measured soil properties from each site were used for the simulation with the DAYMET weather data. Above-and below-ground biomass data were collected and DAYCENT simulated plant production data were compared with measured data to complete the calibration step of the model. If plant production values were predominantly below or above literature values at this stage, additional adjustments to PRDX occurred and sites re-run. An additional five formerly cropped sites were used to validate the plant productivity settings.

Stage 2: Calibration-Validation of N-Leaching.

Nitrogen leaching was fit for a climate similar to Wisconsin using a nitrogen leaching study from Lansing, Michigan that included two N-fertilizer rates: high (245 kg N ha^{-1} yr^{-1}) and low (98 kg N ha^{-1} yr^{-1}) (Frank et al., 2006). An additional three years of unpublished data was supplied by one of the authors of the Michigan study (Frank, personal communication). Adjustments to slope and intercept (fleach1 and fleach3, respectively) for N movement through the saturated profile were used to adjust simulated N-leaching to measured leaching rates from Michigan where fleach1 was decreased from 0.6 to 0.1 and fleach3 was decreased from 0.95 to 0.2.

Stage 3: Soil Properties Sensitivity Analysis

DAYCENT blocking: The equilibrium block was run under the original vegetation of forest. The kill block removed the existing vegetation in cropped and pastured scenarios before initiating agricultural practices (1850). In forested scenarios, the kill block (a burn, removes proportions of C and N similar to logging) took place the year before golf course construction. The agricultural block for the cropped or pastured sites immediately followed the kill-block and continued until the construction of the course (beginning 1851). For the forested sites, no agricultural block existed. The cropping sequence of wheat, corn, and alfalfa through the 1950s and corn, soybean, and wheat through present day was used for cropped sites. The pasture sequence was with moderate grazing. The turfgrass block consisted of golf course construction, turf establishment, and turfgrass growth through present day. The turfgrass block immediately followed the agricultural block for the cropped or pastured sites and for the forested sites it followed a kill block. Courses with significant earth movement had the topsoil moved to piles, followed by contouring of the subsoil and returning the topsoil. Twelve plowing and cultivating events occurred during the construction year to simulate this soil disturbance (weekly plowing events from late May to early August in DAYCENT) compared to five plowing and cultivating events (distributed from late May to early August) in sites that did not have topsoil removal and return.

Measured soil properties were entered first and blocked simulations (as described above) run for all sites. The output data of soil organic C and soil organic N were collected. One measured soil property was replaced with a Web Soil Survey derived soil property, the simulations run again, and the same output data collected. This continued until all eight combinations of measured and Web Soil Survey determined properties were used in simulation (Table 1). Each of these combinations were then compared to the measured values of soil organic C and soil organic N using ANOVA where each combination and the measured value represented a treatment. The combination of soil properties that was least different from the measured values and was the most practical in terms of homeowner or superintendent collection ability was used for subsequent simulations.

Stage 4: Land Use History Sensitivity Analysis.

Forested Sites. Forested sites were simulated under deciduous forest until the year before golf course construction. Sensitivity analysis occurred during the kill block where options were clear-cut, a hot burn for the site, or a medium burn for slash as set in the DAYCENT program. The simulations continued through the turfgrass block. The option that showed the 2013 value of soil organic C and soil organic N closest to the measured values was used in later forested simulations.

Chemical Fertilizer Cropped Sites. Irrigation was not considered because Wisconsin receives sufficient annual rainfall to sustain agronomic crops grown on Alfisols most growing seasons. No-Till and tilled cultivation practices were combined with high and low fertility for corn-soybean rotations on the cropped sites. No-till consisted of using a no-till drill for planting, tilled used a moldboard plow and the no-till drill for planting, high fertility was 250 kg ha^{-1} N (Laboski and Peters, 2012), low fertility included 90.1 kg ha^{-1} N (Laboski and Peters, 2012). Both fertility regimes received 200 kg ha^{-1} N from manure annually before 1950. Manure chemistry, cultivation influence on soil C, and crop productivity was used as set-up in the

DAYCENT program. The cropped site that had the soil organic C and N most like the measured values was used for the final run of the cropped sites.

Manured Cropped Sites. Manured cropped sites became a golf course before 1950, and were likely not fertilized with chemical fertilizer or managed with no till. These sites received manure (200 kg ha^{-1} annually) and tillage for their corn-soybean rotations.

Pastured Sites. Pastured sites were planted to either grass or grass-legume mixtures with low, medium, or high levels of grazing intensities. Grazing intensities and plant productivity were used as set-up in the DAYCENT program. Each grazing intensity and plant combination was simulated and the combination with soil organic C most similar to present day soil organic C was selected for further simulations.

Stage 5: Fine-Adjustment to the Simulations.

Major cities near each of the golf courses were founded between 1830 and 1860. A year of 1850 for conversion to agriculture was deemed a reasonable for these sites. In the 1930s and 1940s a land survey was conducted in Wisconsin to determine the various land uses in each township section, which is also known as the Bordner Survey (University of Wisconsin Digital Collections Center). The Bordner survey served as verification of land use history between the 1930s and opening of the course.

The National Agricultural Statistics Service (2015) was consulted to construct the general cropping sequence for Wisconsin, which was wheat-corn through 1880, oat-corn through 1960, and soybean-corn through present day. The agricultural statistics were further consulted to determine the average size of farms and number of cows on a farm. The number of cows was five in in 1880 and ten in 1920 (NASS, 2015). The quantity of manure produced from the cows plus one team of horses was calculated to determine the amount of manure (Chastain and Camberato; Madison et al.) each farm would have to spread on the cropped fields before 1950 when fertilizer use began to become more common. Manure quantities were set to 15, 35, 46, or 60 g m^{-2} N depending on whether or not the site needed additional C to achieve 2013 quantities of soil organic C. The two lowest rates represent four to eight 455 kg animals.

Tree and crop productivity were adjusted to achieve sufficient above- and belowground biomass C based on literature values (Aber and Federer, 1992; Nadelhoffer and Raich, 1992; Al-Kaisi et al., 2005). Nitrogen deposition was set to constant rates beginning in 1900 - 1.17 g m^{-2} N for the northern sites and southern sites 1.77 g m^{-2} N (Andraski and Bundy, 1990). Nitrogen fixation was set to 0.565 g m^{-2} for all sites (Stevenson, 1982). Sites were further adjusted for early turf fertility or manure (if cropped) as needed to achieve C levels within ~25 Mg ha^{-1} for at least half of the sites within a land use history.

Stage 6: Simplifying Use of DAYCENT with One Land Use History.

Using adjusted tree productivity, the chemical fertilized cropped sites, and the pastured sites, DAYCENT was run by substituting cropping and pasture sequences with forests. The data from the simplified land use history was compared to the actual Wisconsin data (from the actual land use history simulations) using linear regression of simulated versus measured data.

Simulation of N-Leaching.

Ten formerly cropped sites in a gradient of ages were selected to examine DAYCENT predicted N-leaching. Average soil texture, pH and bulk density for the sites fairways and roughs were input into DAYCENT to run site twice through 2013 ï once for each management regime.

Ages in the formerly cropped sites ranged from nine to 97 yrs old, with two sites aged 9-12 yrs, three sites 23-39 yrs old, and three sites 85-97 yrs old. Fertility in fairways ranged from 35 kg ha^{-1} to 149 kg ha^{-1} N annually. Fertility in roughs ranged from 0 to 99 kg ha^{-1} N annually (Table 2). Soil organic C, soil organic N, and N-leaching were selected as the output information for these sites. Site data was examined and the year that the threshold leaching of 5 kg ha^{-1} was reached and soil organic N at that time were recorded.

Statistical Analysis.

Nitrogen leaching output from DAYCENT was plotted with respect to measured values and linear regression performed. For soil property sensitivity analysis (Stage 3), the differences between measured soil organic C and soil organic N and simulated soil organic C and soil organic N data were calculated and ANOVA performed on the differences, with each combination of properties serving as a treatment. The least different option (or most practical if statistically similar) was selected for subsequent stages of DAYCENT fitting. For land use history sensitivity analysis (Stage 4), the statistical analysis proceeded as it did in Stage 3 for each individual land use history (manured cropped, chemical fertilizer cropped, forested, or pastured). In Stage 5, the fine tuning phase, simulated soil organic C and soil organic N were plotted with respect to measured soil organic C and soil organic N and linear regression performed for each land use. Stage 6 (use of forested land use history regardless of actual land use history) simulated data were plotted as simulated soil organic C and soil organic N with respect to measured soil organic C and soil organic N and linear regression performed on the whole data set. Simulated N-leaching data were analyzed to determine management (fairway or rough) influence on simulated soil organic N content at threshold leaching and number of years to that threshold leaching.

Results:

Calibration-Validation.

Stage 1: Plant Biomass Calibration-Validation. Simulated aboveground plant production in the fairway ñidealò scenario ranged from 350 to 400 g m^{-2} C and belowground plant production ranged from 370 to 440 g m^{-2} C. Simulated aboveground and belowground plant production in the roughs ranged from 230 to 270 g m^{-2} C and 210 to 250 g m^{-2} C, respectively. The five calibration and five validation site above- and below-ground biomass many of the sites between the upper and lower bounds of the literature values (Figure 3.2). Calibration and validation data that were below the lower boundary from the literature values were likely lower because of soil texture (i.e. less fertile or soils with low water holding capacity), lack of irrigation, and no to low fertilizer inputs. For instance, Madison 6, Sheboygan 3, and Janesville 3 either did not fertilize their roughs or fertilized at ~25 kg ha^{-1} N annually. Janesville 3 was also a sandy site, containing 56 to 58 g sand 100 g $soil^{-1}$. Racine 3 ranged in sand content from 38 to 60 g sand 100 g $soil^{-1}$, but also had 18 to 40 g 100 g $soil^{-1}$ of silt. Racine 3 irrigated both fairways and roughs while the Madison 6, Sheboygan 3, and Janesville 3 irrigated only the fairways.

Stage 2: Nitrogen Leaching Calibration-Validation. Measured nitrogen leaching from Michigan ranged from 0.5 to 3 g m^{-2} for the high N input treatment (245 kg N ha^{-1}) and was generally less than 0.5 g m^{-2} in the low N input treatment (98 kg N ha^{-1}) (Frank et al., 2006; Frank, personal communication) (Figure 3.3A). DAYCENT simulations of N leaching compared to measured data followed similar magnitudes of separation between the two N fertility rates and similar annual rates of leaching (Figure 3.3B). Linear regression of the simulated and measured data showed good fit between the two data types (R^2=0.52 for high fertility and R^2=0.31 for low fertility) (Figure 3.3A); therefore the adjustments to the slope (fleach1 was decreased from 0.6 to 0.1) and intercept (and fleach3 was decreased from 0.95 to 0.2) driving N leaching in DAYCENT appropriate for use in our Wisconsin sites.

Stage 3: Soil Property Sensitivity Analysis. Modeled soil organic C and N were statistically different from measured soil organic C and N in all soil property combinations used in the simulations (Table 3.1). The soil property combination that resulted in soil organic C closest to the measured value was the combination of measured bulk density, and Web Soil Survey texture and pH; however, the combination was not statistically different from other soil property combinations. The soil property combination that resulted in a soil organic N value closest to the measured value was measured bulk density, Web Soil Survey pH, and Web Soil Survey texture, but was not statistically different from other soil property combinations. Interestingly, the combination of all measured soil properties resulted in the greatest difference between measured and simulated soil organic C and N; however, the measured soil property combination was not statistically different from four combinations for C and five for N. The combination of Web Soil Survey bulk density, measured texture, and measured pH was the most practical for sampling and lab analysis and had simulated C and N values that were not statistically different when all measured soil properties were used to run DAYCENT. The combination of Web Soil Survey, measured texture, and measured pH was therefore used in further simulations.

Stage 4: Land Use History Sensitivity Analysis. Carbon and N from forested site simulation options of clear cut, hot burn, and medium burn for slash were not statistically different from each other (Table 3.3). All three options had C and N contents that were lower than measured values. Clear cut was the most likely option used from a both an economic and practical standpoint. While not statistically different, clear cut had the smallest difference between simulated and measured C and N; so clear cut was used in the final simulations.

Soil organic C and N from chemical fertilizer (cropped 1850 through sometime after 1950) cropped site simulation options (Table 3.3) were lower than measured values, though were not statistically different. The no-till, high fertility option was utilized for the final simulations because no-till resulted in values of simulated soil organic C and N closest to the measured values of soil organic C and N.

The manured cropped sites (cropped 1850 through sometime before 1950) averaged 62.7 Mg ha^{-1} in measured soil organic C and 6.1 Mg ha^{-1} in measured soil organic N. Simulated soil organic C under corn-soybean rotation was not statistically different at 59.6 Mg ha^{-1} (p=0.915), but simulated organic N was statistically lower averaging 3.87 Mg ha^{-1} (p<0.0001).

Pastured simulations, which used combinations of grass or grass-legume with low, medium, or high intensity grazing, were generally above measured values for soil organic C. The low graze, grass combination was statistically different in C from the high graze grass, legume option (Table 3.3). Other pastured options showed C levels similar to one or both of the previously mentioned options. Soil organic N output was similar across all pastured options. The option closest to the measured value was the medium graze, grass-legume mix, and was used for the final simulations since it had the lowest differences in both C and N, even though those differences were not statistically different from other options.

Stage 5: Fine Adjustment to the Simulations. Final adjustment to tree productivity, N-deposition, manure levels, and early N fertility were used to fine tune C and N output. The average measured soil organic C for forested sites was 69.2 and soil organic N was 6.2. The final DAYCENT soil organic C averaged 54.2 Mg ha^{-1} and soil organic N averaged 3.3 Mg ha^{-1}. Linear regression of measured versus simulated soil organic C and N showed slopes of ~1, though simulated values were generally underpredicted, especially for soil organic N (Figure 3.4 A and Figure 3.5 A). The measured C:N ratio was 11.2 and simulated C:N ratio was 16.4.

Measured soil organic C and N for chemical fertilized cropped sites averaged 55.9 Mg ha^{-1} and 5.5 Mg ha^{-1}, respectively. Simulated chemical fertilized cropped site soil organic C averaged 38.8 Mg ha^{-1} and soil organic N averaged 4.9 Mg ha^{-1}. Plotting the measured and simulated data with linear regression between the two showed that chemical fertilized cropped sites generally followed the 1:1 line with R^2 values of 0.423 and 0.368 for soil organic C and N, respectively (Figure 3.4 C and Figure 3.5 C). Measured and simulated C:N ratios were 10.2 and 7.9, respectively.

After including cropping sequence from agricultural statistics and adjusting manure for the manured cropped sites, the overall fit had R^2 values at 0.195 for soil organic C and 0.118 soil organic N and slopes 0.6 and 0.7 (Figure 3.4B and 3.5B). Manured cropped sites had an average measured soil organic C and N of 62.6 Mg ha^{-1} and 4.33 Mg ha^{-1}, respectively. The C:N ratios were 10.2 and 15.2 for the simulated and measured data respectively.

Former pastured site measured soil organic C averaged 71.1 Mg ha^{-1}. Measured soil organic N had a mean of 7.7 Mg ha^{-1}. Mean for simulated soil organic C on former pastured sites was 74.6 Mg ha^{-1} and mean for simulated soil organic N was 4.9 Mg ha^{-1}. Simulated and measured soil organic C and N were plotted and linear regression showed good R^2 values of

0.413 for soil organic C and 0.282 for soil organic N. Soil organic C was distributed both above and below the 1:1 line; however, soil organic N was generally underpredicted (Figure 3.4 D and Figure 3.5 D). Simulated C:N ratio was 15.2 and measured was 9.2. All land use histories were combined into one graph and linear regression performed; however, the relationship was weaker than the individual land use histories plotted separately (Figure 3.6).

Stage 6: Simplification of DAYCENT Use With One Land Use. When simulated as forest (instead of actual former land use), formerly cropped and pastured sites showed no improvement in soil organic C (Figure 7). The slope for the simplified land use was 0.428 and for actual land use was 0.973. The two slopes for soil organic C were statistically different with p=0.022 (t=2.30). For soil organic N, under the forested land use instead of actual cropped or pastured, the slope was 0.813 and for actual Wisconsin data was 0.789. The two slopes for soil organic N were not statistically different with p=0.909 (t=0.115).

N-leaching.

Annual N leaching was simulated from the start of each course through 2013 after simulating earlier history of the course; therefore, leaching data may range from 9 to 39 data points within an individual course. Mean simulated N-leaching had ranged from 0.2 g m^{-2} yr^{-1} to 2.8 g m^{-2} yr^{-1}. The minimum leaching was 0 g m^{-2} yr^{-1} to 0.2 g m^{-2} yr^{-1} and average maximum leaching was 0.36 g m^{-2} yr^{-1} to 5.4 g m^{-2} yr^{-1}.

Soil organic N content when leaching exceeded 0.5 g m^{-2} was not influenced by management (p=0.963). Mean (standard error) soil organic N at threshold leaching was 5.0±0.368 Mg ha^{-1}. Management did not influence the year N-leaching began to consistently be over 0.5 g m^{-2} (p=0.433), and had a mean of 7.4±1.4 yrs. The amount of soil organic N at the time N-leaching was >0.5 g m^{-2} was not correlated to the percentage of N applied (p=0.651).

Discussion:

Web Soil Survey vs Measured Soil Properties Data.

DAYCENT can be used to simulate soil organic C and N with three soil properties that are relatively easy to measure and collect during research sampling. If DAYCENT is to be used for purposes other than research, like making fertilizer recommendations, then soil properties that can be determined from homeowner samples is required. For a homeowner, collecting bulk density would be difficult due to the equipment required, leaving pH and soil texture which are more easily and fairly inexpensively determined by a soil testing lab. Our results showed that Web Soil Survey texture, pH, and bulk density could be used to operate DAYCENT with results that were not statistically different from combinations of Web Soil Survey and measured soil properties. Surprisingly, using all measured soil properties to operate DAYCENT resulted in simulated soil organic C and N values with the greatest differences from measured values of soil organic C and N compared to all other soil property options (Table 3.1).

Universal Land Use History and Uncertainty in Land Use History Timing.

A single, easy to program DAYCENT simulation for land use history (as forest instead of actual land use history) was ineffective for accurately estimating soil organic C, but simple land use did appear to work as well as actual land use data for simulating soil organic N. Soil organic C content and accumulation in Wisconsin turfgrass is influenced by land use history (Chapter 2). Forested sites do not increase in C with time while cropped and pastured sites do (Chapter 2). Since simulated soil organic N did not exactly follow measured soil organic N (Figure 3.5 A ,B, C, and D) it is possible that improved simulated soil organic N would also not work for a single land use history because measured soil organic N accumulates at different rates with land use history. Forested sites decreased in soil organic N and cropped and pastured sites increased in N (Chapter 2).

Land use history was important for measured soil organic C and N and appears to be just as important in simulated soil organic C and N. Measured and simulated soil organic C and N were well correlated. Soil organic C was similar to 1:1 in all land uses, but soil organic N was only similar to the 1:1 line in chemical fertilized cropped sites. In the chemical fertilized cropped sites manuring and adjustments to early turfgrass N inputs aided in achieving a relationship between measured and simulated soil organic N close to a 1:1 relationship. Manured cropped sites did have manure and early turf N inputs; however, uncertainty of beginning the cropping sequence could have led to the differences in soil organic C and N between measured and simulated data.

Forested sites also were under predicted in soil organic N, perhaps due to tree chemistry (*Quercus* spp. have wider C:N ratios than species like *Acer* spp. (Finzi et al., 1998)). Another reason for the lack of better fit between measured and simulated soil organic C and N in formerly forested sites may be that DAYCENT does not simulate the multi-storied nature of a forest.

The differences in soil organic N in former pastured sites may be due to the difference in legume-grass ratio. We used grass-legume mix programed in DAYCENT, and a site with more legumes may result in a different C:N ratio than a site dominated by grasses. Additional manure inputs may aide in accurately simulating soil organic C and N in pastured sites. The grazing options in DAYCENT include manure inputs automatically based on the intensity of grazing, but perhaps additional manure is needed or it was applied during the siteǿs history.

For pastured and cropped sites, the uncertainty of when to initiate agriculture may have led to the differences as well. We used 1850 for both land uses as it appeared to be a reasonable estimate for agriculture initiation; however, if agriculture did not occur until 1875 or 1900 it would lead to differences in soil organic C and N in the simulated data. DAYCENT has successfully been used in all of these environments, producing simulated data for soil organic C and N that was well-correlated with measured data (Li et al., 2006; Kelly et al., 1997; Carter and Stewart, 1996; Ojima et al., 1994; Patwardhan et al., 1995; Carter et al., 1993), but these studies were generally much smaller in scope and worked with only a few sites.

Nitrogen Leaching.

If the threshold for N-leaching is correct and the quantity of soil organic N is correct, cropped sites should not be fertilized at high rates (150 kg ha^{-1} N annually) of N for very long as the y-intercept for cropped sites is ~5.8 Mg N ha^{-1} (Chapter 2). The level of soil organic N at time zero or y-intercept being the same as soil organic N at threshold leaching coincides with the relatively quick onset of elevated N-leaching. The average formerly cropped site used in our study could only be fertilized for ~7 yrs before leaching became a concern, at that point, fertilizer input would need to be reduced in order to minimize leaching losses. The rapid time to N-leaching of 7 yrs is less than the time where reductions in N inputs are recommended by University of Wisconsin-Extension for turfgrass- which is 10 to 12 yrs. The particular amount to reduce N-inputs would need additional research and DAYCENT could prove useful for that parameter as well. However, it is critical that Nitrogen leaching predicted by DAYCENT be verified with measured data from Wisconsin before use is recommended.

Conclusions:

If DAYCENT is to be used for purposes beyond research (for instance management recommendations), the use of easily measured and collected soil properties data is important.

Land use history showed important differences in soil organic C and N, just like measured data. Land use history simulations were well correlated with measured soil organic C in Wisconsin turfgrass; however, the only simulations for soil organic N that were well correlated with measured values were the chemical fertilized cropped sites. Use of a single, easy to simulate, land use history was ineffective for soil organic C. DAYCENT would need additional work fine tuning soil organic N for turfgrass before it could be confidently used to simulate soil organic N, and N-leaching. Similarly, N-leaching needs verification with field data, though it appears potential exists for using DAYCENT to correlate N-leaching over a threshold of 5 kg ha^{-1} with soil organic N.

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Table 3.1. Difference between measured soil organic C or N in turfgrass soils of Wisconsin and DAYCENT simulated soil organic C or N for sensitivity analysis of soil property combinations (texture, pH, and bulk density) (n=260).

Soil Texture Source	pH Source	Bulk Density Source	Measured ï Simulated Soil Organic C	Measured ï Simulated Soil Organic N
			Mg ha^{-1}	
Measured	Measured	Measured	25.9 Aÿ	3.9 A
WSSÀ	Measured	Measured	13.1 B	3.1 B
Measured	Measured	WSS	18.8 AB	3.4 AB
Measured	WSS	Measured	18.6 AB	3.4 AB
WSS	Measured	WSS	12.8 B	3.1 B
WSS	WSS	Measured	12.1 B	3.0 B
Measured	WSS	WSS	19.1 AB	3.5 AB
WSS	WSS	WSS	18.6 AB	3.4 AB
		$LSD_{0.05}$	9.62	0.759

ÀWSS: Web Soil Survey
ÿ Letters within a column denote statistical differences between treatments at $p<0.05$.

Table 3.2. Sites, age, and fertility of sites used for DAYCENT simulations to assess N-leaching from turfgrass in Wisconsin.

Site	Former Land Use	Age	Fairway Fertility	Rough Fertility
			kg ha^{-1} N	
Madison 7	Cropped	9	124	99.0
Madison 8	Cropped	12	111	74.3
Madison 5	Cropped	25	99.0	49.5
Janesville 3	Cropped	32	149	0
Sheboygan 3	Cropped	39	99.0	0
Eau Claire 1	Cropped	38	198	123.8
Eau Claire 2	Cropped	28	198	123.8
Sheboygan 6	Cropped	11	74.3	0
Sheboygan 7	Cropped	10	148.5	125.8
Madison 6	Cropped	19	49.5	24.8

Table 3.3. Difference between measured and DAYCENT simulated values of soil organic C and N from former land use history options for golf courses in Wisconsin.

Option	*n*=number of samples	Measured - Simulated Soil Organic C	Measured - Simulated Soil Organic N
		--------------- Mg ha^{-1}	---------------
		Forested Land Use History	
Hot Burn, All	60	-16.8 AÀ	-2.98 A
Clear Cut	60	-15.9 A	-2.94 A
Medium Burn Slash	60	-16.1 A	-2.94 A
$LSD_{0.05}$		10.5	0.917
		Chemical Fertilized Cropped Land Use History	
HF, NT, CS	100	17.1 A	2.81 A
HF, T, CS	100	21.9 A	3.18 A
LF, NT, CS	100	15.7 A	2.89 A
LF, T, CS	100	22.9 A	3.28 A
$LSD_{0.05}$		17.1	1.25
		Pastured Land Use History	
HG, GL	50	2.60 A	3.35 A
HG, G	50	-4.61 AB	2.95 A
LG, GL	50	-5.01 AB	2.83 A
LG, G	50	-15.5 B	2.29 A
MG, GL	50	-2.74 AB	2.99 A
MG, G	50	-13.3 AB	2.43 A
$LSD_{0.05}$		17.9	1.60

ÀLetters within a column and land use history denote statistical differences at $p<0.05$.
ÿCropped options included: HF=high fertility, NT=no-till, CS=corn-soybean, T=tilled, LF=low fertility. Post WWII cropping means sites were cropped from 1850 through sometime after WWII, with cessation of cropping occurring when planted to turfgrass.
§Pastured options included: HG=high intensity grazing, GL=grass-legume plants, G= grass plants, LG=low intensity grazing, MG= medium grazing.

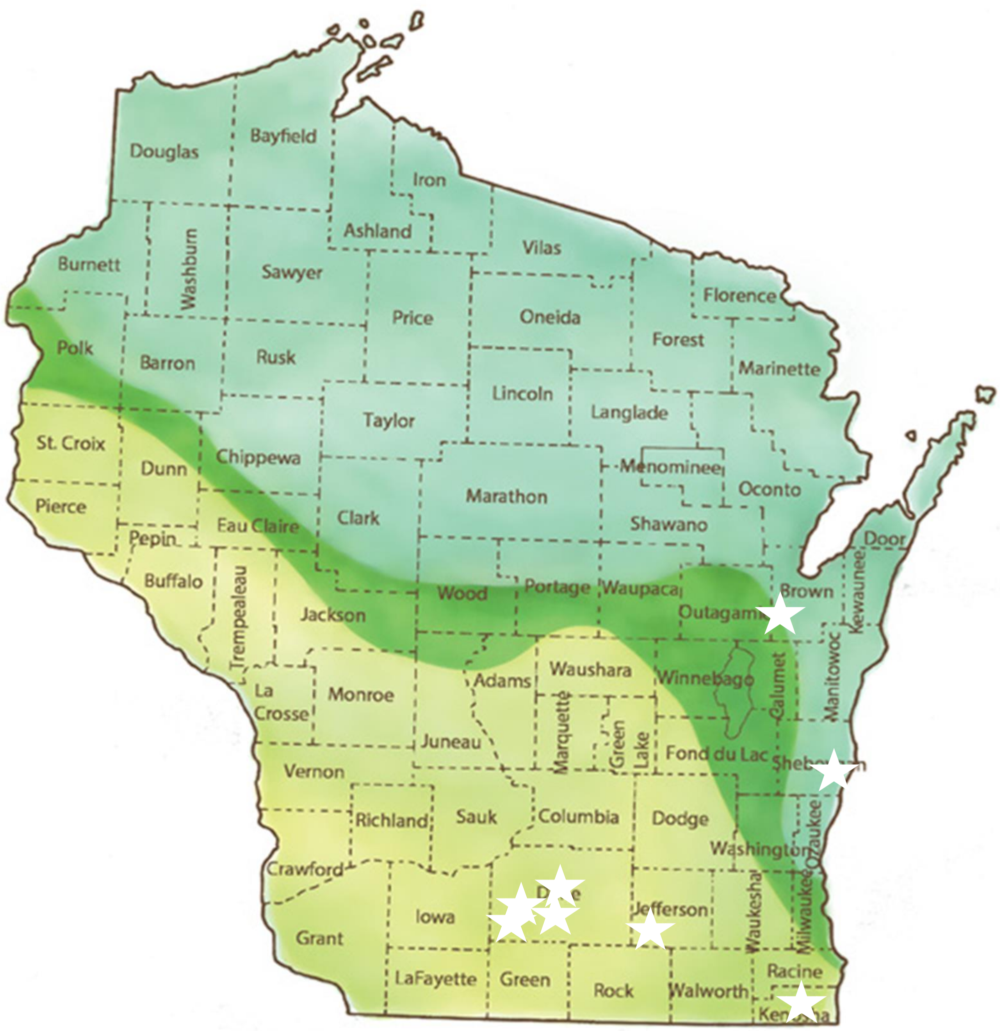

Figure 3.1. Map of Wisconsin golf courses used in DAYCENT N-Leaching simulations.

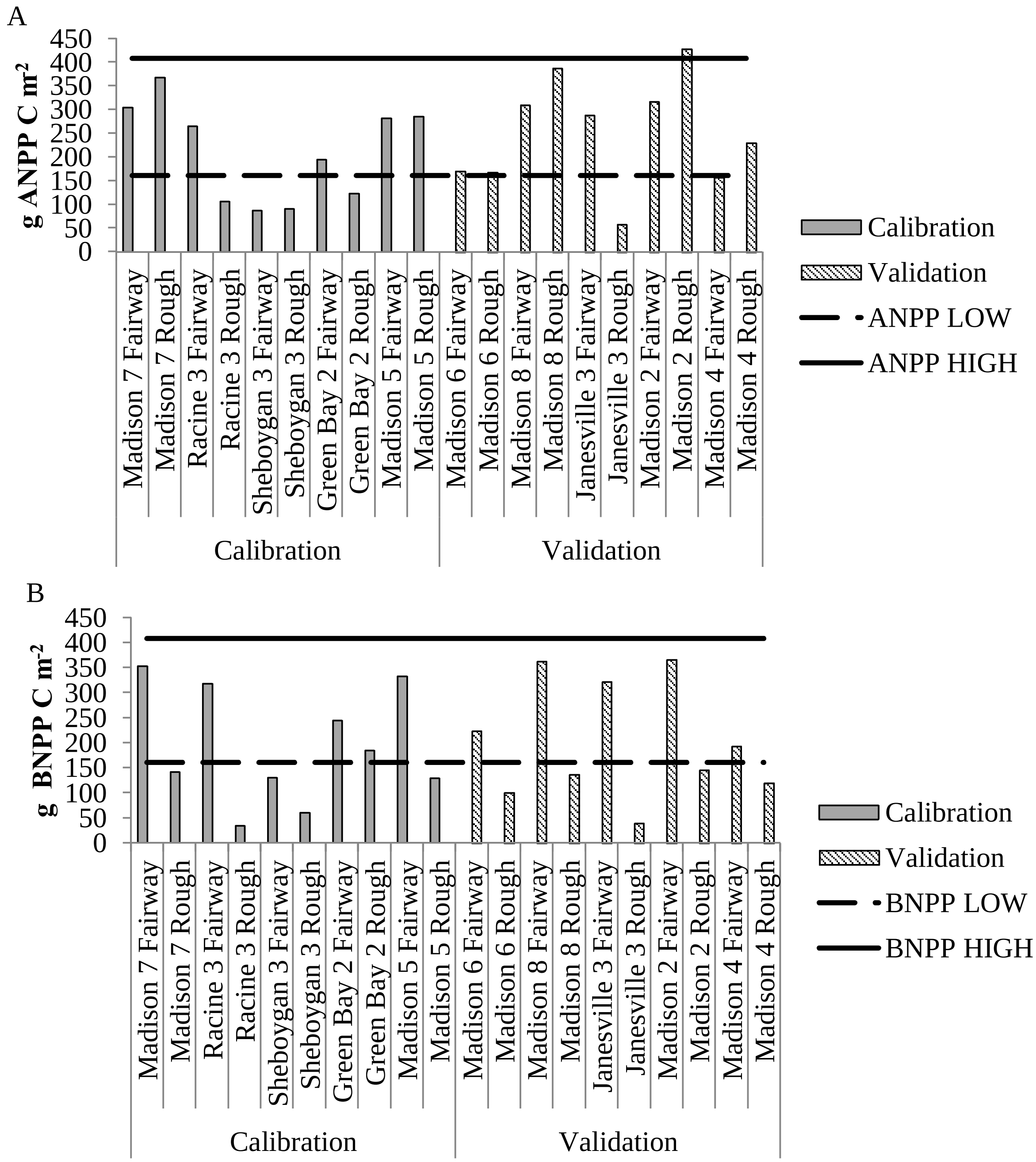

Figure 3.2: DAYCENT simulated (A) Aboveground and (B) Belowground Net Primary Productivity (NPP). Literature values (lines) of aboveground NPP and belowground NPP (Falk, 1976).

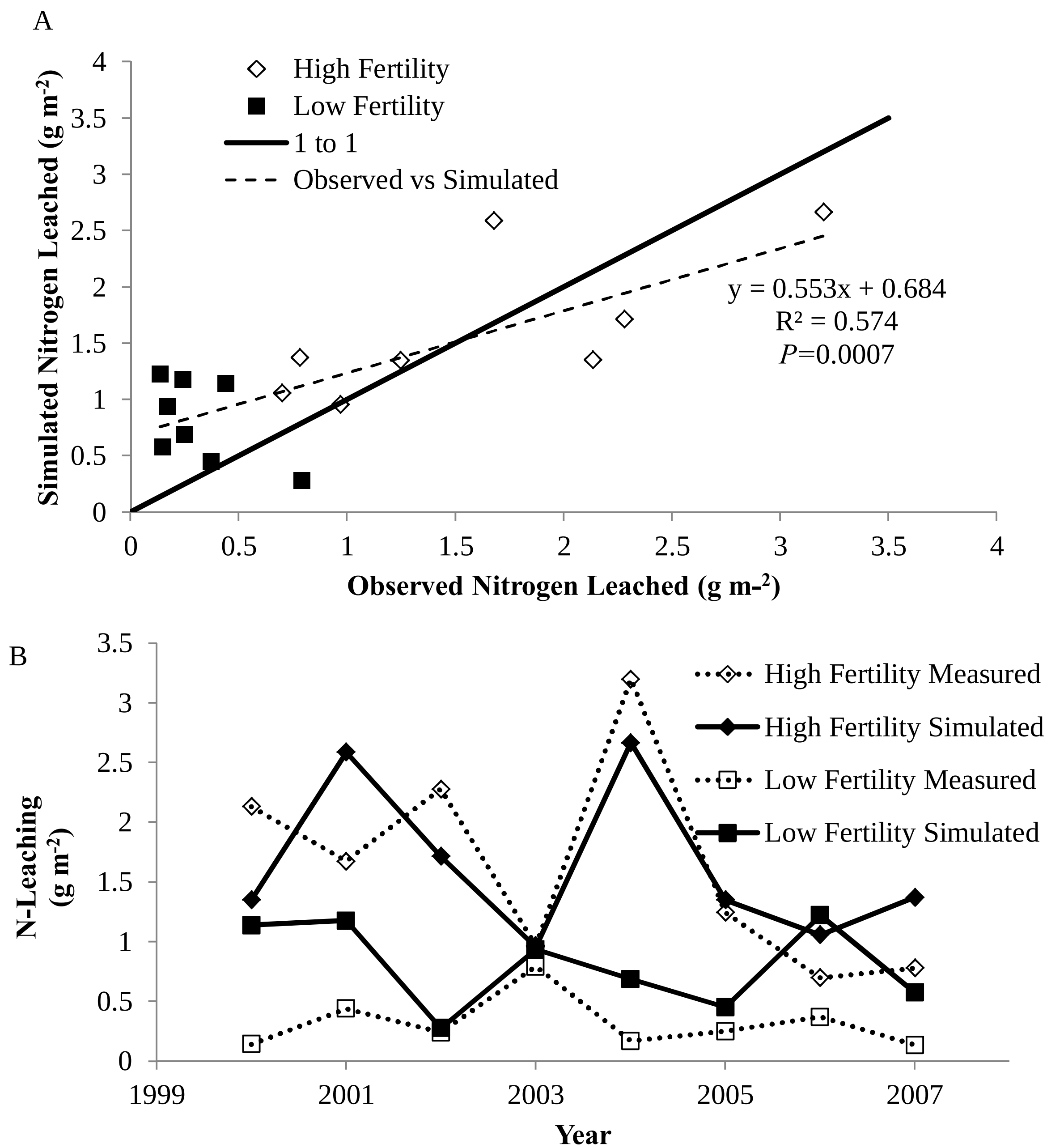

Figure 3.3: (A) Measured versus DAYCENT simulated annual N-leached in Michigan turfgrass study (Frank et al., 2006; Frank, personal communication). (B) Measured and DAYCENT simulated annual N-leached from Michigan turfgrass areas over time. Michigan data used to calibrate DAYCENT for N-leaching in Wisconsin. High fertility was 245 kg N ha^{-1} yr^{-1} and low fertility was 98 kg N ha^{-1} yr^{-1}.

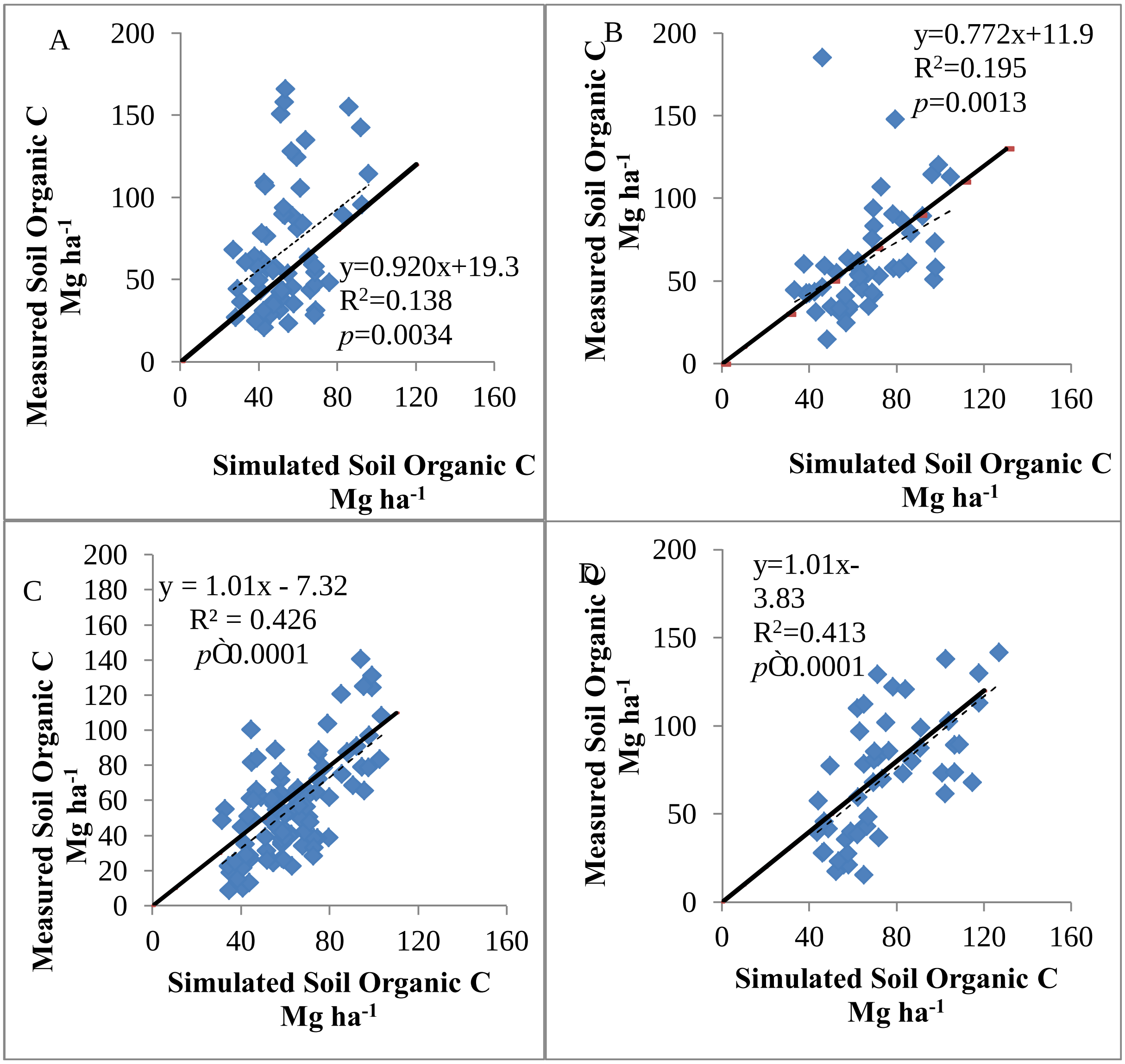

Figure 3.4: Simulated vs measured soil organic C for (A) formerly forested, (B) manured (cropped 1850 through sometime before 1950), (C) chemical fertilized cropped (cropped 1850 through sometime after 1950), (D) formerly pastured for Wisconsin golf courses. Solid black line is the 1:1 line, dashed line is the regression line for the data points. Confidence Intervals for Slopes were: (0.316, 1.52), (0.317, 1.23), (0.775, 1.25), (0.659, 1.36) for A, B, C, and D. Confidence Intervals for Intercepts were: (-14.7. 53.3), (-19.1, 42.9), (-22.7, 7.94), (-31.2, 23.1) for A, B, C, and D.

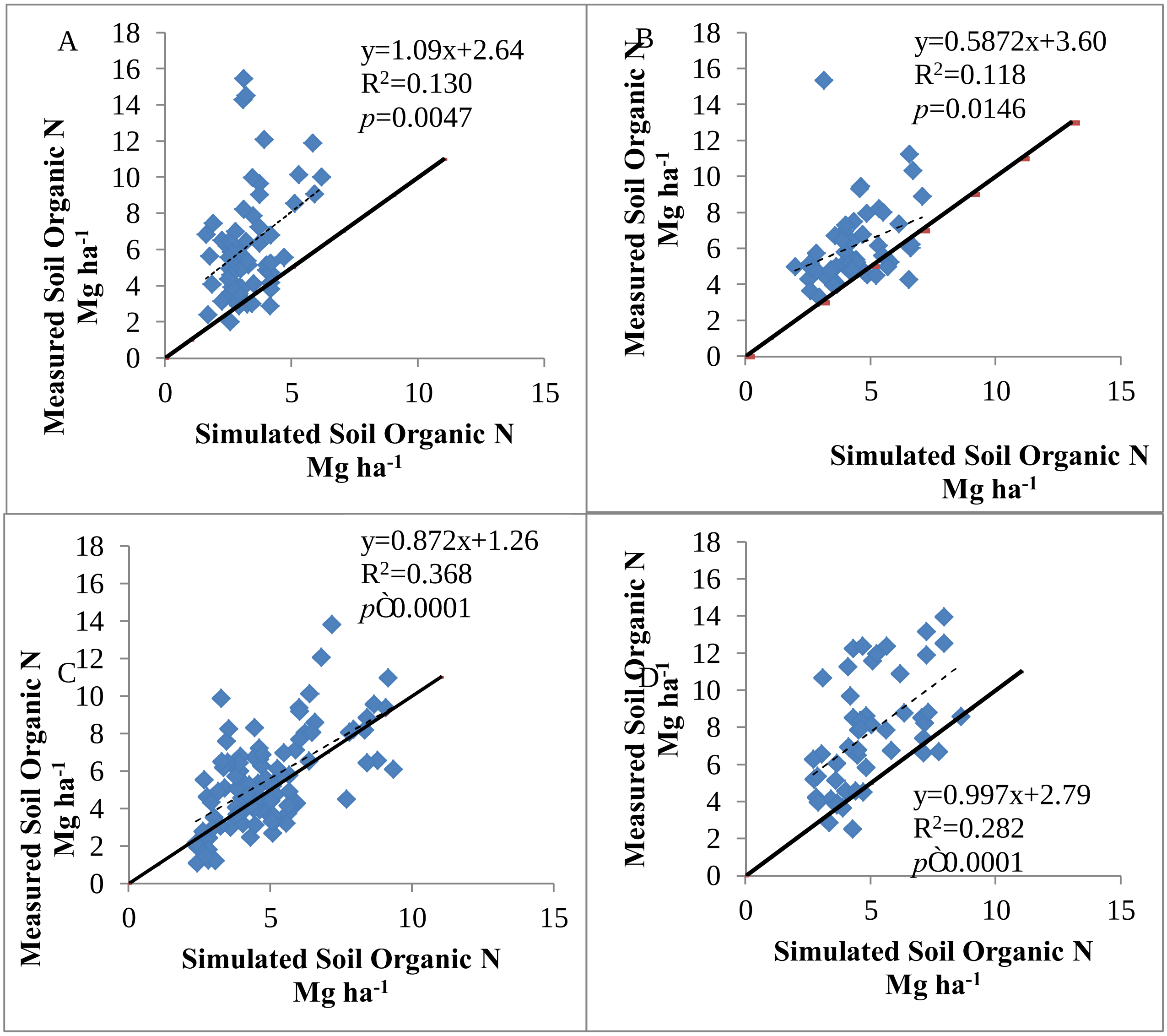

Figure 3.5: Simulated vs measured soil organic N for (A) formerly forested, (B) manured (cropped 1850 through sometime before 1950), (C) chemical fertilized cropped (cropped 1850 through sometime after 1950), (D) formerly pastured for Wisconsin golf courses. Solid black line is the 1:1 line, dashed line is the regression line for the data points. Confidence Intervals for Slopes were: (0.349, 1.84), (0.121, 1.05), (0.643, 1.10), (0.535, 1.45) for A, B, C, and D. Confidence Intervals for Intercepts were: (0.0904, 5.20), (1.49, 5.70), (0.0822, 2.44), (0.385, 5.19) for A, B, C, and D.

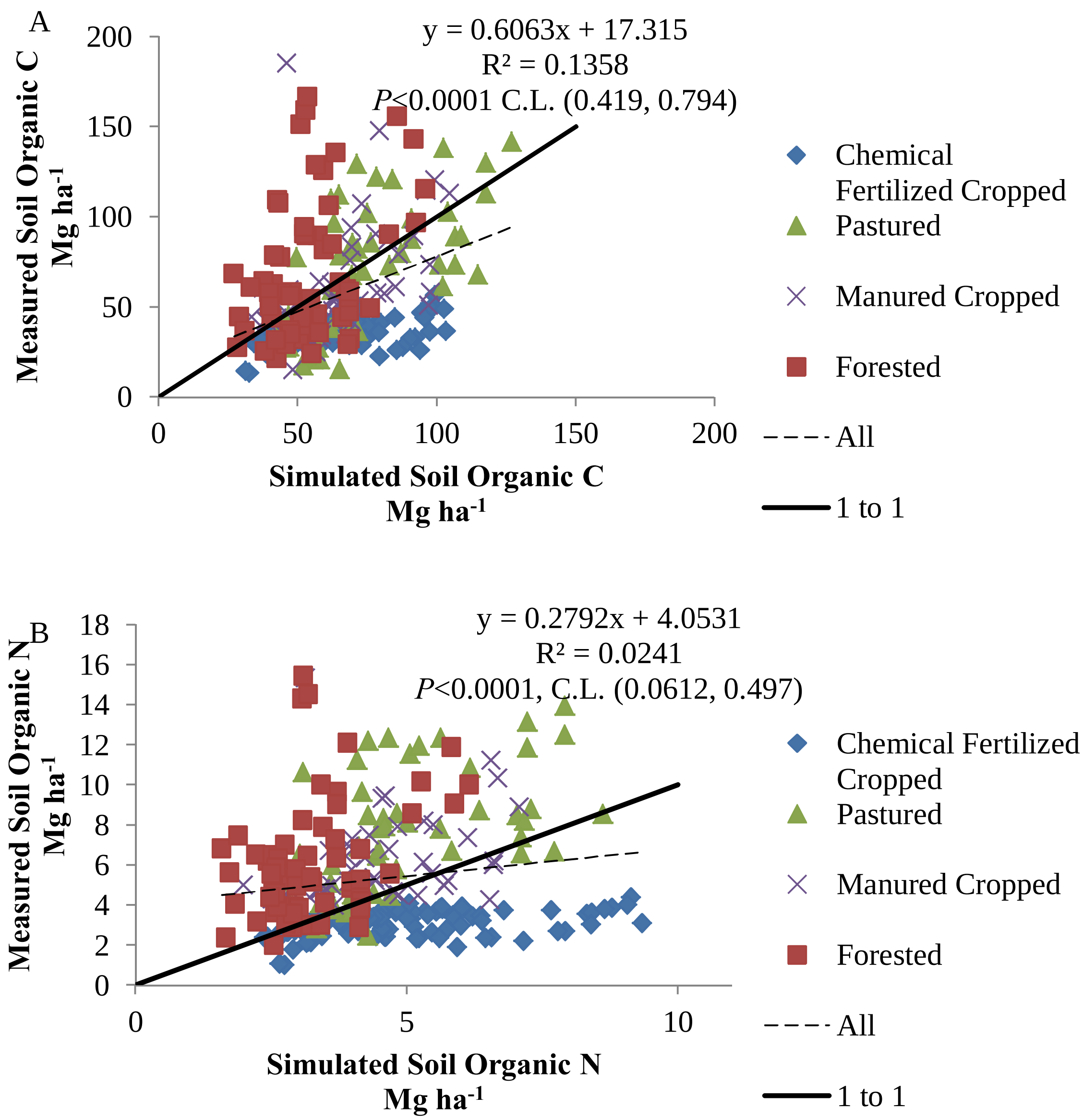

Figure 3.6: Regression of DAYCENT simulated and measured soil organic C (A) and soil organic N (B) across land use history in turfgrass of Wisconsin. C.L.=confidence limit of the slope.

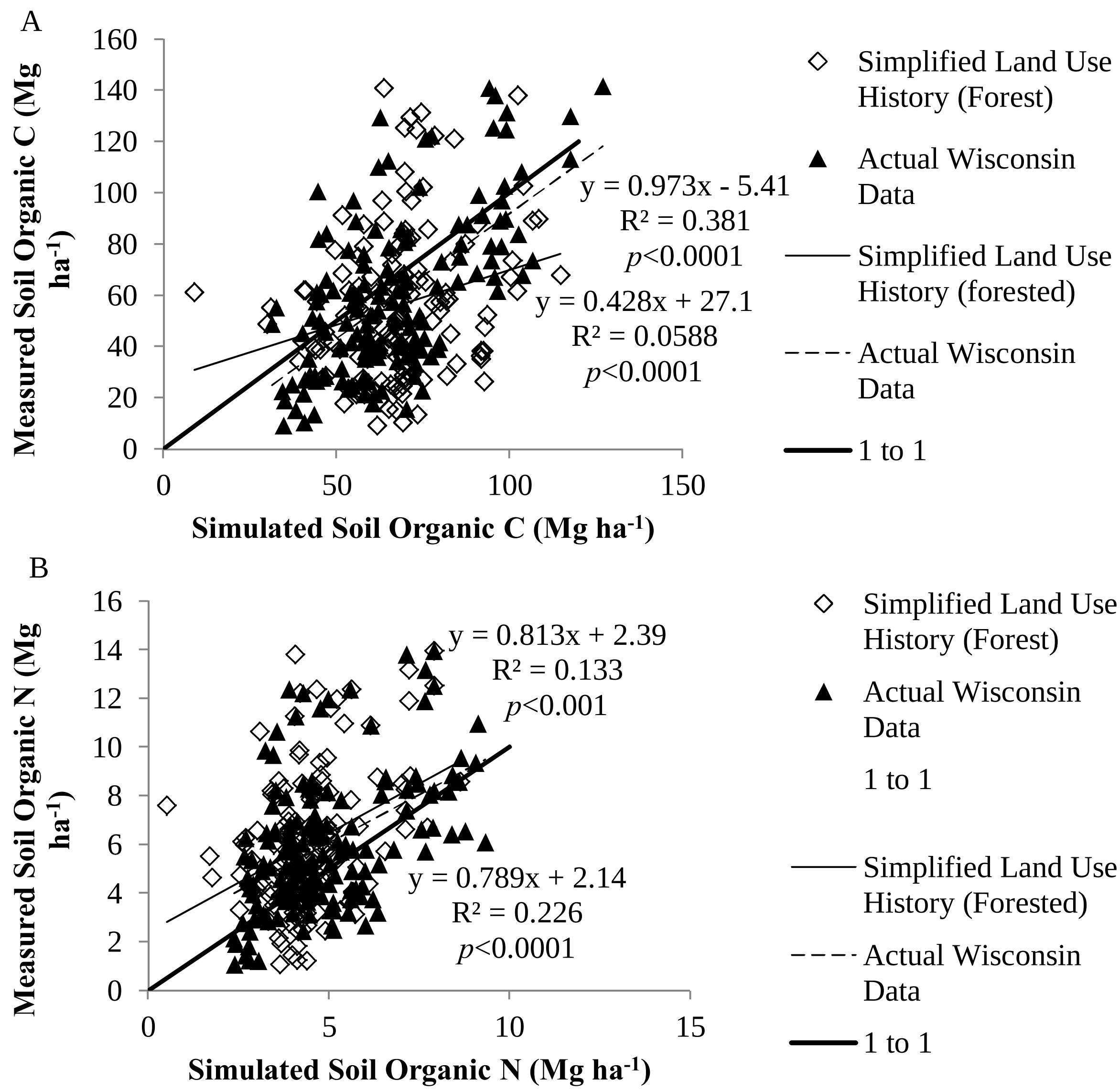

Figure 3.7: Regression of measured and DAYCENT simulated soil organic C (A) and soil organic N (B) when sites were run with actual land use history or as forest, ignoring land use history. DAYCENT was run under forest regardless of land use history for and with actual land use history for the Wisconsin data.

CHAPTER 4

SOIL FRACTIONATION AND COMPARISONS OF C AND N CONCEPTUAL MODELS IN TURFGRASS SOILS OF WISCONSIN

Abstract

Soil organic C and N accumulate in turfgrass areas across the United States, but accumulation of C and N occurs at different rates. Soil fractionation into particle sized C and N pools and use of C and N conceptual models may give insight into why C and N accumulate at different rates in turfgrass soils. Conceptual models of soil organic C and N in the literature have been developed for agricultural and forested ecosystems, but the behavior of C and N was similar across systems. Turfgrass is a blend of both systems in plant type (perennial), management (fertilized and sometimes irrigation) and little soil disturbance once established, so the conceptual models likely apply, with the largest portion of C and N in the particulate organic matter pools. The objectives of this study were to: determine the quantity of soil organic C and N in particulate and mineral-associated pools through physical size fractionation; calculate maximum C and N contents within the clay pool and compare with actual values of soil organic C and N in the clay pool to determine saturation level in soil organic C and N, and compare turfgrass system soil organic C and N with C and N conceptual models from the literature. Soil samples were collected from 15 golf courses in Wisconsin varying in age (9 to 93 yrs old), management and land use history. Soils were fractionated into particulate (coarse and fine) and mineral (silt and clay) associated organic C and N. Silt-associated C and N comprised the largest pool. Soil organic C and N saturation ratios for clay minerals in Wisconsin showed predominantly minerals undersaturated in organic C and N. Particulate organic matter C and N followed conceptual models by increasing with respect to silt plus clay content. However, mineral associated C and N did not increase with increased whole soil C and N, contrary to conceptual models. Because many of the clay minerals were undersaturated with C and N, and mineral associated C and N

appeared to not increase with age since establishment, these turfgrass sites may have been at effective stabilization capacity. Turfgrass areas at an effective stabilization capacity may cycle C and N similar to what a system with a saturated mineral associated C and N pool, meaning increased losses once a system is saturated or at effective stabilization capacity, if N inputs are not decreased.

Soils are a large terrestrial sink for organic C but seem to have a finite storage capacity (Gulde et al., 2008; Stewart et al., 2007). As a soil fills with C, the difference between the capacity and current level is reduced, leading to reduction of C storage ability with that soil (Stewart et al., 2007). Similarly, organic N can become saturated in soils, potentially leading to increased N leaching if N inputs are not adjusted (Lovett and Goodale, 2011). Nitrogen leaching is problematic from the standpoint of eutrophication of surface waters, contamination of groundwater, and the associated human health consequences of drinking water high in NO_3^-. Yet, organic C and N allocation and storage capacity within soil pools under turfgrass is largely unknown.

Turfgrass covers approximately 16 million ha in the United States (Milesi et al., 2005). Soils under turfgrass management typically accumulate C and N with time across the United States (Selhorst and Lal, 2011; Shi et al., 2006; Bandaranayake et al., 2003; Qian and Follett, 2002). Turfgrass soils accumulate C at different rates, depending upon location. Turfgrass sites in Wisconsin accumulated C at a rate of ~0.27 Mg C ha^{-1} yr^{-1} with all sites together (Chapter 2). The value for all Wisconsin site C accumulation was low compared to literature values, which were ~1 Mg ha^{-1} yr^{-1} (Bandaranayake et al., 2003); however Chapter

2 was a much broader study than Bandaranayake et al. (2003), where three soil series were studied under former land use of shortgrass prairie in two locations. Individual soil series accumulated C at rates of 0.8, 0.5, and 0.3 Mg C ha^{-1} yr^{-1} in Wisconsin, with some sites likely even less.

Soil fractionation into various C or N pools may help explain why turfgrass soils may accumulate soil organic C and N at different rates. To our knowledge, only one study has examined allocation of C and N to various pools in turfgrass soil. Yao et al, (2010) used a density fractionation and found that the light fraction comprised ~10% of total C and N, while >80% of the total soil C and N was recovered in the heavy, mineral-associated fraction in turfgrass soils in North Carolina and Nevada. In the North Carolina soils, greater quantities of C and N were mineralizable compared to the Nevada sites. Since the Nevada sites mineralized less C and N, more C and N were stored compared to the North Carolina sites, resulting in differing rates of accumulation. Different rates of accumulation lead to difficulty in predicting accumulation of soil organic C and N in turfgrass.

Conceptual models of soil organic C and N cycling may be useful for understanding turfgrass C and N cycling, but we do not know if conceptual models proposed in other ecosystems may apply to turfgrass systems as turfgrass systems contain substantial biomass and infrequent soil disturbance. These conceptual models are useful for visualizing and explaining how C and N cycling may change as a system saturates in C and N. Studies that have evaluated conceptual models have done so in forests or cultivated land uses (Castellano et al., 2012; Stewart et al., 2007; Stewart et al., 2008). One conceptual model of soil organic C predicts that C accumulates at the same rate, regardless of how much is already in the system, leading to a system that never saturates (Stewart et al., 2007). A second conceptual model of C predicts that the amount of C that is stored is dependent upon the quantity of C in the system, and eventually will reach an asymptote where the system stores only a small portion of the C inputs. The third conceptual model of C proposes that different pools accumulate C differently, where some pools may accumulate C linearly and others may saturate in C (Stewart et al., 2007). These conceptual models of C were tested in agricultural systems of the United States (Stewart et al., 2007; Stewart et al., 2008). Stewart et al. (2008), fractionated soil into the following pools: free particulate organic matter, microaggregate-associated C, silt and clay associated C, and nonhydrolyzable C. Mineral-associated C pools tend to saturate in C while particulate organic matter pools increase linearly in soil organic C in a variety of locations in the United States and Canada (Stewart et al., 2008). Stewart et al. (2007), also proposed that an effective stabilization capacity could be achieved, which was defined as the maximum capacity of C storage due to management and plant inputs of the system, and is at a lower level than C saturation. A system that is at an effective stabilization capacity may behave like a saturated system, but would not necessarily be saturated in the mineral-associated C and N pools.

Building on Stewart et al.ô s (2008) work, Castellano et al. (2012) used similar concepts to form hypotheses regarding N saturation in forested ecosystems of Maryland. Castellano et al. (2012), predicted that as the mineral pool filled, the quantity of N in particulate organic matter and net nitrification would increase, while the C:N ratio of the particulate organic matter pool would decrease. Using A and B horizons of a forested ecosystem to ensure a gradient in saturation, Castellano et al. (2012), tested their model by fractionating the soil into particulate organic matter and mineral associated organic matter (silt plus clay), measuring the N content in each fraction, and using ^{15}N incubations to track the movement of new N in the system. As the mineral associated N increased in N, the particulate organic matter N also increased, and the C:N ratio of particulate organic matter decreased rapidly. Net ammonification and net nitrification both increased as the mineral pool filled with N (Castellano et al., 2012).

To predict saturation of soil C in mineral pools, a number of studies have used linear regression of C in fine soil fractions with respect to the quantity of fine soil fraction, however, calculation of saturation through loading and surface area may give a better estimate of saturation. To find a better method to predict saturation, Feng et al. (2013), calculated organic matter loading to determine maximum saturation for crops, forests, and grasslands and compared the calculated loading to the loading provided by linear regression. In the instances of 2:1 minerals, the calculated loading overpredicted C loading, but in 1:1 mineral loading, the calculated loading under predicted C. The researchers suggested that more detailed information on clay types, surface area of clays or loading of C to clay minerals would likely give better fits for soil organic C loading, rather than general estimates of surface area (15 g m^{-2} for 1:1 minerals; 80 g m^{-2} for 2:1 minerals) and loading (1 mg C m^{-2}) (Feng et al., 2013).

Calculation of organic C and N loading of clay minerals and saturation deficit of C and N may assist in determining why soil C accumulation rates may differ among various sites. The comparison of our data with conceptual models allows for better visualizing behavior of C and N in the turfgrass system and may lend additional insight into differences in C and N accumulation rates. Slow rates of C and N accumulation could be due to saturated mineral-associated C and N pools.

Soil fractionation of C and N into various pools, determination of C and N saturation, and comparison of soil organic C and N with conceptual models from the literature, provides a mechanistic understanding of how turfgrass soils store and accumulate soil organic C and N. The objectives of this study were to: 1) determine the quantity of soil organic C and N in particulate and mineral-associated pools through a physical-size fractionation approach; 2) determine turfgrass soil organic C and N saturation in the clay-associated pool by calculating maximum soil organic C and N loading on clay minerals and determining saturation ratios; 3) use the physical-size fractionation of soil organic C and N pools combined with whole soil organic C and N to compare turfgrass system soil organic C and N accumulation with C and N conceptual models from the literature; 4) Using a wet-sieving procedure, determine the quantity of water stable aggregates in turfgrass soils. Knowing in which pools C and N are stored in turfgrass, the degree of saturation, and the general behavior of C and N through conceptual models may aide our understanding why our Wisconsin turfgrass soils accumulated C and N at lower rates than other turfgrass studies in the literature (Chapter 2).

Materials and Methods:

Soil samples (20 cm depth) were collected from 15 golf courses planted on Alfisols of varying soil texture distributed throughout Wisconsin in spring after turfgrass was actively growing. Turfgrass sites were distributed between northern and southern Wisconsin based on Curtisôwork (1959). Ten to 12 2.5 cm diam soil cores were collected from each of three fairways and three roughs at each site. Soil samples were gently passed through a 4 mm sieve, soil texture determined, and incubations for N mineralization conducted as described in Chapter 2. Six formerly forested sites and nine formerly cropped sites were selected for fractionation, because they were two of the most common previous land uses for golf courses in Wisconsin. The nine cropped sites were selected to give a gradient in age from nine to 93 yrs old (n=54). Former forested sites formed a gradient from 20 to 87 yrs old for a total of 36 samples.

Soils were processed using a physical size-fractionation approach (Castellano et al., 2012) to characterize differences in the distribution of C and N pools with different degrees of association with mineral particles. Subsamples of 25 g from each soil sample were weighed into 125 ml plastic bottles and 100 ml of deionized water added to each. Samples were shaken on a reciprocal shaker to disperse the soil for 16 hr. Samples were rinsed through a 250-µm sieve with deionized water until water was clear and the flow-through portion collected. The portion remaining on the 250-µm sieve was considered coarse particulate organic matter plus sand (cPOM). The flow-through portion was rinsed through a 53-µm sieve until water flow was clear and the new flow-through portion collected. The portion remaining on the 53-µm sieve was considered the fine particulate organic matter plus sand (fPOM). The 53-µm flow-through was allowed to settle for two hrs to allow silt to settle to the bottom of the jar. A vacuum pump (Welch 2511, Niles, IL) with tubing and pipette tip was used to remove the water plus suspended clay particles from the silt fraction. The portion at the bottom of the jar after siphoning was the silt-associated organic matter and the suspended portion was the clay-associated organic matter. All fractions were dried at 65ºC until constant weight and final dry weight recorded.

Before elemental analysis, soils were pulverized to a flour-like consistency in a paint shaker modified to hold 2 ml microcentrifuge tubes with two ball bearings. Eight to 10 mg of each fraction were weighed into sample tins and rolled tightly for analysis. Carbon and nitrogen analysis of each fraction was conducted using Carlo Erba 1500 Series 2 Elemental Analyzer (CE Elantech, Lakewood, NJ). Standards containing 0.27% N and 3.19% C were used to calibrate the instrument.

To determine C and N loading and saturation of soil minerals, linear regression of clay or silt plus clay with the C in the mineral fraction was plotted with the intercept forced through zero or not (e.g Feng et al., 2013; Six et al., 2002). Organic matter loading (or capacity) was calculated for the clay portion of the soil. Loading (mg C or N m^{-2}) was multiplied by mean surface area for the type of mineral, which was then multiplied by the mass (g clay 100 g $soil^{-1}$) of clay-sized (<2.0 µm) soil particles. A regression line forced through zero was plotted for fine soil particles and organic matter C content in the clay pool. The slope then determined maximum carbon loading. Degree of saturation was calculated as the ratio between measured and calculated C contents in the clay pool, where >1 means oversaturated minerals. Surface areas used in calculation were 20 g m^{-2} for kaolinite, 95 g m^{-2} for montmorillonite, 85 g m^{-2} for illite, and 85g m^{-2} for chlorite (Brady, 1990). Organic matter loading can range between 0.5 and 1.0 mg C m^{-2} (Christensen, 2001); however in soil pH > 5.5 loading is lower, ranging from 0.25 to 1 mg C m^{-2}, with a mean of ~0.43 mg C m^{-2} (Mayer and Xing, 2001) which was used in our study. To determine soil organic N loading rate for clay minerals, the C:N ratio for clay in grasslands (Guggenberger et al., 1994) was used to convert C loading to N loading. Nitrogen loading was calculated as described above for C.

To determine clay type, we measured X-ray diffraction using a Scintag PAD V diffractometer. Clay preparation included soaking ~200 g of soil sample in a 1000 mL beaker with deionized water for four hrs and extracting the top 5 cm. Samples were then centrifuged for 20 min at 5.0x1000. Supernatant was decanted and two drops of deionized water were added to the clay pellet to assist spreading sample on the slide for diffraction. Three samples from the three soil regions (Madison and Gundlach, 1993; Hole et al., 1968), which encompass south-central and south east WI (B), central and north central near Wausau, WI (F), and along Lake Michigan, including Green Bay (I) were selected for X-ray diffraction. These sites were randomly selected within a region because of the concentration of courses in the region and because soils from the sites were used in the fractionation. Samples were analyzed from 2° to 60° at 45 kV and 40 mA in 2° increments. Measurements occurred for air-dried samples, ethylene glycol overnight treated samples, and 400°C for 1 hr treated samples. Diffractograms were analyzed using JADE software (Fredrickson, 2013; Fredrickson, personal communication). The weight percentages were then used for calculating maximum C and N contents on clay as previously described.

To compare our data to the Castellano et al. (2012) conceptual model, silt and clay associated N pools were summed to achieve a total mineral-associated N pool. Maximum net potential N mineralization (Chapter 2; Drinkwater, 1996; Bundy and Meisinger, 1994) and aerobic N mineralization to NH_4^+ and NO_3^- (Chapter 2; Hart et al., 1994) from whole soil was determined. Dependent variables of: particulate organic matter N, C:N ratio of particulate organic matter, maximum net potential N mineralization, and aerobic incubation products were plotted against silt plus clay associated N and linear regression run for each variable. A dependent variableôs behavior with respect to increased mineral associated-N was considered to follow the model if the behavior was similar (increase, decrease), and statistically different from zero.

To determine whether or not turfgrass soils of Wisconsin follow the conceptual models proposed by Stewart et al. (2007, 2008), soils were fractionated as previously described. Dependent variables of silt and clay associated C and particulate organic matter C were plotted with respect to whole soil C. Nitrogen in the silt plus clay fraction and particulate organic matter N were also plotted with respect to whole soil N. The behavior of the pools (increase, decrease, linear, logarithmic) was compared with the conceptual models to determine whether or not our data followed the conceptual model or the regression was statistically important.

To determine aggregate stability in our study, soils from fairways and roughs were combined and 20 g of soil were soaked in deionized water for 30 min, placed on a stack of sieves (1 mm, 0.5 mm, and 0.25 mm). Sieves were submerged in water and oscillated vertically once per second for 20 s, ensuring that all soil remained submerged. Aggregates remaining on sieves were dried at 105°C to constant weight (Spaccini and Piccolo, 2012; Kemper and Rosenau, 1986). The percentage of water stable aggregates in each size class (>1 mm, 0.5 to 1 mm, 0.25 to 0.5 mm, <0.25 mm (determined by difference) were analyzed for influence of age, land use history, and relationship with whole soil organic C and N Determination of sand-free aggregates involved adding 10 mL of 5% sodium hexametaphosphate solution and 20 mL of water were added to each of the three aggregate size classes (>1 mm, 0.5 to 1 mm, and 0.25 to 0.5 mm). Each sample was dispersed using a drink mixer, on low speed for 30 s, followed by rinsing the silt and clay components through a 53 µm sieve until water ran clear. Samples were dried at 105°C until constant weight.

Statistical Analysis. Analysis of variance was used to discern differences between log-transformed soil C and N in fractions and determine if land use history, management, or age influenced C and N content in the soil fractions. Soil fraction data were non-normally distributed. Influence of land use history, management, or age on saturation ratio of C and N was determined through ANOVA (analysis was not significantly impacted by transformation of data, so analysis proceeded on non-transformed data). The impact of climate on saturation ratio of C and N was determined with ANOVA on log-transformed C and N saturation ratios. Linear regression was used to determine the relationship between N-mineralization and saturation ratio. Nitrogen mineralization parameters did not meet assumptions of normality, but analysis with log-transformed data did not influence the statistics so non-transformed analysis was reported. To examine the relationships between C and N in the silt and clay associated pools (measured and calculated) with silt plus clay or clay content of the soil, linear regression was used on non-transformed data (statistics were not different when analyzing the transformed data). Linear regression was also used to determine relationships between particulate organic matter, NH_4^+ production, NO_3^- production, and maximum net potential N mineralization and mineral associated N. This final phase of linear regression was not transformed because transformation did not significantly influence the statistical outcome, but one data point that was greater than three times the standard deviation was excluded.

Results:

Soil Characterization. The soils in our study had a mean (and standard error) of 3.3% ±0.11 for organic matter, measured by loss on ignition. The soil pH averaged 7.1 ±0.06. Soil texture averaged 16.0% ±0.56 clay, 36.3% ±1.12 silt, and 47.7% ±1.29 sand. Clay analysis via X-ray diffraction showed predominantly 2:1 type minerals (Table 4.1), with an average 2:1 to 1:1 ratio of 1.9. More northern sites (regions F and I) contained similar types and quantities of clay minerals; while the southern site (region B) was predominantly 1:1 minerals. The X-ray diffraction data facilitated calculation of C and N loading in our sites.

Soil Fractions. Organic C and total N contents differed among soil fractions (Table 4.2). The silt-associated fraction contained approximately three times the concentration of C and four times the concentration of N as the clay-associated fraction. The silt-associated fraction contained about two to three times the C and N concentration of either of the particulate organic matter fractions. Total particulate organic C was 6.77 kg $soil^{-1}$ with fine particulate organic matter comprising ~60% of the total particulate organic matter. Total mineral associated C was 9.35 g kg $soil^{-1}$ with silt-associated C consisting of ~76% of the total mineral C. Total particulate organic matter N and mineral N were 0.392 and 0.614 g kg $soil^{-1}$, respectively. Silt-associated N comprised ~79% of the total mineral N, and fine particulate organic matter was ~57% of the total particulate organic matter N.

Management (p=0.411 for C and p=0.884 for N; Table 4.3) did not influence soil organic C and N distribution among the soil fractions. Land use history was influential for soil organic C and N distribution among soil fractions (p=0.0063 for C and p=0.0109 for N) (Table 4.3, 4.4). Climate was influential for soil organic C in the fractions (p=0.0557), but not soil organic N (Table 4.4). Age of site also did influence soil organic C or N in the soil fractions when analyzed in aggregate (p=0.0163 for C, p=0.0081, for N). Recovery of soil (sum of % bulk soil) ranged from ~95-100% (Table 4.2). The percentage of total soil organic C and N was different between fractions, with the majority of the soil organic C and N in the silt-associated fraction (Table 4.2). Fine particulate organic matter soil organic C and soil organic N increased with site age (p=0.0017, R^2=0.106, log[NfPOM]=-2.81+0.0259*age for N and p=0.00380, R^2=0.0910, log[CfPOM]=0.932+0.00916*age for C), though the fit of the linear regression was weak. Coarse particulate organic matter C and N were not influenced by age of site (p=0.0670 for C and p=0.183 for N). Silt and clay associated C and N pools were not influenced by age (p=0.489 for silt C, p=0.777 for silt N, p=0.309 for clay C, p=0.436 for clay N).

Saturation and Loading. Soil organic C increased at a rate of 0.2 g C kg $soil^{-1}$ per 1% increase of silt plus clay with linear regression not forced through zero (non-transformed data so the slopes may be compared to other studies) (Figure 4.1A). Soil organic N increased at a rate of 0.02 g N kg $soil^{-1}$ per 1% increase of silt plus clay for regression line not forced through zero (Figure 4.1B). Fit of the linear regression line for soil C and N and silt plus clay was generally poor with R^2 values of 0.08 for soil organic C (p=0.026) and 0.075 for soil organic N (p=0.0092).

Calculation of soil organic C loading on clay showed maximum C content to be greater than the linear regression line for the minerals we analyzed (Figure 4.2A). The loading rate for the linear regression line of measured data was 0.19 mg C g $clay^{-1}$ and for calculated C associated with clays was 0.25 mg C g $clay^{-1}$. The loading rate for N based on linear regression of measured data was 0.015 mg N g $clay^{-1}$ and for calculated N on clays the loading was 0.030 mg N g $clay^{-1}$ (Figure 4.2B). The calculated loading of our sites based on the minerals we analyzed showed predominantly undersaturated minerals for both C and N (Figure 4.3 A and B). Carbon saturation ratio calculations showed that ~33.3% of minerals in our sites were oversaturated. About 27% of minerals were within ±0.2 of saturation (with 1 being saturation). Nitrogen saturation ratio showed that 20.0% of clay minerals were oversaturated. Approximately 21% of the clay minerals loaded to within ±0.2 of saturation for N. If the same calculations are made for literature-based Wisconsin clay values, only 22% of the clay minerals are within ±0.2 of saturation for C.

Factors that may influence C and N content in the soil (management, land use history, and age) did not influence the saturation ratio of C or N in our turfgrass soils (p=0.232, 0.944, 0.246 for management, land use history and age and C saturation ratio; p=0.332, 0.634, 0.149 for management, land use history and age and N saturation ratio). Climate did influence the log-transformed C and N saturation ratio (p=0.0035 for C and 0.0387 for N), where southern sites had a mean saturation ratio of 0.452 for C and 0.502 for N while northern sites had mean saturation ratios of 0.857 for C and 0.192 for N. Saturation ratio of C or N did not influence nitrification (Table 4.5), but did influence ammonification. Ammonium production increased as the saturation ratio increased, meaning that as more N was present in the clay pool, additional ammonification could occur. Maximum net potential N mineralization increased as saturation ratio of C and N in the clay pool increased (Table 4.5). Maximum net potential N mineralization increased at rates of 1.7 μg N g soil^{-1} d^{-1} per unit increase in saturation ratio of C and at rates of 1.6 μg N g soil^{-1} d^{-1} per unit increase in saturation ratio of N.

Measured Data and Conceptual Models. The turfgrass systems we measured did not follow the Castellano et al., (2012) conceptual model of N saturation in most aspects of the model. In the model, particulate organic matter N was expected to increase as silt+clay associated N increased, and particulate organic matter N from golf course soils did show a relationship with increased N in the silt+clay pool (p=0.0476) (Figure 4.4A). The Castellano model predicted that particulate organic matter C:N ratio would decrease, and while the C:N ratios of particulate organic matter did decrease respect to increased silt+clay associated N, the decrease was not significant (*p*=0.109) (Figure 4.4B). Net mineralization to N-NH_4^+ was expected to increase with increased silt+clay associated N according to the Castellano model. During the aerobic incubation, N-NH_4^+ increased slightly with increased N in the silt+clay pool; however, this increase was not statistically significant (*p*=0.761) (Figure 4.5A). Aerobic nitrification did not change with increased silt+clay associated organic N (*p*=0.766) (Figure 4.5A). The anaerobic incubation, measuring maximum net potential mineralization increased significantly by 1 µg NH_4^+ g N silt+clay^{-1} (p=0.0497) (Figure 4.5B).

Stewart et al. (2007, 2008)ôs conceptual models of C showed that particulate organic matter pools increase linearly while mineral-associated pools increase and then reach a plateau. Linear regression of clay associated C and silt associated C with whole soil C showed no increase in mineral-associated C pools with respect to whole soil C (for clay: *p*=0.243, for silt: *p*=0.501). Clay and silt associated N pools did not change with respect to whole soil N (for clay: *p*=0.91, for silt: *p*=0.437).

After accounting for sand content in each of the aggregate fractions age was a non-factor in aggregate stability (*p*=0.531 for 0.25 to 0.5 mm, *p*=0.247 for 0.5 to 1mm, and *p*=0.817 for >1 mm). Small sand free aggregates (0.25 to 0.5 mm) were not influenced by land use history (*p*=0.0928). Medium (*p*=0.0379) and large (*p*=0.0610) sand free aggregates were influenced by land use history with forested sites having greater quantities of medium sized sand free aggregates (38.2% for forested and 32.3% for cropped) and fewer in the large size class (26.0% for forested and 35.3% for the cropped sites) compared to cropped sites. The percentage of water stable sand free aggregates of large (>1 mm), medium (0.5 to 1 mm), and small (0.25 to 0.5 mm) in forested land use histories showed no relationship with whole soil N (p=0.599, p=0.151, p=0.730 for large, medium, and small aggregates), or whole soil C (p=0.781, p=0.721, p=0.638 for large, medium, and small aggregates).

Medium and small water stable sand free aggregates in formerly cropped sites showed no relationship with whole soil C (p=0.981 for medium and p=0.0727 for small) or whole soil N (p=0.687 for medium and p=0.0567). The percentage of large water stable sand free aggregates was positively related to whole soil C (p=0.0429, R^2=0.154, wholesoilC=16.9+0.208*%WSA) and whole soil N (p=0.0118, R^2=0.228, wholesoilN=1.65+0.0193*%WSA).

Discussion:

Soil fractionation of soil organic C and N and aggregate stability data support the hypothesis that our sites may be at an effective stabilization capacity. The marginal support from our data supporting conceptual models of soil C and N, and our calculated saturation levels in soil organic C and N add further support to the hypothesis. Effective stabilization capacity is the maximum capacity of C storage due to the management and plant inputs of the system, and is at a lower level than C saturation (Stewart et al., 2007).

Support for effective stabilization capacity through the soil organic C and N fractions.

Our soil organic C and N fractions had concentrations of C and N similar to what would be considered ñstableò (forest, grassland) systems that were more likely to be at an effective stabilization capacity. Our particulate organic matter C (~10 g C kg soil^{-1}) was greater than the observed literature values of 3.79 to 4.69 g C kg soil^{-1} (Plante et al., 2006) in tilled formerly forested sites, ~4 g C kg soil^{-1} in no-tilled fields of the central US (Cambardella and Elliott, 1992), and ~7 g C kg soil^{-1} native grasslands in the central US (Cambardella and Elliott, 1992). Our particulate organic matter N (~0.9 g N kg soil^{-1}) was more like forested sites in the US which contained 0.64 g particulate organic matter N kg soil^{-1}. Because conceptual models propose that C and N saturated systems (Castellano et al., 2012; Stewart et al., 2008; Stewart et al., 2007) continue to increase in particulate organic matter after reaching saturation in other pools, the large quantity of particulate organic matter in our turfgrass system is not surprising if the behavior of sites at effective stabilization capacity is the same as those that are saturated in C and N. Our mineral associated C and N (~15 and 1.2 g kg soil^{-1} for C and N respectively) were similar to no-till agriculture and native grasslands which contained ~12 g C kg soil^{-1} and 1.2 g N kg soil^{-1} in the no-till system and 10.7 g C kg soil^{-1} and 1.08 g N kg soil^{-1} in the native grassland (Cambardella and Elliot, 1992).

Support for effective stabilization capacity through conceptual models of C and N.

Conceptual models, while not supported by our data for saturation appear to support effective stabilization capacity. Stewart et al. (2008) demonstrated that clay associated C (<2 µm) and silt associated C (2 to 53 µm) increased before reaching a plateau while particulate organic C pools increased linearly. If our sites are within the asymptote of C-accumulation for mineral-associated C pools, then the appearance of a lack of C increase with respect to whole soil C would be expected. A second theory that may support the presence of clay minerals undersaturated in C and the lack of an increase in C in the mineral pool with respect to whole soil C, is the concept of an effective stabilization capacity (Stewart et al., 2007).

Particulate organic matter N, especially fine particulate N continued to increase with time, which may also indicate the mineral pool is at capacity. While Castellano et al. (2012) showed that as the mineral N pool approached saturation in deciduous forests, the particulate organic matter pool continued to increase. The silt and clay associated pools of N did appear to be increasing in our turfgrass sites. Since particulate organic matter appears to continue to increase in our study, turfgrass soils in Wisconsin may be at capacity in the mineral pool, according to conceptual models (Castellano et al., 2012; Stewart et al., 2008).

Increased potential N mineralization with increased mineral N was expected based on the research by Castellano et al. (2012). As the mineral-associated C and N pool fills, easily accessible C and N are mineralized, decreasing the C:N ratio of particulate organic matter as a portion of the organic material is returned to the atmosphere as CO_2. This means that as the mineral-associated N pool becomes saturated, the N from the particulate organic matter is mineralized and more likely to be involved in leaching if not immobilized. While our sites did not show an association between C:N ratio of particulate organic matter (although the general trend was a decrease) and filling of the mineral pool, it may be another indicator that our sites contain mineral-associated C and N at effective stabilization capacity. The same holds true for our saturation ratio calculations and N-mineralization.

Support for effective stabilization capacity through saturation calculations.

Saturation calculations showed primarily undersaturated minerals, indicating lack of saturation in our sites. Clay or silt associated pools of soil organic C showed no differences with age or with total soil organic C (p=0.243 for clay; p=0.501 for silt), suggesting that the clay or silt associated pools may be saturated, however, saturation ratios showed that our sites were predominantly undersaturated with respect to mineral associated soil organic C and N. Our calculated maximum loading was 0.25 mg C g clay^{-1} and measured loading (through linear regression) was 0.19 mg C g clay^{-1}. Literature values of calculated loading were 0.15 mg C g^{-1} for 1:1 minerals to 0.80 mg C g^{-1} for 2:1 minerals (Feng et al., 2013); however, measured loadings in the literature (through linear regression) were often between these two values and ranged from 0.26 to 0.63 mg C g^{-1} clay (Feng et al., 2013; Six et al., 2002). Sixty one percent of the clay minerals were 2:1, so according to Feng et al. (2013)ǒs estimates, our loading should be ~0.53 mg C g $clay^{-1}$. We used an average surface area from the literature for each clay type (for instance, kaolinite has an average surface area of 20 m^2 g $clay^{-1}$, and illite 85 m^2 g $clay^{-1}$) for our calculations of loading rather than a mean surface area across 2:1 clays and another for 1:1 clays as Feng et al. (2013) had done. Use of the average surface area for each clay type improved estimates of maximum loading (0.19, 0.25, and 0.53 mg C g $clay^{-1}$ for linear regression of our data, our maximum calculated loading, and loading of our sites based on Feng et al. (2013), respectively). We only included clay (<2 µm) for which loading estimates of C and clay mineral surface areas are determined. Other studies (Feng et al., 2013; Six et al., 2002; Hassink, 1997) included fine silts in their calculations by using the soil fractions <20 µm. Inclusion of fine silts and using surface areas for clay in the calculation would lead to overestimates in C loading, as silts have much smaller surface areas compared to clay.

Silt and clay associated N pools showed no difference with age or total soil organic N (p=0.56, R^2=0.003), which indicates that the mineral pools may also be saturated with N. Calculated loading of N on clay minerals showed many undersaturated minerals. However, similar to C, N may be at a level of effective stabilization capacity for turfgrass, since many clay minerals were observed to be undersaturated with N.

Support for effective stabilization capacity through aggregate stability.

Data on aggregate stability in our turfgrass areas seem to support the conclusion of saturation or effective stabilization capacity. Studies have shown that aggregate stability increases with increased time since disturbance until reaching some maximum quantity of water stable aggregates (Burri et al., 2009; Wick et al., 2009). Because the transition from a forest to turfgrass maintains the perennial nature of the plant species at the site, the lack of relationships between C and N parameters and percentage of water stable sand free aggregates was not surprising. The lack of relationships between the C and N parameters and the water stable sand free aggregates also points to saturation or effective stabilization capacity in the formerly forested turfgrass sites in our study. In US agricultural and native system comparisons, the more stable the ecosystem would be presumed to be considered (forest, prairie) the greater the quantity of water stable aggregates compared to no-till or tilled agriculture (Six et al., 2000). The amount of water stable aggregates increased by 40 to 75% across conventional tillage, no-tilled, and native vegetated sites (Six et al., 2000). In a former landslide area in Switzerland, the amount of stable aggregates in the climax forest was more than twice that of the control site (allowed to revegetate on its own) (Burri et al., 2009). In a reclaimed coal mine chronosequence in Colorado showed that aggregate stability may be achieved relatively quickly (after 10-15 yrs) and levels of stable aggregates that were similar to native vegetation (Wick et al., 2009). The lack of change in aggregate stability suggests a stable system that could be at an effective stabilization capacity for soil organic C and N.

Conclusions:

Soil fractions from turfgrass showed no influence of age or management. The silt pool contained the greatest quantity of soil organic C and N. Particulate organic matter pools increased with age, but mineral-associated pools of C and N did not. Calculated loading of C and N on clay minerals and saturation ratios between calculated and measured levels of C and N showed predominantly undersaturated minerals. Conceptual models compared to our data may support the idea of C and N saturation of Wisconsin turfgrass systems (if turfgrass is within the asymptote of C and N accumulation) where conceptual models predicted that mineral pools cease to increase in C or N, but particulate C and N and mineralization continue to increase once mineral pools are filled. A more likely explanation, considering the undersaturation of C and N in clay minerals, is that turfgrass soils of WI may be at effective stabilization capacity. Data from the soil fractions, comparisons with conceptual models, calculation of saturation, and aggregate stability support the hypothesis that turfgrass soils of WI are at an effective stabilization capacity for soil organic C and N.

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Table 4.1. Clay type and percentages from one turfgrass site from each of three soil regions (Madison and Gundlach, 1993; Hole et al., 1968) in Wisconsin. Clay type was determined through X-ray diffraction for current sites.

Mineralogy Source	Soil Region	% of clay type	Clay Species	Clay Type
Current Sites	B	59.5	Kaolinite	1:1
		23.3	Illite	2:1
		12.2	Chlorite	2:1
		0.50	Montmorillonite	2:1
	F	48.3	Muscovite	2:1
		30.5	Kaolinite	1:1
		21.2	Chlorite	2:1
	I	62.4	Illite	2:1
		21.3	Kaolinite	1:1
		16.3	Chlorite	2:1
Brown and Jackson, 1958; Hiawatha loamy sand	H	13.7	Illite	2:1
		80.2	Montmorillonite	2:1
		6.1	Kaolinite	1:1
Borchardt et al., 1968, Lapeer loam	B	21.0	Mica	2:1
		12.0	Vermiculite	2:1
		12.7	Montmorillonite	2:1
		9.2	Amorphous	
		12.2	Kaolinite	1:1
Borchardt et al., 1968, Saylesville silt loam	B	37.8	Mica	2:1
		9.8	Vermiculite	2:1
		19.9	Montmorillonite	2:1
		4.6	Amorphous	
		5.1	Kaolinite	1:1
Borchardt et al., 1968, Saylesville silt loam	B	31.2	Mica	2:1
		18.2	Vermiculite	2:1
		2.8	Montmorillonite	2:1
		7.4	Amorphous	
		5.9	Kaolinite	1:1
		4.9	Chlorite	2:1
Glenn et al., 1960, Tama silt loam	A	19.1	Illite	2:1
		2.4	Vermiculite	2:1
		52.1	Montmorillonite	2:1
		19.1	Amorphous	

		7.2	Kaolinite	1:1
Fanning and Jackson, 1965, Varna silt loam	A	42.0	Mica	2:1
		5.0	Chlorite	2:1
		9.0	Amorphous	
		30.0	Vermiculite/Montmorillonite	2:1
Mean Literature Clay		33.0	Illite	2:1
		11.5	Vermiculite	2:1
		1.9	Chlorite	2:1
		36.6	Montmorillonite	2:1
		9.9	Amorphous	
		7.3	Kaolinite	1:1

Table 4.2. Mean (and standard error or significant difference as indicated by letters) of soil organic C and soil organic N in soil fractions of Wisconsin golf course soils.

Fraction	C in soil fraction	N in soil fraction	C:N	% fraction soil of bulk soil	% of whole C	% of whole N
	g C kg soil^{-1}	g N kg soil^{-1}				
cPOM	2.72 BC	0.168 B	17.0 (1.26)	18.2 (1.37)	23.3 (2.93)	18.5 (2.29)
fPOM	4.05 B	0.224 AB	13.8 (0.806)	23.3 (1.13)	32.1 (4.93)	27.3 (3.62)
Silt	7.14 A	0.488 A	14.5 (0.881)	42.3 (1.54)	60.6 (8.18)	52.0 (7.47)
Clay	2.21 C	0.126 B	16.1 (1.26)	13.7 (0.750)	16.7 (2.00)	13.6 (1.76)

Table 4.3. Means (and standard error) of the C and N of the size class of soil from Wisconsin golf courses according to management and previous land use history.

Fraction	---g soil organic C kg soil^{-1}----		---g soil organic N kg soil^{-1}---	
	Management			
	Fairway	Rough	Fairway	Rough
cPOMÀ	3.94 (0.536)	5.33 (0.941)	0.321 (0.0537)	0.404 (0.0673)
fPOM	5.77 (0.859)	6.49 (1.15)	0.491 (0.0356)	0.561 (0.112)
Silt	13.2 (2.48)	10.2 (1.48)	1.12 (0.0819)	0.863 (0.135)
Clay	2.97 (0.361)	3.77 (0.538)	0.237 (0.243)	0.295 (0.0474)
	Land Use History			
	Cropped	Forested	Cropped	Forested
cPOM	3.68 (0.450)	6.08 (1.15)	0.276 (0.0431)	0.493 (0.0822)
fPOM	5.05 (0.815)	7.75 (1.27)	0.423 (0.0792)	0.680 (0.123)
Silt	11.5 (2.10)	11.9 (1.92)	0.962 (0.199)	1.03 (0.178)
Clay	3.80 (0.479)	2.73 (0.363)	0.284 (0.0429)	0.240 (0.0367)
	Climate			
	North	South	North	South
cPOM	4.50 (0.886)	4.79 (0.587)	0.348 (0.0632)	0.379 (0.0582)
fPOM	7.05 (1.08)	7.05 (0.895)	0.596 (0.105)	0.445 (0.0866)
Silt	10.9 (1.64)	12.5 (2.56)	0.940 (0.147)	1.05 (0.247)
Clay	3.43 (0.537)	3.31 (0.337)	0.269 (0.0459)	0.264 (0.0363)

ÀcPOM=coarse particulate organic matter, fPOM= fine particulate organic matter.

Table 4.4. Means (and significant differences) for interaction between soil fraction C or N with land use history or climate. Soil fractions were separated through physical fractionation on samples obtained from Wisconsin golf course soils. Non-significant interaction of fractionxclimate N is not displayed.

Soil Fraction	LUH	Soil Organic C	Soil Organic N
cPOM	Cropped	2.24 CD	0.123 CD
cPOM	Forested	3.62 ABC	0.267 ABCD
fPOM	Cropped	3.27 BCD	0.141 BCD
fPOM	Forested	5.56 AB	0.452 ABC
Silt	Cropped	7.07 A	0.424 AB
Silt	Forested	7.23 A	0.603 A
Clay	Cropped	2.65 CD	0.173 ABCD
Clay	Forested	1.68 D	0.0780D

	Climate	Soil Organic C	Soil Organic N
cPOM	North	2.55 C	N/A
cPOM	South	2.91 BC	N/A
fPOM	North	5.00 AB	N/A
fPOM	South	3.17 BC	N/A
Silt	North	6.50 A	N/A
Silt	South	7.93 A	N/A
Clay	North	1.87 C	N/A
Clay	South	2.68 BC	N/A

Table 4.5. Relationship between maximum net potential N mineralization, net NH_4^+ produced during aerobic incubation, net NO_3^- produced during aerobic incubation, and saturation ratio of C or N. Saturation ratio was measured C or N divided by calculated C or N content associated with clays. Calculation of C and N content was based on loading of clay minerals, surface area of clay minerals, and quantity of clay minerals present.

Saturation Ratio	Parameter	Slope	Intercept	R^2	*p*-value
C	Maximum potential net mineralization	1.72	10.9	0.132	0.0043
N	Maximum potential net mineralization	1.56	11.5	0.0939	0.0172
C	Net NH_4^+ mineralized	0.0443	-0.0398	0.0997	0.0140
N	Net NH_4^+ mineralized	0.0462	-0.0299	0.0935	0.0175
C	Net NO_3^- mineralized	-0.0187	0.343	0.00692	0.527
N	Net NO_3^- mineralized	-0.000924	0.325	0.000	0.997

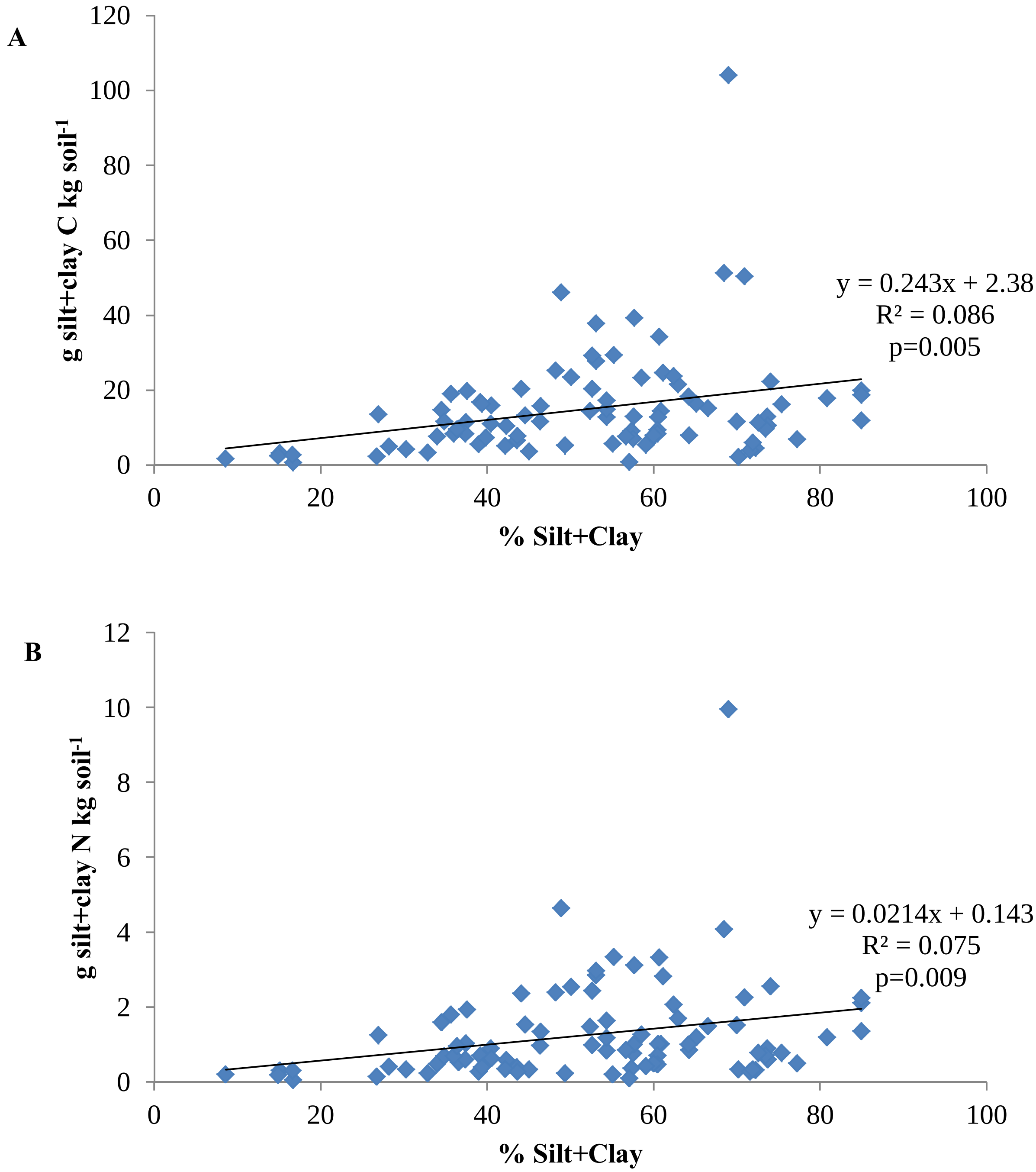


Figure 4.1. Influence of silt plus clay content on soil organic C (A) and soil organic N (B).

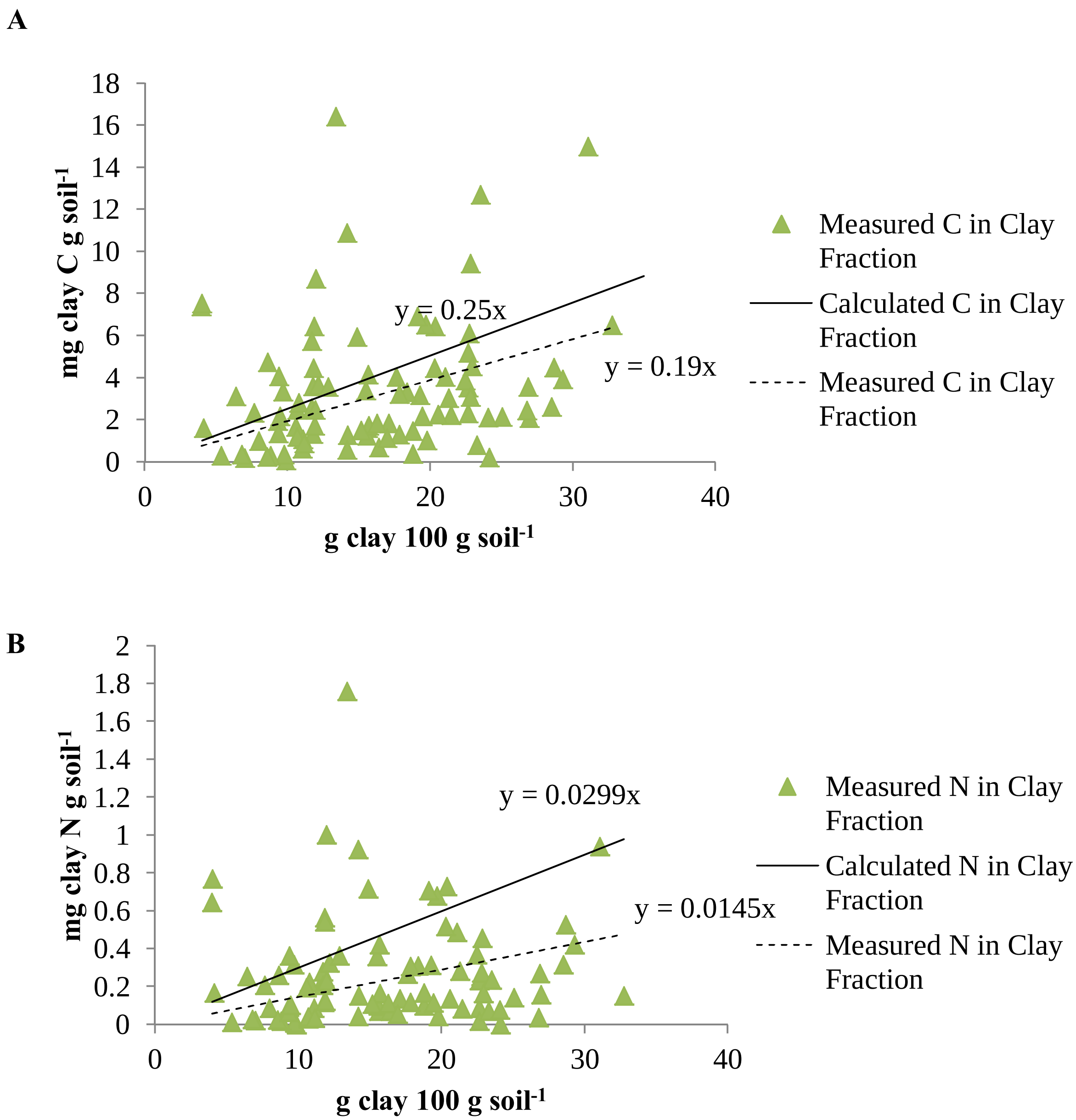
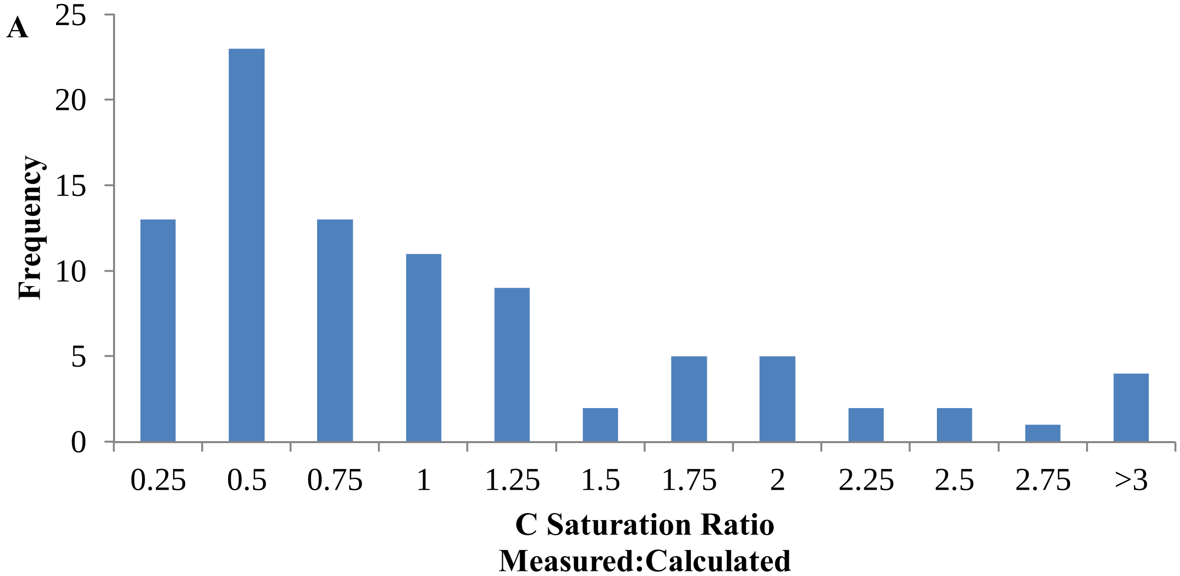


Figure 4.2. Linear regression of calculated maximum soil organic C in clay associated pool and measured soil organic C in the clay associated pool (A) and N (B) as influenced by clay content. Calculated maximum organic C and N slopes indicate loading of C and N on minerals in Wisconsin sites, determined by X-ray diffraction. Slope of measured data regression also indicates loading.



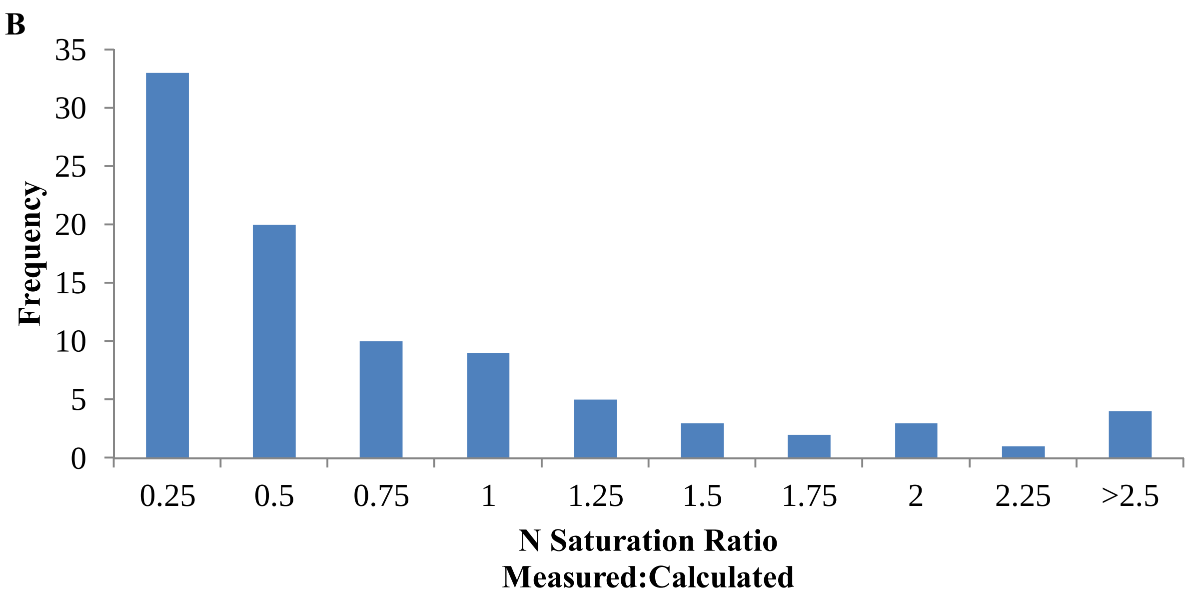


Figure 4.3. Frequency of C (A) and N (B) saturation ratios in Wisconsin turfgrass soils. Measured C and N associated with clay and calculated C and N associated with clay was based on percentage of clay minerals determined by X-ray diffraction, clay surface area, and loading of C or N on the clay mineral.

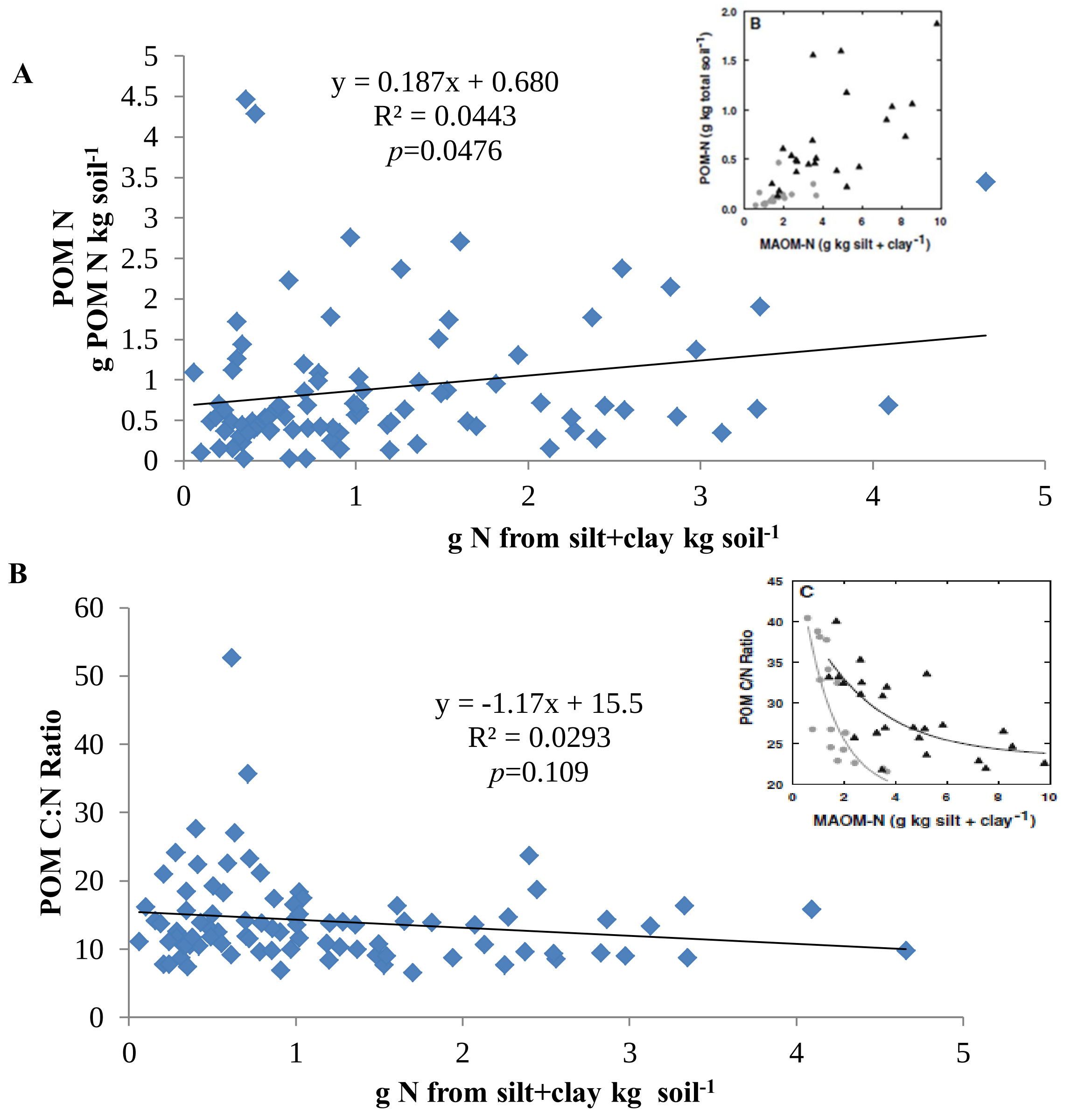


Figure 4.4. Influence of N-associated with soil minerals on POM N (A), POM C:N Ratio (B) in Wisconsin golf courses. POM is particulate organic matter. Inlays in Figure A and B are from Castellano et al., 2012.

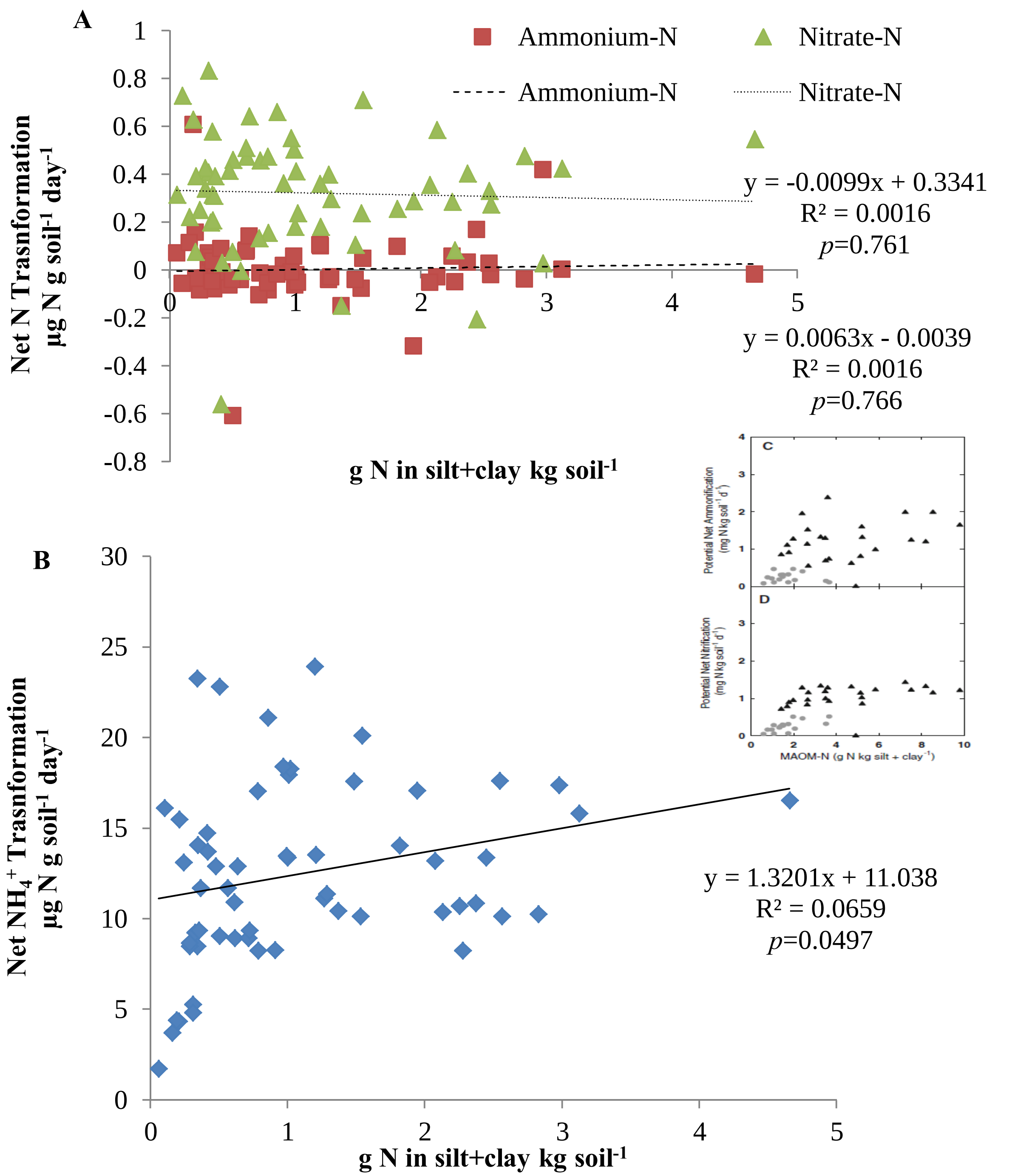


Figure 4.5. Mineral associated N influence on aerobic mineralization to NH_4^+ and NO_3^- (A) and anaerobic mineralization to NH_4^+ (B) in Wisconsin turfgrass. Inlay from Castellano et al., (2012).

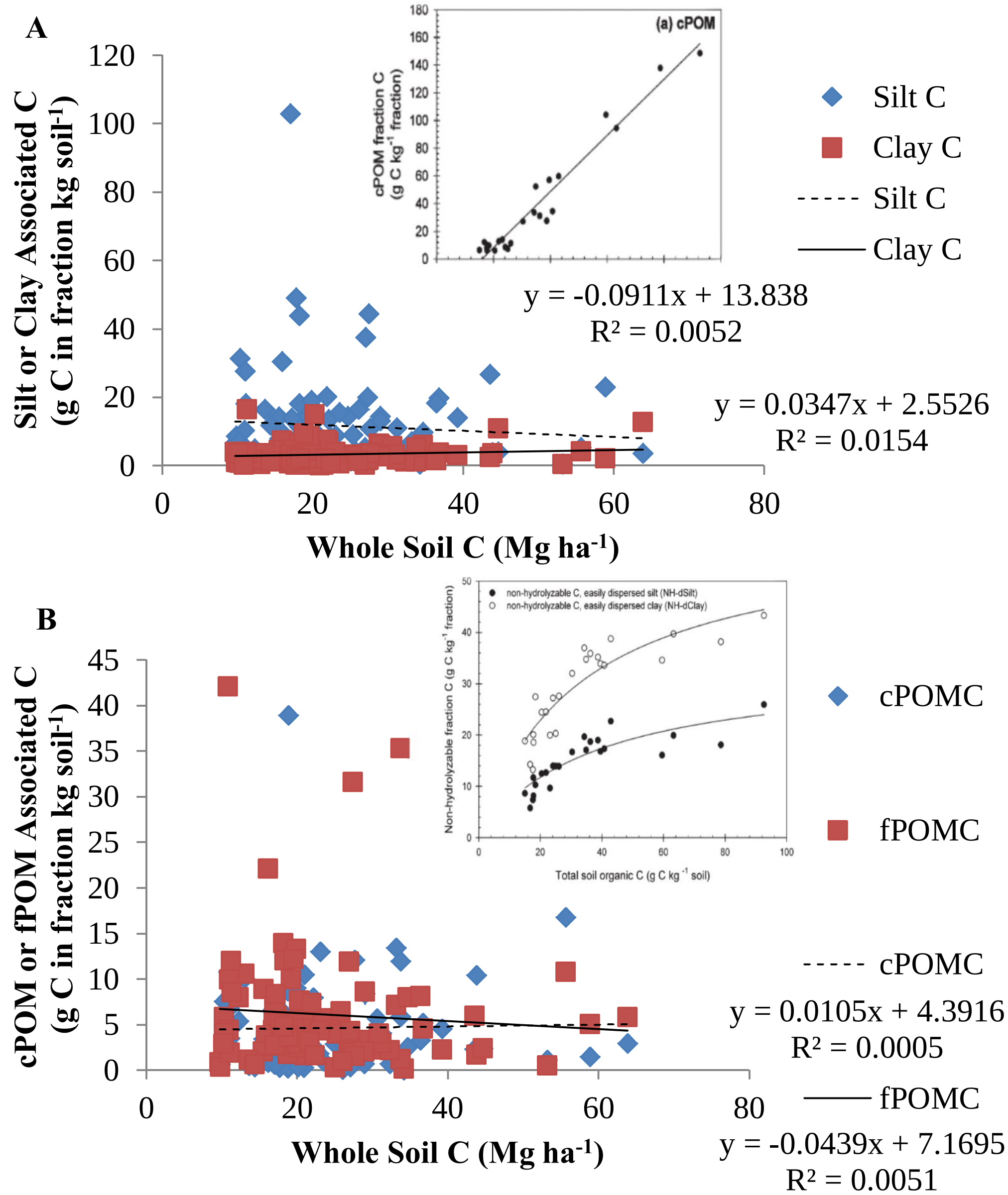


Figure 4.6. Relationship between mineral associated C and particulate associated C (POM) in Wisconsin golf courses. Inlays from Stewart et al.,(2008). Lowercase c and f before POM indicate coarse or fine.

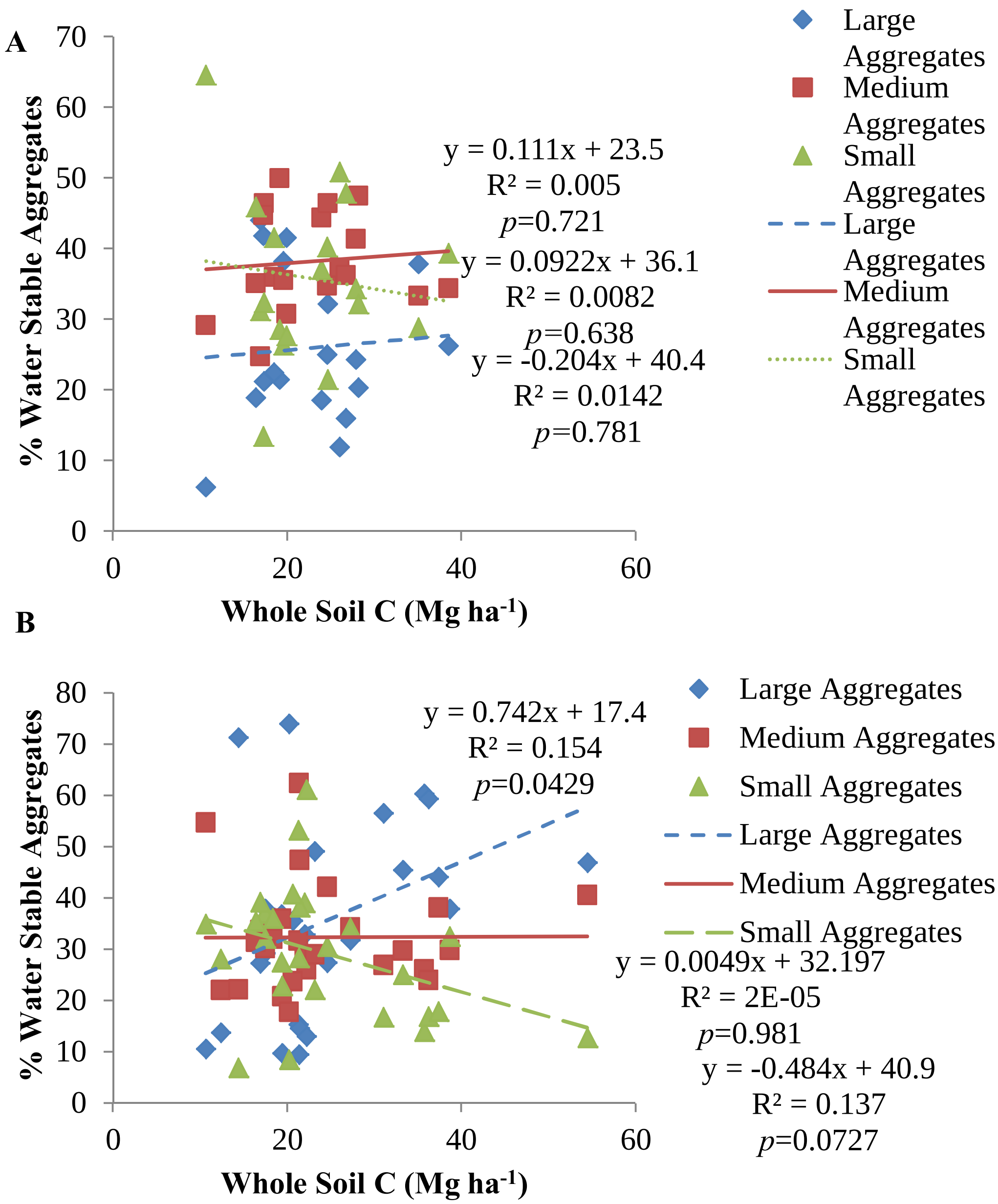


Figure 4.7. Percentage of water stable aggregates from Wisconsin golf courses and relationship with whole soil C in (A) formerly forested and (B) formerly cropped sites.

CHAPTER 5

SUMMARY AND CONCLUSIONS

Turfgrass areas in the US have increased to encompass an area equivalent to the size of Wisconsin. With this rapid expansion and large area covered by turfgrass, it is important to study C and N cycling in the turfgrass system. As a turfgrass system ages and accumulates soil organic C and N, they may lose N through N-leaching if N fertilizer is not reduced. Loss of N through N-leaching leads to environmental effects of eutrophication of surface waters, hypoxia in the Gulf of Mexico, and human health effects from consuming ground water contaminated with NO_3^- such as methemoglobinemia. Current University of Wisconsin Extension recommendations are to fertilize with N at 150 kg ha^{-1} for the first 10 to 12 yrs, and then reduce the N-input rate by half. Those recommendations do not take into account sites that are low in soil organic N, which may need longer duration of the recommended rate before reduction, nor do those recommendations take into account high soil organic N sites where the reduction in N-input should occur earlier. Through better understanding of C and N cycling in turfgrass, a long term goal is to make fertilizer recommendations that maintain turfgrass health while being protective of the environment.

The first step to being able to make those better fertilizer recommendations was to quantify soil organic C and N in turfgrass systems of Wisconsin. Soil organic C in our Wisconsin turfgrass sites accumulated slowly (0.27 Mg ha^{-1}) with time when all sites were analyzed together. This slow accumulation rate is in contrast to a number of other turfgrass studies which suggest turfgrass soil organic C accumulation is around 1 Mg ha^{-1} (Selhorst and Lal, 2011; Shi et al., 2006; Bandaranayake et al., 2003; Qian and Follett 2002); however, these studies were more limited in scope. The lower accumulation rate in our study can partly be explained by variability in soil series and land use history. Soil organic C over time analyzed by soil series showed that certain soil series may accumulate C at rates similar to literature values; however, others do not and when all the individual soil series are analyzed together, the result is a lower R^2 value and slower accumulation rate. A second source of variability that likely contributed to the slow soil organic C accumulation rate was variability in land use history. Some land use histories do not change in soil organic C with time, while others increase, and those that do increase, may increase at different rates. The important factor to consider when evaluating soil organic C or N in turfgrass is that variability. If the variability is not accounted for, or only a few sites are studied because they are of a certain land use or soil type, then the result may be that turfgrass C sequestration potential may be overestimated at a large scale when extrapolations are made from small data sets.

DAYCENT, a computer model for soil organic C and N, can accurately simulate soil organic C in turfgrass (Banadaranayake et al., 2003). DAYCENT could potentially be used as a tool to estimate soil organic N and then make fertilizer recommendations that would minimize N-leaching potential, while maintaining turfgrass quality. DAYCENT needs inputs of soil texture, pH, and bulk density, in addition to weather and site history. To facilitate ease of data collection for instances when DAYCENT would be used outside of research (for instance to make fertilizer recommendations), Web Soil Survey data may be an acceptable substitute. We found that Web Soil Survey properties could be used as a substitute for measured properties; however, in our further simulation we did use measured properties that would be easy to measure by soil labs from homeowner samples (soil texture and pH) which could serve as good starting point for a new client. In either case, neither were statistically different from each other. Land use history could not be simulated as a simple, easy to program one (forested) rather than the more complicated actual land use history. Actual land use history simulations were well correlated with measured soil organic C with slopes of ~1 in Wisconsin turfgrass, but soil organic N values were typically underpredicted (the exception being chemical fertilized cropped sites).

Nitrogen leaching output from DAYCENT was also evaluated for chemical fertilizer cropped sites based on a threshold leaching of 5 kg N ha^{-1} yr^{-1}. Chemical fertilizer cropped sites reached threshold leaching at 5.0 Mg soil organic N ha^{-1} (after ~7 yrs of turfgrass management). If these simulations are correct, N-fertilizer could be reduced much earlier than the 10 to 12 yrs after turfgrass establishment. At this time further research would be needed to determine how much to reduce fertilizer inputs and still maintain turfgrass quality and reduce N-leaching. DAYCENT would need additional work fine tuning soil organic N for turfgrass before it could be confidently used to simulate soil organic N, and N-leaching. Similarly, N-leaching needs verification with field data, though it appears potential exists for using DAYCENT to correlate N-leaching over a threshold of 5 kg ha^{-1} with soil organic N.

Soil fractionation, conceptual models, and calculation of saturation in soil organic C and N give a more mechanistic view of how soil organic C and N are stored and cycled. Our turfgrass sites in Wisconsin did not follow the conceptual models proposed by Castellano et al. (2012) or Stewart et al. (2007), which may indicate saturation if our turfgrass sites are within the asymptote of C and N accumulation. However, calculation of loading and saturation ratios of soil organic C and N were predominantly undersaturated, leading to the conclusion that our sites were likely at an effective stabilization capacity. Data from aggregate stability showed that the percentage of water stable aggregates did not change (or in the case of cropped sites only the large 1 to 4 mm aggregates) with whole soil C or N. It was expected that if sites were not at effective stabilization capacity that the percentage of water stable aggregates would increase as whole soil C and N increased. Since aggregate stability did not change, it lends further support to the theory that our turfgrass sites are at an effective stabilization capacity. Turfgrass sites at an effective stabilization capacity would likely lose soil organic C and N similar to saturated system.

Future study should work to close the gap between measured and simulated soil organic N. Nitrogen leaching data from DAYCENT should be verified with measured N-leaching from our Wisconsin sites. If fertilizer inputs of N are reduced using DAYCENT and threshold levels of leaching, whether or not turfgrass maintains acceptable quality must be determined. Use of younger turfgrass sites may show differences in saturation of soil organic C and N and could give additional support to turfgrass sites in Wisconsin reaching effective stabilization capacity quickly. Finally, not all Wisconsin turfgrass is planted on Alfisols, so the study of other soil orders, such as Mollisols, may add further insight into turfgrass soil organic C and N cycling in turfgrass.

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APPENDIX A

GOLF COURSE SUPERINDENTENT QUESTIONNAIRE

Golf course questionnaire

1. Year course was built: front vs back nines?
2. Fertility regime for roughs, please include number of years this regime has been used?
3. Fertility regime for fairways, please include number of years this regime has been used?
4. Please describe previous fertility regimes for your fairways and roughs as far back as possible (preferably to the year of course beginning).
5. Irrigation regime for roughs? Fairways?
6. Aerification?
7. Pesticide applications for the aforementioned playing surfaces?
8. How much contouring/building-up occurred during the construction process, if known?
9. Previous land use?

Soil type and order:
Address:
Superintendent name: