CARBON AND NITROGEN CYCLING IN URBAN ECOSYSTEMS

A Dissertation
Presented to the Faculty of the Graduate School
of Cornell University
in Partial Fulfillment of the Requirements for the Degree of
Doctor of Philosophy

by
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Chapter 1: Urban areas are growing in size and importance; however we are only beginning to understand how the process of urbanization influences ecosystem dynamics. In particular, there have been few assessments of how the land use history and age of residential soils influence carbon and nitrogen pools and fluxes, especially at depth. In this study, we used one-meter soil cores to evaluate soil profile characteristics and carbon and nitrogen pools in 32 residential home lawns that differed by previous land use and age, but had similar soil types. These were compared to soils from 8 forested reference sites.

Chapter 2: The rapid increase in residential land area in the United States has raised concern about water pollution associated with nitrogen fertilizers. Nitrate (NO$_3^-$) is the form of reactive N that is most susceptible to leaching and runoff; thus, a more thorough understanding of nitrification and NO$_3^-$ availability is needed if we are to accurately predict the consequences of residential expansion for water quality. In this study, we evaluated potential net nitrification and mineralization, microbial respiration and biomass, and soil NO$_3^-$ and NH$_4^+$ pools in residential home lawns and forests.

Chapter 3: We previously created a mass balance for residential lawns, however, a major N flux was missing - gaseous losses from denitrification. Using recent advances in instrumentation, we were able to measure field-relevant rates of denitrification from lawns. We calculated annual denitrification rates of 14.0 ± 3.6 kg N/ha/yr for the lawns in this study, which suggests that denitrification
is an important means of removing reactive N from the residential landscape. Further work is required before these findings can be generalized to a wider range of residential lawns and soils.

Chapter 4: Economic and political realities present challenges for implementing an aggressive climate change abatement program in the United States. A high efficiency approach will be essential. In this synthesis, we compared carbon budgets and evaluated carbon mitigation costs for nine counties across the northeastern United States that represent a range of biophysical, demographic and socioeconomic conditions.
BIOGRAPHICAL SKETCH

Steve Raciti earned his Bachelor’s of Arts from Vassar College in 2003, where he double majored in Environmental Studies and Geology and minored in Chemistry. While at Vassar College, he worked as a Forest Ecology Intern at the Cary Institute of Ecosystem Studies, where he assisted scientists studying the prevalence of Lyme in ticks and the distribution of gypsy moth populations in forest communities. For his senior thesis, he spent two field seasons working with scientists at the US EPA Atlantic Ecology Division exploring the impact of nutrient enrichment, shoreline development, and other anthropogenic stressors on juvenile winter flounder abundance in Narragansett Bay, RI.

After finishing college Steve was accepted to Cornell University, where he earned his Master’s of Science from the Department of Natural Resources and participated in the IGERT Program in Biogeochemistry and Environmental Biocomplexity. For his master’s thesis, he used stable isotopic techniques to compare nitrogen cycling and retention in lawns and forests at the Baltimore Ecosystem Study Long Term Ecological Research area. In 2006, he worked as an urban forestry specialist for the U.S. Forest Service, Chesapeake Bay Program Office. At the Chesapeake Bay Program, he helped develop a workshop and guide to assist communities in evaluating and enhancing tree canopy cover; worked on finding “points of opportunity” in state stormwater regulations for the integration of natural vegetation as best management practices; and worked with developers and municipalities to document the opportunities for, and obstacles to, using low impact development practices in the region.

For his doctoral degree, Steve continued his research in urban ecosystems. He examined carbon and nitrogen stocks and transformations across a chronosequence of residential soils; measured denitrification in lawns; calcu-
lated carbon dioxide emissions and sinks for counties in the northeastern United States; and explored how local-scale differences in biophysical, demographic, and socioeconomic conditions alter the range and cost of opportunities available for mitigating those emissions. Steve will continue his research in urban ecology and biogeochemistry as a postdoctoral research associate at Boston University
This dissertation is dedicated to my mother, Roberta (Sandi) Raciti. She has always been my biggest supporter and her unwavering confidence in me has been a continuous source of strength and inspiration.
ACKNOWLEDGEMENTS

I would like to thank my advisors, Dr. Timothy Fahey and Dr. Peter Groffman. Without their assistance and encouragement, this work would not have been possible. I would also like to thank my committee members, Dr. A. Martin Petrovic and Dr. Joseph Yavitt, for their sound advice and support. Additional thanks to Dr. Richard Pouyat, Dr. Ian Yesilonis, Dr. Amy Burgin, David Lewis and Dan Dillon for their guidance and logistical support; and thanks to my collaborators in this work, including Mary L. Cadenasso, Frederick J. Carranti, Charles Driscoll, Timothy J. Fahey, David Foster, Peter M. Groffman, Morgan Grove, Philip S. Gwyther, Brian Hall, Jennifer C. Jenkins, Julian Jenkins, Steven Hamburg, Brandon W. Peery, Steward T. A. Pickett, Richard V. Pouyat, Christopher Neill, Scott Ollinger, Jarlath O’Neill-Dunne, Steward Pickett, Erin Quigley, Ruth Sherman, R. Quinn Thomas, David Weinstein, Peter Woodbury, Tim Vadas, Matt Vadeboncoeur, Geoff Wilson, Peter Woodbury and William Yandik.

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1.1 Introduction

With recent shifts in public attitudes across the United States concerning the problem of global climate change, momentum is building for aggressive action to mitigate greenhouse gas (GHG) emissions. At the same time, economic realities present challenges for financing an aggressive climate change abatement campaign; hence it is imperative that cost-effective strategies for reducing GHG emissions be identified and pursued. This task is made more difficult by the complex suite of local and regional factors that influence the abatement potential and cost-effectiveness of various mitigation approaches. These factors include biophysical features such as climate, soils, topography and vegetation; demographic factors such as population density and distribution; features of the existing infrastructure including transportation networks, heat and power supplies, housing, commerce and industry; and the governance structures in which policies must be positioned.

The objective of this study is to describe how variation in this suite of factors influences the current carbon balance (i.e., net carbon dioxide fluxes) and the feasibility of approaches for decreasing CO₂ emissions in the northeastern United States. We compare carbon budgets and mitigation opportunities across nine representative northeastern counties to illustrate some of the key features influencing the choice of strategies in this region. We focus on CO₂, which accounts for 77% of anthropogenic GHG emissions (IPCC 4AR 2007) and 85%
of US GHG emissions (US GHG Emissions Inventory 2006), as the key GHG challenging society’s resolve to respond to the climate change threat. A major emphasis in our analysis is the contribution of land use to carbon emissions, sinks, and mitigation opportunities. Recent analyses illustrate that land use options may provide cost effective carbon sequestration in the United States (Lubrowski et al. 2005). While several CO$_2$ emissions analyses have been conducted at larger scales (DOE 1997, Metz et al. 2001), few have been done at the local scale, where much of the planning and implementation of emissions reduction goals will likely take place (Nelson and deJong 2003). We hope that this synthesis stimulates productive dialogue among policy-makers, educators and society at large; and offers motivation and guidance for municipalities who will set goals to decrease carbon emissions in response to regional and international initiatives.

1.2 Study Area

The region chosen for this study encompasses the states involved in the Regional Greenhouse Gas Initiative (RGGI), an early cap-and-trade system designed to decrease CO$_2$ emissions from the northeastern United States. Under RGGI, a cap on CO$_2$ emissions from the electric power sector has already been applied with the goal of a 10% decrease by 2018 (http://www.rggi.org/about). The Northeast is heavily populated and urbanized and currently emits more greenhouse gases than all but five nations: China, Russia, India, Japan and the U.S. (EIA data compiled by UCS). Within this region we chose eight counties plus the independent city of Baltimore for detailed study; hereafter all nine locations will be referred to as counties (Figure 1.1). These counties exhibit a
wide range of demographic and land-use characteristics from highly urbanized to heavily forested (Table 1.1). Six of the counties encompass intensive research sites in the National Science Foundation’s Long Term Ecological Research (LTER) Network (http://www.lternet.edu/).

Figure 1.1: Map of the study area, which includes the states participating in the Regional Greenhouse Gas Initiative (RGGI). Detailed carbon budgets and mitigation analyses were conducted for the highlighted counties.
Table 1.1: Demographic and land use information for the nine selected counties.

<table>
<thead>
<tr>
<th>County</th>
<th>Area (km²)</th>
<th>Pop.</th>
<th>Pop. Density (#/km²)</th>
<th>—— Land Use (%) ——</th>
</tr>
</thead>
<tbody>
<tr>
<td>Coos, NH</td>
<td>4,740</td>
<td>33,111</td>
<td>7</td>
<td>87</td>
</tr>
<tr>
<td>Grafton, NH</td>
<td>4,532</td>
<td>81,743</td>
<td>18</td>
<td>87</td>
</tr>
<tr>
<td>Tompkins, NY</td>
<td>1,273</td>
<td>96,500</td>
<td>76</td>
<td>43</td>
</tr>
<tr>
<td>Chittenden, VT</td>
<td>1,605</td>
<td>146,571</td>
<td>91</td>
<td>73</td>
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<tr>
<td>Worcester, MA</td>
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<td>Baltimore, MD</td>
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<tr>
<td>Essex, MA</td>
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<tr>
<td>Middlesex, MA</td>
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<td>1,467,016</td>
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<td>46</td>
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<tr>
<td>Baltimore City, MD</td>
<td>207</td>
<td>639,493</td>
<td>3055</td>
<td>8</td>
</tr>
</tbody>
</table>

1.3 Methods in Brief

In general, we followed protocols developed by Vadas et al. (2007) to circumscribe boundary conditions and to make emissions and sequestration estimates for the counties. Because utility data on heat and power supplies are not generally available at the county scale, we adjusted state-level data on the basis of population, employment, housing statistics, and typical energy usage profiles for each housing type. Further, our analysis excludes emissions from air travel and indirect emissions associated with the manufacture of imported goods, which would be challenging to incorporate without violating our county-level boundary conditions, designed to prevent double-counting of emissions across geographic areas. For land areas classified as forested by the USDA Forest Service, Forest Inventory and Analysis (FIA) data (http://fia.fs.fed.us/) were used to estimate changes in forest carbon stocks, except for Baltimore City and Baltimore County, where more detailed forest and non-forest biomass data were available (Jenkins and Riemann 2003, Nowak et al. 2001). Estimates of max-
imum expected biomass of existing forestland were made using PnET-CN, a
simulation model that estimates carbon, water, and nitrogen dynamics in north-
eastern forests (Aber et al. 1997). To estimate carbon sequestration by afforesta-
tion we assumed that all inactive agricultural land (USDA Ag Census 2009) is
available for afforestation. To estimate sequestration potential, we used FIA es-
timates of forest carbon for 26-30 year old forest plots in each state (using data
from 2002 - 2008) and divided the total biomass by the median age to provide an
estimate of mean annual sequestration. For the carbon budgets we used com-
mon, widely available data sources when possible to 1) standardize our com-
parisons across counties and 2) to ensure that our calculations would be easily
repeatable, so they might serve as a model for calculating CO$_2$ budgets and mit-
igation opportunities for other regional counties. Detailed methods and data
sources can be found in Vadas et al. (2007) with a few major adjustments as
follows:

1. Biofuels. We used statistical and geospatial methods to estimate land
availability for switchgrass, short rotation willow, soybean, and corn with-
out competing with current agricultural production or forest land. Based
on the total area of pasture, hay, and grassland in each county (Homer
et al. 2004) we discounted land in Federal ownership, land with slopes
greater than 15%, and land currently in pasture or hay production based
on the 2007 Census of Agriculture (www.agcensus.usda.gov). We as-
sumed that only 20% of the total available land area would actually be
used for bioenergy feedstock production. Because yield data for dedi-
cated bioenergy feedstocks are only available from a few locations in the
Northeast, we estimated potential yields based on an integrated index of
soil and climate characteristics called the National Commodity Crop In-
We developed regression equations to predict maize yield based on this index ($r^2 = 0.65$). Assuming a 1:1 relationship between grain and stover, this regression relationship was adapted to predict aboveground biomass yield of maize which was used as a proxy for switchgrass and willow yield.

2. Commercial-scale wind energy. We used 200m resolution simulated wind resource data to evaluate the potential for commercial wind power generation across counties in New England. Our analysis focused on terrestrial wind resources. Areas with class 3 (6.4 m/s mean speed at 50 meters height) or greater wind power potential were considered commercially viable sites for wind power generation. Developed areas were excluded as potential sites. Information on land availability for wind power generation was determined using the 2001 National Land Cover Database (Homer et al. 2004). Wind resource data (obtained from MassGIS) were originally developed by Truewind Solutions, LLC under contract to AWS Scientific, Inc., as part of a project funded by the Connecticut Clean Energy Fund, the Massachusetts Technology Collaborative (MTC), and Northeast Utilities System.

3. Light bulb replacement. We assumed a mean total usage of 30 bulb-hours per day for compact fluorescent light (CFL) replacement bulbs in an average housing unit (EIA RECS 2005).

4. Residential photovoltaics. We assumed that half of all single family dwellings would be realistic candidates for photovoltaic systems (Anders and Bialek 2006).

5. Electric grid fuel mix. For all electricity based energy savings and new renewable power generation, we assumed that emission reductions would
displace emissions from the current mix of fuels in the regional electricity grid (EPA eGRID 2008).

1.4 Carbon Budgets

The counties included in this study span a wide range of population densities, from 7/km² in Coos County, NH, to over 3,000 /km² in Baltimore City, MD. Net CO₂ fluxes from the counties were strongly and positively correlated with population density (Figure 1.2), despite differences in per capita CO₂ emissions among the counties (Figure 1.3). Current rates of carbon sequestration in vegetation and soils tended to be inversely related to population density ($r^2 = 0.63$); however, this pattern was not robust at the lower population densities because counties with population densities less than about 200/km² differed little in sequestration rates. Based on NLCD data (Homer et al. 2004), all counties with population densities in this range have developed land areas of less than 12% and most of that developed area falls into the open space and low intensity development land-use categories (NLCD 2001). These data suggest that opportunities for unaided sequestration in vegetation and soils are not greatly diminished until the developed proportion of the landscape exceeds about 10 - 15%.

Not surprisingly, most of the counties were net sources of CO₂ to the atmosphere as emissions from fossil fuel combustion exceeded sequestration in vegetation and soils (Figure 1.2). The exceptions were the two most rural, forested counties in northern New Hampshire where sequestration in growing forests exceeded CO₂ emissions. Thus, most of the northeastern United States is a source of atmospheric CO₂ with the strength of the source varying primarily
Figure 1.2: Net carbon flux plotted against population density. Net-zero emissions, where anthropogenic emissions are roughly in balance with sequestration in vegetation and soils, coincides with a county population density of about 30/km² (inset). It’s important to note that our analysis excludes emissions from air travel and indirect emissions associated with the manufacture of imported goods.

with human population density and only sparsely populated forested counties acting as net C sinks.

Net-zero emissions of CO₂ in the northeastern region coincide with a population density of about 30/km², based on the regression between net emissions and population density (Figure 1.2). This value represents the population density at which emissions in the Northeast are roughly in balance with sequestration in vegetation and soils. Contrast this value with the mean population
Figure 1.3: Annual per capita CO2 emissions for the counties. County population densities increase from left to right.

density of the region of 134 /km² (US Census 2000). Note also that this figure ignores air travel (which constitutes nearly 3% of US CO₂ emissions) as well as imported food and goods (Bin and Dowlatabadi 2005). These indirect emissions are caused by demand for goods within the Northeast that is met by production in other parts of the world (see for example Helm et al. 2007). The implication is that unaided sequestration in forests and soils cannot offset existing emissions from the region.

The future potential for unaided natural sequestration to offset regional CO₂
emissions is less promising when patterns of forest regrowth in the Northeast are examined. Forests in the region are maturing and their ability to sequester additional C will likely decline unless policies and practices shift considerably (Hurtt et al. 2002). Our ability to predict future forest carbon is complicated by factors that may increase or decrease C storage capacity, including changes in temperature, growing season length, water regime, management practices, frequency of fire, drought, and severe weather events, outbreaks of invasive and native insect pests, changes in tree species composition, and CO$_2$ fertilization effects. Our model only accounts for existing conditions and presumes that a maximum potential biomass will be reached, though growing evidence suggests that even old growth forests can continue to accumulate C (Keith et al. 2009, Luyssaert et al. 2008). The model results suggest that forest biomass in northeastern counties is generally approaching the maximum biomass expected under present conditions (Figure 1.4). Indeed, forests in 2/3 of the counties may have attained 75% or more of their predicted maximum biomass. Further, sequestration in northeastern forests is threatened by a number of invasive pests and pathogens (Lovett et al. 2006) that could significantly reduce forest biomass over coming decades. Finally, forest cover in the region is declining as real estate development expands in both suburban and rural areas (Stein et al. 2005). Thus, if current trends in land use continue, future C sequestration potential will be reduced and some previously stored C in vegetation and soil will be released to the atmosphere.
Figure 1.4: Aboveground forest biomass in existing forests as a percentage of the predicted maximum under current climate conditions. Note that maximum potential biomass may be under and over-predicted by the model and future changes in temperature, growing season length, water regime, management practices, frequency of fire, drought, and severe weather events, outbreaks of invasive and native insect pests, changes in tree species composition, and CO2 fertilization effects may increase or decrease carbon storage potential. There is increasing evidence that even old growth forests can continue to sequester carbon.
1.5 Per Capita Emissions

Per capita emissions among the counties ranged from 2,900 kg C/person for Chittenden County, VT to 4,666 kg C/person for Baltimore County, MD (Figure 1.3). Although this range is much smaller than the international variation that is explained largely by GDP (Aldy 2006), it raises questions about local controls on per capita emissions. Population density is a weak predictor of total per capita emissions and of per capita emissions from the transportation, residential, commercial, and industrial sectors individually ($r^2 < 0.13$ for all). Land cover, expressed as percent forest, agriculture, and developed land, varies widely among the counties (Table 1.1), and is a similarly weak predictor of total and individual sector emissions ($r^2 < 0.20$ for all relationships). Differences in per capita emissions are explained by other factors, such as the carbon intensity of fuels used for heating and electricity generation, socioeconomic factors, availability of mass transit, local climate, and economic and industrial history.

The transportation sector accounted for the largest share of emissions from every county (35% - 47%), except Baltimore City (26%, Figure 1.3). Per-capita transportation emissions ranged from 923 kg C/person in Baltimore City, MD to 1,643 kg C/person in neighboring Baltimore County, MD. The greater availability of public transportation and closer proximity to places of employment play a role in Baltimore City’s lower transportation emissions. More than 28% of working Baltimore City residents used public transportation, walked, or used alternate means of transportation (19.4%, 7.1%, and 1.6%, respectively) to get to work (US Census ACS 2006-2008). Compare these values to neighboring Baltimore County where fewer than 8% of working residents used these forms of transportation for their daily commute. Socioeconomic conditions also contribute to
lower per-capita transportation emissions; nearly 20% of Baltimore City residents live below the poverty line and the unemployment rate is twice as high as in neighboring Baltimore County (U.S. Census ACS 2006-2008). Lower income combined with higher access to public transportation contribute to lower vehicle ownership rates in the City, where 30% of households have no personal vehicle access, compared to just 6% of households in Baltimore County. The other county with comparatively low per-capita transportation emissions is Tompkins County, NY, which is dominated by the small city of Ithaca. Ithaca is itself dominated by Cornell University, which provides strong incentives to discourage single-occupancy vehicle (SOV) commuting. More than 8% of Tompkins County commuters use public transportation and an even greater percentage (19.4%) walk or use other alternative means of transit. Still, a most striking pattern is the similarity in per-capita transportation emissions across counties with dramatically different population densities and landcover patterns. This emphasizes that rural forested counties, mixed forested and agricultural counties, and highly suburbanized and urbanized counties all have a high dependence on personal vehicle usage and a generally low reliance on public and alternative transportation.

The residential sector accounted for the second largest share of emissions in each of the counties (except Baltimore City where it ranked first), accounting for 25.4% to 34.6% of total emissions. Emissions ranged from 760 kg C/person in Chittenden County, VT to 1,259 kg C/person in Coos County, NH. This wide range in residential CO$_2$ emissions results from myriad variables, including local climate, housing mix, and the carbon intensity of fuels used for heating and electricity generation. For instance, Baltimore City has lower per-capita emissions than Baltimore County despite similar fuel mixes for heating and electric-
ity generation. These differences result in part from the greater proportion of attached houses, multi-family dwellings, and apartment buildings in Baltimore City (86%) versus Baltimore County (53%). These housing units are smaller and generally require less energy to heat and cool than detached single family houses; in 2005 the average single-family detached house in the United States used 13,200 kW of electricity versus 9,300 kW for single-family attached houses and only 7,100 kW for apartments in buildings with 5 or more units (EIA RECS 2005).

Electricity usage comprised a large and highly variable percentage of residential carbon emissions across the counties and influenced many of the patterns in residential emissions. Chittenden County, VT has unusually low per-capita residential emissions due to its extensive use of renewable energy (50%, mostly hydroelectric) and nuclear power (34%) for electricity generation (VT-DPS 2007). These relatively low carbon intensity electricity sources result in residential electricity emissions of only 82 kg C/person or just 11% of total residential emissions. On the other hand, Baltimore City and Baltimore County had the highest per-capita residential electricity use (0.049 and 0.056 BBTU/person including system losses) and the highest accompanying emissions from electricity use at 746 kg/person and 846 kg/person, respectively - ten times higher than in Chittenden County, VT. The warmer climate in Maryland and the state’s heavy reliance on coal for electricity generation explain this sharp contrast. Baltimore City averages 1,220 cooling degree days per year (CDD/yr) versus just 371-551 CCD/yr for the other counties (NCDC 2000), which leads to higher electricity use for home cooling. The milder climate also stimulates a greater proportion of homes to rely on electric heat, which is a relatively inefficient source; more than 36% of Maryland residents heat their homes with electric-
ity compared to fewer than 10% in the New England states (EIA RECS 2005). The Massachusetts and New Hampshire counties fall between the extremes of the Vermont and Maryland counties, with annual electricity related emissions of 410 - 520 kg/person. Tompkins County, NY falls toward the lower range of electricity related emissions due to the lower carbon intensity of the upstate New York power grid (28% renewable and hydroelectric, 27% nuclear, and 16% natural gas, EPA eGRID 2005) and relatively high proportion of attached and multifamily housing units (53%), which are largely concentrated in the city of Ithaca (Census ACS 2006-08).

Fossil fuel burning for space and hot water heating accounted for the largest proportion of residential CO$_2$ emissions in the upstate New York and New England counties, making up 59% to 89% of emissions compared to less than 35% of residential emissions in the Maryland counties. Natural gas and heating oil are favored in the colder climate of New England (6,800-7,500 Heating Degree Days compared to 4,700 HDD in Baltimore City and County).

Per-capita industrial emissions were much greater in Baltimore City and Baltimore County (927 and 1,058 Mg C/person) than in other counties (355 to 636 Mg C/person). Baltimore is home to a major port and has historically been a center for industry in the region, despite a major industrial decline in the second half of the 20th century. Finally, differences in per-capita emissions from the commercial sector are not explained by any obvious factor; for example, the highest and lowest per capita commercial sector emissions are observed in the two northernmost rural counties in our study.
1.6 Mitigation Opportunities

The nine counties in this study represent a wide variety of biophysical, demographic, political, and economic conditions, which in turn influence the feasibility of different approaches for reducing CO$_2$ emissions. Generally speaking, in counties where forests and inactive agricultural land are abundant, a variety of land-based strategies offer opportunities to sequester emissions in vegetation and soils, provide feedstocks for biofuel production, or space to accommodate alternative energy technologies. In more urbanized counties where available land is limited and expensive, the most cost-effective CO$_2$ mitigation strategies will include energy efficiency practices and energy saving technologies. In all cases, a range of locally tailored management and technology options can offer substantial emissions reductions at little or no long term cost.

1.6.1 Zero Cost Mitigation Opportunities

We identified and evaluated a range of zero cost mitigation opportunities (Figure 1.5) based on the criterion that they pay for themselves over time due to income generated or energy cost savings over the lifetime of the strategy. For simplicity we have assumed a simple payback period, so the actual costs to implement these strategies may be higher than represented in some cases. On the other hand, if carbon emissions are priced and energy prices rise, our predicted costs may be overestimated. Our suite of zero cost mitigation opportunities includes land-intensive alternative power sources such as sustainable fuelwood from forests and utility scale wind power (assuming class 3 or greater wind potential at installed sites). In the residential sector, zero cost opportunities include
Figure 1.5: Zero cost mitigation opportunities pay for themselves over time due to income generated or energy cost savings over the lifetime of the strategy. The y-axis shows mitigation potential as a percentage of current emissions. County population densities increase from left to right.
energy efficient lighting (replacing incandescent bulbs with compact fluorescent bulbs), increased home insulation, programmable thermostats, lowered thermostat temperature settings for heating, sealing air leaks, boiler maintenance or replacement, and US EPA Energy Star certified refrigerators and air conditioning units (Vadas et al. 2007). Although many of the zero cost opportunities are applicable to the commercial and industrial sectors, we focus our analysis on the residential sector together with land-use change opportunities.

Rural counties in our study area (< 100 persons/km²) could offset 27% to 1,027% of current emissions at no increased cost through sustainable harvest of fuelwood from existing forests and installation of commercial scale wind energy farms at favorable sites (Figures 1.5 and 1.6). Our wind energy analysis focused on terrestrial wind resources, which are most abundant in hilly and mountainous terrain; however, the region also possesses abundant off-shore wind resources (Figure 1.6, inset). Due to low population densities, abundant forest cover and favorable topography for wind energy, greater emissions reductions can be achieved by developing wind and forest resources in the region’s most rural counties than from the home and commercial energy savings measures we evaluated. These strategies could provide significant carbon offsets in suburban counties (up to 116,000 Mg C/yr in Essex County, MA), though they represent a smaller proportion of total county emissions (0.4% to 3.3%). In Baltimore City, which has a small land area and a high population density, these opportunities were not significant.

For all counties, regardless of population density and available land area, the suite of home, commercial, and industrial energy saving opportunities can be substantial, representing a potential 29 - 37% reduction of county emissions at
Figure 1.6: The large map shows the percentage of each county’s undeveloped land area with class 3 (6.4 m/s at 50 meters height) or greater wind potential. The inset map shows the spatial distribution of these land areas. Information on land availability (i.e., developed vs. undeveloped) was based on the 2001 National Land Cover Data (NLCD). The original wind resource data were developed by Truewind Solutions, LLC under contract to AWS Scientific, Inc. as part of a project jointly funded by the Connecticut Clean Energy Fund, the Massachusetts Technology Collaborative (MTC), and Northeast Utilities System. Our analysis focused on terrestrial wind resources, which are greatest in hilly and mountainous regions, but it should be noted that off-shore wind resources are abundant (inset map).
zero cost. The largest potential for zero-cost home energy savings was for space and water heating, where a combined 9 - 13% reduction in county emissions could be achieved by sealing air leaks, increasing insulation in older homes, lowering thermostats to 65°F (versus 70°F), using programmable thermostats, boiler maintenance, and replacement of outdated boilers. Other space and water heating upgrades would bring even greater energy savings, but would require greater upfront costs. For instance, augmenting conventional home heating systems with geothermal systems could reduce heating energy expenditures by about 50% (Office of Geothermal Technologies 1998) leading to an 8 - 11% reduction in county CO₂ emissions. Such systems would cost $7,500 for a typical house (Office of Geothermal Technologies 1998) and have a simple payback period of 7 - 11 years based on 2008 energy prices in the region. Similarly, residential solar hot water systems (flat plate collectors) could decrease water heating costs by about 50%, with upfront costs of $3,250 and a payback period of 14 to 22 years for a typical home. This could reduce county emissions by another 1.7 - 2.3%. Replacing the 30 incandescent light bulbs contained in the average home (IEA 2003) with more energy efficient lighting is the next largest zero-cost opportunity to reduce emissions in the residential sector; county emissions could be reduced by another 2.4 - 3.4% with CFL bulbs. For larger buildings, including offices, hospitals, schools and hotels, combined heating and power (CHP) can reduce county emissions by 0.6 - 2.4% if installed at favorable sites (Box 1). Enabling computer energy saving features on the nearly 50% of commercial sector computers that are currently set to run constantly during the day (Weber et al. 2000) would decrease total county emissions by another 1.1 - 2.4%. Surprisingly, installing LED exit signs at commercial businesses (at a cost of $90 per unit), can result in relatively large energy savings and decrease county emissions by an-
other 0.8 - 1.7%. In total, the zero cost mitigation options we evaluated could decrease or offset a high proportion of county emissions. For example, the rural counties that contain small cities, represented here by Tompkins County, NY (Ithaca) and Chittenden County, VT (Burlington), could offset more than half of their emissions. At higher population densities, energy efficiency strategies and technologies are the most cost effective options and could offset as much as 34% of county emissions.

### 1.6.2 Combined Heat and Power

Combined heat and power (CHP) is the use of a generator to produce electrical power while using the waste heat for another purpose, such as space heating or absorption refrigeration. Using waste heat can result in energy efficiencies as high as 85% compared to 35% for conventional systems (Midwest CHP Application Center [MCHPAC] 2009). CHP also offers opportunities to switch to low carbon fuels (such as natural gas), which can provide additional reductions in carbon emissions. CHP technology is now available in smaller, more scalable configurations; however, its feasibility needs to be evaluated on a case-by-case basis.

“Spark spread” is a widely accepted initial indicator of CHP economic feasibility and is defined as the cost comparison between electrical power and a chosen combustible fuel source. A high spark spread (typically greater than $12/MMBtu) indicates a good candidate for CHP systems (MCHPAC 2009). Other factors that contribute to economic feasibility include a good balance of thermal and electrical load, larger building size, and long operating hours.
(MCHPAC 2009). Good candidates for CHP include office buildings, hospitals, colleges, schools, hotels, and certain industrial operations (MCHPAC 2009).

To assess the economic viability of CHP systems we first performed a spark spread analysis comparing natural gas and electricity prices in the nine counties. The results showed high spark spreads in all counties ($25.08/MMBtu to $34.56/MMBtu), with the exceptions of Baltimore City and Baltimore County, where the spark spreads were much lower ($12.66/MMBtu), but still promising. For these two counties, other factors (good electrical/thermal balance, large building size, long operating hours, state or federal incentive programs) will be necessary for favorable payback timeframes.

A second analysis was performed to determine the potential carbon reductions that would result if natural gas powered CHP systems were installed in all high potential buildings in each county (hospitals, educational facilities, office buildings, and lodging). The total square footage of each building type in each county was estimated by scaling regional building square footage estimates (EIA CBECS 2003) by county population or student population for educational buildings (Census 2010). Next, average electrical, heating, and hot water intensities per square foot (EIA CBECS 2003) were used to calculate the annual energy usage for each building type. The fuel required for each CHP system was estimated using the total electrical load and a conservative heat rate (Midwest CHP Center 2010). These energy requirements were converted to tons of CO₂ emissions (EPA 2010). The net CO₂ reduction was obtained by summing current electrical and thermal CO₂ emissions and subtracting the CO₂ released from hypothetical CHP systems (Table 1.2). These calculations demonstrate that CHP can provide substantial, cost-effective emissions reductions in candidate
buildings, particularly in counties where the spark spread is high.

Table 1.2: Potential Carbon Emissions Reductions for CHP Installation (tons C). Potential carbon emissions reductions that would result if natural gas powered CHP systems were installed at all high potential buildings in each county (hospitals, educational facilities, office buildings, and lodging).

<table>
<thead>
<tr>
<th>County</th>
<th>Educational Facilities</th>
<th>Hospitals</th>
<th>Office Buildings</th>
<th>Lodging</th>
<th>Total as Percent of County Emissions</th>
</tr>
</thead>
<tbody>
<tr>
<td>Coos, NH</td>
<td>129</td>
<td>705</td>
<td>321</td>
<td>994</td>
<td>1.78%</td>
</tr>
<tr>
<td>Grafton, NH</td>
<td>1,565</td>
<td>1,410</td>
<td>1,682</td>
<td>1,679</td>
<td>1.89%</td>
</tr>
<tr>
<td>Tompkins, NY</td>
<td>4,699</td>
<td>235</td>
<td>1,354</td>
<td>583</td>
<td>2.42%</td>
</tr>
<tr>
<td>Chittenden, VT</td>
<td>2,518</td>
<td>235</td>
<td>2,503</td>
<td>1,371</td>
<td>1.56%</td>
</tr>
<tr>
<td>Worcester, MA</td>
<td>7,498</td>
<td>3,759</td>
<td>8,579</td>
<td>2,125</td>
<td>0.77%</td>
</tr>
<tr>
<td>Baltimore County, MD</td>
<td>9,346</td>
<td>1,645</td>
<td>9,926</td>
<td>1,611</td>
<td>0.61%</td>
</tr>
<tr>
<td>Essex, MA</td>
<td>6,526</td>
<td>4,229</td>
<td>8,061</td>
<td>2,296</td>
<td>0.80%</td>
</tr>
<tr>
<td>Middlesex, MA</td>
<td>18,365</td>
<td>7,989</td>
<td>24,500</td>
<td>4,078</td>
<td>1.02%</td>
</tr>
<tr>
<td>Baltimore City, MD</td>
<td>7,484</td>
<td>3,994</td>
<td>8,314</td>
<td>1,234</td>
<td>0.92%</td>
</tr>
</tbody>
</table>

1.6.3 Non-zero Cost Mitigation Opportunities

The non-zero cost opportunities we evaluated (Figure 1.7) included terrestrial sinks (e.g. preservation of forest land and afforestation of inactive agricultural land) and biofuel crops (willow and switchgrass for solid fuels, corn ethanol and soybean biodiesel for transportation fuels). We also evaluated the potential for photovoltaic systems in the residential, commercial, and industrial sectors. Lastly, there are many opportunities to decrease emissions in the transportation sector. These transportation opportunities include improved fuel efficiency for passenger vehicles, use of hybrid electric buses, increased carpooling and use of public transportation, traffic system upgrades, and utilizing local waste oil for fuel. Among the counties evaluated, only Tompkins County had completed a
Figure 1.7: Non-zero cost mitigation opportunities. At present (in the absence of government subsidies and strong carbon markets) these opportunities do not fully recoup their initial investment costs. The y-axis shows mitigation potential as a percentage of current emissions. County population densities increase from left to right.
detailed transportation analysis, and we use that case study to demonstrate the potential for mitigation in the transportation sector.

The rural counties (under 100 persons/km²) could offset a large percentage of current emissions (18% - 424%) by protecting and expanding forest carbon sinks, and in some cases, by growing bioenergy crops. Among the opportunities evaluated, forest preservation can protect the existing largest potential C offset for rural counties, representing 18% - 417% of annual emissions or 77,000 to 511,000 Mg C/yr. However, these future carbon sinks face strong development pressure. For instance, the Maryland Department of Planning predicts that if current trends continue the State will lose 9% of its forest cover by 2020 (Weber et al. 2006). This trend holds true for the Northeast as a region, where approximately 9% of existing forests are predicted to be subsumed by urban development between 2000 and 2050 (Nowak et al. 2005). Relatively high land values in much of the Northeast combined with the low value of carbon offsets in existing markets mean that at present, forest preservation is a non-zero cost strategy (Sohngen and Mendelsohn 2003). The next largest non-zero cost carbon offset available to rural counties is afforestation of non-forest land (Plantinga et al. 1999), which could offset 0.3 - 13% of current county emissions. In the suburban counties (>180 persons/km², excluding Baltimore City), the combined potential for forest preservation and afforestation is significant in magnitude (101,000 to 312,000 Mg C/yr) and could offset 1.9% to 11% of current emissions. However, it is likely that land prices and development pressure in these areas will be highest. For instance, Nowak et al. (2005) predict that four of the most developed northeastern states (Rhode Island, New Jersey, Massachusetts, and Connecticut) will each be more than 60% urban by the year 2050.
In the residential, commercial and industrial sectors, grid-connected photovoltaic systems have the potential to reduce county-level CO$_2$ emissions by 9.5 - 19%, but the initial costs are high and payback periods exceed the lifetime of the systems unless large subsidies are available to reduce costs. Initial system costs are approximately $37,500 for 5kW residential systems and $112,500 for 15 kW commercial systems (Vadas et al 2007) with associated energy savings and credits (from selling electricity back to grid) ranging from $800 - $1,000/yr and $1,900 - $2,800/yr, respectively. At current electricity prices, and assuming no government subsidies, this would mean a payback period of 37 to 63 years depending on the county and the application. If the capital costs associated with PV systems were to decrease, or energy prices, carbon credits, and/or efficiency of PV cells were to increase, then PV would become a more cost-effective opportunity for the region. A number of states in the region offer economic incentives that cover a significant proportion of the upfront costs; however, the total amount of money set aside for such incentives has been relatively small (Barbose et al. 2008). For example, New York State offers incentives that cover approximately 40 - 45% of the typical installed cost of a residential or commercial system, but the total funds currently budgeted for such incentives is only $13.8 million (NYSERDA.org), which would cover fewer than 900 residential systems.

### 1.6.4 Transportation Sector Mitigation Opportunities

To illustrate potential mitigation opportunities in the transportation sector we highlight the case of Tompkins County, NY. Much of our analysis is derived from the Ithaca Tompkins County Transportation Council’s (TCTC) 2025 Long
Range Transportation Plan (2004), which includes opportunities to decrease greenhouse gas emissions. In the plan, the TransCAD travel demand model (Caliper Corporation 2008) was used to generate and distribute trips along the road network, which included all state, county, and important local roads, and to simulate the results of proposed transportation upgrades.

A broad suite of mitigation opportunities applies to the transportation sector (Figure 1.6), including changes in land development patterns to support mixed-use and other environment-friendly zoning practices (Banister 1999). The impact of these land use planning activities was tested with a number of indicators, including congestion and vehicle miles travelled (VMT). Under proposed land use planning scenarios, the model predicted a 2% decrease in VMT at peak travel times. If this outcome is generalized to include off-peak travel times, it would mean an approximately 2% decline in county emissions compared to the business as usual scenario.

Other mitigation opportunities in the transportation sector include improving passenger vehicle fuel efficiency, producing transportation biofuels, increased carpooling to work, increased use of public transit, traffic signal upgrades, upgrading the county bus fleet to hybrid-electric drive-trains, and utilizing in-county waste oil for fuel (Table 1.3). The opportunity with the largest potential to decrease emissions (of those evaluated) is increasing passenger vehicle fuel efficiency in the county. Current passenger vehicle fuel efficiency is estimated at 27 miles per gallon (MPG), and an increase to 35 MPG would offset transportation related CO$_2$ emissions by 6.6%. Increasing effective fuel efficiency to 50 MPG would lead to an 18.9% decrease in transportation related emissions. The county could provide some incentives to encourage the use
of fuel efficient vehicles, such as enhanced parking privileges or special travel lanes for hybrid, plug-in hybrid, and electric vehicles. Increased carpooling to work could reduce transportation emissions by up to 7.4%, assuming that the majority of persons who drive alone to work participate. Increasing bus ridership in the county by approximately 1/3 (or 1,000,000 annual rides) would decrease transportation emissions by 1.3%. A mix of smaller upgrades, including traffic signal upgrades in the city of Ithaca (0.6%), hybrid electric buses (0.2%), and using available county waste-oil as fuel (0.1%) could further reduce transportation emissions. Transportation emissions could be offset another 0.8% by growing corn and soybeans for ethanol and biodiesel based upon a scenario that avoids deforestation and competition with existing agricultural production. Taken in total, these improvements have the potential to decrease transportation sector emissions by 16.9 - 29.2% and total county emissions by 6.3 to 11% (Table 1.3). The transportation mitigation portfolio for other counties would vary as a consequence of differences in current transportation systems and other factors but clearly incentives for increased passenger vehicle fuel efficiency will dominate the mitigation opportunities region wide.

1.6.5 Other Patterns in Mitigation Cost and Potential

A number of regional and local conditions contribute to differences in potential mitigation costs and emissions benefits (Figure 1.8) among the counties, particularly the mix of fuels used for heating and electricity generation, local climate (e.g. cooling and heating degree days), and fuel prices. The payback period for photovoltaic systems; energy efficient lighting, air-conditioning, and appliances; and commercial wind installations are all dependent on the market price
Table 1.3: Summary of transportation mitigation opportunities for Tompkins County, NY. Range for TOTAL row is based on 35 MPG vs 50 MPG vehicle fuel efficiency scenarios.

<table>
<thead>
<tr>
<th>Transportation Mitigation</th>
<th>(Mg C/yr)</th>
<th>% Transport Emissions</th>
<th>% Total Emissions</th>
</tr>
</thead>
<tbody>
<tr>
<td>Vehicle fuel efficiency to 50 MPG</td>
<td>20,774</td>
<td>18.90%</td>
<td>7.30%</td>
</tr>
<tr>
<td>Vehicle fuel efficiency to 35 MPG</td>
<td>7,226</td>
<td>6.60%</td>
<td>2.50%</td>
</tr>
<tr>
<td>Increased Carpooling to work</td>
<td>8,200</td>
<td>7.40%</td>
<td>2.90%</td>
</tr>
<tr>
<td>Increased Bus Ridership</td>
<td>1,417</td>
<td>1.30%</td>
<td>0.50%</td>
</tr>
<tr>
<td>Traffic signal upgrades</td>
<td>670</td>
<td>0.61%</td>
<td>0.24%</td>
</tr>
<tr>
<td>Biodiesel</td>
<td>472</td>
<td>0.43%</td>
<td>0.17%</td>
</tr>
<tr>
<td>Ethanol</td>
<td>391</td>
<td>0.35%</td>
<td>0.14%</td>
</tr>
<tr>
<td>Hybrid Electric Buses</td>
<td>189</td>
<td>0.17%</td>
<td>0.07%</td>
</tr>
<tr>
<td>Waste oil as fuel</td>
<td>73</td>
<td>0.07%</td>
<td>0.03%</td>
</tr>
<tr>
<td>TOTAL</td>
<td>18,620 - 32,169</td>
<td>16.9% - 29.2%</td>
<td>6.6% - 11.3%</td>
</tr>
</tbody>
</table>

of electricity (EIA 2009), which is highest in New York and Massachusetts (15-16 cents/kWh) and lowest in Maryland (10-11 cents/kWh). These factors can combine to create dysfunction in the economic incentive structure for carbon abatement. For example, as described earlier, Baltimore County and City have the highest carbon intensity residential heating systems in the region (because of low efficiency and high coal reliance), yet they have the lowest economic incentive (largest simple payback time) for mitigation opportunities (Figure 1.8). Clearly, correction of this sort of economic distortion, such as through a rise in the cost of carbon credits under RGGI, will be needed.

1.7 Conclusions

Clearly, no single carbon mitigation strategy will be cost-effective for all locations in the Northeast region; however, by implementing a range of locally tailored management and technology options, substantial emissions reductions
Figure 1.8: Summary of technological mitigation opportunities for rural (< 100/km²), suburban, and urban counties (Baltimore City). The height of each bar indicates the mean carbon offset potential as a percentage of current emissions. The color indicates the expected payback period, with payback periods increasing from bottom to top.

can be achieved at low cost. For largely rural counties, forest preservation, afforestation, fuelwood harvest, and utility-scale wind power can provide the largest and most cost-effective mitigation opportunities. For urban and suburban counties, energy efficiency measures and energy saving technologies will be most cost-effective. Many of the mitigation opportunities presented here are effectively “zero cost”, as the energy saved or income generated will equal or exceed initial capital costs over time. These zero cost options could effectively offset or decrease emissions by as much as 31 - 1,064% in counties across the region. And if the non-zero cost options are included, emission can be reduced
by another potential 14 - 443%. In both cases, the greatest emissions reductions are possible in rural counties where carbon sequestration in vegetation and soils may already exceed current emissions. If the transportation sector opportunities explored in Tompkins County are indicative of what can be achieved in other counties, then county-wide emissions could be reduced by an additional 6.6 - 11.3%.

Despite these promising findings, fully implementing even the zero-cost mitigation opportunities will be difficult or impossible without strong leadership, effective policies, and greater public support for reducing carbon dioxide emissions. The potential clearly exists to dramatically alter the carbon mitigation landscape in the United States by taking best advantage of a suite of existing technologies and future breakthroughs in such areas as building design, alternative energy vehicles, biofuels and cellulosic ethanol, photovoltaics and carbon capture. The efficient implementation of these technologies will be best accomplished by tailoring policies to the local level, and the patterns and trends explored in the present study can provide a blueprint for strategies to achieve this objective.
CHAPTER 2
ACCUMULATION OF CARBON AND NITROGEN IN RESIDENTIAL
SOILS WITH DIFFERENT LAND USE HISTORIES

2.1 Introduction

Recent data suggest that for the first time in history, the world’s urban population has exceeded the world’s rural population and that urban areas will account for all population growth over the next four decades (UN WUP 2007). Between 1982 and 1997 the amount of urbanized land in the United States increased by almost 50% (Fulton et al. 2001). In the Chesapeake Bay region of the U.S., developed land area is expected to increase an additional 80% by 2030 if current trends continue (Goetz et al. 2004). There is no doubt that urban areas are growing in size and importance; however, we are only beginning to understand how the process of urbanization influences ecosystem dynamics (Kaye et al. 2006).

Urbanization has profound, but highly variable and poorly characterized effects on soils (Pouyat et al. 2010). Soils in urban landscapes are generally thought of as low in fertility, highly disturbed and heterogeneous - findings supported by research that has focused on highly compacted areas and human-constructed soils along streets (e.g. Craul and Klein 1980, Short et al. 1986). However, in urban landscapes as a whole, soils that are largely undisturbed or of high fertility have also been found (Pouyat et al. 2007, Pouyat et al. 2010). There is increasing evidence that urban soils provide important ecosystem services by acting as a sink for atmospheric nitrogen (N) deposition (e.g. Raciti et al. 2008), offsetting carbon emissions (e.g. Golubiewski 2006, Townsend-Small
and Czimczic 2010), and providing stormwater treatment (e.g. Zhu et al. 2004, Dietz and Clausen 2006). The influence of urbanization on soil C and N maybe be particularly great in former agricultural areas, which are often depleted in C and N (Tilman and Knops 2000, Post and Kwon 2000) at the time of development, but may accumulate C and N over time (Compton and Boone 2000, Tilman and Knops 2000, Post and Kwon 2000, Pouyat et al. 2006). There is increasing interest in home lawns, which are the dominant vegetation cover in residential areas (Milesi et al. 2005). Previous studies have found a wide range of soil conditions and processes in home lawns and there is thus considerable uncertainty about their impact on ecosystem processes and services (Petrovic 1990, Easton and Petrovic 2008, Groffman et al. 2009).

There is particular uncertainty about carbon and nitrogen dynamics in residential soils at depth (Galbraith et al. 1999, Pouyat et al. 2006), i.e., below the upper 15 cm where most research has been done. Obtaining deep soil cores (e.g. one-meter or greater) from residential areas is difficult. The heterogeneity of residential systems necessitates working at a diversity of sites, which requires the permission of many property owners; digging permits must be obtained; and coordination with local utilities is required to mark the location of power, water, sewage and gas lines. These obstacles increase the time and cost to sample an appropriate population of sites in residential environments. A major source of uncertainty is the nature and extent of soil profile modification during construction and home lawn establishment (DeKimpe and Morel 2000). There have been few assessments of how disturbance, land use history and age of residential soils influence carbon and nitrogen pools and fluxes.

In this study, we used one-meter soil cores to evaluate soil profile character-
istics and carbon and nitrogen pools in 32 residential home lawns that differed by previous land use and age, but had the same soil type. These were compared to 8 forested reference sites with the same soil type. The work was part of U.S National Science Foundation funded long-term ecological research (LTER) studies of the Baltimore metropolitan area (Pickett et al. 2008). Our specific objectives were to evaluate 1) soil carbon and nitrogen storage in residential lawns relative to the natural ecosystem type that they replace, 2) if this storage varied with previous land use (forest versus agriculture) and time since development, and 3) the nature and extent of residential soil profile disturbance. We chose residential sites that were relatively young (generally between 4 and 60 years old), because previous research suggests that lawn soils may aggrade in C and N most quickly in the years immediately following land conversion (Qian and Follett 2002, Golubiewski 2006, Pouyat et al. 2009). At the outset of the study, we hypothesized that 1) the lawn chronosequence would reveal evidence of C and N accumulation with time since development, 2) land use history would be an important predictor of soil C and N with former forest sites having greater soil C and N than former agricultural sites, 3) land use history would be a less important predictor of soil C and N among older residential sites, and 4) some sites would show strong evidence of soil disturbance from development.
2.2 Methods

2.2.1 Site Description

Soil cores were obtained from residential and forest sites in the Baltimore, MD USA metropolitan area. This area has a temperate climate with warm, humid summers (1,220 cooling degree days), cold winters (4,720 heating degree days) and mean annual precipitation of approximately 1,060 mm distributed relatively evenly throughout the year (NCDC 2009). Soil cores were obtained from 32 residential properties and 8 forest sites. The residential sites were mostly within the Gwynns Falls Watershed (76° 30’ , 39° 15’ and approximately 17 km²), which has a population of 356,000 people with sub-watershed densities ranging from 2,200 to 19,400 persons/km². Forest soil cores were taken from permanent forest plots of the Baltimore Ecosystem Study (BES) LTER, which have been described in detail elsewhere (Groffman et al. 2006, 2009). These remnant forests are over 100 years old with soils that were comparable in type and texture to those underlying the residential study sites (NRCS 1976, 1998).

2.2.2 Residential Site Selection

To aid the site selection process we used neighborhoods in the Baltimore City metropolitan area that have been mapped using HERCULES, a high resolution land cover classification system designed to assist in the study of human-ecological systems (Cadenasso et al. 2007). Using HERCULES and additional data sources, we identified residential sites that were similar except for single factors that we hypothesized to be important predictors of ecosystem dynamics.
These factors included land use history (agriculture and forest), development density (i.e. housing density; low and med/high), and housing age (4 to 58 yrs old) (Table 2.1). Housing age was acquired from the Maryland Property View database (MD Dept of Planning 2007). Prior land use was determined based on land use change maps developed by integrating aerial photos from 1938, 1957, 1971, and 1999 into a geographic information system (Wehling 2001). Once a list of residential parcels meeting the predefined criteria were identified, we sent mailings to property owners chosen at random from each of the factor groups with the goal of recruiting 40 property owners for a 3 year study (of which this work is a part). We had recruited 32 property owners at the time that soil cores were obtained.

Table 2.1: The study design, which included 32 residential properties that differed by previous land use, housing age, and development density, but had the same soil type. Ag = agriculture.

<table>
<thead>
<tr>
<th>Factors</th>
<th>Comparisons</th>
<th>Observations</th>
</tr>
</thead>
<tbody>
<tr>
<td>age</td>
<td>Regression</td>
<td>32</td>
</tr>
<tr>
<td>land use history</td>
<td>Ag vs Forest</td>
<td>10 &amp; 22</td>
</tr>
<tr>
<td>structure density</td>
<td>Low vs Medium</td>
<td>9 &amp; 23</td>
</tr>
</tbody>
</table>

2.2.3 Sample Collection

Soil coring took place over a one month period during the summer of 2007. For residential sites, we overlaid a grid onto a map of each property and randomly chose two locations for coring. Locations beneath impervious surfaces (buildings, walkways, driveways) or within close proximity to belowground pipes and power lines were excluded and another random location identified. Undisturbed one-meter soil cores were extracted from each of these locations using
a 3.3 cm diameter soil corer. Cores were enclosed in plastic sleeves with end-caps, put into coolers, and transported to the laboratory where they were stored at 4 °C until they could be processed. Coring in the forested reference plots followed a similar procedure, with two cores taken from random locations at each site. In total, 80 intact soil cores were collected from 32 residential properties and 8 forest sites.

2.2.4 Sample Processing

Digital photos were taken of each soil core followed by a visual inspection to determine horizon depths and Munsell color. Soil cores were also inspected for obvious signs of disturbance such as buried horizons, lithologic discontinuities, or human artifacts of less than 3.3 cm (the diameter of the soil core). Cores were divided into four soil depth intervals (0 to 10 cm, 10 to 30 cm, 30 to 70 cm, and 70 to 100cm) and sorted to remove coarse roots and rocks (> 2 mm). The roots and rocks were dried at 105 °C, weighed and set aside. Rock volume was determined by mass and an assumed density of 2.7 g/cm³. Subsamples of homogenized soil from each depth interval were analyzed for soil dry weight and percent moisture (48 hrs at 105 °C). Bulk density (BD) was calculated as BD = (Total Dry Mass - Rock Mass) / (Total Volume - Rock Volume). Soil texture was obtained by the hydrometer method (Gee and Bauder 1986). Total C and N were obtained by flash-combustion / oxidation using a Thermo Finnigan Flash EA 1112 elemental analyzer. For all data, the density of C in a unit area (1 m²) was calculated as $C = C_f B_D (1 - \delta_{2mm}) V$, where $C$ is carbon density, $\delta_{2mm}$ is the fraction of material larger than 2 mm diameter, $B_D$ is bulk density, $C_f$ is the fraction by mass of organic C, and $V$ is the volume of the soil core (Post et al.,
2.2.5 Statistical Analysis

We used multivariate analysis of variance (MANOVA) to test for the main effects of housing age, development density, depth, and land use history on each of the measured response variables (soil C, N, BD) with appropriate transformations to meet assumptions of normality. In the MANOVA, housing age was a continuous variable, while development density, depth, and land use history were categorical variables. Regression analysis was used to test for relationships between housing age and total soil C, total soil N, and BD. A two-tailed t-test was used to test differences in whole-core (0 to 100 cm, normalized by soil mass) C, N, and BD between residential sites and forested reference sites. Repeated measures ANOVA was used to test for differences between residential and forest soils across depth intervals. To complement our visual assessment of soil disturbance, we conducted a quantitative test for abrupt changes in soil texture and bulk density. For residential soils, we defined an abrupt change as a change in BD or soil texture between two adjacent soil depth intervals that was more than two standard deviations from the mean change measured in the undisturbed forested reference soils. All statistical analyses were performed using SAS JMP version 8 statistical software (SAS institute 2009).
2.3 Results

2.3.1 Soil Profile Analysis

There was no statistically significant difference in soil BD (Figure 2.1d) between residential and forested reference soils at the whole-core level (BD = 1.18 ± 0.02 and 1.19 ± 0.03) or with depth. Among residential soils, development density and housing age were not significant predictors of BD. Soils at 31 of the 40 sites (residential and forest) had loam texture (using the USDA classification system). The remaining sites had sand loam, silt loam, and clay loam texture. There were no significant differences in soil texture between or among residential and forest soils; this held true for all soil depth intervals. We found no evidence of redox-morphic features, which suggests that these soils were generally well-drained. A visual inspection of the soil profiles revealed signs of recent disturbance at a few residential sites, including evidence of fill material and buried soil horizons - most likely the result of construction and/or landscaping activities. However, the majority of residential soil profiles did not show strong visual evidence of disturbance. Our quantitative analysis, which looked for abrupt changes in soil BD and texture, supported this interpretation. Five of the 32 residential sites had soil cores that showed abrupt changes in soil texture. Among those five sites, two also showed abrupt changes in BD.

2.3.2 Carbon Density

Mean soil C density from 0 to 100 cm (Figure 2.1a) was significantly higher in the residential soils than in the forested reference soils (6,945 ± 632 g/m²)
Figure 2.1: Comparison of carbon density, nitrogen density, carbon-to-nitrogen ratio, and bulk density between residential and forest soils at 0 to 100 cm depth (a, b, c, and d; n = 32 and n = 8 for residential and forest, respectively). * P < 0.05; ** P < 0.01.

versus 5,443 ± 361 g/m², p = 0.04). Among the residential sites, housing age (p < 0.01) and land use history (p < 0.01) were significant predictors of soil carbon density with increased age and previous agricultural land use associated with greater soil C. Development density (i.e. housing density) was not a significant predictor of carbon density (p = 0.34). For residential sites built on agricultural lands, there was a positive linear relationship between carbon density and age (r² = 0.49, p = 0.04, Figure 2.2a). The slope of the regression line suggests a C accumulation rate of 82 g/m²/yr for residential sites built on agricultural land (Figure 2.2a). There was no significant trend in soil C with age for residential sites built on forest land (r² = 0.02). We found significant differences between residential and forested reference soils at 10 to 30 cm and 30 to 70 cm depth intervals (p = 0.02, p < 0.01, Figure 2.3a). Differences in soil C at our shallowest
Figure 2.2: Regression of housing age against soil carbon and nitrogen density (a and b, respectively) at 0 to 100 cm depth for residential sites that were in agriculture prior to development (n = 10). A dashed line in each panel indicates the mean carbon and nitrogen concentrations measured in forested reference sites with the same soil type (n = 8).
Figure 2.3: Comparison of carbon and nitrogen density (a and b) and concentration (c and d) between residential and forest soils across four depth intervals (0 to 10 cm, 10 to 30 cm, 30 to 70 cm, and 70 to 100 cm; n = 32 and n = 8 for residential and forest sites, respectively). * P < 0.05; ** P < 0.01.

(0 to 10 cm) and deepest (70 to 100 cm) depth intervals were not statistically significant. C concentration (%) showed a similar pattern (Figure 2.3c).

### 2.3.3 Nitrogen Density

Mean soil N from 0 to 100cm (Figure 2.1b) was greater in residential than forested reference soils (552 ± 34 g/m² versus 403 ± 49 g/m², p = 0.02). For residential soils, age (p < 0.01) and land use history (p < 0.01) were significant predictors of soil N density; older residences on agricultural land were associ-
ated with higher soil N densities. Development density was not a significant predictor of soil N. On former agricultural land, there was a positive linear relationship between nitrogen density and age ($r^2 = 0.48$, $p = 0.04$). The slope of the regression line suggests a N accumulation rate of 8.3 g/m$^2$/yr for residential sites built on agricultural land (Figure 2.2b). In contrast, the residential sites built on forest land showed no significant time-related trend ($r^2 = 0.01$). We found significant differences in soil N density between residential and forested reference soils at all depth intervals (0 to 10 cm, 10 to 30 cm, 30 to 70 cm) except for 70 to 100 cm ($p = 0.03$, $p < 0.01$, $p < 0.01$, and $p = 0.41$, Figure 2.3b). Patterns in N concentration (%) showed a similar pattern (Figure 2.3d).

### 2.3.4 C:N

There was no statistically significant difference in soil C:N (Figure 2.1c) between residential and forested reference soils (12.6 ± 0.6 and 13.5 ± 1.42); much of the difference in arithmetic means is attributable to two forested sites with particularly high soil C:N. If soil cores from these two sites are excluded, the mean C:N for forested reference soils is the same as for residential soils (12.5). Linear regression analysis corroborates these findings; C and N were highly correlated ($r^2 = 0.61$, $p < 0.01$) with no statistically significant difference in slope between residential and forested sites (Figure 2.4).
2.4 Discussion

Our results indicate that residential soils contain higher C and N densities than nearby forest soils in our study area (Figures 2.1a and 2.1b) and that many of these residential soils appear to be accumulating C and N over time (Figures 2.2a and 2.2b). We considered the possibility that soil compaction in residential areas was driving part of this trend, but there were no significant differences in BD between residential and forest soils (Figure 2.1d). For residential areas built on agricultural land, housing age appears to be a predictor of soil C and N (Figures 2.2a and 2.2b) with the slope of the regression lines suggesting accumulation rates of 82 g C/m²/yr and 8.3 g N/m²/yr. This C accumulation rate is similar in magnitude to rates measured for golf courses (90 - 100 g C/m²/yr,
Qian and Follett 2002) and conservation reserve program lands (110 g C/m$^2$/yr, Gebhart et al. 1994). These results also support the findings of Pouyat et al. (2006), who found soil C densities in residential sites comparable to northeastern forests and possibly higher than forests in the Middle-Atlantic region. In a similar vein, Golubiewski (2006) found that urban greenspaces were associated with substantially greater soil C pools than native grasslands in a semi-arid system. Taken together, these findings support the hypothesis proposed by Pouyat and colleagues (2006, 2009) that anthropogenic management practices have the potential to overwhelm the native environmental factors that control soil C storage.

Turfgrass (i.e. lawn) is the dominant vegetation cover in residential systems and may play an important role in organic matter and N accumulation. For instance, recent studies show that lawns have dynamic soil C and N fluxes with considerable potential for organic matter accumulation and N retention (Qian and Follett 2002, Kaye et al. 2005, Golubiewski 2006, Raciti et al. 2008). Raciti et al. (2008) found evidence for rapid uptake and incorporation of N inputs into lawn biomass, and later, into soil organic matter (SOM) pools. The strong positive relationship between C and N in residential soils in this study (Figure 2.4) suggests that C may be accumulating along with these N inputs, presumably at ratios commonly found in SOM (approximately 14:1, Cleveland and Liptzin 2007). A study of golf courses by Qian and Follett (2002) showed sharp increases in soil C and N over the first 25-30 years of a chronosequence and continued accumulation of C and N up to 45 years after golf course establishment. Lawns have longer growing seasons than deciduous trees and tend to receive greater nutrient and water inputs due to irrigation and fertilizer application (e.g. Petrovic 1990, Law et al. 2004), factors that may further drive C and N accumulation.
in residential soils.

While C and N densities were higher in residential soils as a whole, our regression analysis did not reveal a significant trend in soil C and N with housing age among residential sites on former forest land. This was in contrast to results from sites on former agricultural land. Several potential explanations could contribute to this finding. First, our data suggest that residential soils on former forest land have higher initial (i.e. immediately following development) C and N densities than former agricultural sites. Among residential sites that were less than 20 yrs old, mean C densities were 6,171 ± 677 g/m² on former forest versus 4,692 ± 474 g/m² on former agricultural sites. Though differences in development practices cannot be ruled out as a causal factor, lower carbon densities among young, former agricultural sites suggest that agricultural soils experienced carbon losses prior to development. C and N depletion is common in agricultural soils, with eventual recovery following agricultural abandonment (e.g. Compton and Boone 2000, Tilman and Knops 2000, Post and Kwon 2000). Tilman and Knops (2000) found that their mid-western agricultural sites had lost 75% of their original soil N and 89% of their soil C; they predicted that, in unmanaged conditions, it would take 180 - 230 years for C and N to recover after agricultural abandonment. A change from agricultural to residential land use may lead to a faster recovery of soil C and N. For instance, Golubiewski (2006) suggested a short (approximately 25 year) time frame for the recovery of soil C and N after development on shortgrass steppe soils. Our results also suggest a rapid recovery of soil C and N for post-development agricultural soils.

Based on previous BES research, we assembled a nitrogen mass balance for the residential sites (Table 2.2) to gain a better understanding of how external N
inputs compared to estimated rates of N accumulation. We found that estimated N fertilizer inputs (Law et al. 2004) were remarkably similar in magnitude to N accumulation rates on residential sites with agricultural land use history. This suggests that residential sites on previously agricultural land have a high capacity to retain fertilizer N inputs, at least for several decades. While leaching losses from the BES sites (Groffman et al. 2009) were higher in lawns than in forests, they represented only a fraction of total N inputs (94.7 kg N/ha/yr for atmospheric deposition and fertilizer N combined), which further suggests a high capacity for N retention. A $^{15}$N tracer study found that a pulse of $^{15}$N-labelled NO$_3^-$ added to Baltimore area lawns was rapidly incorporated into microbial biomass, fine roots, aboveground vegetation, thatch, and soil organic matter (Raciti et al. 2008). Given the high level of concern about water pollution from residential areas, further research into the mechanisms for N retention in residential areas, and how they can be enhanced, is warranted.

Table 2.2: Nitrogen mass balance for residential lawns in the Baltimore metropolitan area.

<table>
<thead>
<tr>
<th>N Inputs</th>
<th>Residential</th>
<th>Forest</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Atmospheric Dep (kg N/ha/yr)</td>
<td>11.2</td>
<td>11.2</td>
<td>Groffman et al. 2004</td>
</tr>
<tr>
<td>Fertilizer (kg N/ha/yr)</td>
<td>83.5</td>
<td>N/A</td>
<td>Law et al. 2004</td>
</tr>
<tr>
<td>N Losses</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Leaching (kg N/ha/yr)</td>
<td>14.1</td>
<td>4.4</td>
<td>Groffman et al. 2009</td>
</tr>
<tr>
<td>(4 yr average)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>N Sequestration</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>N accumulation in SOM (kg N/ha/yr)</td>
<td>83.0</td>
<td>N/A</td>
<td>Current study</td>
</tr>
</tbody>
</table>

While it is logical that conversion of agricultural land use to home lawns will result in accumulation of soil C and N, it is less clear why former agricultural
soils may ultimately accumulate more C and N than former forest soils. One possible explanation is that there are inherent differences between formerly forest and formerly agricultural soils (structure, organic matter quality, micronutrients, aeration, and overall fertility) that allow agricultural soils to accumulate C and N at greater rates when under residential land use. Data from the 40+ years old residential sites support this suggestion; mean soil C and N densities were higher in former agricultural sites (9,947 ± 1,054 g C/m² and 742 ± 44 g N/m²) than former forest sites (6,277 ± 898 g C/m² and 486 ± 54 g N/m²) of similar mean age. This explanation is also supported by results from forested systems. For instance, Compton and Boone (2000) found significantly higher C and N in forest soils on lands that had previously been cultivated compared to those that had been woodlots (i.e. no mineral soil disturbance). These results are consistent with the idea that the most fertile and productive sites in the landscape were originally used for agriculture and that this inherent difference is now being expressed under residential land use. A second possibility is that differences in management, such as the frequency and quantity of irrigation and fertilizer addition, are driving these outcomes. However, a survey of homeowners from this study (unpublished) shows that current management practices are similar across our two land use histories and age classes; although historical management practices are unknown for most of these properties. Finally, it is possible that differences in construction practices at the time of development contributed to these outcomes, but the lack of obvious soil profile disturbance suggests that this is not likely.

Our data reveal evidence of C and N accumulation at depth (Figures 2.3a and 2.3b), which is difficult to reconcile with the relatively shallow rooting habit of most cool-season turfgrasses (20 - 45 cm, Wu 1985, Carrow 1989). Though
at least one turfgrass variety commonly grown in our study area (tall fescue, Festuca arundinacea) has the potential for deep root systems, the majority of turfgrass root biomass is concentrated in the upper 0 - 15 cm (Wu 1985). While we cannot establish a clear mechanism for this result, we can offer some hypotheses. One possibility is that dissolved organic carbon (DOC) and dissolved organic nitrogen (DON) are important sources of SOM accumulation at depth (Kalbitz et al. 2000) and that management of turf grass promotes this process. Another possibility is that tree dominated systems have a greater proportion of soil organic carbon (SOC) concentrated in the surface 0 to 20 cm than grassland systems due to higher and more recalcitrant C inputs at the soil surface and that we are seeing a legacy of this effect in the former forest sites (Jobbagy and Jackson 2000).

Soils at most residential sites did not show obvious signs of disturbance, which contradicts common assumptions about residential soils (DeKimpe and Morel 2000, Showstack 2010). For the most part, residential soils had well developed surface horizons with considerable soil organic matter and little visual evidence of disturbance. Our statistical analysis corroborated these findings; only 5 of 32 sites showed evidence of abrupt changes in soil texture or bulk density across soil depth intervals. Soil profiles at some sites appeared to contain fill material or buried horizons; however this was more the exception than the rule. Construction practices surely play a role and may differ by location, building type, time period, and even by individual builder, so it is difficult to know how representative our findings are. Our study, which used two randomly selected locations on each residential parcel, was not designed to answer the question of how soil disturbance varies across an individual parcel. Future research that focuses on variation within residential parcels (rather than
between them) could help answer some remaining questions, such as how soil disturbance varies with distance from the house, utility lines and landscaping features.

2.5 Conclusions

This study provides evidence for high C and N densities in residential soils relative to nearby forested soils of the same type. Results from our chronosequence of sites suggest that former agricultural areas, in particular, have the potential to accumulate C and N rapidly after development. Rates of N accumulation in former agricultural sites were similar in magnitude to estimated fertilizer N inputs, confirming a high capacity for N retention (Raciti et al. 2008). Residential sites on former forest land appear to have higher initial (i.e. immediately following development) C and N densities and also show evidence of C and N accumulation. Soils at most of the sites did not show obvious signs of disturbance, a finding that is at odds with popular assumptions about residential soils and requires further exploration. These data suggest that soils in residential areas have a significant capacity to sequester C and N in the years following development. Given the large area of these soils, they are undoubtedly significant in regional C and N balances.
CHAPTER 3
NITRATE PRODUCTION AND AVAILABILITY IN RESIDENTIAL SOILS

3.1 Introduction

Between 1982 and 1997 the amount of urbanized land in the United States increased by almost 50% (Fulton et al. 2001). In the Chesapeake Bay watershed of the U.S., developed land area is expected to increase an additional 80% by 2030 if current trends continue, consuming 5% of wetlands, 14% of forests and 23% of agricultural lands (Goetz et al. 2004). There is no doubt that urban areas are growing in size and importance; however, we are only beginning to understand how the process of urbanization influences ecosystem dynamics (Kaye et al. 2006).

The most visible changes that accompany urbanization are increased impervious surface area (e.g. buildings, roads and sidewalks) and replacement of natural vegetation or agricultural fields with lawns. Impervious surfaces in the United States are estimated to cover an area nearly the size of Ohio (Elvidge et al. 2004), and the area in lawns is estimated to be even larger (163,800 km$^2$), and continues to grow rapidly (Milesi et al. 2005). In the state of Maryland, which encompasses our study area, more than 10% of the terrestrial surface area is covered by lawns (based on Milesi et al. 2005).

The rapid increase in residential land area has raised concern about water pollution associated with fertilizers and pesticides used for lawn establishment and maintenance (Morton et al. 1988, Gold et al. 1988, 1990, Petrovic 1990, Milesi et al. 2005). This concern is heightened in areas that receive high levels
of atmospheric nitrogen (N) deposition. However, recent studies suggest that
urban watersheds have a high capacity for N retention, indicating that there are
strong sinks for N within these watersheds (Baker et al. 2001, Groffman et al.

We will need a better understanding of N cycling in urban ecosystems if we
are to accurately predict the consequences of residential expansion for water
quality. In particular, we will need to understand how land use history, spa-
tial patterns of development, and land management practices influence rates
of nitrification, nitrate (NO$_3^-$) consumption and NO$_3^-$ losses. We will also need
a better understanding of soil N dynamics at depth, where the uncertainty is
greater still (Galbraith et al. 1999, Pouyat et al. 2006), because most N cycling
work in developed areas has focused on the upper 15 cm of soil.

In this study, we evaluated nitrification and NO$_3^-$ availability in one meter
depth soil cores from 32 residential home lawns and eight forested reference
sites with similar soils. The work was part of U.S National Science Founda-
tion funded long-term ecological research (LTER) in the Baltimore metropolitan
area (Pickett et al. 2008). Soil profile characteristics and soil C and N pools from
this sampling are discussed in Chapter 2. Here, we investigated the effects of
land use history, housing density, housing age, and management practices on
net nitrification and the availability of NO$_3^-$ for leaching and runoff. A more
thorough understanding of nitrification and NO$_3^-$ availability will be crucial for
explaining the extraordinarily high variability in measured N losses from resi-
dential and turfgrass systems, which range from nearly zero to more than 50%
of applied N (Petrovic 1990). We hypothesized that NO$_3^-$ availability and net ni-
trification would be 1) greater and more variable in residential soils than forest
soils, 2) greater in residential soils that were formerly in agriculture than those formerly in forest, and 3) particularly great in residential soils that are fertilized and irrigated.

3.2 Methods

3.2.1 Site Description

Soil cores were obtained from 32 residential and 8 forest sites in the Baltimore, MD USA metropolitan area. This area has a temperate climate with warm, humid summers (1,220 cooling degree days), cold winters (4,720 heating degree days) and mean annual precipitation of approximately 1,060 mm distributed relatively evenly throughout the year (NCDC 2009). The residential sites were mostly within the Gwynns Falls Watershed (76° 30’, 39° 15’ and approximately 17 km²), which has a population of 356,000 people with sub-watershed densities ranging from 2,200 to 19,400 persons/km² (Law et al. 2004). Forest soil cores were taken from permanent forest plots of the Baltimore Ecosystem Study (BES) LTER, which have been described in detail elsewhere (Groffman et al. 2006, 2009). These remnant forests are over 100 years old with soils that were comparable in type and texture to those underlying the residential areas of our study (NRCS 1976, NRCS 1998).
3.2.2 Residential Site Selection

To aid the site selection process we used neighborhoods in the Baltimore City metropolitan area that have been mapped using HERCULES, a high resolution land cover classification system designed to assist in the study of human-ecological systems (Cadenasso et al. 2007). Using HERCULES and additional data sources, we identified residential sites that were similar except for single factors that we hypothesized to be important predictors of ecosystem dynamics. These factors included land use history (agriculture and forest), housing density (low and med/high), and housing age (4 to 58 yrs old). Housing age was acquired from the Maryland Property View database (MD Dept of Planning 2007). Prior land use was determined based on land use change maps developed by integrating aerial photos from 1938, 1957, 1971, and 1999 into a geographic information system (Wehling 2001). Once residential parcels meeting the predefined criteria were identified, we sent mailings to property owners chosen at random from each of the factor groups with the goal of recruiting 40 property owners for a three-year study (of which this work is a part). We had recruited 32 property owners at the time soil cores were obtained. A yard management survey of the homeowners was used to assign sites to classes of fertilizer application (yes or no) and irrigation (yes or no).

3.2.3 Sample Collection

Soil coring took place over a one month period during the summer of 2007. For residential sites, we overlaid a grid onto a map of each property and randomly chose two locations for coring. Locations beneath impervious surfaces (build-
ings, walkways, driveways) or within close proximity to belowground pipes and power lines were excluded and another random location identified. Undisturbed one-meter soil cores were extracted from each of these locations using a 3.3 cm diameter soil corer. Cores were enclosed in plastic sleeves with end-caps, put into coolers, and transported to the laboratory where they were stored at 4 °C until they could be processed. Coring in the forested reference plots followed a similar procedure, with two cores taken from random locations at each site. In total, 80 intact soil cores were collected from 32 residential properties and eight forest sites.

### 3.2.4 Sample Processing

Photographs were taken of each soil core followed by a visual inspection to determine horizon depths and Munsell color. Soil cores were also inspected for obvious signs of disturbance such as buried horizons, lithologic discontinuities, or human artifacts less than the diameter of the core. Cores were divided into four soil depth intervals (0 to 10 cm, 10 to 30 cm, 30 to 70 cm, and 70 to 100 cm) and sorted to remove coarse roots and rocks (> 2 mm). The roots and rocks were dried at 105 °C, weighed and set aside. Rock volume was estimated from mass and an assumed density of 2.7 g/cm³. Homogenized soil from each depth interval was analyzed for soil dry weight and percent moisture (48 hrs at 105 °C). Bulk density (BD) was calculated as BD = (Total Dry Mass - Rock Mass) / (Total Volume - Rock Volume). Soil texture was measured by the hydrometer method (Gee and Bauder 1986). Total C and N were measured by flash-combustion / oxidation using a Thermo Finnigan Flash EA 1112 elemental analyzer. Subsamples of homogenized soil from each depth interval were set aside to determine
1) initial KCl exchangeable NO$_3^-$ and NH$_4^+$, 2) net nitrification, net mineralization and basal respiration, and 3) microbial biomass C and N.

Exchangeable inorganic N (NO$_3^-$, NH$_4^+$) was extracted from approximately 10 g (dry mass) of soil with 40 ml of 2 M KCl. Samples were agitated for 60 min at 200 rpm on an orbital shaker table and then left undisturbed for 2 hours. The supernatant liquid from each sample was then collected and filtered through Whatman number 42 filter paper into nalgene bottles. Samples were analyzed colorimetrically for NO$_3^-$ and NH$_4^+$ concentration using a Lachat Flow Injection Analyzer.

Rates of potential net N mineralization, nitrification, and respiration were measured in a 10-day laboratory incubation of soils at room temperature. Soils were placed in 946-mL glass jars with lids fitted with septa for gas sampling. After 10 days, the headspace of the jars was sampled by syringe, and the gas samples were analyzed for carbon dioxide (CO$_2$) by thermal conductivity gas chromatography. Inorganic N was extracted as described above. Mineralization was calculated as the accumulation of total inorganic N, nitrification was calculated as the accumulation of NO$_3^-$, and respiration was calculated as the accumulation of CO$_2$ over the course of the 10-day incubation.

Microbial biomass C and N were measured using the chloroform fumigation-incubation method (Jenkinson and Powlson, 1976). Field moist soils were fumigated to lyse microbial cells in the samples. The fumigated samples were then inoculated with fresh soil, allowing microorganisms to regrow using the dead cells as substrate. The flushes of CO$_2$ and 2 M KCl-extractable inorganic N (NH$_4^+$ and NO$_3^-$) released by the cells during the incubation were assumed to be proportional to the C and N in the microbial biomass of the origi-
inal sample. A proportionality constant (0.45) was used to calculate biomass C from the CO$_2$ flush, which was measured by thermal conductivity gas chromatography.

### 3.2.5 Statistical Analysis

We used multivariate analysis of variance (MANOVA) to test for the main effects of housing age, housing density, soil depth, land use history, microbial biomass C and N, total soil C and N, and BD on each of the measured response variables (net nitrification, net N mineralization, exchangeable NO$_3^-$ and exchangeable NH$_4^+$) with appropriate transformations to meet assumptions of normality. In the MANOVA, housing age was a continuous variable, while housing density, depth, and land use history were categorical variables. Regression analysis was used to test for possible linear or curvilinear relationships between response variables and significant main effects variables. A two-tailed t-test was used to test for whole-core (normalized by soil mass) differences between residential and forest soils in net nitrification, net mineralization, and respiration. Repeated measures ANOVA was used to test for differences between residential and forest soils across depth intervals. We used ANOVA to test for the effects of fertilizer application (yes or no) and irrigation (yes or no) on net nitrification, net N mineralization, and exchangeable NO$_3^-$ and NH$_4^+$. Because few homeowners regularly watered we did not have enough degrees of freedom to test for interaction effects between fertilizer application and irrigation. All statistical analyses were performed using SAS JMP version 8 statistical software (SAS Institute 2009).
3.3 Results

3.3.1 Net Nitrification, Net N Mineralization and Basal Respiration

Whole-core net nitrification rates, normalized by soil mass to one-meter depth, were significantly higher in the residential soils than the forest soils (128.6 ± 15.5 mg/m²/day versus 4.7 ± 2.3 mg/m²/day, p < 0.001, Figure 3.1a). Net nitrification rates were also significantly higher in residential soils at all individual depth intervals down to one meter (p < 0.01) on a per gram of soil basis (Figure 3.2a).

Among residential soils, microbial biomass N, net mineralization, total soil N and total soil C were significantly correlated with net nitrification rates (p < 0.01, except microbial biomass C with p = 0.03). Basal respiration, microbial biomass C, land use history, housing age, and housing density were not significant predictors of net nitrification in residential soils. Regression analysis revealed a significant linear relationship between: microbial biomass N and net nitrification (R² = 0.51, p < 0.01 Figure 3.3a); net mineralization and net nitrification (R² = 0.64, p < 0.01, Figure 3.3b); and total soil N and net nitrification (R² = 0.48, p < 0.01, Figure 3.3c) in residential soils.

Whole-core net N mineralization rates were significantly higher in residential soils than forest soils (123.5 ± 21.2 mg/m²/day versus 8.2 ± 16.9 mg/m²/day, p < 0.001, Figure 3.1b). When measured on a per gram of dry soil basis, net N mineralization was significantly higher in residential soils at the 10 to 30 cm depth interval, but differences between residential and forest soils were
Figure 3.1: Comparison of net nitrification, net N mineralization, basal respiration, exchangeable NO$_3^-$, exchangeable NH$_4^+$, microbial biomass N and C between residential lawn and forest soils at 0 to 100 cm depth (a - g; n = 32 and n = 8 for residential and forest respectively). * P < 0.05; ** P < 0.01.
Figure 3.2: Comparison of net nitrification, net N mineralization, basal respiration, exchangeable NO$_3^-$, exchangeable NH$_4^+$, microbial biomass N and C (a - g, respectively) between residential lawn and forest soils across four depth intervals (0 to 10 cm, 10 to 30 cm, 30 to 70 cm, and 70 to 100 cm; n = 32 and n = 8 for residential and forest respectively). * P < 0.05; ** P < 0.01.
Figure 3.3: Among residential lawn soils only, regressions of net nitrification against microbial biomass N, net N mineralization, and total soil N (a - c, respectively) and net N mineralization against total soil N (d) at four depth intervals (0 to 10 cm, 10 to 30 cm, 30 to 70 cm, and 70 to 100 cm; n = 32). * P < 0.05; ** P < 0.01.

not significant at other depth intervals (Figure 3.2b). Among the other factors we measured, only total soil N was a significant predictor of net N mineralization (p < 0.001) in residential soils. There was a significant linear relationship between total soil N and net N mineralization (R2 = 0.60, p < 0.001, Figure 3.3d).

Whole-core basal respiration rates were not significantly different between residential and forest soils (p = 0.23, Figure 3.1c). When comparing rates at individual depth intervals, only the 10 to 30 cm depth interval had significantly higher respiration rates in residential soils than forest soils (p < 0.01, Figure 3.2c).
3.3.2 KCl Exchangeable NO$_3^-$ and NH$_4^+$

At the whole core level, KCl exchangeable NO$_3^-$ was significantly higher in residential soils than forest soils ($3.8 \pm 0.5$ g/m$^2$ versus $0.7 \pm 0.3$ g/m$^2$, p < 0.001, Figure 3.1d). Exchangeable NO$_3^-$ was higher in residential soils at all individual depth intervals (p<0.01, Figure 3.2d). Exchangeable NH$_4^+$ was not significantly different between residential and forest soils at the whole-core level (p = 0.53, Figure 3.1e); however, it was significantly lower in residential soils at the 0 to 10 cm depth interval ($3.5 \pm 0.7$ g/m$^2$ versus $8.1 \pm 1.2$ g/m$^2$, p < 0.01, Figure 3.2e), with no significant differences at other depth intervals.

Among residential soils, land use history (former forest or agriculture) and housing density were significant predictors of KCl exchangeable NO$_3^-$ (p < 0.001, p < 0.01). Residential sites that had higher housing density or were formerly in agriculture, were more likely to have higher concentrations of KCl exchangeable NO$_3^-$ when other experimental factors were taken into account. None of the experimental factors was a significant predictor of exchangeable NH$_4^+$ among residential sites.

3.3.3 Microbial Biomass C and N

Microbial biomass N at the whole-core level was higher in residential soils than forest soils (p < 0.01, Figure 3.1f); however, microbial biomass C was not significantly different between the land use types (p = 0.80, Figure 3.1g). Microbial biomass N was significantly higher in residential soils at the 30 to 70 cm and 70 to 100 cm depth intervals (p < 0.01, Figure 3.2f) and microbial biomass C was significantly higher in residential soils at the 10 to 30 cm and 30 to 70 cm depth
intervals (p < 0.01, Figure 3.2g). Land use history was a significant predictor of microbial biomass N among soils at residential sites (p < 0.01). Residential sites that were formerly in agriculture had higher microbial biomass N. Among the factors that we tested, there were no significant predictors of microbial biomass C. Overall microbial biomass C:N was lower in residential soils than forest soils (10.8 ± 0.4 versus 13.8 ± 0.5).

3.3.4 Yard Management

A survey of the homeowners in our study (unpublished data) revealed that half of our homeowners applied fertilizer or had it applied for them; only one quarter irrigated. Fertilizer application and irrigation were not significant predictors of net nitrification, net mineralization, exchangeable NO$_3^-$ or exchangeable NH$_4^+$. Our survey covers recent management practices, so the effects of past management cannot be accounted for.

3.3.5 Nitrogen Mass Balance

We estimated a N mass balance for residential and forest sites (Table 3.1). On average, residential sites have received greater N inputs than forest sites (Law et al. 2004, Groffman et al. 2004) and had higher N leaching losses (Groffman et al. 2009). Net and gross rates of NO$_3^-$ production and consumption were higher in residential sites than forest sites, as were pool sizes of exchangeable NO$_3^-$ (Raciti et al. 2008). Turnover rates for NO$_3^-$ were rapid in both systems (Raciti et al 2008). Estimated rates of N sequestration in residential systems (Chapter 2)
were similar in magnitude to fertilizer N inputs (Law et al. 2004).

Table 3.1: Nitrogen mass balance for residential lawns in the Baltimore metropolitan area.

<table>
<thead>
<tr>
<th></th>
<th>Residential</th>
<th>Forest</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>N Inputs</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Atmospheric Dep (kg N/ha/yr)</td>
<td>11.2</td>
<td>11.2</td>
<td>Groffman et al. 2004</td>
</tr>
<tr>
<td>Fertilizer (kg N/ha/yr)</td>
<td>83.5</td>
<td>0</td>
<td>Law et al. 2004</td>
</tr>
<tr>
<td>N Losses</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Leaching (kg N/ha/yr)</td>
<td>14.1</td>
<td>4.4</td>
<td>Groffman et al. 2009 (4 yr average)</td>
</tr>
<tr>
<td>NO3- Production</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Net NO3 Production (kg/ha/yr)</td>
<td>469.4</td>
<td>17.1</td>
<td>This study</td>
</tr>
<tr>
<td>Gross NO3 Production (kg/ha/yr)</td>
<td>13,870</td>
<td>5,256</td>
<td>Raciti et al. 2008</td>
</tr>
<tr>
<td>NO3- Consumption</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Gross NO3 Consumption (kg/ha/yr)</td>
<td>13,432</td>
<td>4,453</td>
<td>Raciti et al. 2008</td>
</tr>
<tr>
<td>Pool Size</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Exchangeable NO3 (kg N/ha/yr)</td>
<td>37.7</td>
<td>7.0</td>
<td>This study</td>
</tr>
<tr>
<td>Pool Turnover</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>NO3- Turnover Rate (days)</td>
<td>0.16</td>
<td>0.23</td>
<td>Raciti et al. 2008</td>
</tr>
<tr>
<td>N Sequestration</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>N accumulation in SOM (kg N/ha/yr)</td>
<td>83.0</td>
<td>N/A</td>
<td>Raciti et al. submitted. (Residential sites with agricultural land-use history)</td>
</tr>
</tbody>
</table>

3.4 Discussion

3.4.1 Predictors of NO3− production and availability

Net nitrification and exchangeable NO3− (a proxy for NO3− availability) were significantly higher in residential sites than forested reference sites (Figures 3.1a, 3.1d, 3.2a, 3.2d), which may be a cause for concern with regard to runoff and leaching potential. However, it should be noted that the values we measured in residential soils were quite low overall and within the range measured elsewhere for deciduous forests. For instance, Verchot and colleagues (2001)
measured higher net nitrification rates under beech and maple stands in the Catskills of NY than we measured in our residential sites in Baltimore (2.2 ± 0.7 (beech) and 2.5 ± 1.4 (maple) µ N/g dry soil/day versus 0.7 ± 0.1 µ N/g dry soil/day for 0 to 10 cm residential soils). Exchangeable NO$_3^-$ pools were also similar between the residential sites in this study and the Catskills beech and maple stands. Verchot and colleagues (2001) also studied oak-dominated stands in the Catskills that had particularly low net nitrification and available NO$_3^-$, similar to the oak-dominated forest sites in this study (Groffman et al. 2006). There is increasing evidence that certain tree species (particularly oaks) are associated with low rates of NO$_3^-$ production and availability (e.g. Verchot et al. 2001, Lovett et al. 2004) which may help to explain the very low rates of net nitrification and pools of exchangeable NO$_3^-$ that we observed in our forest sites. Thus, the land use effect that we observed may have as much to do with low rates of activity in our forest sites as with high rates in our residential sites. Still, our results suggest that residential land use change has increased pools and production of reactive N, which has clear implications for water quality in the region (Boesch et al. 2001, Galloway et al. 2003).

Among residential sites, there were several significant, positive predictors of net nitrification, including microbial biomass N, net mineralization, and total soil N. These factors are indicative of a dynamic microbial population with access to large pools of soil organic matter and N, so it is unsurprising to see higher rates of net nitrification associated with them. The lower C:N of microbial biomass in residential soils may also be indicative of a more active N cycling community in residential soils, while the relatively high C:N of forest microbial biomass, combined with near zero (or negative) net nitrification may indicate suppressed nitrification (Lovett et al. 2004). We might expect a negative correla-
tion between microbial biomass C and nitrification as this variable can serve as an index of microbial demand for N. However, while microbial biomass N was a strong predictor of net nitrification, microbial biomass C was not.

Higher housing density and agricultural land use history were predictors of greater NO$_3^-$ availability in residential soils. The association of greater NO$_3^-$ availability on former agricultural sites may be a legacy effect of soil disturbance, a history of fertilizer addition, or these sites may be inherently more N rich than formerly forested sites. These former agricultural sites also tended to be higher in total soil N (see Chapter 2). Higher housing density may be an indicator of greater site disturbance, which is known to increase NO$_3^-$ availability and loss (Vitousek et al. 1979). If these trends hold across a larger sample of residential sites they may be useful indicators of N leaching or runoff potential in developed watersheds.

Surprisingly, fertilizer addition and irrigation were not significant predictors of available NO$_3^-$. This may reflect our study design, which does not account for historic management practices, and also the transient availability of inorganic N after fertilization application. Turfgrass studies have shown that lawns have a high capacity to sequester N, though that capacity is dependent upon a host of factors and may be overwhelmed by very high fertilizer application rates (see reviews by Petrovic et al. 1990 and Easton and Petrovic 2008). In a $^{15}$N tracer study we found that when a pulse of $^{15}$N-labelled NO$_3^-$ was added to Baltimore area lawns, it was rapidly incorporated or retained in microbial biomass, fine roots, aboveground vegetation biomass, thatch, and soil organic matter (Raciti et al. 2008). We found that for simulated atmospheric deposition events overall $^{15}$N retention was similar (and possibly higher) in lawns than in forested refer-
ence sites.

Other environmental factors, such as soil moisture and temperature, may play a role in variable $\text{NO}_3^-$ production and availability both among residential sites and between residential and forest sites. Soil temperatures are consistently higher in grass plots than forest plots, particularly in the summer months (Groffman et al. 2009, Savva et al. 2010), but there were no consistent differences in soil moisture among our plots. Residential soils may thus have higher rates of decomposition and microbial activity than forest soils, which can deplete soil C levels and increase N availability (Hart et al. 1994). This increase in decomposition and C loss, along with fertilizer inputs in at least some of our sites, likely lead to changes in microbial community composition, increases in microbial biomass N and decreases in microbial C:N and produced the high nitrification rates that we observed in residential sites relative to the forest sites.

We measured significantly greater net nitrification and $\text{NO}_3^-$ availability in residential soils, compared to forest soils, at depths as great as one meter (Figures 3.2a and 3.2d). This $\text{NO}_3^-$ is below the rooting zone for many cool season turfgrasses, which suggests a greater likelihood of leaching. However, some species of turfgrass in our study area can be deeply rooted (e.g. Kentucky blue grass, *Poa pratensis*, Wu 1985) and studies of water use in turfgrasses have shown that some species are capable of drawing on deeper water sources and even of hydraulic lift (Carroll 1989, Huang 1999). There was also evidence of greater microbial biomass C and N at depth in residential soils (Figures 3.2f and 3.2g), which can provide another active pathway for $\text{NO}_3^-$ sequestration.
3.4.2 Nitrogen Mass Balance

We estimated a nitrogen mass balance for our residential sites to gain a better understanding of how NO$_3^-$ production and availability compared to N inputs, losses, and internal cycling (Table 3.1). One striking finding is that pools of available NO$_3^-$ are quite large (37.7 kg N/ha) relative to N inputs (11.2 kg N/ha/yr and 83.5 kg N/ha/yr for atmospheric deposition and fertilizer application, respectively) and outputs (14.1 kg N/ha/yr as leaching). Net NO$_3^-$ production (469.4 kg N/ha/yr) was an order of magnitude larger still. However, these seemingly large NO$_3^-$ pools and fluxes represent only a small fraction of total N cycling in the system. Gross rates of NO$_3^-$ production (13,870 kg N/ha/yr) and consumption (13,432 kg N/ha/yr) were orders of magnitude larger than the available NO$_3^-$ pool, so it is not surprising that turnover of this pool was rapid (less than 4 hours) and leaching losses comparatively small. It is clear that residential soils can support rapid N processing and significant N retention, but further study will be necessary if we are to better understand which processes (e.g. plant uptake, microbial uptake, nitrification, denitrification) are most important for N retention and how we can manage residential systems for better environmental performance.

3.5 Conclusions

Our results suggest that residential land use change has increased pools and production of reactive N, which has clear implications for water quality in the region. However, the results contradict some common assumptions about residential soils. While net nitrification and exchangeable NO$_3^-$ were significantly
higher in residential soils than forest soils, these measures of NO$_3^-$ production and availability were still notably low - comparable to deciduous forest stands in other studies. A second unexpected result was that current homeowner management practices did not predict NO$_3^-$ availability or production, which may reflect the transient availability of inorganic N after fertilizer application in these dynamic soil ecosystems. Housing density and land use history were significant predictors of NO$_3^-$ availability in residential soils. If these factors prove to be predictive across a wider range of sites, they may be useful indicators of NO$_3^-$ availability, leaching, and runoff potential at the landscape scale. Finally, the N mass balance makes it clear that residential soils can support rapid processing and retention of N inputs. Further research will be necessary to understand which N cycling processes are most important for N retention in residential ecosystems and how we can manage them to minimize their impact on the environment.
4.1 Introduction

The developed land area of the United States has grown more than 400% in the past half century (Brown et al. 2005) and continues to grow rapidly (Goetz et al. 2004, Jantz et al. 2005, Brown et al. 2005). The most visible changes that accompany development are increased impervious surface area (e.g. buildings, roads and sidewalks) and replacement of natural vegetation and agriculture fields with lawns (i.e. turfgrass). Impervious surfaces in the United States are estimated to cover an area nearly the size of Ohio (Elvidge et al. 2004), and the area in lawns is estimated to be even larger (Milesi et al. 2005).

The rapid increase in residential land area has raised concerns about pollution associated with fertilizers and pesticides, which are used for lawn establishment and maintenance (Morton et al. 1988, Gold et al. 1988, 1990, Petrovic 1990, Milesi et al. 2005). Concern is particularly high in coastal regions, where nitrogen pollution has contributed to numerous water quality problems, including harmful algal blooms, decreased biodiversity, fisheries declines, and “dead zones” in places like the Gulf of Mexico and Chesapeake Bay (Carpenter et al. 1998, Kemp et al. 2005, Paerl et al. 2006).

It is clear that residential systems receive higher N inputs than natural systems; however, recent research suggests that residential areas may have a high capacity for N retention (Baker et al. 2001, Wollheim et al. 2005, Groffman et
al. 2009). Lawns, which are the dominant vegetation cover in residential areas, may play an important role in this outcome. While lawns can have significant N losses, especially when over-fertilizered (Morton et al. 1988, Petrovic 1990, Qian et al. 2003), they have been shown to have considerable potential for organic matter accumulation and N retention (Gold et al. 1990, Qian and Follett 2002, Kaye et al. 2005, Golubiewski 2006, Raciti et al. 2008).

In recent experiments, we have measured N inputs, outputs, and transformations in residential ecosystems with a particular focus on lawns. In a $^{15}$N tracer study we found that when a pulse of nitrate (NO$_3^-$) was added to Baltimore area lawns to simulate atmospheric deposition, it was rapidly incorporated or retained in microbial biomass, fine roots, aboveground vegetation biomass, thatch, and soil organic matter (Raciti et al. 2008). Overall $^{15}$N retention was similar, and possibly higher, than in forested reference sites. In a second study, we evaluated soil carbon and nitrogen pools in 32 residential home lawns down to one-meter of depth (Chapter 2). In that study we found evidence for rapid accumulation of carbon (C) and N in residential soils, particularly those with a history of agricultural land-use. When we studied NO$_3^-$ production and consumption in the same lawns (Chapter 3) we found that net nitrification and exchangeable NO$_3^-$ were significantly higher in residential soils than forest soils in that study, but these measures of NO$_3^-$ production and availability were still notably low - comparable to deciduous forest stands in other studies (Lovett et al. 2004). Finally, while N inputs to our developed watersheds were high (Law et al. 2004), measured leaching losses were a relatively small fraction of total inputs (Groffman et al. 2009). Using information gathered from these experiments, we created a mass balance for our lawn-dominated residential systems (Chapter 3), however, a major N flux was missing from our balance.
- gaseous losses of N\textsubscript{2} from denitrification - a process that has been notoriously difficult to measure in terrestrial ecosystems (Kulkarni et al. 2008).

Using a combination of laboratory and field measurements, we sought to close the N budget in our residential system. There is general consensus on the proximate controls of denitrification in terrestrial systems (e.g. soil temperature, moisture, oxygen, organic matter, and available NO\textsubscript{3}\textsuperscript{-}). As an anaerobic, primarily heterotrophic process, denitrification in terrestrial environments tends to occur in small areas (hotspots) and during brief periods (hot moments) of activity when these controls converge (Parkin 1987, McClain et al. 2003). In addition to influencing the total rate of denitrification, these factors influence the relative importance of the end products of denitrification, including N\textsubscript{2}O, which is an important greenhouse gas (Kulkarni et al. 2008). Two problems have hindered attempts to estimate meaningful rates outside the laboratory: 1) an inability to measure direct N\textsubscript{2} fluxes due to high atmospheric concentrations and 2) an inability to measure and monitor soil oxygen (O\textsubscript{2}) concentrations in the field (Groffman et al. 2006c). Using recent advances in instrumentation (Butterbach-Bahl et al. 2002, Burgin et al. 2010), we were able to overcome these problems to measure field-relevant rates of denitrification from lawns.

### 4.2 Materials and Methods

#### 4.2.1 Site Description

The lawns used in this study were located in the Baltimore metropolitan area in association with the Baltimore Ecosystem Study (BES, http://beslter.org), a
component of the U.S. National Science Foundation Long Term Ecological Research (LTER) network, and have been described elsewhere (Groffman et al. 2009).

The two lawns (referred to as UMBC1 and South Campus) were located on the grounds of University of Maryland Baltimore County, but were not turfgrass research plots. Campus lawns were chosen because of their known, long-term management regimes, similarities in landuse history (former agriculture) and their representativeness of typical residential lawns. Each lawn (or plot) contained a mixture of tall fescue (Festuca arundinacea spp. L.) and fine fescue (Festuca spp.). The UMBC1 lawn, which was less actively managed, also had significant white clover (Trifolium repens) cover. UMBC1 was fertilized each spring at a rate of 98 kg N/ha using LESCO 14-14-14, applied in two applications, approximately two weeks apart. Mowing was done at 2-3 week intervals (dependent on rainfall and subsequent growth) during the spring, summer, and fall seasons to a height of approximately 10 cm. This lawn received no irrigation and clippings were left in place. South Campus was fertilized each spring and fall at 98 kg N/ha (196 kg N/ha/yr total). Fertilizer was applied over two applications each season, approximately two weeks apart, using LESCO 18-24-12 in the spring and LESCO 25-5-10 in the fall. The lawn was mowed weekly during the growing season to a height of approximately 10 cm. Management of our lawns were similar to moderate (UMBC1) and high maintenance (South Campus) lawns in the study area. A lawn management survey by Law et al. (2004) found that lawn fertilizer inputs in the area ranged from zero to more than 300 kg N/ha/yr with a mean of 83.5 kg N/ha/yr in their suburban watershed (0.13 ha mean lot size). The soils at UMBC1 and South Campus have been classified as Joppa (loamy-skeletal, siliceous, semiactive, mesic Typic Hapludult) and
Brandywine (sandy-skeletal, mixed mesic Typic Dystrudept), respectively. Soil pH was between 5.7 and 6.0 for both lawns when measured at the beginning of the study. Atmospheric N deposition in the Baltimore metropolitan area is estimated at 11.2 kg N/ha/yr (Groffman et al., 2004).

4.2.2 Instrumentation

In early August of 2008, each lawn was instrumented with three Apogee diffusion-head soil O$_2$ sensors (SO-100 series, Apogee Instruments, Logan, UT) with the diffusion heads at 7 cm depth. The sensors in each lawn were connected to a Campbell Scientific CR-800 datalogger (Campbell Scientific, Logan UT), which was programmed to measure soil O$_2$ hourly over the course of approximately one year (with monthly interruptions to recharge and replace batteries at each installation). Both lawns have been continually monitored for soil temperature and moisture as part of the Baltimore Ecosystem Study network of long-term study sites (Groffman et al. 2009).

4.2.3 Sample Collection and Processing

Soil cores were collected to a depth of 10 cm using a 5 cm diameter slide-hammer corer (AMS Equipment Corp.). All cores were transported to the laboratory in coolers and then stored at 4 °C until they could be processed. A subset of soil cores from each time point were used to determine soil dry weight, percent moisture, and bulk density. Soil dry weight and percent moisture were determined by the change in mass of 10 g field moist soil after 48 hrs at 105 °C. Bulk
density (BD) was calculated as \( BD = \frac{(\text{Total Dry Mass} - \text{Rock Mass})}{(\text{Total Volume} - \text{Rock Volume})} \). Rock volume was determined by mass and an assumed density of 2.7 g/cm\(^3\).

### 4.2.4 Direct Measurement of N\(_2\), N\(_2\)O, and CO\(_2\) Flux Rates from Intact Lawn Soil Cores

Denitrification measurements were made using the Nitrogen-Free Atmospheric Recirculation Method (N-FARM) system (Burgin et al. 2010), which is based on those built by Swerts and colleagues (1995) and Butterbach-Bahl and colleagues (2002). Intact soil cores were encased in 10 stainless steel tubes (each acting as an independent incubator) connected to a gas-tight flow injection system built from Swagelok connections (Swagelok, Crawford Fitting Co., Solon, OH). These were mounted in-line with a Shimadzu GC8A gas chromatograph (Kyoto, Japan) with a thermal conductivity detector (TCD) to measure N\(_2\), N\(_2\)O, CO\(_2\), and O\(_2\). The stainless steel tubes were housed in a plexiglass box that was continuously flushed with Argon gas at 100 mL min. Using this system, the ambient soil atmosphere could be replaced with a N\(_2\)-free helium (He) and O\(_2\) atmosphere using a series of gentle vacuum-flush cycles (Butterbach-Bahl et al. 2002, Burgin et al. 2010). By lowering N\(_2\) concentration in the soil atmosphere (to below the 7.5 ppmv detection limit of our TCD), we were able to measure small fluxes of N\(_2\) gas exchange from intact soil cores. The system allowed us to vary soil O\(_2\) concentrations by adjusting the He:O\(_2\) ratio used during the vacuum-flush cycle. Regular system pressure checks, empty chambers (2 blanks per incubation), and incubation of killed soil cores (autoclaved twice
over 3 days) were used to ensure that measured fluxes were genuine rates and not due to system leaks.

4.2.5 Field Fluxes

In the field, soil:atmosphere fluxes of CO$_2$ and N$_2$O were measured using 29 cm (inner diameter) polyvinyl chloride (PVC) cylinder chambers with gas sampling ports (Bowden et al. 1991, Groffman et al. 2006b). Just before sampling, these chambers were mounted on PVC base rings installed to 5 cm depth and flush with the soil surface. These low-profile base rings allowed mowing to take place as usual between sampling intervals. At 0, 10, 20 and 30 minutes following placement of the chamber on the base, 9-mL gas samples were collected from gas sampling ports in the center of the chamber top by syringe. Samples were transferred to evacuated glass vials which were stored at room temperature prior to analysis by gas chromatography with electron capture (N$_2$O) or thermal conductivity (CO$_2$) detection.

4.2.6 Experiments

The first experiment tested the effects of O$_2$ concentration on N$_2$ and CO$_2$ flux rates. Using intact soil cores, N$_2$ and CO$_2$ fluxes were measured at 20%, 5%, and 0% O$_2$ concentrations in the N-FARM. Each soil O$_2$ concentration was tested in a separate incubation using fresh soils (n = 4) to avoid unknown affects that might arise from subjecting soil cores to prolonged incubation. Headspace N$_2$ and CO$_2$ concentrations were measured after 2, 6, and 24 hrs of incubation time.
so linear flux rates could be calculated. Our second experiment tested the effect of soil moisture and \( \text{NO}_3^- \) availability on \( \text{N}_2 \) and \( \text{CO}_2 \) flux rates at 0% \( \text{O}_2 \). Wet soil cores (30% volumetric soil moisture, \( n = 4 \)) were collected during a rain event and compared with dry soil cores (18% soil moisture, \( n = 4 \)) collected prior to the rain event. Since similarly low \( \text{N}_2 \) flux rates were obtained for both conditions, a third incubation was conducted at 0% \( \text{O}_2 \) after injecting 10 ml of \( \text{NO}_3^- \) solution (approximately 100 mg N/kg dry soil) into four “wet” soil cores. Headspace \( \text{N}_2 \), \( \text{N}_2\text{O} \) and \( \text{CO}_2 \) concentrations were measured after 2, 6, and 24 hrs of incubation time.

Our third experiment tested for changes in soil \( \text{N}_2 \), \( \text{N}_2\text{O} \), and \( \text{CO}_2 \) fluxes following a field fertilization experiment. On May 13, 2009, a 3 x 30 m section of the UMBC1 site was amended with LESCO 24-0-11 fertilizer at a rate of 49 kg N/ha using a calibrated spreader. Soil cores were collected from the recently fertilized and “ambient” (i.e. control) areas of the plot. The first N-FARM incubation was performed at 20% \( \text{O}_2 \) using ambient (\( n = 4 \)) and fertilized (\( n = 4 \)) soil cores taken 24 hours after fertilizer addition. The incubation was performed again at 2% soil \( \text{O}_2 \) using fresh cores. The experiment was repeated for cores taken 9 days after fertilizer addition. The fertilized area of UMBC1 included three gas rings that were used to measure \( \text{N}_2\text{O} \) and \( \text{CO}_2 \) fluxes in the field. Field fluxes were measured prior to fertilizer application, and 24 hours and 9 days after application (i.e. at the same time that soil cores were collected).

We performed two more N-FARM incubations to test the effects of soil saturation (e.g. from a major rain storm) on denitrification rates in fertilized soils. Field fertilized soil cores (collected 9 days post-fertilization) were saturated by slowly dripping water onto them until no further water could be absorbed.
These cores were incubated at 20% and 2% O\textsubscript{2} (n = 8 for each O\textsubscript{2} concentration).

### 4.2.7 Calculations and Statistics

Linear regression (accumulation of N\textsubscript{2}, N\textsubscript{2}O, or CO\textsubscript{2} versus time) was used to calculate flux rates from N-FARM incubations. We used empty chambers (two per incubation) and killed soil cores to correct for any N\textsubscript{2} accumulation that might have originated from system leaks or degassing from small soil pores. Flux rates were expressed on a per-area basis by dividing by the ground surface area of each soil core. Rates presented on an annual basis were calculated using a mean growing season length of 231 days for Baltimore, MD (National Climactic Data Center 2009). Field fluxes (using PVC cylinder chambers described earlier) were calculated from the linear rate of change in gas concentration, the chamber internal volume and soil surface area (Groffman et al. 2006b, 2009). Statistical differences between experimental treatments were tested using analysis of variance (ANOVA) followed by Tukey’s Honestly Significant Difference (HSD) post-hoc test. All statistical analyses were performed using SAS JMP version 8 statistical software (SAS institute 2009).

Using a combination of laboratory and field measurements we estimated annual N\textsubscript{2} fluxes in residential lawns (Table 4.1). Annual N\textsubscript{2} fluxes were estimated by apportioning flux rates from appropriate N-FARM incubations to the proportion of the growing season when field-measured soil moisture and O\textsubscript{2} conditions fell within that range. For example, we measured mean N\textsubscript{2} flux rates equivalent to 2.95 ± 1.65 kg N/ha/yr in UMBC1 and South Campus soils at 20% soil O\textsubscript{2} and 18% to 30% soil moisture. Based on soil O\textsubscript{2} and moisture measure-
ments at the sites, similar conditions were found during 95% of the year, so a rate of 2.95 kg N/ha/yr was assumed for this time period annually. Conversely, we measured very high N\(_2\) flux rates (equivalent to 224 ± 40 kg N/ha/yr) at 20% O\(_2\) in saturated, fertilized lawn soils. Based on maximum soil moisture readings at the sites (38%), we estimated that soils at 35% and greater soil water content would be approaching saturated conditions. Saturated conditions were experienced for 5% of the average year (based on 5 years of soil moisture data). By this method, total annual denitrification was estimated at 14 ± 3.6 kg N/ha/yr.

Table 4.1: Annual denitrification rates calculated for study lawns based on laboratory measured N2 flux rates and field measured soil moisture conditions.

<table>
<thead>
<tr>
<th>Volumetric Soil Moisture</th>
<th>Proportion of Year</th>
<th>N(_2) Flux (kg N/ha/yr)</th>
<th>Notes</th>
</tr>
</thead>
<tbody>
<tr>
<td>8-35%</td>
<td>0.95</td>
<td>2.95 ± 1.65</td>
<td>Fertilized lawns 18-20% O(_2), pH = 5.7 - 6.0</td>
</tr>
<tr>
<td>35%+</td>
<td>0.05</td>
<td>223.9 ± 40.1</td>
<td>Rate for saturated soils</td>
</tr>
<tr>
<td><strong>Annual N(_2) Flux</strong></td>
<td></td>
<td><strong>14.0 ± 3.6</strong></td>
<td></td>
</tr>
</tbody>
</table>

4.3 Results

We measured in situ soil O\(_2\) concentrations hourly for one year at BES lawn plots and found that concentrations stayed within a relatively small range (Figures 4.1a and 4.1b). Soil O\(_2\) generally stayed near atmospheric concentrations (20%), occasionally dropping as low as 17%. A three week period with two major storm systems (4.3 cm and 10.4 cm of precipitation) failed to cause a significant drop in bulk soil O\(_2\) concentrations at our sites.
Figure 4.1: Macro-scale soil oxygen concentrations in the study lawns (UMBC1 and South Campus) over a one year time period. Values for each lawn represent the mean of three sensors at 7 cm depth.
When we used laboratory incubations to test for the effects of soil O$_2$ concentration on N$_2$ and CO$_2$ flux rates (at 20% field moisture), we found that CO$_2$ fluxes (i.e. respiration) decreased predictably as O$_2$ concentrations were lowered ($p < 0.02$ for all differences); however increases in mean N$_2$ fluxes (denitrification) were not statistically significant due to high variability within treatments (Figure 4.2a and 2b). N$_2$ fluxes were low across the O$_2$ concentration gradient, with means ranging from $4.26 \pm 3.91 \mu g N/m^2/hr$ at 20% O$_2$ to $28.9 \pm 17.0 \mu g N/m^2/hr$ at 0% O$_2$.

We used changes in the field moisture of lawn soils (collected before and during a rain event) to measure the influence of soil moisture on N$_2$ and CO$_2$ flux rates at 0% soil O$_2$ and found that while arithmetic means were higher for flux from wet soils (30% field moisture) than dry soils (18% field moisture), these differences were not statistically significant ($59.8 \pm 13.9 \mu g N/m^2/hr$ vs. $28.9 \pm 17.0 \mu g N/m^2/hr$ and $9.1 \pm 1.5 mg C/m^2/hr$ vs $6.0 \pm 1.8 mg C/m^2/hr$, Figures 4.3a and 4.3b). When a pulse of labile NO$_3^-$ was added to wet soils incubated at 0% O$_2$, N$_2$ flux rates increased dramatically from $59.8 \pm 13.9 \mu g N/m^2/hr$ to $2216 \pm 509 \mu g N/m^2/hr$ ($p < 0.001$). CO$_2$ fluxes from wet soils amended with NO$_3^-$ were significantly higher than CO$_2$ fluxes from dry soils, but not significantly higher than fluxes in unamended wet soil cores.

Fertilized and unfertilized (“ambient”) soil cores collected 24 hours after fertilization in the field (applied at 49 kg N/ha) did not show significantly different N$_2$ and CO$_2$ fluxes when incubated at 20% and 0% O$_2$ (Figures 4.4a and 4.4b). When this experiment was repeated with soil cores collected 9 days after fertilization (Figure 4.4a), we found that soils incubated at 2% O$_2$ had higher denitrification rates than those incubated at 20% soil O$_2$, however, differences between
Figure 4.2: N₂ and CO₂ fluxes (A and B) from intact lawn soil cores at approximately 20% volumetric soil moisture across three soil oxygen concentrations (20%, 5%, and 0%). Different letters indicate significant differences across treatments (p < 0.05, n = 4).
Figure 4.3: N\textsubscript{2} and CO\textsubscript{2} fluxes (A and B) from intact lawn soil cores at 0% oxygen. Bars represent soil cores under ambient conditions prior to rainfall (approximately 20% volumetric soil moisture), ambient conditions after rainfall (approximately 30% volumetric soil moisture), and injected with a NO\textsubscript{3}\textsuperscript{-} solution after rainfall. Different letters indicate significant differences across treatments (p < 0.05, n = 4).

Fertilized and ambient soil cores were not statistically different at a given O\textsubscript{2} concentration. We found a predictable pattern of lower CO\textsubscript{2} fluxes at lower O\textsubscript{2} concentrations.
concentrations (Figure 4.4b), but no differences in CO$_2$ fluxes between ambient and fertilized soils at a given O$_2$ concentration. Soil moisture was approximately 20% at both sampling intervals.

When we saturated field fertilized soil cores (to simulate soil conditions after a major storm event), we found tremendously high N$_2$ fluxes at 20% and 2% soil moisture.
O\textsubscript{2} (Figure 4.5a). N\textsubscript{2} fluxes in saturated soils were 4039 ± 723 µg N/m\textsuperscript{2}/hr and 2959 ± 254 µg N/m\textsuperscript{2}/hr at 20% and 2% O\textsubscript{2}, respectively, compared to 53.3 ± 30 µg N/m\textsuperscript{2}/hr and 243 ± 98 µg N/m\textsuperscript{2}/hr in field moist soils across the 24 hr and 9 day time intervals.

Field measured fluxes of N\textsubscript{2}O and CO\textsubscript{2} showed no significant differences 24 hrs after fertilization (Figures 4.6a and 4.6b), but after 9 days fluxes had increased significantly compared to pre-fertilization rates (2.00 ± 0.62 µg N/m\textsuperscript{2}/hr and 43.4 ± 5.5 mg C/m\textsuperscript{2}/hr versus 0.07 ± 0.33 µg N/m\textsuperscript{2}/hr and 23.0 ± 3.2 mg C/m\textsuperscript{2}/hr).

4.4 Discussion

At the outset of this study we hypothesized that 1) soil O\textsubscript{2}, moisture, and available NO\textsubscript{3}\textsuperscript{−} were the most important controls on denitrification in residential lawns; 2) denitrification would vary temporally based on these controls; and 3) N\textsubscript{2} flux rates would be low most of the time with large pulses following fertilizer addition and precipitation events. While our results generally confirm these hypotheses, the threshold levels of soil O\textsubscript{2}, moisture, and NO\textsubscript{3}\textsuperscript{−} availability required to see significantly rates of denitrification in lawns (relative to N inputs) were greater than expected. Soil oxygen concentrations were less dynamic (Figures 4.1a and 4.1b) and N\textsubscript{2}O fluxes even more dynamic than expected (Figure 4.7).

Soil moisture has been linked to soil O\textsubscript{2} in a number of studies (Sexstone et al. 1985, Burgin et al. 2010), so we expected to measure significant drops in soil O\textsubscript{2} concentrations following large storm events; however this was not the case.
Figure 4.5: N\textsubscript{2} and CO\textsubscript{2} fluxes (A and B) at 20% and 2% oxygen for lawn soil cores collected 9 days after fertilization in the field, then saturated with water in the laboratory. Different letters indicate significant differences across treatments (p < 0.05, n = 8).
Figure 4.6: \( N_2 \) and \( CO_2 \) fluxes (A and B) from in-situ gas sampling chambers in each of the lawn plots. Different letters indicate significant differences across treatments \( (p < 0.05, n = 6) \).
Figure 4.7: Laboratory measured N$_2$O fluxes for each sampling interval (0 - 2 hrs, 2 - 6 hrs, and 6 - 24 hrs) for individual fertilized, saturated soil cores.

Soil O$_2$ concentrations (measured hourly by three diffusion-head O$_2$ sensors at each site) never dropped much below ambient atmospheric concentrations over our study period, which included several major storm events (Figures 4.1a and 4.1b). One storm system brought more than 10 cm of rainfall over a three day period (NCDC 2009), yet soil O$_2$ concentrations did not change significantly at the sites. These findings suggest that the soils were well-aerated and well-drained. A lack of redoxymorphic features in these soils further supports this
conclusion.

We expected denitrification rates to be low when soil O$_2$ concentrations were high, but this was not always the case. Under wet or saturated conditions, soils amended with fertilizer or NO$_3^-$ had high N$_2$ fluxes, even when soil O$_2$ concentrations were at 20% ($4039 \pm 723 \, \mu g \, N/m^2/hr$, Figure 4.5a). These findings make it clear that high soil O$_2$ concentrations, at least when measured at the macro-scale, do not necessarily indicate low denitrification potential. The inverse, however, may still hold true. For instance, Burgin and colleagues (2010) found that macro-scale soil O$_2$ strongly controlled denitrification rates in a wet riparian site with dynamic O$_2$ concentrations. This was not true of the dry riparian site, where bulk soil O$_2$ concentrations stayed relatively high. Since denitrification is primarily a microbial process that requires anoxic conditions (Kulkarni et al. 2008), our findings suggest that wet lawns soils contain enough anoxic microsites to carry out rapid denitrification when NO$_3^-$ is available, even when macro-scale soil O$_2$ concentrations remain high.

Lawn soils, which tend to have high SOM, microbial biomass, and labile C and N, have correspondingly high rates of respiration (Qian and Follett 2002, Raciti et al. 2008, Qian et al. 2010, Figures 4.1b and 4.6b). Since respiration is the primary oxygen-consuming process in soils, we might expect anoxic microsites to form readily in lawn soils, especially under warm, moist conditions that favor high soil respiration. These results suggest that lawns can support extremely high denitrification rates, but only under particular circumstances, such as high rainfall events following fertilizer application. These findings also suggest that our soil O$_2$ sensors do not accurately represent O$_2$ concentrations at soil microsites. The N$_2$O fluxes we measured in the field suggest that lawns are not
a dominant source of N$_2$O emissions in our study system. When we fertilized part of the UMBC1 site, we expected to see an immediate and rapid increase in N$_2$O emissions (one of the intermediary gases in the denitrification process), but N$_2$O fluxes remained low after 24 hrs (Figure 4.6a). We saw significantly higher N$_2$O flux rates 9 days after fertilizer application, but these rates (less than 0.2 ng N/cm$^2$/hr) are well-below the mean rates measured in lawn and forest plots at BES LTER (Groffman et al. 2009), which calls into question whether the low, but significantly higher, N$_2$O fluxes are a result of the experimental treatment or merely a result of natural variation in the system.

Results from the N-FARM incubations suggest that the balance of N$_2$O production and consumption may be a key factor in controlling N$_2$O emissions from lawns (Figure 4.7). Fluxes from wet, fertilizer-amended soils support the hypothesis that soil moisture is a key factor controlling N$_2$O fluxes in lawn soils (Groffman et al. 2009), but these results also reveal a more complicated relationship between N$_2$O production and consumption (Figure 4.7). When fertilized soils were saturated with water (or when a NO3 solution was added to wet soils), we saw extraordinarily high rates of N$_2$O production for the first few hours of the experiment, followed by rapid N$_2$O consumption. In some cases, virtually all of the accumulated N$_2$O had been consumed before the next sampling interval (4 or 18 hours later). These findings indicate a lag time between accelerated N$_2$O production and counter-balancing increases in N$_2$O consumption, which may have important implications for overall N$_2$O flux rates from fertilized soils. Soil properties and conditions that influence diffusion may cause some sites to have very high N$_2$O emissions, particularly after rainfall events or following fertilizer application. Thus, like studies in the western U.S. (Kaye et al. 2004, Hall et al. 2008), we cannot yet conclude that lawns are an insignificant
source of N$_2$O in the area.

Table 4.2: Nitrogen mass balance calculated for study lawns.

<table>
<thead>
<tr>
<th>N Inputs</th>
<th>Residential</th>
<th>Forest</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Atmospheric Dep (kg N/ha/yr)</td>
<td>11.2</td>
<td>11.2</td>
<td>Groffman et al. 2004</td>
</tr>
<tr>
<td>Fertilizer (kg N/ha/yr)</td>
<td>83.5</td>
<td>0</td>
<td>Law et al. 2004</td>
</tr>
</tbody>
</table>

| Hydrologic N Losses                  |             |        |                                             |
| Leaching (kg N/ha/yr)                | 14.1        | 4.4    | Groffman et al. 2009 (4 yr average)        |

| Gaseous N Losses                     |             |        |                                             |
| N$_2$O (kg N/ha/yr)                  | < 1         |        | Groffman et al. 2010                       |
| Denitrification as N$_2$ (kg N/ha/yr)| 14.0        | N/A    | This study combined with long-term soil     |
|                                       |             |        | moisture records at Baltimore Ecosystem   |
|                                       |             |        | Study (BESLTER.org).                      |

| N Sequestration                      |             |        |                                             |
| N accumulation in soils (kg N/ha/yr)| 83.0        | N/A    | Residential sites with agricultural land-  |
|                                       |             |        | use history (Raciti et al. accepted).      |

4.4.1 Closing the Nitrogen Budget

Using a combination of laboratory and field measurements, we calculated an annual denitrification rate of 14 ± 3.6 kg N/ha/yr for the lawns in this study (see Calculations and Statistics and Table 4.1). Despite the significant uncertainty in this estimate, our calculations suggest that denitrification is an important part of the N budget for residential lawns (Table 4.2). The N$_2$ flux we calculated was comparable in magnitude to atmospheric N deposition (11.2 kg N/ha/yr), NO$_3^-$ leaching losses (14.1 kg N/ha/yr) or approximately 15% of fertilizer N inputs (83.5 kg N/ha/yr). This rate is also within the range of estimates obtained by Horgan and colleagues using a $^{15}$N mass balance approach, though a significant fraction of the $^{15}$N tracer used in those studies could not be accounted for at the end of the experiments (2002, 2002b). While the annual denitrification rate
calculated in this study was significant, there is evidence that soils may be an even larger N sink in residential landscapes, particularly for sites with a history of agricultural land-use prior to development (Chapter 2). Our results suggest that denitrification is an important means of removing reactive N from the residential landscape. However, further work is required before these findings can be generalized to a wider range of residential lawns and soils.
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