Impact of urbanization on biogeochemical cycling in western Florida

by

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Abstract

Many city residents may benefit from the ecosystem services that urban forests provide. An important supporting process to ecosystem services is nutrient cycling. Very few studies examining the impacts of urbanization on nutrient cycling and soil carbon storage have taken place in the coastal regions of the southeastern United States, one of the fastest growing regions in the United States. The focus of this study was to investigate the impacts of urbanization on soil biogeochemical cycling in western Florida (panhandle) across six land covers. The land covers in this study include rural slash pine (*Pinus ellottii*) plantations, rural naturally regenerating forests (slash pine or mixed oak dominated), urban forest fragments (slash pine or mixed oak dominated) and urban lawns.

Soil organic carbon content in urban forests has mean values of 24.5 kg cm^{-2} in the 0 to 90 cm mineral soil and both increases and remains unchanged from that found for rural forests depending on the dominant overstory vegetation type. Urbanization appears to stimulate soil microbial biomass and activity (potential carbon mineralization rates) in the forest sites. Urban forest pine dominated sites are found to have 2.5 times as much N mineralized on a daily basis as natural forest pine sites and urban forest oak sites are found to have 1.9 times as much N mineralized as natural forest oak sites based on the mean daily rates averaged over the two year study. Urban forest sites and urban lawns sites in the study area are not significantly different in their soil carbon and nitrogen contents in the mineral soil (0 to 90 cm), microbial biomass carbon and

nitrogen contents, potential carbon mineralization rates, and potential net total N mineralization rates. Potential net nitrification rates in unfertilized lawns in this study are significantly higher than those in urban forests, with nitrification composing the entire potential net N mineralized in urban lawn soils.

Urban sites are on average warmer by 0.63 ^{o}C ($p=0.0101$) than rural sites in the winter and tend to have a more narrow temperature range than the rural forested sites. These alterations of the forest floor temperature is not reflected in different mass loss or nutrient release patterns in decomposing litter among contrasted land covers (urban forest oak versus natural forest oak and urban forest pine versus natural forest pine). Foliar mass loss was measured over an 82 week period that was characterized by periods of extended drought, which may have masked an urbanization effect.

Oak and pine dominated stands do have distinct differences in biogeochemical cycling. This may suggest that future urbanization studies in this region need to account for the overstory composition of a forest stand, as this may reflect differences in stand origin that then may impact biogeochemical cycling and the overall effect of urbanization.

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1. Introduction

With half of the world's population living in urban areas, there is intense interest in the scientific community about the role of urban green space in providing ecosystem services to city residents. The societal benefits of an ecosystem can be categorized into provisioning services, cultural services and regulating services and a supporting ecosystem process to all three of these services is nutrient cycling (MA 2003). The focus of this research is to gain insight into how urbanization impacts carbon (C) and nitrogen (N) cycling. Our approach is to study the indirect effects of urbanization by comparing unmanaged forest fragments within the urban matrix (i.e. urban forests) to nearby rural forests and to study the direct effects of urbanization by comparing urban forests to urban lawns. In the literature, the term "urban forest" can also include managed forests (street trees, trees along buildings, trees in lawns and parks) in addition to unmanaged forest patches contained within the urban matrix. Unmanaged urban forests can further be defined as remnant (never been cleared for urban use) or emergent (have been cleared for urban use, abandoned, and then reverted to a tree-covered site) (Zipperer, 2002). Within this introduction, the value and costs of urban green space is overviewed. A brief introduction of the direct and indirect effects of urbanization on soils is also provided. Specific objectives and an outline of the research are provided at the end of the introduction.

Ecosystem services and disservices of urban green space

Commonly cited benefits of urban trees are improvement of air quality, reduction of noise pollution, reduction in building heating and cooling costs, sequestration of C, and provision of habitat for wildlife (see reviews by Nowak et al. 2010; Escobedo et al., 2011). Furthermore, research using hedonic valuation of urban forests and green spaces in North America and Europe suggests that property values of homes increase with decreasing distance to urban green areas (Tyrvainen, 1997; Morancho, 2003; Conway et al., 2010). Tree cover has been found to increase the property value of homes (Des Rosiers et al., 2002; Donovan and Butry, 2010), however above average density of vegetation close to the home caused housing prices to decrease (Des Rosiers et al., 2002).

In the eastern and southeastern United States, it would seem intuitive that trees in an urban environment would be a valuable resource and increase resident's quality of life. Public-private programs such as the MillionTreesNYC (New York City Parks Department and New York City Restoration Project) and TreeCityUSA (Arbor Day Foundation) are based upon this reasoning. However, ecosystem disservices (or costs) are also associated with urban forests. Escobedo et al. (2011) provides examples of the disservices of urban forests, such as financial costs (i.e. tree pruning, removal and replacement), social nuisances (i.e. production of allergens, rufgia for vector-spread diseases), and environmental costs (i.e. alteration of nutrient cycles, introduction of invasive species). Relationships between property values and proximity to green space may depend on how safe individuals feel in that green space, especially in areas where the green space is unmanaged or overgrown.

Recent research has stressed a lack of empirical evidence in regards to urban forests reducing greenhouse-gas emissions and air pollution concentrations (Pataki et al., 2011). In Florida, urban forests offset 3.4 percent and 1.8 percent of the total carbon dioxide emissions in Gainesville and Miami-Dade County, respectively (Escobedo et al., 2010). Urban forests in this study were reported as being moderately effective at sequestering carbon dioxide relative to other strategies. Interestingly, an invasive, exotic tree was responsible for over 30 percent of the net C sequestered in the Miami-Dade County. As for air pollution, the impact of trees on pollution reduction is complicated as tree species, tree cover, location of the tree and precipitation patterns all impact how effective urban forests are at air pollution removal (Escobedo, 2007). In some instances, urban trees may increase air pollution as trees can produce volatile organic compounds (Chameides et al., 1988) and street trees can trap pollutants under the tree crown (Gromke, 2011).

Although it is impossible to generalize the benefits and costs of green space across all cities in the United States, the possibility exists that the social benefits of urban forests may outweigh their environmental and economic costs. A review of human health case studies reveals that the positive contributions of urban green space to human health and well-being are numerous and, in particular, can make a difference in the behavior of children (Tzoulas et al., 2007). One thing that may be true: when green space is gone, many city residents miss it. An example of this can be found in select areas within the New York City metropolitan area, where former sidewalks now contain picnic tables sitting atop concrete that is painted green.

The cultural benefits of green space include green space as forested areas and as areas covered by turfgrass. Turfgrass land cover (residential, commercial, and institutional lawns, parks, golf courses and athletic fields) accounts for an estimated 1.9 % of the total continental United States area (Milesi et al., 2005), whereas urban lands account for an estimated 3.5 to 4.9% (Nowak et al., 2001). Soil C storage in turfgrass ecosystems, which is a regulating ecosystem benefit, must be balanced with the C costs of turf maintenance from mowing, fertilization, and irrigation. Perhaps the greatest concern that arises from the conversion of forested land to turfgrass is alterations to the N cycle. Urban lawns in Baltimore, Maryland, have been found to lose more N from the soil via nitrate leaching compared to urban forests (Groffman et al., 2009). A worst case scenario would involve this contributing to nitrate pollution in stream and ground water.

Direct and indirect effects of urbanization

Soils are important to ecosystems in that they sustain plant and animal life by providing physical support for plant growth, cycling nutrients, storing significant amounts of C, and regulating water. Activities associated with urbanization can drastically modify soils through removal of the topsoil, additions of fill material, clearing of native vegetation and sealing of the soil surface (Pouyat et al., 2003). Another direct effect of urbanization is in post-development management practices of urban land cover, such as the mowing, irrigation, and fertilization of lawns. Soils and vegetation communities in an urban matrix can remain physically undisturbed. Hence, indirect effects of urban land use change involve changes in the abiotic and biotic environment, such as changes in climate, pollution inputs, and exotic species presence (Pouyat et al., 2003). Little research exists on how both the direct and indirect effects of urbanization impact key ecosystem process, such as aboveground net primary productivity, soil C content and foliar litter decomposition, plant available N and soil nitrate production.

Research purpose and objectives

Very few studies examining the impacts of urbanization on the aforementioned ecosystem process have taken place in the coastal regions of the southeastern United States. Compared to other regions in the country, the southeast has a large percentage of urban land (7.5 % of the region's area or 4.6 million hectares) and the greatest rate of urban growth from 1990 to 2000 (an increase of 1.1 million hectares) (Nowak et al., 2010). This study takes place within and near Apalachicola, Florida, which is in an area with a low population density (i.e. an urban cluster). The effect of urbanization is often determined after major development has occurred and, with this study, we have an opportunity to establish baseline data as well as study how these data change as this coastal area transitions into a post-developed state. This investigation meets the need for new research in underrepresented environments.

Previous work by Nagy et al. (2013) determined the five most prominent land use, land covers centered on Apalachicola and Eastpoint, along the mainland coastline (to 10 km inland) in Franklin County. Those five prominent land covers included forested wetlands, natural regenerated forests (hereafter "natural forests"), pine plantations, urban lawns, and urban forests. Urban lawns and forests are the only urban land covers; the other three land covers are rural. This study focused on the land covers of natural forest, slash pine (*Pinus elliotti* Engelmann) plantation, urban forest and urban lawn. In order to minimize the variation within a land cover designation, all forested sites and urban lawn sites are located on soils with similar parent material, sites in the natural and urban forest land covers are further categorized by their dominant overstory vegetation (oak or pine), and all sites within a land cover span across the same range in soil drainage classifications.

The primary objectives of this study are to gain insight into how urbanization:

1. indirectly alters soil organic carbon (SOC) content via conversion from rural (natural forest and pine plantation) to urban forest cover and directly alters SOC content through conversion of forests into lawns;
2. impacts the decomposition of mixed species foliar litter, the nutrient release patterns in decomposing litter, foliar litter quality, and forest floor temperatures in urban forests compared to natural forests and pine plantations; and
3. indirectly alters plant available N and nitrification rates in surface soil via conversion from rural to urban forest cover and directly alters plant available N and nitrification rates through conversion of forests into lawns.

General outline

The following research is organized into three major topics: SOC storage, flux of C and nutrients from foliar litterfall and decomposition, and plant available N in surface soils. Chapter 2 addresses the question of whether the conversion of natural forested ecosystems to urban forests and of urban forests to lawns will affect overall soil C pools. Soil organic carbon content is partly dependent on site productivity and litterfall inputs to the soil surface. Furthermore, N and P regulate C storage to some degree through limitation of primary productivity. Therefore, aboveground net primary productivity (NPP), aboveground standing crop biomass, N and phosphorus (P) content of foliar litterfall, and soil N content were measured in order to explain differences or similarities between land covers. Urbanization studies focusing on SOC storage are also reviewed in this chapter and the C storage values found in this research are put into context with results from other urbanization studies.

How urbanization affects foliar litter decomposition rates and nutrient release patterns in decaying litter compared to rural forests is the focus of Chapter 3. Soils of the southeastern coastal plain are sandy and nutrient poor and an accumulation of foliar litterfall on the forest floor could immobilize essential nutrients. For example, the immobilization of nutrients in the forest floor is suggested as a primary reason for the cessation of vegetative biomass production at approximately 26 years of age within slash pine plantations in northern Florida (Gholz et al., 1985). Within the literature there is great uncertainty as to how urbanization impacts decomposition rates in forested areas. The uncertainty associated with the effects of urbanization on decay rates is likely due in part to the large variety of anthropogenic disturbances associated with urbanization that exist within and between cities. Furthermore, these changes associated with urbanization may have opposing effects on decomposition processes. Increased temperatures in urban forests (urban heat island effect) (Pouyat and Carreiro, 2003) and decreased foliar litter quality (Carreiro et al., 1999) have been postulated to influence decay rates in urban forests compared to rural forests. For this research, forest floor temperatures and initial litter quality parameters are measured in order to understand the mechanisms behind differences in decay and nutrient cycling across the urban and rural sites.

One of the most important microbial mediated processes in surface soils is N mineralization and immobilization, as most of soil N is stored in organic matter and unavailable to plants. There is a concern that alterations to the soil abiotic and biotic environment (i.e. higher temperatures, increased carbon dioxide, increased presence of exotic species, N deposition) from urbanization may alter the N conserving mechanisms of microbes and N mineralization rates in these soils. Chapter 4 describes the quantification of actual and potential net N mineralization rates across all land covers in the study. Microbial biomass was also measured for the land covers. As microbial biomass C and N can be sensitive to management and environmental change, we expected alterations in net N mineralization and nitrification to also be reflected in these microbial properties.

Chapter 5 is a summary and conclusion to the preceding chapters. Direction for future research will also be discussed. Chapter 6 is the Appendix and contains 11 tables. Appendix A through G provides additional data for Chapter 2. Appendix H, I and J provides additional information for Chapter 3.

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2. Impacts of urbanization on soil organic carbon and nitrogen storage in coastal land covers in western Florida

Abstract

As natural forests in the southeastern United States are converted into urban forests and lawns, there is a need to understand how soil organic carbon (SOC) content is altered. Many challenges to urban soil studies exist as SOC content can vary in a complicated manner with land cover patterns. Furthermore, there are very few studies that directly compare the soil carbon content of urban forest soils to that of their pre-urban state. In this study, we investigate the effects of urbanization on SOC content of sandy soils in the lower coastal plain of western Florida (panhandle), where parent material is fairly homogenous. We measured soil carbon and nitrogen contents to a depth of 90 cm in naturally regenerating forests, slash pine plantations, urban lawns and urban forests. Additionally, the effects of soil drainage and overstory plant community (pine versus oak) are also taken into account in this study. Soil organic carbon content in urban forests has mean values of 24.5 kg cm^{-2} in the 0 to 90 cm mineral soil and both increases and remains unchanged from that found for rural forests depending on the dominant overstory vegetation type. Oak dominated urban forests have higher SOC contents in the mineral (0 to 90 cm) soil and mineral soil plus forest floor than oak dominated natural forests. Pine dominated forests do not show a clear difference in SOC content between urban and rural (natural and pine plantation) forests. Both pine and oak dominated urban forested sites have higher aboveground net primary productivity rates than their natural forest counterparts. However, the higher

aboveground productivity in the urban forest pine dominated sites is largely from increased woody production and this does not appear to affect SOC contents. Whereas the trend of urban forest oak sites towards higher inputs of foliar C and N (on a kg m^{-2} yr^{-1} basis) into the forest floor may have promoted higher mineral soil C and N contents compared to the natural forest oak sites. Urban lawns had similar soil carbon and nitrogen contents in the mineral soil (0 to 90 cm) to urban forest soils. Soil drainage did not have a consistent meaningful effect on soil carbon content or aboveground net primary productivity.

Introduction

Very few studies examining the impacts of urbanization on soil organic carbon (SOC) have taken place in the southeastern United States. Compared to other regions in the country, the southeast has a large percentage of urban land (7.5 % of the region's area or 4.6 million hectares) and the greatest rate of urban growth from 1990 to 2000 (an increase of 1.1 million hectares) (Nowak et al., 2010). Recent estimates on carbon stored in urban and developed lands and the amount of carbon lost due to urbanization over a period of 62 years did not include a dataset originating from the southeastern United States (SUS) to test model predictions for SOC pools (Zhang et al., 2012). The authors state that "further model validation was limited by the lack of more field observational and experimental data in the SUS" (Zhang et al., 2012). Furthermore, there is a need for quantitative data on SOC stored in urban land covers (lawn and forests) to improve carbon storage estimates at a city and regional level. For instance, Pouyat et al. (2006) used SOC estimates for timberland soils in the southeast to estimate soil carbon stored in urban forests in this region. This paper is a primary reference for estimating carbon stored in soils of urban areas. Therefore, it is also worthy to investigate if SOC in urban forests increases, decreases or stays the same from that found for natural forests and pine plantations in the SUS.

Within the SUS, a study in western Florida found that unmanaged urban forest systems had similar SOC contents as naturally regenerating rural forests and pine plantations (Nagy et. al., 2013). Outside of the SUS, urban forests (intact forest fragments within the city) were also found to have similar SOC contents as rural forests along an urban to rural gradient in Puerto Rico (Cusack 2013), in the New York City metropolitan area (Pouyat et al., 2002) and in the Boston metropolitan area (Rao et al., 2013). In contrast, urban remnant forests had SOC contents 70% higher than those of rural forests in Baltimore, Maryland (Pouyat et al., 2009). Urban remnant forests are unmanaged urban forests that have never been cleared for urban use (Zipperer, 2002). As for differences among unmanaged urban forests and urban lawns, SOC contents were found to be similar in western Florida (Nagy et al., 2013) and in Baltimore (Pouyat et al., 2009). However in a separate study in Baltimore, urban lawns were found to have significantly higher SOC densities compared to urban remnant forests when both urban land covers occurred on the same soil series (Raciti et al., 2011).

Overall, in the contiguous United States and Puerto Rico, five studies have been found to compare rural forests and unmanaged urban forests and three studies that compare urban forest remnants to urban lawns. In a meta-analysis of the effects of land cover (including pasture, plantation, native forest, secondary forest, crop) change on soil carbon stocks, 74 studies covering 537 observations were used (Guo and Gifford, 2002). The authors concluded that the quantity of available data was not sufficiently large enough and, as a result, the readers could only formulate working hypotheses for future investigations based on the results of the meta-analysis. The study of how urban carbon stocks differ from rural forest carbon stocks is in its infancy and this is one of several challenges facing this area of research.

Other challenges involve the variety of experimental designs employed to study the effects on SOC of converting rural forest land into urban forest land cover and urban forest to urban lawn. First is the lack of a common depth of soil sampled among the studies. The synthesis of urban research mentioned previously only considered differences found in the total depth of soil sampled, which could range from the top 10 cm of soil to the top 100 cm of soil with and without forest floor carbon. For example, Pouyat et al. (2002) found no differences in SOC content in urban and rural forests for the upper 10 cm of mineral soil plus forest floor. However, when the forest floor contribution was removed from the analysis, urban forests were found to have significantly higher SOC contents in just the 0 to 10 cm mineral soil depth compared to the same depth in the rural forests.

Second is the use of unplanned comparisons in post-hoc analyses. In the Florida study mentioned above, urban land covers (forest and lawn) were compared to three other land covers common to the region (Nagy et al., 2013). Included in these land covers were wetland forests, whose high SOC content dominated the unplanned comparisons between land covers. Wetland soils also dominated the differences between the SOC concentrations of eight different land covers commonly found in central Florida (Ahn et al., 2009). As wetland land covers may not represent a pre-urban land cover, planned comparisons between urban land covers and its pre-urban state may increase the statistical power such that more subtle, yet environmentally significant, differences due to urbanization can be observed.

The next set of challenges involves variables that are sometimes out of the control of the researcher. Hence, the third challenge involves how soil variation can be confounded with land cover patterns (see Groffman et al., 2006). In Puerto Rico, urban remnant forest soils were located at low elevations (20 – 160 meters above sea level), where rural forests were located at high elevations (260 and 640 meters above sea level) (Cusak 2013). In Baltimore, high fertility soils were found in urban areas and low fertility soils tended to be located in the rural fringe (Groffman et al., 2006). As a result, the authors found that the inherent fertility of a soil impacted soil properties, such as nitrogen mineralization rates, more than land cover change. Lastly, time since land conversion and the previous land cover condition (i.e. forest or agricultural land) can determine C and N accumulation rates in urban lawns (Raciti et al., 2011), which could increase the variation in carbon densities found between sites in the urban lawn land cover category.

Differences in terminology regarding urban land cover types can also complicate urban studies. The term "urban forest" can include managed forests (street trees, trees along buildings, trees in lawns and parks) and unmanaged forest patches contained within the urban matrix. These unmanaged forest patches may have never been cleared for urban use (remnant urban forests) or have been cleared for urban use, abandoned, and then reverted to a tree-covered site (emergent urban forests) (Zipperer, 2002). The term "urban lawn" can also include a range of management strategies and plant species. Several papers were not included in the aforementioned synthesis because their definition of urban land covers was too vague, such that the reader could not determine if the urban land cover had a forest, turfgrass, or no vegetation cover. In the context of this study, urban forests are remnant or emergent urban forests that have complete canopy closure. Urban lawns are dominated by a mix of turfgrass and broadleaf weedy species and are maintained through regular mowing (Zipperer et al., 1997).

The purpose of this research is to build upon the investigations of Nagy et al. (2013) to gain a better understanding of how urbanization indirectly alters SOC content via conversion from rural to urban forest cover and directly alters SOC content through conversion of forests into lawns along the mainland coast of western Florida. Based on trends in their data, we hypothesize that urban forests will have similar SOC contents to urban lawns and will have higher SOC contents than rural forests (naturally regenerating forests and pine plantations) in both the surface mineral soil (0 to 7.5 cm) and in the 0 to 90 cm depth. In order to explain differences or similarities between land covers, we also measured aboveground net primary productivity (NPP), aboveground standing crop biomass, nitrogen (N) and phosphorus (P) content of foliar litterfall, and soil N content. Soil organic carbon is partly dependent on site productivity and litterfall inputs to the soil surface. Furthermore, N and P regulate C storage to some degree through limitation of primary productivity.

To address some of the concerns raised from previous urbanization studies, we compare SOC among land covers at specific depths and for the entire mineral surface soil (0 to 90 cm) and mineral soil (0 to 90 cm) plus forest floor. Urban sites are located within city limits and rural sites are located within 30 km of their urban counterparts. Planned comparisons are used to specifically focus on indirect (urban forest versus rural forest) and direct effects (urban forest versus urban lawn) of urbanization. The direct effects of urbanization involve changes in management or actual disturbance of the land cover, whereas the indirect effects of urbanization involve an alteration of the abiotic (i.e. temperature and pollution levels) and biotic (i.e. invasive species presence) environment with land cover change (Pouyat et al., 2003). The soil parent material is similar across the rural and urban sites. To minimize the high natural variation in SOC found in the region, we further define the forested land covers by dominant overstory vegetation (pine or oak). In addition, sites within a land cover include similar ranges of drainage classifications based on field observations of depth to seasonal high water table.

Methods

Study site description

The study was conducted within and outside the city limits of Apalachicola and Eastpoint, Franklin County, Florida, from 2010 to 2012. In 2010, Apalachicola and Eastpoint had populations of 2,242 and 2,337 people, respectively (US Census Bureau, 2010). Apalachicola covers an area of 4.87 km2 and has a population density of 460.5 people km^{-2}. Eastpoint is 19.0 km2 in size and has a population density of 123.1 people km^{-2}. Apalachicola was considered an urban cluster by the US Census Bureau in 2010 (US Census Bureau, 2010). Watersheds in the Apalachicola and Eastpoint area have impervious surface percentages that do not exceed 15 % of the watershed land area

(Nagy et. al., 2012). The study area is located in a subtropical climate, with average summer temperatures of 27.2 ^{o}C and average winter temperatures of 12.4 ^{o}C. The study area typically receives 143.5 cm of precipitation per year, with the majority of the precipitation occurring in July, August and September (National Climatic Data Center, 2012). For the study period, precipitation data was obtained for the study area from the National Climate Data Center Station at the Apalachicola Airport, Apalachicola, Florida.

Previous work by Nagy et al. (2013) determined the five most prominent land use, land covers centered on Apalachicola and Eastpoint, along the mainland coastline (to 10 km inland) in Franklin County. Those five prominent land covers included forested wetlands, natural regenerated forests (hereafter “natural forests”), pine plantations, urban lawns, and urban forests. Urban lawns and forests are the only urban land covers; the other three land covers are rural. This study focused on the land covers of natural forest, pine plantation, urban forest and urban lawn. Forested wetlands are not included in this study because they are not a common pre-urban land cover in this area. For this study, urban forest sites are urban remnant or regenerated forests on vacant lots within city limits and both urban and natural forest sites are further defined as oak dominated or pine dominated forest sites.

To identify potential sites for this study, digital orthoimagery quarter quandrangles (DOQQs) with 1 m spatial resolution were used (Florida Department of Environmental Protection, 2004) along with the Franklin County digital soil survey (Soil Survey Staff, 2008). Potential forested sites were identified as areas at least 0.16 ha in size and had canopy closure. The soil data layer was used to identify soil drainage classes (based on the soil series information), which gave indication to the vegetation community at that site. All potential sites from nine km east and 30 km west of Apalachicola (which includes Eastpoint) and five km from the coastline were explored. Landowner consent narrowed the pool of potential sites. Furthermore, rural forested sites were selected from potential sites based on the additional criteria that the site had not been burned in the last five years and would not be burned in the next three years and the soil and vegetation had not been recently disturbed or harvested. Sites within each land cover that rated the highest in regard to employee safety at the site and accessibility during all times of the year were selected for study.

Seventeen circular plots (0.04 ha) were placed in the study area and include urban forest oak dominated (n=4), urban forest pine dominated (n=3), natural forest oak dominated (n=4), natural forest pine dominated (n=3) and pine plantation (n=3) land covers. Forested sites were established in July 2010. Five additional sites were added in June 2012 to incorporate the urban lawn land cover into the study. Urban lawns were only measured for soil carbon and nitrogen content. The urban lawns were at least 0.04 ha, contained at most 6.3 stems ha^{-1}, and had not undergone recent soil disturbance. These sites were not irrigated, fertilized or limed, but were mowed approximately once every month. After traversing each site, plot center was positioned such that each 0.04 ha plot would be located within a soil that had similar parent material, vegetation, drainage class and topography.

The distribution of soil types and plant communities in the study area are highly complex as slight changes in ground elevation result in different soil drainage classes and different plant communities (Table 1.1, Appendix A). Within the oak land covers and urban lawn land covers, sites span a range of soil drainage classifications from very poorly drained to moderately well drained. For the pine land covers, sites range from very poorly drained to somewhat poorly drained. Pine dominated vegetation communities with closed canopies could not be found on moderately well drained sites. All of the soils in the study area are derived from sandy marine sediments (Sasser et al., 1994). Drainage classes and soil series were determined using the current soil survey for Franklin County (Soil Survey Staff, 2008) and onsite verification. Very poorly and/or poorly drained soils include Leon (sandy, siliceous, thermic Aeric Alaquods), Rutlege (sandy, siliceous, thermic Typic Humaquepts) and Scranton (siliceous, thermic, Humaqueptic Psammaquents) series. Somewhat poorly drained soils are Mandarin (sandy, siliceous, thermic Oxyaquic Alorthods) series. Moderately well drained soils are Resota (thermic, coated Spodic Quartzipsamments) series.

The majority of urbanization in Franklin County, excluding the barrier islands, has occurred in the pine or hardwood dominated forests along the mainland coastline. Natural plant communities in the study area include wet and mesic flatwoods, mesic and xeric hammocks, and sandhill communities (Florida Natural Areas Inventory, 1990). Common overstory trees on the wetter sites include slash pine (*Pinus elliotti* Engelmann), southern magnolia (*Magnolia grandiflora* Linnaeus), sweetbay magnolia (*M. virginiana* Linnaeus), swamp laurel oak (*Quercus laurifolia* Michaux) and water oak (*Q. nigra* Linnaeus). Typically slash pine and live oak (*Q. virginiana* Miller) are found on the mesic sites, whereas the drier sites (xeric hammocks and sandhill communities) can have live oak, sand live oak (*Q. geminata* Small), laurel oak (*Q. hemisphaerica* Bartram ex Willdenow), turkey oak (*Q. laevis* Walter), myrtle oak (*Q. myrtifolia* Willdenow), southern magnolia, slash pine, longleaf pine (*P. palustris* Miller) and shortleaf pine (*P.*

echinata Miller). Many of the urban forests within Apalachicola and Eastpoint are remnants of these natural plant communities. However, camphortree (*Cinnamomum camphora* (L.) J. Presl) is a non-native tree species observed within the urban forests and not within the natural forests and pine plantations observed in this study. Pine plantation sites are dominated by slash pine and were likely flatwood sites before being converted to a plantation.

Aerial photographs of the study area from 1953 to 2004 were used to track land cover changes for all sites (Florida Department of Environmental Protection, 2004; Florida Department of Transportation, 2013; US Department of Agriculture, 2013). Aerial photographs from 2004 reflected the condition of each site during the study period from 2010 to 2012. Aerial photographs of the study area from 1953, 1969, 1984 and 2004 indicate that the urban lawns sites have been managed as a lawn in an urban setting since 1953. Six of the seven urban forest sites are remnant forest sites. The somewhat poorly drained urban forest oak dominated site had a small house on the site in 1953 and by 1969 the house was removed and trees were growing on site. This site had complete canopy closure by 1984 and no soil disturbance was observed in 2010. The remaining six urban forest sites appeared to be undisturbed regenerating forest in an urban setting in 1953. The majority of these six sites had an open forest canopy and a scrubby midstory that progressed toward a closed canopy by 1984 (Appendix B). All of the natural forest sites appear to be naturally regenerating forest with a closed canopy in 1953. Aerial photos in 1969 and 1973 of the study area indicate little change in the vegetation cover for these natural forest sites.

All three pine plantation sites appear to have unmanaged forest or woodland cover in 1953. Aerial photographs indicate that the very poorly drained and poorly drained pine plantation sites remained under this cover type in 1969 and in 1973. These two sites were planted with slash pine in 1984 (J. Pitts, personal communication, 2010). Trees growing in the somewhat poorly drained pine plantation site in 2012 have 28 rings at breast height. Aerial photographs indicated that this site was under pine plantation cover in 1984.

Soil and forest floor sampling and analysis

Two soil and forest floor sub-samples were collected from each of the 17 forested sites and the five urban lawn sites during the summer of 2012. Sampling locations for forest floor and soil samples were within a 10 m radius from the plot center within each site and were chosen in areas that best represented the characteristic topography and vegetation of that site. A bucket auger was used to collect mineral soil material from two sampling locations within each circular plot or site. The mineral soil in each plot was sampled at four depths (0-7.5 cm, 7.5-30 cm, 30-60 cm, 60-90 cm). Mineral soil samples were air dried and sieved through a 2-mm sieve. Sieved samples were analyzed for total C and N concentrations using dry combustion on a LECO 2000 CNS analyzer (LECO Corporation, 3000 Lakeview Ave., St. Joseph, MI). Sieved soil samples were also measured for pH in a 1:1 soil to water suspension. Soil bulk density was calculated using the methods of Black and Hartge (1986). An intact soil core of a known volume was taken at the surface (0-7.5 cm) and at approximately 19 cm, 45 cm and 75 cm to estimate the bulk density of the soil within each depth (0-7.5 cm, 7.5-30 cm, 30-60 cm, 60-90 cm). Soils were oven dried at 105^{o}C and live roots were removed from the soil material. Bulk density was calculated for the <2 mm mineral soil fraction by accounting for the weight and volume of any coarse fragments. Coarse fragments did not exceed 5 % of the volume of the bulk density samples. Bulk density and the depth of each sampling were used to convert C and N concentrations (mg g^{-1}) to C and N contents (kg m^{-2}).

Two forest floor samples were also collected at the 17 forested plots by removing all dead organic material within a 0.1 m^2 frame to the mineral soil surface. Forest floor depth was calculated for each sample by taking the average of four depth measurements. Each forest floor sample was oven dried at 65^{o}C until their mass did not change. The forest floor mass was recorded, so that forest floor carbon could be calculated on a content basis. A subsample was finely ground and analyzed for C and N concentrations using a LECO 2000 CNS analyzer.

Forest characteristics

A wedge prism was used to estimate basal area per hectare of each of the 17 forested circular plots. All stems within each of the forested plots (0.04 ha) that had diameter at breast height (DBH) greater than 10 cm were measured for DBH during the winter of 2011 and 2012. A circular sub-plot (0.0025 ha) located at the center of each plot was used to measure trees that had a DBH of 5 to 10 cm. Each stem was identified to genus and specific epithet. Standing crop biomass was calculated using DBH measurements from winter 2012 in allometric regression equations for total tree biomass (foliage, bark and wood, Appendix C). Diameter at breast height measurements from 2011 and 2012 were used in allometric equations to calculate aboveground NPP (see below). If equations did not exist for a species found in this study, an equation was found for a similar species or species group. Allometric regression equations developed by Swindel et al. (1979) for slash pine in Florida were used for pine species in this study. Allometric regression equations developed by Clark et al. (1985) for hardwood species in the Gulf and Atlantic coastal plains were used for overstory hardwoods in this study. Allometric equations developed by Brantley (2008) for privet (*Ligustrum sinese* Lour.) in the SUS were used for understory hardwood species in this study.

Forest productivity and foliar nutrient analysis

Aboveground net primary productivity was estimated for the forested sites as the difference between woody biomass in winter 2012 and winter 2011 plus average annual litterfall. Woody biomass was calculated using DBH measurements (stems > 5 cm) from winter 2011 and winter 2012 in allometric regression equations for woody (bark plus wood) biomass (Appendix D). Litterfall was collected from three 0.25 m2 traps in each site on a monthly basis over a two year period (September 2010 to August 2012). Traps had mesh screen bottoms that were approximately 30 cm above the ground. Monthly litterfall was oven dried at 65oC and separated by component (foliar, woody and reproductive) and the mass of the foliar litter component was then recorded. Monthly foliar litter production values (g m^{-2}) were estimated by dividing the oven-dried foliar mass by the area of the litterfall traps. Yearly foliar litter production was estimated by summing the monthly foliar litter masses for each year and then averaging the two years together for each plot.

Sub-samples from each monthly foliar litterfall collection were measured for leaf area using a Delta-T video imaging system (Delta-T Devices LTD, Burwell, Cambridge, England). Leaf area was divided by the oven dried mass of the sub-sample and then multiplied by the monthly foliar litterfall production. Leaf area index (LAI) was estimated by summing these monthly values (m^2 m^{-2}) for each year and then averaging the two years together for each plot.

A sub-sample from the monthly foliar litterfall collections over an 18 month collection period (April 2010 to August 2012) was taken and finely ground for C and N analyses (see section on forest floor analysis for methods). In addition, P was extracted from the finely ground foliar litterfall subsamples via dry ashing followed by an acid digestion using the vanadomolybdate procedure (Jackson, 1958). Extracts were then analyzed for P spectrophotometrically. Nitrogen and phosphorus concentrations were averaged across all monthly collections to calculate an average nutrient concentration (mg kg^{-1}) for each plot. Monthly C, N and P contents (g m^{-2}) were calculated by multiplying the C, N and P concentrations by the monthly foliar litterfall production rates. For each plot, C, N and P contents (g m^{-2} yr^{-1}) were estimated by averaging duplicate months and then summing the monthly content values of each element over a 12 month period.

Statistical analysis

All data were analyzed by SAS version 9.2 (SAS Institute, Cary, NC). Treatment effects on soil C, soil N, standing crop biomass, basal area, forest productivity, and foliage nutrient concentrations and content were tested using analysis of variance with land cover and soil drainage as the primary treatment variables. Contrast statements were used for the following planned comparisons for differences among the forest sites (a-e) and for differences among the forested and lawn sites (f):

a. Urban forest pine versus natural forest pine;
b. Urban forest oak versus natural forest oak;
c. Natural forest pine versus pine plantation;
d. Urban sites (oak + pine) versus rural sites (natural oak + natural pine + pine plantation);
e. Pine (urban +natural+plantation) versus oak (urban + natural); and
f. Urban forest (oak + pine) versus urban lawn.

Differences among means were considered statistically significant at $\alpha \leq 0.05$ and differences with $\alpha \leq 0.10$ were reported for informational purposes. The model assumptions of normality were not violated. In a few circumstances, the data was logarithmically transformed to meet the assumptions of equality of variance.

Results

Precipitation

All of the sites in the Florida Panhandle underwent a drought during the study period from July 2010 through May 2012 (Figure 2.1). Annual precipitation was 41.3 cm below the historical average from July 2010 through June 2011 and 29.9 cm below the historical average from July 2011 to May 2012. A tropical storm in June 2012 added 31.3 cm of precipitation to the study area over a 48 hour period. This single storm event brought the study area out of a deficit and into a surplus of 14.3 cm above the historical average for July 2011 to June 2012.

Forested site characteristics

Overall, the forested land covers are similar in regard to basal area and standing crop biomass (Table 2.2 and 2.3). Based on information from the aerial photographs, urban forests and natural forests are similar in age. Pine dominated sites have a higher stem (≥ 10 cm) density compared to oak dominated sites ($p=0.0048$). There is a trend of pine plantation sites having a higher stem density ($p=0.0791$), a smaller average stem DBH ($p=0.0048$) and a lower standing crop biomass ($p=0.0690$) compared to natural forest pine sites. The slash pine plantations measured in this study have comparable values for basal area, standing crop biomass, mean DBH, and stem density to other slash pine plantations of a similar age (17 to 34 years old) in Florida (see Appendix E, Gholz and Fisher, 1982, Harding and Jokela, 1994, Clark et al., 1999, Shan et al., 2001).

Per unit of land area, hardwoods compose on average 43 % of the natural forest pine stems, 36 % of the urban forest pine stems and 10 % of the pine plantation stems (Table 2.2). For the natural and urban forest oak dominated sites, hardwoods compose on average 90 and 80 % of the stem density of each forest site, respectively.

Camphortree (*C. camphora*) is a non-native tree to Florida and is present in three of the four urban forest oak sites and one of the three urban forest pine sites (Appendix A).

Both rural pine sites (pine plantation and natural forest) and urban pine sites maintain a dominant overstory of slash pine (*P. ellottii*). For most of the oak sites, the transition from a natural to urban site involves a shift in overstory species from a laurel oak (*Q. hemisphaerica, Q. laurifolia*) and live oak (*Q. virginiana, Q. geminata*) dominated overstory to a laurel oak (*Q. hemisphaerica, Q. laurifolia*) and water oak (*Q. nigra*) dominated overstory with minor components of slash pine (*P. ellottii*) and camphortree (*C. camphora*). The moderately drained urban and natural forest oak sites have similar overstory composition.

Forest productivity and foliar nutrients

Urban forests have higher aboveground NPP compared to natural forests (p=0.0116), even though aboveground standing crop biomass and stand age are similar between these land covers (Table 2.2 and 2.3). Both foliar (p=0.0463) and woody biomass (p=0.0116) productivity is higher in urban versus rural forests. It appears that a higher aboveground NPP in urban forests oak sites is largely due to a higher foliar productivity compared to natural forest oak sites (p=0.1004). Furthermore, a higher aboveground NPP in urban forest pine sites is largely due to a higher woody productivity compared to natural forest pine sites (p=0.0225). Leaf area index is also significantly higher in urban versus natural forest sites (p=0.0345).

The slash pine plantations measured in this study have similar values for aboveground NPP and foliar productivity to other slash pine plantations in Florida (see

Appendix E, Gholz and Fisher, 1982, Clark et al., 1999, Shan et al., 2001). Annual woody growth rates for trees within urban forest pine dominated sites (0.44 ± 0.06 cm yr^{-1}, mean ± standard error, n=81) and oak dominated sites (0.47 ± 0.05 cm yr^{-1}, n=86) in this study are similar to those found for trees in urban remnant forests (0.56 ± 0.10 cm yr^{-1}, mean ± standard error, n=398) in Gainesville, Florida (Lawrence et al., 2012). Camphortree has the highest annual woody growth rate among the urban trees in this study, with rates of 0.69 ± 0.20 cm yr^{-1} (mean ± standard error, n=12) in urban forest oak forests and rates of 0.99 ± 0.38 cm yr^{-1} (n=8) for urban forest pine forests (see Appendix F for the annual woody growth rates for common overstory tree species in all five land covers).

The concentrations of N and P in foliar litterfall are not significantly different between urban and rural forests (Table 2.4 and 2.5). The C:N ratios of foliar litter were trending towards lower values in urban pine dominated forests than rural pine dominated forests (p=0.0670). Litterfall nitrogen contents are significantly higher in urban than rural forests (p=0.0468). This is driven more by the change in N content between urban and rural oak forests (p=0.1353) than in the change between urban and rural pine forests (p=0.4802). Even though the N concentration was not different between urban and rural oak sites (p=0.9042), the somewhat higher foliar productivity (p=0.1004, Table 2.2 and 2.3) results in the N content of urban oak sites trending towards higher values compared to natural forest oak sites. No differences are observed between urban and rural land covers in regard to P content and concentration. Differences in C content reflect those observed for foliar productivity.

The seasonal pattern of foliar litterfall C, N and P contents on a monthly basis is shown in Figure 2.2. The peaks of higher inputs of N and P to oak dominated sites occur in concert with peaks in C input during the spring (March, April, May). Oaks in scrubby flatwoods in Florida have been found to drop their leaves, flower, and produce new leaves within several weeks in the spring (Abrahamson and Hartnett, 1990). Pine sites have a completely different seasonal pattern of leaf drop, where the majority of leaves fall from August through December. All of the pine sites have similar seasonal patterns or foliar litterfall productivity (g C m^{-2} $month^{-1}$). In contrast, the urban oak sites have their peak month in litterfall C content one or two months earlier than the natural oak sites (March versus April or May in 2011 and 2012). The large value for C, N and P in July 2012 for all land covers, except natural forest pine, may be due to a pulse of litterfall from a tropical storm that came through the study area on June 25 and 26 that dropped a total of 31.3 cm of rain in a 48 hour period.

Oak dominated sites have higher N and P concentrations and contents compared to pine dominated sites (Table 2.4 and 2.5). Among the 17 forested sites, there is a strong positive correlation between N and P content in the foliar litter ($p<0.0001$, Figure 2.3a), as pine and oak sites tend to have low and high N and P contents respectively. As nitrogen availability in a forested site increases, the N content in the litterfall increases (Birk and Vitousek, 1986). Therefore, the nitrogen and phosphorus content in litterfall likely reflects the availability of that nutrient at the forested site and, as the nutrient content of litterfall increases, site productivity should also increase. Foliar litterfall productivity increases by 90.5 ± 31.1 g m^{-2} yr^{-1} with every one g m^{-2} yr^{-1} increase in N ($p<0.0001$) and by 483 ± 369 g m^{-2} yr^{-1} with every one g m^{-2}

yr^{-1} increase in P (p=0.0142) content in litterfall (Figure 2.3b, 2.3c and Appendix G). This relationship between nutrient content in litterfall and foliar litterfall productivity is similar between oak and pine sites for N (p=0.3706) and for P (p=0.3436). Pine dominated systems have a higher foliar productivity at a given concentration of N in litterfall compared to oak dominated systems (p=0.0371). Pine dominated systems do not have higher litterfall production rates at a given level of P compared to oak dominated systems (p=0.4789).

Soil pH and bulk density

Urbanization, pine plantation development and vegetation cover type does not appear to impact soil bulk density in the surface 7.5 cm of soil (Table 2.6 and 2.7). Since no obvious differences are observed in the surface soil bulk density between land covers, it is uncertain what mechanisms are driving differences in bulk density at the 7.5 to 30 cm depth. The trend of higher mean bulk density values for the 7.5 to 30 cm depth of urban lawns compared to urban forests (p=0.0599) may be from a legacy effect of past land use. Certain land covers do have distinctly different soil pH. Urban forest oak sites are trending towards higher soil pH in the upper 7.5 cm of the soil profile compared to natural forest oak sites (Table 2.6 and 2.7). Additionally, oak dominated forest sites have higher soil pH than pine dominated forest soils in the 0 to 7.5 cm soil depth (p=0.0333) and are trending towards larger values in the 7.5 to 30 cm depth (p=0.0952). Urban lawns have significantly higher soil pH compared to urban forest soils throughout all soil depths measured. Depending on the depth, lawn soil pH is on average 1.9 to 2.2 units larger than urban forest soil pH. Nagy et al. (2013) also reported that the pH of urban lawn soil (0 – 90 cm) was on average 6.0 ± 0.3 compared to 3.7 ± 0.1 of slash pine plantations in the study area. Although lime additions were not recently applied to the lawn sites in this study, it is likely that historic use of lime in the past resulted in the lawn sites having higher soil pH values compared to the forest sites.

Soil organic carbon and nitrogen content

Urban forest oak dominated sites have higher SOC densities in the top 90 cm of mineral soil (p=0.0308) and mineral plus forest floor (p=0.0320) compared to the natural forest oak dominated sites (Table 2.8 and 2.9, Figure 2.4). This difference in total SOC content to 90 cm is driven by significantly higher contents in the 7.5 to 30 cm (p=0.0311) and 60 to 90 cm (p=0.0481) depths of the urban forest oak sites. Soil organic carbon is trending toward higher values in the 0 to 7.5 cm depth (p=0.1067) and the 30 to 60 cm depth (p=0.0754) of the urban forest oak sites compared to the natural forest oak sites. The results from a comparison of SOC contents of natural forest pine sites versus urban forest pine sites do not show an obvious pattern. Urban forest pine sites have higher SOC content than natural pine sites at the 30 to 60 cm depth (p=0.0582), but not at any other depth.

Carbon stored in the forest floor ranged from 9.8 to 26.7 % of the total soil carbon (forest floor plus 0 to 90 cm mineral soil) in each forest land cover. Samples were taken during mid-summer and therefore capture a value for forest floor carbon that is close to its minimum annual value. Pine sites are observed to have significantly higher carbon contents in the forest floor compared to oak dominated sites (p=0.0396, Table 2.8 and 2.9, Figure 2.4). Pine sites also have forest floors that are 3.88 ± 3.08 cm (95 %

confidence interval, p=0.0191) thicker and have a mass 4.16 ± 4.00 kg m^{-2} (p=0.0435) larger than oak dominated sites.

Similar to the soil organic carbon results, soil nitrogen content is significantly higher in all soil mineral depths for urban forest oak sites compared to natural forest oak sites (Table 2.10 and 2.11, Figure 2.5). There is a higher variability associated with nitrogen content in the urban forest sites, such that the data was log transformed for the 7.5 to 30 cm, the 30 to 60 cm and the 60 to 90 cm depths. The lawn sites are not currently fertilized, which may explain the similar nitrogen content in urban lawns to the other land covers.

Soil drainage classification

Out of 43 two-way ANOVA tests which examined differences among land covers for soil characteristics, forested site characteristics, foliar nutrient analyses, and aboveground biomass and productivity, only four tests indicate that drainage classification has a significant effect on the response variable. The results of the tests are that drainage classification has an effect on aboveground standing crop biomass (p=0.0418), forest floor carbon to nitrogen ratio (p=0.0252) and soil carbon content at the 30 to 60 cm depth (p=0.0173) and 60 to 90 cm depth (p=0.0393). Out of these four tests, only two showed differences among drainage classes following a Tukey's post hoc test. Very poorly drained sites have higher carbon content in the 30 to 60 cm soil depth and lower forest floor carbon to nitrogen ratios compared to somewhat poorly drained sites. In a study of 141 soils in central Florida, Vasques et al. (2010) found that there was no significant difference among SOC content of very poorly drained, poorly drained, somewhat poorly drained, moderately well drained and well drained soil classes.

Discussion

The SOC densities found in this study for pine plantations, natural forests and urban lawns are within the range of those found for similar soils or land covers (Table 2.12). The SOC values for urban forests in this study are among some of the highest found in the literature cited in Table 2.12. The standard error associated with those values for urban forests is also high compared to other studies. The high standard error is likely due to the small sample size within each land cover in this study. Pouyat et al. (2009) is included in Table 2.12 as it is the only other urban study in the eastern United States that has comparable depth measurements of SOC to this study for urban remnant forests and lawns.

In this study, SOC content in urban forests both increases or remains unchanged from that found for rural forests depending on the dominant overstory vegetation type. Oak dominated urban forests have higher SOC contents in the mineral (0 to 90 cm) soil and mineral soil plus forest floor than oak dominated natural forests. Pine dominated forests do not show a clear difference in SOC content between urban and rural (natural and pine plantation) forests.

Soil organic carbon content is in part dependent on NPP, which reflect additions of organic materials to the soil. As for aboveground NPP, both pine and oak dominated urban forested sites have higher rates than their natural forest counterparts. However, the higher aboveground productivity in the urban forested sites is largely from increased (not statistically significant, p=0.10) foliar production in the urban oak forest sites and higher woody production in the urban pine forest sites. Higher aboveground woody production in the urban forest pine sites compared to natural forest pine sites does not appear to affect SOC contents. However, the trend of urban forest oak sites towards higher inputs of foliar C and N (on a kg m^{-2} yr^{-1} basis) into the forest floor may have then promoted higher mineral soil C and N contents compared to the natural forest oak sites.

Previous work in western Florida found that urban forests have the largest aboveground NPP of all the land covers measured, however it was not significantly greater from that of nearby natural forests and pine plantation land covers (Nagy et al., 2013). The use of planned comparisons by this study versus unplanned comparisons by the previous work may have increased the power to distinguish between land covers.

Urban vegetation is often suggested as having higher NPP compared to native vegetation (Groffman et al., 2006; Byrne 2007, Lorenze and Lal, 2009) or, in general, has high productivity. Kaye et al. (2005) reported that irrigated and fertilized urban lawns have a much higher aboveground NPP compared to native shortgrass steppe grassland in Colorado. This study examined a direct effect of urbanization on aboveground NPP, where urbanization involved a change in land management (increased water and fertilizer additions) compared to the native system. The indirect effects of urbanization, such as higher carbon dioxide production, warmer temperatures (Ziska et al., 2004) and lower ozone concentrations (Gregg et al., 2003), have been found to have a positive impact on a single plant species' productivity in urban versus rural areas. The results of this paper then are relatively consistent with these studies in suggesting that the indirect effects of urbanization, which involve alterations in the abiotic environment, can improve urban plant productivity.

Oak and pine dominated stands have distinct differences in biogeochemical cycling. Oak dominated forest sites have higher foliar N and P contents entering the forest floor on a yearly basis, smaller forest floor C contents, and higher surface soil pH values compared to pine dominated forest sites. In studies of the effects of a single tree species on soil biogeochemical cycling, differences between forest stands dominated by a single tree species may not be due to the effect of tree species alone, but instead may be more related to variation in forest stand management, stand age, site history, and soil type (Binkley, 1995, Reich et al., 1997, Mueller et al., 2012). In regard to this urbanization study, it is important to account for the overstory composition of a forest stand as it reflects differences in stand origin that then impact biogeochemical cycling and the overall effect of urbanization. These differences in biogeochemical cycling may allow oak sites to respond to the indirect effects of urbanization through increases in SOC, whereas pine sites do not.

Oak dominated systems may be more sensitive or susceptible to the indirect effects of urbanization compared to pine dominated systems. For instance, the presence of camphortree is more apparent in the urban oak versus the urban pine dominated forests. Additionally, this sensitivity is observed in a shift in overstory species composition and different soil pH between urban forest oak sites and natural forest oak sites that are not seen between urban and rural pine dominated forested sites. The shift in overstory species may have caused an earlier leaf drop in the urban oak sites compared to the natural forest oak sites.

Urban lawns in this study have similar SOC and N contents in the upper 0 to 90 cm of mineral soil as urban forests. The results of this research support a hypothesis by Pouyat et al. (2009) that SOC content in residential lawns can be equivalent to native forest soils in temperate regions since turf grasses may grow without supplements. Temperate lawns that are more intensely managed respond differently in regard to soil C and N storage. For instance, in Baltimore, MD, residential mineral soils (0-100 cm) that were managed under fertilized and irrigated turf were found to have higher SOC and N contents than nearby urban remnant forests on similar soil types (Raciti et al., 2011). Furthermore, urban lawns in Baltimore that are fertilized and irrigated have a high potential for N retention compared to urban remnant forests (Raciti et al., 2008).

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Table 2.1. Location, plant community and soil information for each of the study sites within the oak dominated forest, pine dominated forest and lawn land covers in western Florida.

Land cover	Plant community	Soil drainage	Location UTM UPS NAD 1983 (meters)		
Natural Forest Oak	mesic hammock	very poorly drained	16 R	684434	3293733
	xeric hammock	poorly drained	16 R	684474	3293715
	mesic hammock	somewhat poorly drained	16 R	684631	3293865
	xeric hammock	moderately well drained	16 R	683428	3293155
Urban Forest Oak	urban forest	very poorly drained	16 R	693266	3289183
	urban forest	poorly drained	16 R	694288	3289516
	urban forest	somewhat poorly drained	16 R	694037	3289783
	urban forest	moderately well drained	16 R	703440	3291945
Urban Lawn	urban lawn	very poorly drained	16 R	694529	3290007
	urban lawn	poorly drained	16 R	694300	3289432
	urban lawn	somewhat poorly drained	16 R	694225	3289886
	urban lawn	somewhat poorly drained	16 R	694116	3290519
	urban lawn	moderately well drained	16 R	694845	3289967

Natural Forest Pine	wet flatwood	very poorly drained	16 R	684574	3293648
	wet flatwood	poorly drained	16 R	684613	3293696
	mesic flatwood	somewhat poorly drained	16 R	684675	3293950
Pine Plantation	pine plantation	very poorly drained	16 R	683627	3293600
	pine plantation	poorly drained	16 R	683657	3293509
	pine plantation	somewhat poorly drained	16 R	713654	3296434
Urban Forest Pine	urban forest	very poorly drained	16 R	693263	3289280
	urban forest	poorly drained	16 R	690576	3291264
	urban forest	somewhat poorly drained	16 R	691995	3290655

Table 2.2. Summary of land use characteristics and productivity values in naturally regenerating forests, urban forests and pine plantation sites in western Florida. Standing crop biomass, mean diameter at breast height (DBH) and stem density includes stems ≥ 10 cm DBH. Foliar productivity represents the mean productivity from 2010-2012. Woody biomass, annual woody growth rate and aboveground net primary productivity (NPP) are from 2012. Standard errors are in parentheses.

Response Variable	**Natural Forest Oak**	**Urban Forest Oak**	**Natural Forest Pine**	**Urban Forest Pine**	**Pine Plantation**
Basal area (m^2 ha^{-1})	18.94 (3.02)	27.55 (1.33)	37.50 (4.05)	28.32 (2.02)	35.97 (6.67)
Standing crop biomass (g m^{-2})	22,993 (8,076)	23,527 (4,256)	32,559 (3,375)	31,042 (9,516)	15,152 (4,703)
Mean DBH (cm)	21.27 (1.61)	24.96 (1.48)	26.80 (1.93)	25.87 (2.96)	18.52 (0.64)
Stem density (stems ha^{-1})					
Total	500 (49)	494 (61)	675 (38)	650 (109)	917 (183)
Hardwoods	450 (40)	394 (21)	292 (51)	233 (110)	92 (92)
C. camphora	nd[a]	75.0 (27.0)[b]	nd	58.3 (58.3)[b]	nd
Productivity (g m^{-2} yr^{-1})					

Foliar	523.7 (30.9)	686.4 (81.9)	638.4 (41.8)	779.7 (78.7)	615.1 (66.6)
Woody biomass	535.3 (69.9)	845.1 (109.4)	493.3 (74.8)	1,096 (305)	663.9 (202.4)
Aboveground NPP	1,059 (57)	1,531 (173)	1,132 (98)	1,876 (383)	1,279 (258)
Annual woody growth rate (cm yr^{-1})	0.27 (0.03)	0.47 (0.05)	0.19 (0.02)	0.44 (0.06)	0.31 (0.03)
Leaf Area Index	4.84 (0.12)	6.45 (1.42)	2.42 (0.35)	3.58 (1.36)	1.66 (0.36)

[a]*Cinnamomum camphora* is not detected (nd) at the field sites.

[b]*Cinnamomum camphora* is in three of the four urban forest oak sites and one of the three urban forest pine sites.

Table 2.3. Results of the contrast statements for differences among land covers in site characteristics and aboveground productivity. Contrasts with a p-value ≤ 0.10 are reported. Natural forest oak = NFO; Natural forest pine = NFP; Pine plantation = PP; Urban forest oak = UFO; and Urban forest pine = UFP.

Response Variable	**Contrast**	**Point Estimate**	**Margin of Error**	**p-value**	**Significance**
DBH (cm)					
	NFP - PP	8.28	5.86	0.0190	*
Stem density					
	NFP – PP	-241	276	0.0791	
	Pine forest - Oak forest	293	178	0.0048	*
Standing crop biomass (g m^{-2})					
Total-tree	NFP - PP	17,407	T[a]	0.0190	*
Productivity (g m^{-2} yr^{-1})					
Foliar	UFO - NFO	162.6	193.6	0.1004	

	Urban forest - Rural forest	144.7	141.8	0.0463	*
Woody biomass	UFP - NFP	602.6	495.4	0.0225	*
	Urban forest - Rural forest	417.2	302.9	0.0124	*
Aboveground NPP	UFO - NFO	472.5	570.6	0.0938	
	UFP - NFP	743.82	658.9	0.0310	*
	Urban forest - Rural forest	561.9	389.0	0.0116	*
Leaf area index					
	Urban forest - Rural forest	2.13	1.94	0.0345	*
	Pine forest – Oak forest	-3.65	2.05	0.0030	*

[a]Data are transformed (T) and a 95% confidence interval is not symmetric around the mean

Table 2.4. Nutrients in foliar litterfall in naturally regenerating forests, urban forests and pine plantation sites in western Florida. Nutrient concentration is an average of all of the monthly nutrient concentrations during 18 months of collection, from March 2011 to August 2011. Nutrient content is how much N and P is returned to each site on a yearly basis in the foliar litterfall. Standard errors are in parentheses.

Measured variable	Natural Forest Oak	Urban Forest Oak	Natural Forest Pine	Urban Forest Pine	Pine Plantation
Nutrient concentration (mg kg^{-1})					
N	7,200 (445)	7,286 (588)	3,649 (372)	4,617 (831)	3,770 (317)
P	1,007 (208)	1,225 (153)	568 (112)	490 (83)	520 (24)
Nutrient ratios					
C:N	73.9 (4.9)	74.0 (7.2)	177.1 (14.9)	133.7 (25.5)	157.7 (16.1)
C:P	623 (140)	443 (58)	1,127 (225)	1,279 (259)	1,217 (81)
Nutrient content (g m^{-2} yr^{-1})					
N	3.60 (0.16)	5.05 (0.92)	2.23 (0.46)	2.98 (0.90)	2.06 (0.37)
P	0.499 (0.099)	0.666 (0.111)	0.330 (0.066)	0.312 (0.100)	0.266 (0.038)
C	263 (19)	342 (40)	341 (18)	386 (38)	314 (29)

Table 2.5. Results of the contrast statements for differences among land covers for foliar nutrients. Only contrasts with a p-value ≤ 0.10 are reported. Natural forest oak = NFO; Natural forest pine = NFP; Pine plantation = PP; Urban forest oak = UFO; and Urban forest pine = UFP.

Response Variable	Contrast	Point Estimate	Margin of Error	p-value	Significance
Nutrient concentration (mg kg^{-1})					
N	Pine forest – oak forest	-3,456	1,123	<0.0001	*
P	Pine forest – oak forest	-605.7	335.3	0.0034	*
Nutrient ratios					
C:N	UFP – NFP	-43.4	45.4	0.0670	
	Pine forest – oak forest	85.4	29.3	0.0001	*
C:P	Pine forest – oak forest	696.3	370.9	0.0027	*
Nutrient content (g m^{-2} yr^{-1})					
C	Urban forest - Rural forest	59.6	69.0	0.0920	

N	Urban forest - Rural forest	1.44	1.36	0.0468	*
	Pine forest – oak forest	-2.19	1.44	0.0088	*
P	Pine forest – oak forest	-0.345	0.188	0.0031	*

Table 2.6. Soil pH and bulk density by depth in naturally regenerating forests, urban forests, urban lawns and pine plantation soils in western Florida.

Land Use	**pH**				**Bulk density (g cm^{-3})**			
	0 – 7.5 cm	**7.5 – 30 cm**	**30 – 60 cm**	**60 – 90 cm**	**0 – 7.5 cm**	**7.5 – 30 cm**	**30 – 60 cm**	**60 – 90 cm**
Urban Lawn	5.8 (0.2)	6.1 (0.3)	6.5 (0.3)	6.2 (0.3)	1.10 (0.05)	1.34 (0.05)	1.31 (0.06)	1.30 (0.06)
Natural Forest Oak	3.6 (0.1)	3.8 (0.1)	3.9 (0.1)	4.0 (0.1)	1.23 (0.01)	1.30 (0.01)	1.33 (0.03)	1.39 (0.03)
Urban Forest Oak	4.4 (0.5)	4.2 (0.2)	4.5 (0.4)	4.7 (0.3)	1.15 (0.02)	1.27 (0.05)	1.33 (0.01)	1.40 (0.06)
Natural Forest Pine	3.2 (0.1)	3.5 (0.1)	3.6 (0.2)	3.6 (0.2)	1.16 (0.02)	1.31 (0.06)	1.36 (0.05)	1.30 (0.05)
Urban Forest Pine	3.4 (0.1)	3.4 (0.1)	3.8 (0.5)	3.7 (0.4)	1.18 (0.10)	1.14 (0.10)	1.30 (0.16)	1.21 (0.10)
Pine Plantation	3.7 (0.1)	4.0 (0.3)	3.9 (0.2)	4.0 (0.3)	1.20 (0.01)	1.35 (0.02)	1.25 (0.05)	1.36 (0.05)

Table 2.7. Results of the contrast statements for differences among land covers for soil pH and bulk density by depth. Only contrasts with a p-value ≤ 0.10 are reported. Natural forest oak = NFO; Natural forest pine = NFP; Pine plantation = PP; Urban forest oak = UFO; and Urban forest pine = UFP.

Response Variable	Contrast	Point Estimate	Margin of Error	p-value	Significance
pH					
0 – 7.5 cm	UFO - NFO	0.9	0.7	0.0217	*
	Urban lawn – Urban forest	1.8	0.5	<0.0001	*
	Oak forest – Pine forest	0.7	0.4	0.0155	*
7.5 – 30 cm	Urban lawn – Urban forest	2.2	0.5	<0.0001	*
	Oak forest – Pine forest	0.4	0.5	0.0952	
30 – 60 cm	Urban lawn – Urban forest	2.0	0.6	<0.0001	*
60 – 90 cm	Urban lawn – Urban forest	1.9	0.3	<0.0001	*
Bulk density (g cm^{-3})					
7.5 – 30 cm	UFP - NFP	-0.17	0.19	0.0774	
	Urban lawn – Urban forest	0.13	0.14	0.0599	

Table 2.8. Soil carbon content (kg m^{-2}) by depth in naturally regenerating forests, urban forests, urban lawns and pine plantation soils in western Florida.

	Carbon (kg m^{-2}) by depth increment					Total Carbon (kg m^{-2})	
Land Use	**Forest floor**	**0 – 7.5 cm**	**7.5 – 30 cm**	**30 – 60 cm**	**60 – 90 cm**	**0 – 90 cm**	**Forest floor + Soil 0 – 90 cm**
Urban Lawn	NA	3.03 (0.35)	4.51 (0.55)	3.23 (0.76)	2.18 (0.31)	12.95 (1.18)	12.95 (1.18)
Natural Forest Oak	2.23 (0.78)	1.89 (0.40)	1.26 (0.34)	2.06 (0.37)	1.27 (0.34)	6.49 (1.05)	8.71 (0.48)
Urban Forest Oak	2.68 (0.99)	4.03 (1.34)	5.35 (2.31)	8.32 (5.04)	7.05 (4.42)	24.74 (12.63)	27.42 (13.17)
Natural Forest Pine	5.55 (1.18)	4.55 (1.13)	3.94 (2.22)	2.06 (1.40)	4.66 (0.71)	15.21 (3.01)	20.76 (3.96)
Urban Forest Pine	4.86 (1.26)	4.56 (1.36)	3.41 (0.39)	9.82 (6.32)	7.64 (3.75)	24.30 (10.06)	29.16 (9.03)
Pine Plantation	3.60 (0.50)	1.73 (0.76)	1.14 (0.41)	3.48 (1.23)	4.96 (1.46)	11.31 (3.76)	14.92 (4.21)

Table 2.9. Results of the contrast statements for differences among land covers for soil C content (kg m^{-2}) by depth. Only contrasts with a p-value ≤ 0.10 are reported. Natural forest oak = NFO; Natural forest pine = NFP; Pine plantation = PP; Urban forest oak = UFO; and Urban forest pine = UFP.

Response Variable	Contrast	Point Estimate	Margin of Error	p-value	Significance
Soil carbon content (kg C m^{-2})					
Forest floor	Pine forest – Oak forest	2.35	2.21	0.0396	*
0 – 7.5 cm	UFO - NFO	2.14	2.65	0.1067	
	NFP - PP	2.81	3.08	0.0704	
7.5 – 30 cm	UFO - NFO	4.08	3.64	0.0311	*
30 – 60 cm	UFO - NFO	6.26	6.99	0.0754	
	UFP - NFP	7.76	8.07	0.0582	
	Urban forest - Rural forest	6.75	4.93	0.0111	*
60 – 90 cm	UFO - NFO	5.77	5.71	0.0481	*
0 – 90 cm	UFO - NFO	18.25	16.26	0.0308	*
Forest floor + Soil (0 – 90 cm)	UFO - NFO	18.71	16.83	0.0320	*

Table 2.10. Soil nitrogen content (kg m^{-2}) by depth in naturally regenerating forests, urban forests, urban lawns and pine plantation soils in western Florida.

	Nitrogen (kg m^{-2}) by depth increment					Total Nitrogen (kg m^{-2})	
Land Use	**Forest floor**	**0 – 7.5 cm**	**7.5 – 30 cm**	**30 – 60 cm**	**60 – 90 cm**	**0 – 90 cm**	**Forest floor + Soil 0 – 90 cm**
Lawn	NA	0.198 (0.034)	0.187 (0.044)	0.088 (0.022)	0.089 (0.006)	0.562 (0.065)	0.562 (0.065)
Natural Forest Oak	0.051 (0.019)	0.043 (0.017)	0.014 (0.005)	0.022 (0.010)	0.026 (0.013)	0.104 (0.027)	0.155 (0.026)
Urban Forest Oak	0.068 (0.030)	0.210 (0.071)	0.211 (0.090)	0.292 (0.143)	0.209 (0.085)	0.922 (0.355)	0.990 (0.366)
Natural Forest Pine	0.086 (0.017)	0.103 (0.023)	0.091 (0.037)	0.087 (0.033)	0.092 (0.019)	0.373 (0.056)	0.459 (0.070)
Urban Forest Pine	0.083 (0.015)	0.143 (0.051)	0.115 (0.035)	0.274 (0.190)	0.195 (0.095)	0.688 (0.297)	0.772 (0.286)
Pine Plantation	0.048 (0.007)	0.044 (0.027)	0.014 (0.003)	0.066 (0.026)	0.104 (0.012)	0.229 (0.013)	0.277 (0.008)

Table 2.11. Results of the contrast statements for differences among land covers for soil N content (kg m^{-2}) by depth. Only contrasts with a p-value ≤ 0.10 are reported. Natural forest oak = NFO; Natural forest pine = NFP; Pine plantation = PP; Urban forest oak = UFO; and Urban forest pine = UFP.

Response Variable	Contrast	Point Estimate	Margin of Error	p-value	Significance
Soil nitrogen content (kg N m^{-2})					
0 – 7.5 cm	UFO - NFO	0.167	0.127	0.0140	*
7.5 – 30 cm	UFO - NFO	0.198	T[a]	0.0016	*
	NFP - PP	0.077	T	0.0272	*
30 – 60 cm	UFO - NFO	0.270	T	0.0016	*
60 – 90 cm	UFO - NFO	0.183	T	0.0041	*
0 – 90 cm	UFO - NFO	0.818	0.503	0.0038	*
Forest floor + Soil (0 – 90 cm)	UFO - NFO	0.834	0.505	0.0034	*

[a]Data are transformed (T) and a 95% confidence interval is not symmetric around the mean

Table 2.12. Soil organic carbon (SOC) densities (to 90 or 100 cm) for mineral soils or land covers that are comparable to this study. Standard error is in parenthesis.

Land use, forest-type or soil type	Location	SOC (kg C m^{-2})	Depth (cm)	n	Source
Florida spodosols	Florida	10.4 (0.8)	0-100	244	Stone et al. (1993)
Florida soils	central Florida	7.49[a]	0-100	139	Vasques et al. (2010)
Slash pine plantation	western Florida	11.31 (3.76)	0-90	3	this study
Pine plantation	western Florida	8.8 (1.6)	0-90	10	Nagy et al. (2013)
Slash pine plantation (control)	northern Florida	14.08 (NA)[b]	0-100	3	Shan et al. (2001)
Longleaf-slash pine	Contiguous US	10.4 (NA)[b]	0-100	NA[c]	Johnson and Kern (2003)
Natural forest oak dominated	western Florida	6.49 (1.05)	0-90	4	this study
Natural forest pine dominated	western Florida	15.21 (3.01)	0-90	3	this study
Natural forest	western Florida	7.3 (0.9)	0-90	11	Nagy et al. (2013)
Urban forest oak dominated	western Florida	24.74 (12.63)	0-90	4	this study
Urban forest pine dominated	western Florida	24.30 (10.06)	0-90	3	this study
Urban forest	western Florida	15.9 (4.4)	0-90	14	Nagy et al. (2013)
Urban remnant forest	Baltimore, MD	12.1 (1.8)	0-100	5	Pouyat et al. (2009)
Urban lawn	western Florida	12.95 (1.18)	0-90	5	this study

Urban lawn	western Florida	10.7 (2.6)	0-90	14	Nagy et al. (2013)
Turfgrass	Baltimore, MD	11.0 (0.9)	0-100	19	Pouyat et al. (2009)

[a]Value is the median SOC value.

[b]Standard error is not provided.

[c]Soil organic carbon data is from the National Soil Characterization Database of the USDA Natural Resource Conservation Service. The authors accessed this database in 1997 and retrieved soils data from 136,510 samples.

Figure 2.1. Monthly mean precipitation (cm) for the study area, measured at Apalachicola Airport, Apalachicola, Florida. Data is reported for (a) the long term (1900-2010) and the study period (July 2010 to June 2012) precipitation and for (b) the departure of the monthly precipitation during the study period from the long term monthly mean.

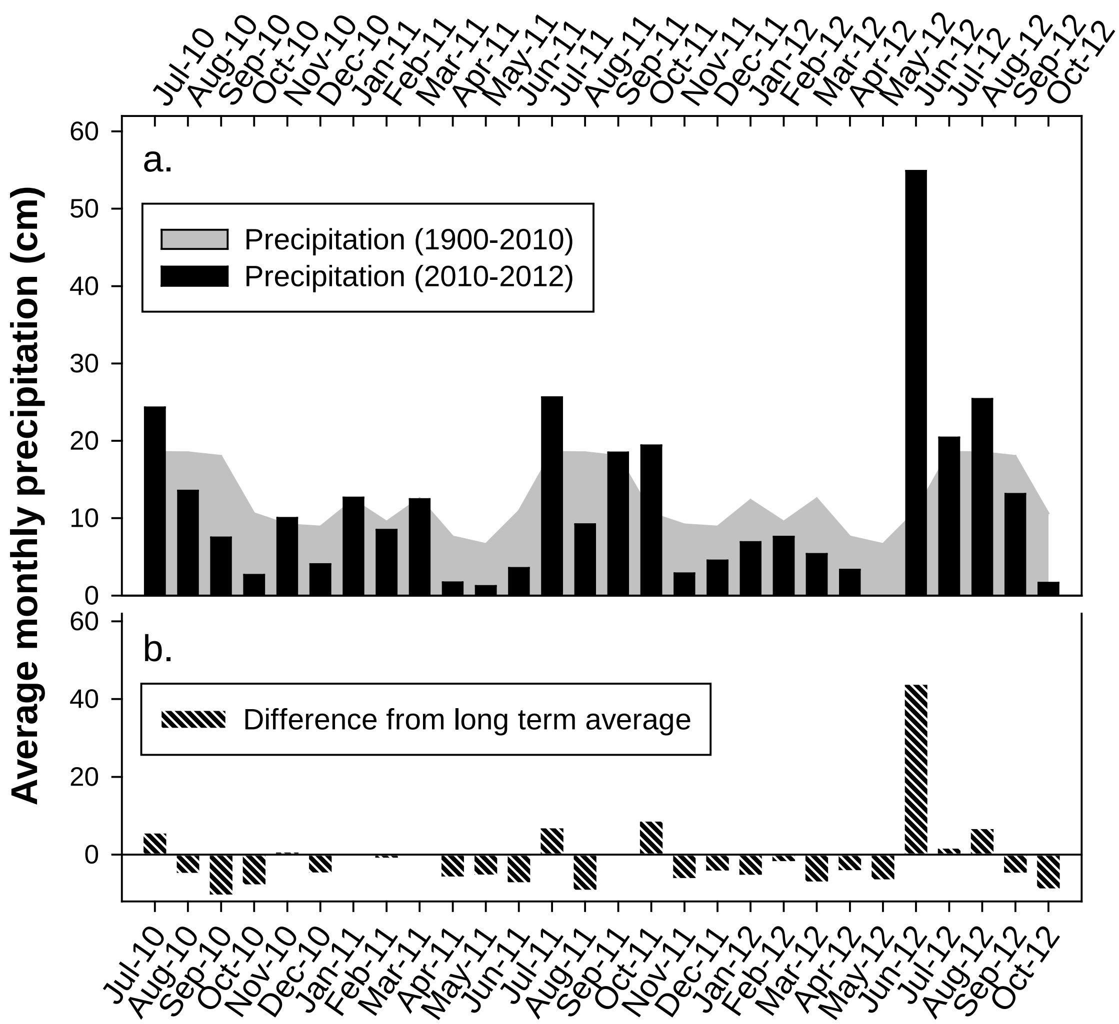


Figure 2.2. Seasonal patterns of (a) N, (b) P, and (c) C foliar nutrient content (g m^{-2}) on a monthly basis in oak dominated naturally regenerating forests and urban forests. Seasonal patterns of (d) N, (e) P, and (f) C foliar nutrient content (g m^{-2}) on a monthly basis in pine dominated naturally regenerating forests, urban forests and plantation sites in western Florida. Nutrient content is provided for 18 months of collection, from March 2011 to August 2011.

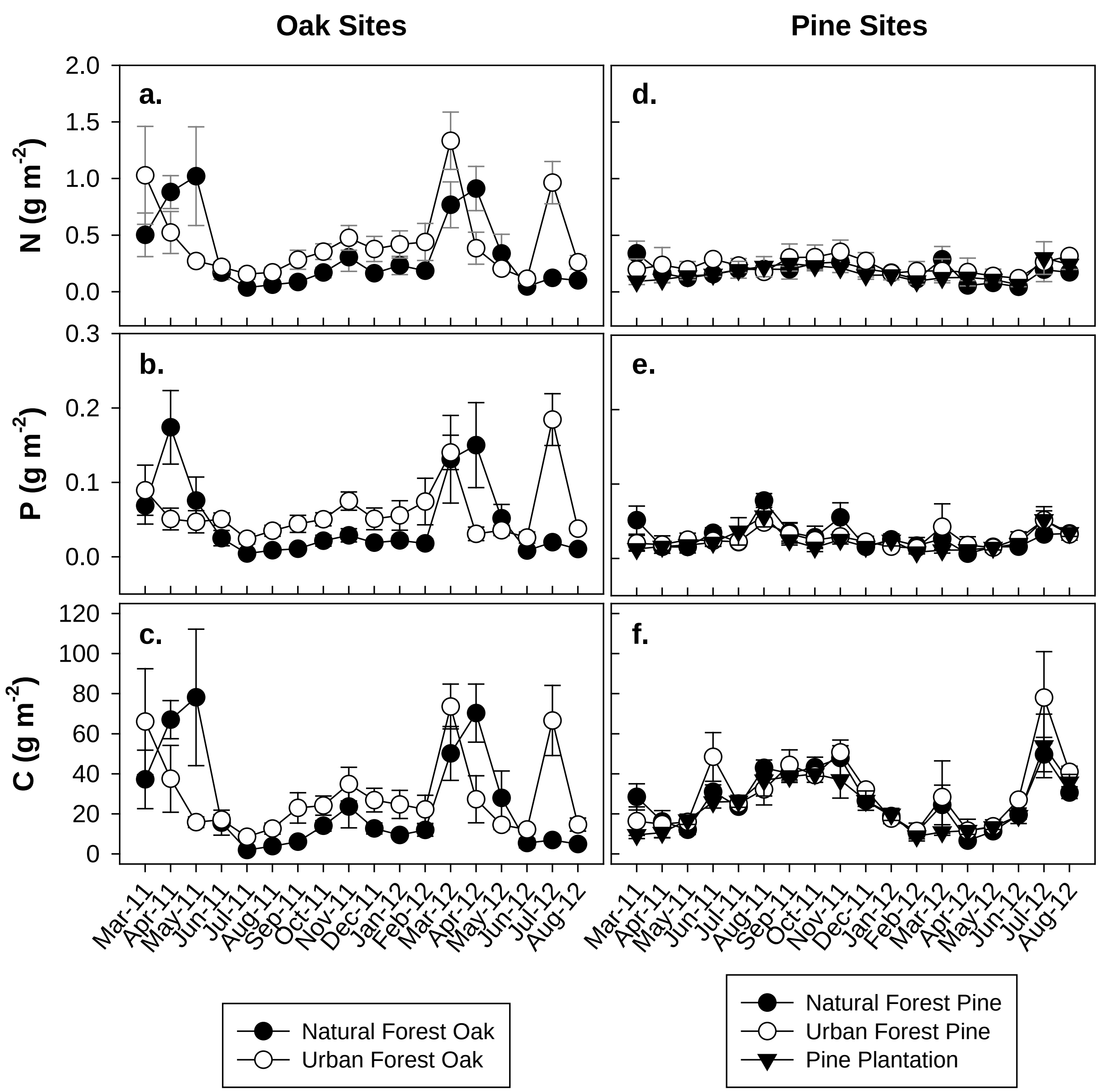


Figure 2.3. Correlation (a) between mean N and P contents in litterfall and regression relationships between mean (b) N and (c) P contents in litterfall, vegetation type and foliar litterfall productivity rates in naturally regenerating forests, urban forests and pine plantation sites in western Florida (n=17). The regression relationship for (b): foliar productivity = 213 + 90.5(Litterfall N) + 191(VegType) + 22.2(VegType*Litterfall N). The regression relationship for (c): foliar productivity = 323 + 483(Litterfall P) + 104(VegType) + 342(VegType*Litterfall P). Vegetation type (VegType) is 0 for oak and 1 for pine.

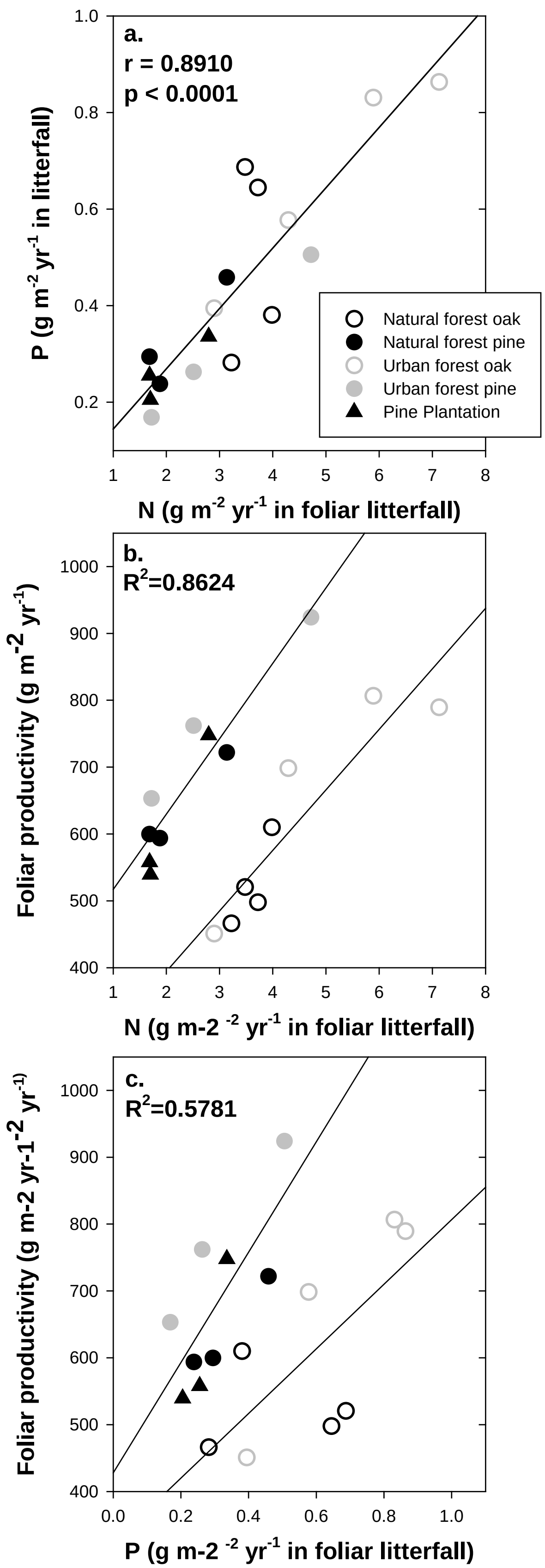

a.
r = 0.8910
p < 0.0001
P (g m-2 yr-1 in litterfall)
N (g m-2 yr-1 in foliar litterfall)
Natural forest oak
Natural forest pine
Urban forest oak
Urban forest pine
Pine Plantation
b.
R2=0.8624
Foliar productivity (g m-2 yr-1)
N (g m-2 -2 yr-1 in foliar litterfall)
c.
R2=0.5781
Foliar productivity (g m-2 yr-1-2 yr-1)
P (g m-2 -2 yr-1 in foliar litterfall)

Figure 2.4. Soil organic carbon content (kg m^{-2}) for the forest floor, mineral soil (0 – 90 cm) and forest floor plus mineral soil in naturally regenerating forests, urban forests and pine plantation sites in western Florida. Land covers are clustered to compare urbanization effects within oak sites, pine sites, and urban sites (lawn and forest).

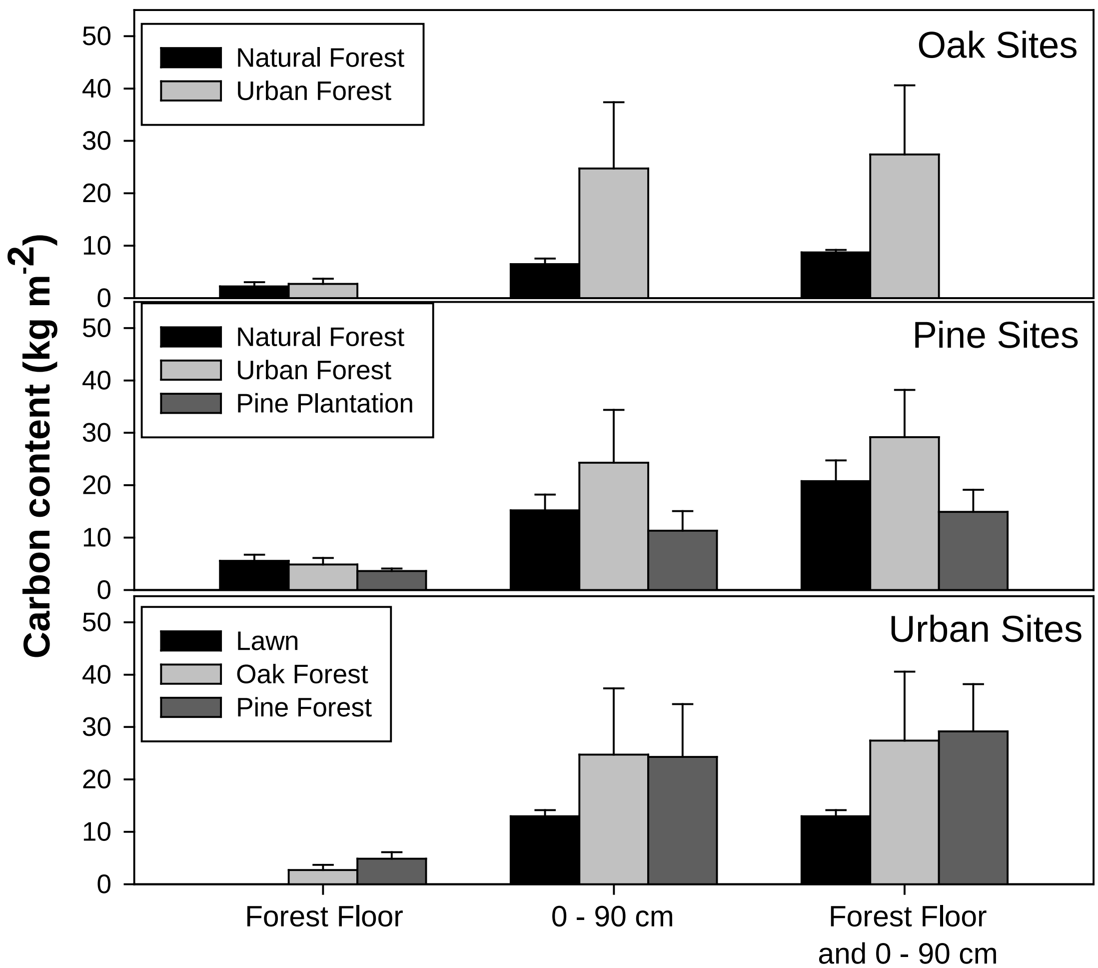
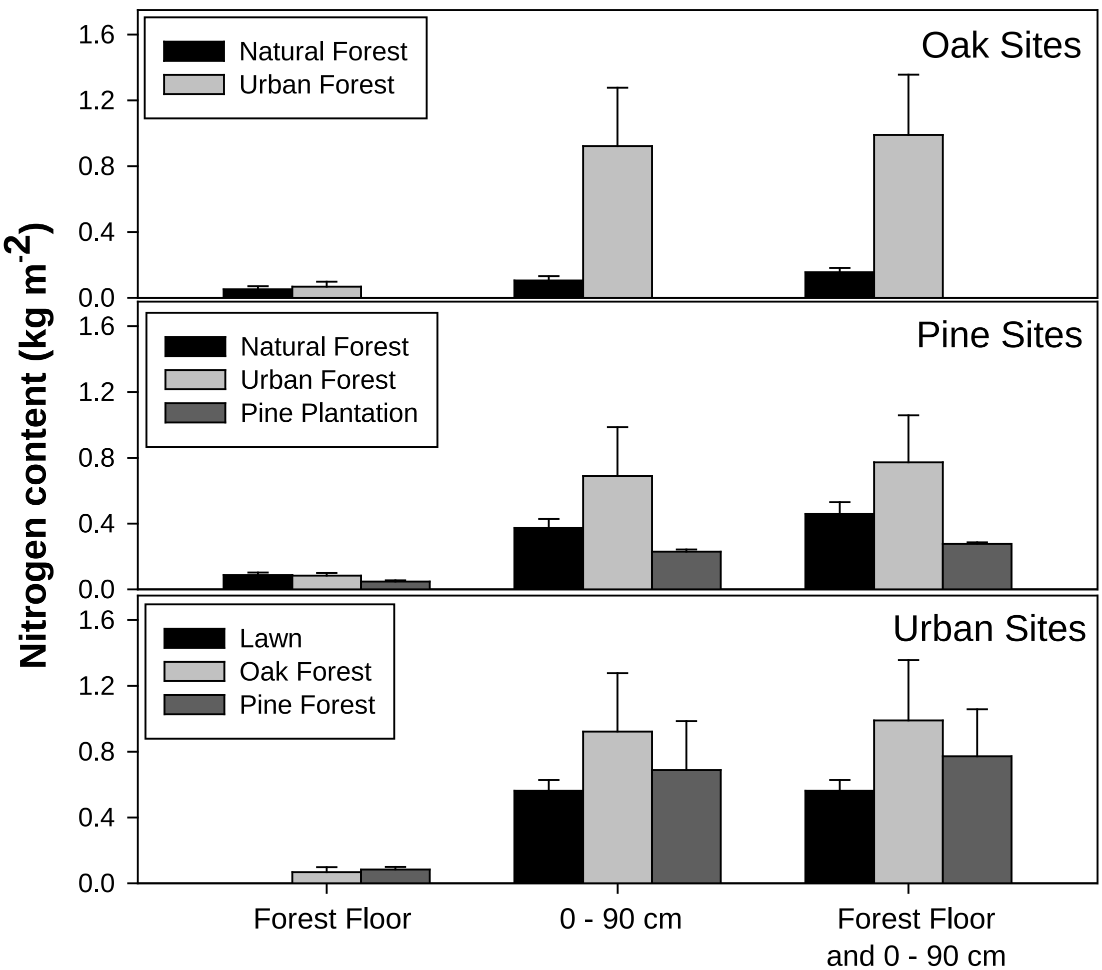


Figure 2.5. Soil nitrogen content (kg m^{-2}) for the forest floor, mineral soil (0 – 90 cm) and forest floor plus mineral soil in naturally regenerating forests, urban forests and pine plantation sites in western Florida. Land covers are clustered to compare urbanization effects within oak sites, pine sites, and urban sites (lawn and forest).



3. Impact of urbanization on leaf litter decomposition and nutrient dynamics

Abstract

Many city residents may benefit from the ecosystem services that urban forests provide. An important supporting process to ecosystem services is nutrient cycling and it is uncertain how urbanization may alter nutrient cycling compared to rural or natural forests. The purpose of this research is to gain insight into how urbanization impacts the decomposition of mixed species foliar litter, the nutrient release patterns in decomposing litter, foliar litter quality, and forest floor temperatures in a developing coastal area in western Florida. The land covers in this study include rural slash pine (*Pinus ellottii*) plantations, rural naturally regenerating forests (slash pine or mixed oak dominated), and urban forest fragments (slash pine or mixed oak dominated). Urban forest floor temperatures are on average warmer by 0.63 ^{o}C in the winter ($p=0.0101$) and tend to have a more narrow temperature range than those of the rural forested sites. These alterations of the forest floor temperature is not reflected in different mass loss or nutrient release patterns in decomposing litter among contrasted land covers (urban forest oak versus natural forest oak and urban forest pine versus natural forest pine). Foliar mass loss was measured over an 82 week period that was characterized by periods of extended drought, which may have masked an urbanization effect. Urbanization does not appear to impact litter quality indices (carbon to nitrogen and lignin to nitrogen ratios) of mixed foliar species litter, but does appear to improve litter quality of single species foliar litter. Foliar decay rates are correlated to initial lignin to nitrogen ratios and to initial nitrogen and phosphorus concentrations due to the distinct difference in litter quality between oak and pine dominated sites. The higher initial litter

quality of the mixed species litter from the oak dominated sites (both urban and natural) compared to the pine dominated sites also resulted in a different nutrient release patterns in oak versus pine sites. This may suggest that future urbanization studies in this region need to account for the overstory composition of a forest stand, as this may reflect differences in stand origin that then may impact biogeochemical cycling and the overall effect of urbanization.

Introduction

Urban forests can provide many services to city dwellers, including a reduction of noise pollution, improved aesthetics of a neighborhood, and habitat for wildlife (see review by Nowak et al., 2010). The importance of urban forests in regards to air pollution remediation and direct carbon sequestration has recently been questioned (Pataki et al., 2011). It is uncertain how far individuals will go to insure that urban trees are maintained, even if there is evidence to suggest that green space or planted trees in a neighborhood can increase housing prices (Saphores and Li, 2012). As a study in California, concluded: “Los Angeles residents may want additional trees, they are unwilling to pay for them” (Saphores and Li, 2012). One thing that may be true: when green space is gone, many city residents miss it. An example of this can be found in select areas within the New York City metropolitan area, where former sidewalks now contain picnic tables sitting atop concrete that is painted green.

Understanding ecosystem processes within urban forests is important in order to improve management of these areas as well as to gain insight into what city dwellers may or may not lose when urban forests are gone. The societal benefits of an ecosystem can be categorized into provisioning services, cultural services and regulating services and a supporting ecosystem process to all three of these services is nutrient cycling (MA 2003). The purpose of this research is to gain insight into how urbanization impacts foliar litter decomposition and nutrient cycling in a developing coastal area in western Florida. The effect of urbanization is often determined after major development has occurred and, with this study, we have an opportunity to establish baseline data as well as study how these data change as this coastal area transitions into a post-developed state.

How urbanization affects decomposition rates compared to a rural, naturally regenerating forest is of interest from a plant growth and forest regeneration standpoint as well as from a carbon cycling standpoint. A common growth limiting factor to trees in the southeastern United States is nitrogen. With the suppression of fire, which was a historical driver of nutrient cycling in the southeast, the decomposition of foliar litter is a major source of nitrogen and other nutrients to trees within urban forests and rural forests where burning is restricted. Soils of the southeastern coastal plain are sandy and nutrient poor and an accumulation of foliar litterfall on the forest floor could immobilize essential nutrients. For example, the immobilization of nutrients in the forest floor is suggested as a primary reason for the cessation of vegetative biomass production at approximately 26 years of age within slash pine plantations in northern Florida (Gholz et al., 1985). Furthermore, decomposition rates along with foliar litterfall production rates influence the forest floor depth. Forest floor depth was found to vary greatly between urban and rural forests in the New York City metropolitan area. In turn, this impacted seedling germination and survival and ultimately was expected to alter long-term patterns of forest regeneration (Kostel-Hughes et al., 1998). Finally, the role of soil respiration in the carbon cycle should not be underestimated. Soil respiration releases approximately 12.5 times the amount of carbon per year as the burning of fossil fuels and 83 times that of the net destruction of vegetation on a global basis (Schlesinger and Andrews, 2000).

A recent review of the literature found seven papers that were published since the late 1970's that address the impact of urbanization on foliar decomposition by comparing decomposition rates of foliar litter collected at urban and rural forested sites. Two of the studies measured litter decomposition in the laboratory under controlled conditions (Cotrufo et al., 1995; Carreiro et al., 1999) and the others investigated litter decomposition *in-situ*. All of the studies used single species litter and monitored decomposition over time periods that varied from five months to 36 months. Decay rates were found to be faster (Nikula et al., 2010; Pouyat et al., 1997), slower (Inman and Parker, 1978; Cotrufo et al., 1995; Carreiro et al., 1999; Pavao-Zuckerman and Coleman, 2005) or the same (Pouyat and Carreiro, 2003) for urban versus rural foliar litters. It is uncertain how urbanization effects decay rates across different cities.

The uncertainty associated with the effects of urbanization on decay rates is likely due in part to the large variety of anthropogenic disturbances associated with urbanization that exist within and between cities. Furthermore, these changes associated with urbanization may have opposing effects on decomposition processes. Several factors exist in urban forests that can potentially decrease foliar decay rates from those in rural areas, such as lower litter quality in urban litters due to higher concentrations of lignin and other recalcitrant components (Carreiro et al., 1999).

Higher heavy metal (Cu, Pb, and Zn) concentrations in urban soils can lead to lower microbial populations (Cotrufo et al., 1995) and slow foliar litter decay rates compared to unpolluted rural areas (Inman and Parker, 1978; Cotrufo et al., 1995). In contrast, an increase in urban temperatures, the urban-heat island effect, has been suggested to increase decomposition rates in urban versus rural forested areas (Pouyat and Carreiro, 2003).

Urban forests can also be subjected to an increased exotic species presence (Zipperer and Guntenspergen, 2009). In general, the presence of exotic plants has been shown to increase decomposition rates compared to uninvaded areas (Ehrenfeld, 2003). In a reciprocal litterbag experiment, Pouyat and Carreiro (2003) found that rural litter placed in urban sites decomposed faster than rural litter on rural sites due to the higher number of exotic earthworms and increased temperatures found in the urban location.

The objective of this study is to measure foliar litter decomposition rates (decay constants and mass remaining over time) and nutrient immobilization and mineralization patterns in rural forests and unmanaged urban forests in coastal western Florida. The aforementioned urban litter decomposition studies used a single species foliar litter to investigate the impacts of urbanization on foliar decomposition rates (Inman and Parker, 1978; Cotrufo et al., 1995; Pouyat et al., 1997; Carreiro et al., 1999; Pouyat and Carreiro, 2003; Pavao-Zuckerman and Coleman, 2005; Nikula et al., 2010). One distinct difference in the methodology of this decomposition study is the use of site specific mixed litters in the litterbags. As mass losses of mixed litters have been found to deviate from those predicted by the mass losses of the individual litters decomposing alone (Gartner and Cardon, 2004), the inclusion of mixed litter may help to further clarify the effect of urbanization on forest carbon and nutrient cycling.

The urban forests of this study are located in Apalachicola and Eastpoint, Florida. Both towns have low population densities and the United States Census Bureau defines the area as an urban cluster. In order to understand the mechanisms behind differences in decay and nutrient cycling across the urban and rural sites, forest floor temperatures and initial litter quality parameters were also measured. Differences in the microenvironment between urban and rural forests were estimated by monitoring the mass loss over time of a common substrate, wooden (*Betula papyrifera*) popsicle sticks (Baker et al., 2001). Even though the urban forests are in towns with low population densities, we expect the urban environment to be altered from that of the rural environment and that alteration to be reflected in different decomposition rates and nutrient cycling patterns.

Methods

Study site description

The study was conducted within and outside the city limits of Apalachicola and Eastpoint, Franklin County, Florida. In 2010, Apalachicola had a population density of 460.5 people km^{-2} and Eastpoint had a population density of 123.1 people km^{-2} (U.S. Census Bureau, 2010). Watersheds in the Apalachicola and Eastpoint area have impervious surface percentages that do not exceed 15 % of the watershed land area (Nagy et. al., 2012). The study area is located in a subtropical climate, with average summer temperatures of 27.2 ^{o}C and average winter temperatures of 12.4 ^{o}C. The study area typically receives 143.5 cm of precipitation per year, with the majority of the precipitation occurring in July, August and September (National Climatic Data Center, 2012).

Five forested land covers are the focus of this study: pine plantation, urban forest pine dominated, urban forest oak dominated, natural forest pine dominated, and natural forest oak dominated. Slash pine *(Pinus elliotti* Engelmann) is the dominant overstory tree on the pine dominated forested sites, whereas live oak (*Quercus virginiana* Miller), sand live oak *(Q. geminata* Small), laurel oak (*Q. hemisphaerica* Bartram ex Willdenow), swamp laurel oak (*Q. laurifolia* Michaux) and/or water oak (*Q. nigra* Linnaeus) are common overstory species on the oak dominated sites. Nagy et al. (2013) determined the prominent land covers for this region and the land covers chosen for this study are a refined subset of forested land covers that occur on an urbanization gradient. For this study, urban forest sites are unmanaged forest patches within the urban matrix. Many of the urban forests within Apalachicola and Eastpoint are remnants of natural plant communities common to the region. Natural plant communities in the study area included wet and mesic flatwoods, mesic and xeric hammocks, and sandhill communities (for community descriptions, see Florida Natural Areas Inventory, 1990). Pine plantation sites were likely wet or mesic flatwood sites before being converted to a plantation.

The distribution of soil types and plant communities in the study area are highly complex as slight changes in ground elevation result in different soil drainage classes and different plant communities. To identify potential sites for this study, digital orthoimagery quarter quandrangles (DOQQs) with 1 m spatial resolution were used

(LABINS, 2004) along with the Franklin County digital soil survey (Soil Survey Staff, 2008). Potential forested sites were identified as areas at least 0.16 ha in size, had canopy closure and the soil and vegetation had not been recently disturbed or harvested. The soil data layer was used to identify soil drainage classes (based on the soil series information), which gave indication to the vegetation community at that site. All potential sites from nine km east and 30 km west of Apalachicola (which includes Eastpoint) and five km from the coastline were explored. Forested sites were selected using the same criteria as listed for the potential sites along with the requirement that the site had not been burned in the last five years and would not be burned in the next three years.

In July 2010 seventeen circular plots (0.04 ha) were established in the study area (Chapter 2). Each of the oak land covers had four replicate sites, with each site having a different soil drainage classification (very poorly drained, poorly drained, somewhat poorly drained or moderately well drained). For each of the pine land covers, sites were found on very poorly drained, poorly drained and somewhat poorly drained soils for a total of three replicates per pine land cover. Pine dominated vegetation communities with closed canopies could not be found on moderately well drained sites. All of the soils in the study area were derived from sandy marine sediments (Sasser et al., 1994). Drainage classes and soil series were determined using the current soil survey for Franklin County (Soil Survey Staff, 2008) and onsite verification.

In a previous study, detailed information on site location, vegetation communities, and plant and soil characteristics were collected (Chapter 2). Based on previous measurements, urbanization did not have an impact on forest floor depth, forest floor mass, standing crop biomass or stem densities (stems with diameters at breast height ≥ 10 cm) (Table 3.1). Foliar productivity is higher in urban forest oak sites compared to natural forest oak sites, yet it is not significantly different between urban and natural forest pine sites. Oaks sites do have a shift in overstory tree composition from natural to urban forests, whereas pine sites do not (Table 3.1). Additionally, camphortree (*Cinnamomum camphora* (L.) J. Presl) is a non-native tree species located within the urban forests and not within the natural forests in this study. Distinct differences are observed between oak dominated and pine dominated sites. Hardwoods dominate the stem densities (stems > 10 cm diameter at breast height) in the oak dominated sites, whereas hardwoods are a smaller component in the pine dominated sites. Pine sites do have larger forest floor masses and deeper forest floor depths compared to oak dominated sites.

Decomposition and litter quality

A decomposition study using litterbags was conducted from March 2011 to May 2012. Leaf litter collection for the decomposition study began in September 2010 and continued through February 2011. Foliar litterfall was collected in 0.25 m^2 traps and tarpaulins that were approximately 30 and 100 cm above the ground, respectively. Leaves were collected each month, air dried, sorted by species and then weighed. For each site, the quantity of litter from each plant species was determined as a relative proportion by weight of all foliar litter collected from the site. Plant species that occupied a dominant proportion of the total litterfall were included in the litterbags. The relative proportions of these dominant plant species were used to form weighted average mixtures to fill the litterbags (Table 3.2, Appendix H). Twenty litterbags (30.5 by 45.7 cm with 6-mm openings on the upper side and 1-mm openings on the lower side) were filled with 10 g of plot-specific species composition leaves and then placed on the forest floor at each of the 17 forested sites. When multiple species were placed in the same litter bag, species specific foliar litter was weighed separately to insure that each litter bag contained the appropriate contribution of species. As a result, litterbags within a site contained the same mixture of species. In addition, twenty litterbags filled with three popsicle sticks (approximately 10 g) were placed on the forest floor of each of the forested sites. Wooden (Betula papyrifera) popsicle sticks were used as a common substrate across the sites in order to assess the effect of the microenvironment on decomposition mass loss rates (Baker et al., 2001).

Litterbags and popsicle sticks were collected at time 0-, 2-, 4-, 6-, 12-, 18-, 30-, 42-, 60-, and 82-weeks. The time 0 collection was weighed before deployment and after collection. This allowed for the calculation of a correction factor for the materials lost due to transport and placement at the sites. Once litterbags were collected, the foliar litter remaining in the bag was removed, dried at 65^{o}C, weighed to determine percent mass remaining, and then ground. A subsample was taken and ashed at 550^{o}C for 12 hours to calculate mass remaining on an ash free basis. Popsicle sticks were cleaned free of sediment, oven dried at 65^{o}C and weighed for each collection time. Popsicle sticks were used to track mass loss over time and were not used to assess nutrient mineralization and immobilization patterns. For litterbags and popsicle sticks, the decomposition rate constant (k) was calculated from the decay curve using the following equation (Swift et al., 1979):

$\ln (M_0 / M_t) = - k * t$

where,

M_0 = mass of litter at time 0,

M_t = mass of litter at time t,

t = time of incubation (in years),

k = decomposition rate constant.

The decomposition rate constant for the litterbags was calculated over the entire collection period (82 weeks). The decomposition rate constant for the popsicle sticks was calculated from week 12 to week 82 (70 weeks), since the popsicle sticks across all land uses did not lose mass from week 0 to week 12.

Carbon (C), nitrogen (N), phosphorus (P) and lignin concentrations were measured at time 0 for litter quality analyses and initial content. The nutrients were also measured during the remaining collection times to determine N and P mineralization and immobilization dynamics during decomposition. Total N and C were determined using a PerkinElmer Series II CHNS/O Analyzer 2400 (PerkinElmer Corp., Norwalk, CT). Phosphorus was extracted from the samples via dry ashing followed by an acid digestion using the vanadomolybdate procedure (Jackson, 1958). Extracts were then analyzed spectrophotometrically. Lignin was determined using the acid detergent fiber method (Van Soest and Wine, 1968). Lignin and nitrogen concentrations estimated at time 0 were used to calculate initial lignin to N ratios.

In addition to the litter quality analysis of the mixed species litter, litter quality of individual tree species was determined over a two year period. Foliar litterfall was also collected on a monthly basis from September 2011 through February 2012, oven dried

(65 ^{o}C), and sorted by species to mirror the collection that occurred the previous year (September 2010 through February 2011). Slash pine (*P. ellottii*) needles, laurel oak (*Q. laurifolia*) leaves and live oak (*Q. virginiana*) leaves from each monthly collection were combined to make a composite sample for each collection year (2010/2011 and 2011/2012) for a particular site. These three species were chosen as they were common species in the study area. Furthermore, sufficient mass of each of these species from the litter collections was available for lignin, C, N and P analyses. Slash pine samples were taken from urban and natural forest pine dominated sites that are very poorly drained, poorly drained and somewhat poorly drained. Laurel oak samples and live oak samples were taken from urban and natural forest oak dominated sites that are somewhat poorly drained and poorly drained.

Precipitation and temperature

Precipitation data were obtained for the study area from the National Climate Data Center Station at the Apalachicola Airport, Apalachicola, Florida. Forest floor temperature was monitored on the forested sites by installing one portable temperature recorder (Onset Computer Corporation, Bourne, MA) at each site. Temperature probes were placed on the forest floor material under similar conditions as the litterbags and in a location that was under full canopy closure. Temperature was taken every hour from March 2010 to May 2012 and data were downloaded from each recorder when litterbags were collected. Mean, minimum and maximum temperatures were calculated for each month temperature was recorded. Daily temperature ranges (daily maximum minus daily minimum) were calculated and then averaged over each month to estimate a monthly average temperature range. In addition, monthly values for mean, maximum and minimum temperatures as well as the temperature range were averaged across seasons to obtain representative values for fall, winter, spring and summer. After the data collection was completed, it was observed that the moderately well drained urban forest oak dominated site exhibited unusually high maximum temperatures compared to the other oak dominated sites. After the study was completed in Florida, all of the probes from the oak sites were placed in a similar outdoor environment in Auburn, AL, during the summer. The probes recorded similar hourly temperatures.

Statistical analysis

All data were analyzed by SAS version 9.2 (SAS Institute, Cary, NC). The non-linear (NLIN) procedure was used to calculate rates of mass loss for the litterbags and for the popsicle sticks at each of the sites. Treatment effects on k, mass and nutrients remaining at week-82, and week-0 litter quality parameters were tested using analysis of variance (ANOVA) with land use and drainage class as the primary treatment variables. Contrast statements were used for the following planned comparisons for differences among sites:

a. Urban forest pine = natural forest pine
b. Urban forest oak = natural forest oak
c. Natural forest pine = pine plantation
d. Urban sites (oak + pine) = rural sites (natural oak + natural pine + pine plantation
e. Pine (urban +natural+plantation) = oak (urban + natural)

To discern whether N and P percent remaining in the decomposing foliar litter varied over time from the initial content at week 0, paired t-tests were used to test for differences between week 0 and each of the nine collections. For the paired t-test, the power of the t-test is low because the sample size is small. Sample size (i.e. the number of paired differences) is three for pine dominated sites and four for oak dominated sites. During week-82, animals damaged the litterbags at two of the natural forest oak sites and one of the urban forest oak sites. As a result, there were only a few significant changes over time.

To test for the effect of urbanization on individual species litter quality, an ANOVA was used with land use (urban forest or natural forest), litter species (slash pine, laurel oak, or live oak), and the interaction between land use and species as the primary treatment variables. A separate ANOVA was used for each collection year (2010/2011 and 2011/2012). A Tukey's honest significance test was used to look for differences among urban and natural forest litter quality parameters if the interaction was not significant ($p > 0.05$) and the land use or species variable was significant.

An analysis of variance procedure was used to test for differences in forest floor temperatures between land covers within each season using all 17 forested sites and a subset (n=16) of the forested sites. A subset of the sites is analyzed so as to determine whether one of the urban forest sites is an outlier in regards to its maximum temperature data. This urban forest oak site has maximum monthly temperatures that are on average 6.55 0C higher than those found in the three other urban forest oak dominated sites. When ANOVA results were significant for either the subset or the entire set of data, the contrast statement comparing urban sites versus rural sites (d.)

was used In order to test for an urban heat island effect. Since the climate in the panhandle of Florida is subtropical, the months included in each season were defined as follows: Summer = June, July, August; Fall = September, October, November; and Winter=December, January, February; Spring=March, April, May.

For both the planned and unplanned comparisons, differences among means are considered statistically significant at $\alpha \leq 0.05$ and differences with $0.05 \leq \alpha \leq 0.10$ are reported for informational purposes. The model assumption of normality is not violated across all tests performed. In a few circumstances, the data is transformed using the natural log function or the square root function to meet the assumption of equality of variance. Normality is met among transformed data.

Results and Discussion

Precipitation and forest floor temperatures

All of the sites in the Florida panhandle underwent a drought during the study period from July 2010 through May 2012 and from July 2012 to October 2012 (Figure 3.1). Annual precipitation was 41.3 cm below the 110 year historical average from July 2010 through June 2011 and 29.3 cm below the historical average from July 2011 to May 2012. A tropical storm in June 2012 added 32 cm of precipitation to the study area over a 48 hour period. This single storm event resulted in bringing the study area out of a deficit and into a surplus of 14.7 cm above the 110 year average for July 2011 to June 2012. From July to October 2012, the study area was 5.1 cm below the 110 year historical average.

The monthly mean, maximum, minimum and range of temperatures under which decomposition was taking place were recorded over the 82 weeks of decomposition. Monthly and seasonal values for the mean, maximum, minimum and range of temperatures for a subset of the data (n=16) are displayed in Table 3.3 and Figures 3.2 and 3.3. The moderately well drained urban forest oak site exhibited unusually high maximum temperatures compared with the other urban forest oak sites and compared to its corresponding moderately well drained natural forest oak site. The removal of this site changes the magnitude of the difference between the rural and urban maximum temperatures, but it does not change the overall conclusions (Table 3.4). Of the 12 contrasts that are found significant ($p \leq 0.10$) by using the subset of data, 10 are found significant using the full data set. Hereafter, all comparisons of temperature data are using the subset of data.

Mean temperatures for spring, summer and fall are not significantly different between urban and rural sites. However, urban sites are on average warmer by 0.63 ± 0.50 ^{o}C (95% confidence interval, p=0.0101, Table 3.4) than rural sites in the winter. Urban sites had higher minimum temperatures and lower maximum temperatures across most of the seasons compared to the rural sites. As a result, the urban sites tend to have a more narrow temperature range than the rural sites.

Decomposition of foliar litter

The mixed species decomposition rate constants across the 82 week period range from 0.293 for pine plantation sites to 0.615 for natural forest oak sites (Table 3.5, Figure 3.4). The mean mass remaining in the litterbags at week 82 ranges from 25.4%

in natural forest oak sites to 51.2% in pine plantation sites (see Appendix H for mass remaining at each collection time). Mass remaining at week 82 is log transformed to meet the assumptions of constant variance. No significant differences in decomposition rate constants are found between the urban forest oak sites and the natural forest oak sites (p=0.4690), the urban forest pine sites and the natural forest pine sites (p=0.7808), as well as the natural forest pine sites and the pine plantation sites (p=0.4368). Additionally, no significant differences in mass remaining at week 82 are found between the urban forest oak sites and the natural forest oak sites (p=0.9008), the urban forest pine sites and the natural forest pine sites (p=0.8969), as well as the natural forest pine sites and the pine plantation sites (p=0.7371).

Mixed species foliar litter in oak sites have larger k values compared to those of pine sites (p=0.0006). At week 82, pine sites have on average 1.76 times (1.12 to 2.78 times, 95% confidence interval, p=0.0196) more mass remaining than oak sites (Table 3.6). Foliar litter of deciduous species has been found to decompose twice as fast as that of evergreen species during the first year of decomposition (Cornelissen, 1996; Prescott et al., 2004). The lower mass of foliar litter remaining at week-82 in the oak dominated sites compared to pine dominated sites in this study corresponds well with the forest floor characteristics across pine and oak dominated land covers (Table 3.1). Pine dominated sites have a larger forest floor mass and depth compared to oak dominated sites (Chapter 2). Pine dominated sites also have a higher rate of foliar productivity compared to oak dominated sites (Chapter 2), and this likely also increases the forest floor depth and mass in these sites.

Direct comparisons of the k values found in the rural sites in this study to others are problematic due to the differences in the duration of decay periods, methodology (single-species litter versus mixed litters) ecosystem type and site management. For example, the k values for the decomposition of slash pine litter over a three year period in a frequently burned longleaf pine-wiregrass ecosystem in Georgia was 0.097 (Hendricks et al., 2002), a value lower than that found for the slash pine plantation sites in this study. For the purposes of this study, it is more useful to compare percent mass remaining at set time periods from the start of decomposition. Gholz et al. (1985) monitored slash pine litter decomposition in a 35 year old slash pine plantation in northern Florida during an unusually dry year and found 85% of the mass remained at 52 weeks. These results are comparable to this study that found approximately 80 to 81.5 % of the mass remained in slash pine plantations between 42 and 60 weeks of decomposition during a drought period. Myrtle oak (*Q. myrtifolia*) leaf litter is found to have approximately 65% of its mass remaining after 52 weeks in a Florida scrub oak community (Hall et al., 2006), which was similar to the 64 and 61% mass remaining between 42 and 60 weeks for the urban forest oak sites in this study.

Decomposition of popsicle sticks

Differences in the decay rates of wooden popsicle sticks have been used to assess the impact of edaphic factors on decomposition rates, since the influence of litter quality has been removed (Baker et al., 2001). Decomposition rate constants for the popsicle sticks calculated from week-12 to week-82 range from 0.361 for pine plantation sites to 0.939 for urban forest oak sites. In week 60, the urban and natural forest land uses are beginning to show a greater amount of separation in the popsicle stick decomposition patterns than had been observed in the previous 42 weeks (Figure 3.5). However, the separation did not persist through week 82. Similar to the litterbag decomposition rates, urban and rural sites are not significantly different in k values (p=0.1495) and popsicle mass remaining at week 82 (p=0.5726). Hence, similar decay constants and mass remaining at week 82 among urban sites and rural sites for the popsicle sticks indicated that edaphic conditions during a drought period are similar between rural oak and urban oak forest sites and rural pine and urban pine forest sites.

Also consistent with the litterbag decomposition results, the popsicle sticks in oak sites have a larger k value than the pine sites (p=0.0073, Table 3.6). This indicates that oak forest floors are a more conducive site for decomposition during a time of drought. For three months between week-60 and -82, all sites received a surplus of precipitation (Figure 3.1). This appears to have improved conditions for decay, which likely resulted in both oak and pine sites having similar popsicle mass remaining at week-82 (p=0.6485).

Litter quality of mixed foliar species

Differences in lignin, N and P concentrations of the site specific litter mixes placed in the litterbags at time 0 are tested among the five land covers (Table 3.5, 3.6). The only significant difference between urban and rural sites is the lower lignin concentration of the foliar mix in the urban forest oak dominated sites compared to the natural forest oak sites (p=0.0010). The lignin to N, C to N and C to P ratios are calculated from the lignin, N, and P concentrations of the site specific litter mixes.

These ratios, especially the C to N and lignin to N ratios, are useful indices of litter quality and are used to gain insight into the influence of litter quality on decomposition rates. The lignin to N, C to N and C to P ratios of the mixed foliar litter are similar between natural oak and urban oak forest sites and natural pine and urban pine forest sites. Differences among pine plantation sites and natural forest pine sites are also non-significant for the three litter quality ratios.

Oak sites have significantly higher litter quality compared to pine sites (Table 3.5, 3.6). The mixed foliar litter from the pine sites have larger values for the lignin to N ratio ($p=0.0010$), C to N ratio ($p=0.0005$) and C to P ratio ($p=0.0003$) than the mixed foliar litter from the oak sites. This is primarily due to pine sites having lower initial litter N concentrations ($p=0.0003$) and P concentrations ($p=0.0002$) compared to the oak sites. Nitrogen concentrations as well as the lignin to N, C to N and C to P ratios are log transformed to meet the assumptions of equal variance for the ANOVA and planned comparisons.

Across all of the forested sites ($n=17$), the decomposition rate constant (k) is significantly positively correlated to initial nitrogen concentration ($r = 0.808$, $p<0.0001$) and to phosphorus concentration ($r=0.695$, $p=0.0020$), but is not related to initial lignin concentration ($r=0.333$, $p=0.1917$, Figure 3.6). Overall, lignin is not a useful indicator of decay during the 82 weeks of decomposition and this may be a result of a narrow range of lignin values for the mixed foliar litters found across all of the sites (147.0 to 178.4 g kg^{-1}). As a result of the positive correlation with nutrient concentration, k values are negatively correlated to the initial lignin to N ($r=-0.687$, $p=0.0023$), C to N ($r=-0.713$, $p=0.0013$) and C to P ($r=-0.678$, $p=0.0028$) ratios. The strong correlation between lignin to N ratio and k is largely due to the significantly higher N concentrations and k values in oak sites compared to pines sites.

Litter quality of individual litter species

Individual species litter quality was measured over two collection periods (Fall 2010 to Spring 2011 and Fall 2011 to Spring 2012, Figure 3.7). Within each collection period, an ANOVA was used with land cover (urban forest or natural forest), litter species (slash pine, laurel oak, or live oak), and the interaction between land cover and species as the primary treatment variables. For both years, the effect of litter species is significant for all comparisons. A Tukey's post hoc test indicates that laurel oak (*Q. laurifolia*) and live oak (*Q. virginiana*) foliar litter have significantly higher nitrogen concentrations than the slash pine (*P. ellottii*) foliar litter for both years. During the first year of the study, laurel oak and slash pine foliar litter have similar lignin concentrations that are higher than that of live oak. In the second year, lignin concentration is significantly different among the three litter types and increases along the following gradient: live oak litter < slash pine litter < laurel oak litter. Despite the differences in lignin concentration, both oak foliar litters have similar lignin to N ratios. The lignin to N ratio for slash pine foliar litter is significantly higher than both oak foliar litters during both years.

The interaction between land use and species is non-significant for all comparisons; therefore the apparent effect of urbanization is similar across all three species. Land use has a significant effect on litter quality in the first year of collection, with individual species litter in urban forests having lower lignin to N ratios compared to those species in natural forests (p=0.0103). These differences are partly driven by the significantly lower lignin concentration (p=0.0002) and higher N concentration (p=0.0152) of the individual species foliar litter in the urban forest compared to those in the natural forest. This effect of land use is not as strongly expressed in the second year, as the lignin to N ratio, N concentration and lignin concentration of individual species litter in urban forests is not significantly different than those species in natural forests (p=0.0751). Variation in litter quality across time for a single species foliar litter along an urban to rural gradient was found in the New York Metropolitan area. Urban and rural red oak (*Q. rubra* L.) had similar lignin to N ratios in 1989 (Pouyat and Carreiro, 2003), but in 1990 the foliar litter from urban sites had higher ratios than those found in rural sites (Carreiro et al., 1999).

As urban forested sites often experience higher rates of N deposition compared to rural sites, it is suggested in the literature that trees in urban areas would also have higher N concentrations in green leaves and foliar litterfall compared to trees growing in rural areas. This theory originates from studies across historic N pollution gradients, such as those in southern California (Fenn 1991) and in the southeastern US (Boggs 2005), where single species foliage have been found to have higher N concentrations in more polluted forested sites than less polluted sites. Indeed, foliar N concentration of a single species (slash pine, Chen et al., 2010) and of a mix of several tree species (either broadleaves or pine, Fang et al., 2011) have been found to be significantly higher in urban areas compared to rural areas in subtropical China. However, red oak foliar litter in urban and rural forests did not exhibit significantly different N concentrations (Carreiro et al., 1999; Pouyat and Carreiro, 2003) across a measurable

N pollution gradient in the New York City metropolitan area (Lovett et al., 2000). Different plant species may exhibit different sensitivities to N pollution and environmental conditions, as has been observed for the foliar N concentration of several tree species to changes in elevation, temperature, precipitation and N deposition (Aber et al., 2003).

Even if single species are sensitive to the effects of urbanization on foliar litter quality, the response of this single species may not impact the overall litter quality signature of the mixed foliar litterfall occurring within the urban sites. Slash pine, laurel oak and live oak are major contributors to the litterfall in both rural and urban sites in this study; however these species are not the only contributors to the litterfall (Table 3.2). The lower lignin and higher N concentration observed for the single species litterfall in urban sites versus rural sites did not translate into consistent differences between sites in the mixed foliar litterfall (in the time zero litterbags, Table 3.6). A recent study from the Boston metropolitan area, found “[p]atterns in foliar N appeared to be driven more strongly by changes in species composition rather than phenotypic plasticity across the urbanization gradient” (Rao et al., 2013). Understanding the role of the urban environment in litter N concentration is complicated by changes in species composition along an urban to rural gradient (Rao et al., 2013).

Nitrogen mineralization and immobilization patterns in decomposing litter

All of the litters exhibited very similar patterns in N release and immobilization over the first 60 weeks of decomposition, with a period of N loss around week four and an immobilization period that lasted from week six to week 60 (Figure 3.8, see Appendix

I for N remaining at each collection time). From week 6 to week 60, the mean percent N remaining for each land use is not significantly different from 100 (except for the natural forest oak dominated cover in week-12). By week 82, the oak dominated sites and the natural forest pine sites have percent N remaining values that are not significantly different from 100 % but they appear to be trending towards N mineralization ($0.05 \leq p \leq 0.10$). The urban forest pine dominated land cover has a mean percent N remaining below 100%; however it does not have a percent N remaining different from 100 ($p=0.2230$). The pine plantation sites are solidly in an immobilization phase at week-82. The power of the paired t-tests between N remaining in week 0 and the other collection periods is low because the sample size is too small to overcome the higher variability within these land covers.

Mean values for nitrogen remaining at week 82 ranges from 44.9% for urban forest oak sites to 102.6% for pine plantation sites. An analysis of the data at week-82 indicates urban forests have lower percent N remaining compared to rural forests ($p=0.0238$) and pine dominated forests have a higher percent N remaining compared to oak sites ($p=0.0106$). Due to the high variability in both urban forest land covers, a square root transformation is required to meet the assumption of equal variance in order to test for differences between land covers for week-82 percent N remaining. These differences indicate that urban forests are not as N limited as rural forests. Furthermore, it appears that oak sites are not as N limited as pine sites, regardless of whether they are in an urban or rural setting.

Berg and Staaf (1981) detailed three phases of N release in decomposing litter: an initial leaching phase, followed by immobilization, and then finally mineralization.

The duration of the study may determine whether a mineralization phase is observed. Several decomposition studies in the literature as well as this study did not record a final mineralization phase, but instead are dominated by immobilization of N at the end of the study period. For example, in a decomposition study along an urban to rural gradient stretching across New York to Connecticut, both rural and urban red oak (*Q. rubra*) litter decomposing in their site of origin showed immobilization of N over a 22 month decay period even though samples had lost 60% of their mass (Pouyat and Carreiro, 2003). Another study found that European aspen (*Populus tremula* L.) decomposing in urban and rural sites in Finland was dominated by N immobilization over a 13 month study period (Nikula et al., 2010). Immobilization of N has also been observed when myrtle oak (*Q. myrtifolia*) was decomposing over 36 months in natural scrub oak stands in Florida (Hall et al., 2006) and when slash pine litter was decomposing over 25 months in 35 year old pine plantation stands in northern Florida (Gholz et al., 1985).

In a global-scale experiment investigating the N release patterns of litter decomposing in 21 sites covering seven biomes over a 10 year period, N immobilization was tied to both the initial litter N concentration and the C to N ratio of the decomposing material (William et al., 2007). Net N release started when the average C to N ratio of the decomposing material was less than 40, with a range around this number from 31 to 48. Litters with higher initial N concentrations released N at much lower mass remaining values than those litters that had very low initial N concentrations. From week six through 60 in this study, the oak dominated forests have mean C to N ratios greater than 38 and the pine dominated forests have mean C to N ratios greater than 102 (see Appendix I), which may explain why all of the sites are dominated by N immobilization during this time period. At week-82, it appears that the oak dominated sites with mean C to N ratios between 28 and 50 are close to reaching the mineralization phase. The mean C to N ratio for all the pine sites is above 79. Therefore it is uncertain how close the pine sites are to reaching the mineralization phase.

During the initial leaching phase (week four) of N observed in this study, the amount of N remaining in the litter dropped on average between 27 and 56 % of the original amount across the different land uses. By week 6, the amount of N remaining in the litter returned to values close to 100 % for all of the land uses. Typically, when N is observed to be immobilized in decomposing litter, it is likely that microbes conserve N from the decomposing material. However, very little mass is lost from week four to week six, therefore microorganisms likely accessed N exogenous to the litter to convert it into microbial biomass. An increase in N in foliar litter found on the surface of the forest floor has been shown to come from atmospherically deposited N (Kochy and Wilson, 1997). Fungi have been found to translocate soil-derived N to the litter layer (Hart and Firestone, 1991; Frey et al., 2003), such that an absolute increase in N content of the decomposing litter was measured (Hart and Firestone, 1991). For leaves found in close proximity to each other, without a fungal presence, nitrogen has also been found to move between microbial communities growing on different sets of leaves (Schimel et al., 2007). The amount of N transferred was found to be controlled by the microbes growing on the source leaves. Therefore, N could have been added from sources above and below the leaves by both abiotic and biotic means in this study.

Phosphorus mineralization and immobilization patterns in decomposing litter

As for P remaining over time, the oak sites are dominated by immobilization from week two through week 42 as values are not significantly different from 100 (Figure 3.9a, see Appendix J for P remaining at each collection time). By week 60, both oak sites display mineralization of P. Paired t-tests indicate that natural forest oak sites and urban forest oak sites have percent P remaining lower than 100 % ($p=0.0029$ and $p=0.0026$, respectively). The trend of P mineralization continues through week-82. In comparison, the pine sites tend to have a lot more variability in percent P remaining throughout the study period (Figure 3.9b). The immobilization trend is particularly strong for the pine plantation sites, with mean percent P remaining being statistically similar to 100 % or trending towards values larger than 100% in week-4 ($p=0.0879$), in week-28 ($p=0.2349$), and in week-82 ($p=0.0776$). The percent P remaining for the natural forest pine sites and the urban forest pine sites tend to oscillate around 100 % with periods of immobilization and mineralization for both land covers.

Mean values for phosphorus remaining at week 82 ranges from 47.9% for urban forest oak sites to 148.9% for pine plantation sites. An analysis of the percent P remaining at week-82 indicates that urban forest oak and urban forest pine sites are not significantly different from their natural forest counterparts ($p=0.2380$ and $p=0.4502$, respectively). Pine plantation sites at week-82 had significantly more phosphorus remaining in the decomposing litter compared to that remaining in the decomposing litter in the natural forest pine sites ($p=0.0005$, Table 3.6). Pine dominated sites at week-82 have a higher percent P remaining in the decomposing litter compared to oak dominated sites ($p=0.0002$).

The three stages of nutrient dynamics (leaching, immobilization and mineralization) exhibited by N in decomposing litter has also been observed for P in decomposing litter (Blair, 1988). Similar to the slash pine plantations in this study, immobilization of P during decomposition of pine needles has been observed in slash pine plantations in Florida (Gholz et al., 1985) and in loblolly plantations growing in sandy, nutrient poor soils in North Carolina (Sanchez, 2001). At the end of 12 and 24 months of decay, slash pine needles decomposing in a 35 year old slash pine plantation site in Florida had 125 % and 190 % of the original P remaining, respectively. At the end of three years of decay, loblolly pine needles decomposing in a plantation site had around 100 % of the original P remaining and a C to P ratio of 1000 (Sanchez, 2001). Decomposition of chestnut oak (*Q. prinus*) in the Appalachian mountains in North Carolina did not show a consistent net release of P while decomposing and at the end of three years of decay had a final C to P ratio of 508 and a percent P remaining around 100 (Blair, 1988). Additional studies which examined oak litter decomposition and P dynamics in the southeastern coastal plain could not be found.

Forest decomposition studies have reported critical C to P ratios around 550 for loblolly pine (*P. taeda*) litter (Saggar et al., 1998) and between 360 and 480 for hardwood litter (Gosz et al., 1973) to initiate P mineralization. For this study, decomposing oak litter had mean C to P ratios of 318 and 339 in the natural forest sites and urban forest oak sites, respectively, at week-82 (Appendix J). With the percent P remaining significantly below 100 % in both of these land uses at week-60 and for urban forest oak sites in week-82, it is likely that P is mineralized in the oak sites by the end of the study period. Although the percent P remaining in natural forest pine sites is lower than 100 at week-60 and the urban forest sites have values below 100 at week-82, all of the pine sites had C to P ratios above 738 during this time. Carbon to phosphorus ratios are also very high (>1500) for all of the pine sites during week 18. It is likely that any significant decrease of P below 100 observed during the decomposition period for the pine sites is due to leaching rather than microbial mediated mineralization of P.

Summary and conclusions

Urbanization does not appear to impact decay rates and N and P immobilization/mineralization patterns of decomposing litter in this study. This may indicate that the change in the abiotic and biotic factors that occur from a low population density urban environment do not produce enough of an effect that can be identified over the natural variation associated with decomposition in this region. For example, litter quality of individual litters in urban areas in this study can be lower (higher lignin to N ratio) than rural forests but an urbanization effect on lignin to N ratio in the mixed foliar litters placed in the litterbags is not observed. Litter quality of mixed litters appears to be dominated by changes in species composition between sites. It is important to note that this research took place during a drought that persisted during the study period (2010 – 2012). Therefore, low available moisture in the forest floor across all sites may have limited biological activity and masked any effect that urbanization may have had on the forest floor microbial community, and, subsequently, decomposition. For instance, even though urban sites have lower maximum temperatures, higher minimum temperatures and narrower temperature ranges compared to rural sites during most of the 82 weeks of decomposition, it appears that this difference in temperatures during a drought period are not reflected in higher decay rates.

Drought conditions did not mask the effect of overstory species on decomposition rates and nutrient dynamics in decomposing litter over 82 weeks of decay. Oak dominated sites had higher litter quality (as measured by lignin to N, C to N ratios, and N and P concentrations) compared to pine dominated sites. As a result of the interactions between high litter quality and more favorable edaphic conditions (as indicated by decay rates of popsicle sticks), the decomposition rate of oak litter is faster than that of pine litter over the 82 weeks of decay. The difference between oak and pine dominated sites in their nutrient release patterns is also likely due to the higher initial nutrient concentrations in the oak litter compared to the pine. In summary, both N and P are largely immobilized over the first 42 weeks of decomposition across all sites, with periods of leaching occurring during week-4 for N (oak and pine) and around week-18 for P (pine only). Oak sites appear to be trending towards the mineralization of N and P at week-60 and have P mineralization at week-82. The pine sites are likely dominated by immobilization of N and P from week-42 through week-82, with some losses of P occurring via leaching in week-60.

Nitrogen and phosphorus are important limiting nutrients in terrestrial ecosystems and the availability of these nutrients commonly limits the growth of pine forests in the southern United States (Allen, 1987). The immobilization of N and P in forest floor materials can be considered at best a slow-release fertilizer and at the worst a mechanism for long-term storage of this nutrient. Although N and P are largely immobilized over the 60 weeks of decomposition for the oak sites, it appears that both nutrients after 82 weeks may be mineralized and plant accessible over time. It is likely for the pine sites in this study that N and P are stored in the forest floor and that release is very slow and plant access is limited.

The differences between oak and pine dominated sites in regards to decomposition rates, nutrient dynamics in decomposing litter, and litter quality seem to fit the classic hypothesis that hardwoods improve soil quality, increase nutrient cycling, and promote active soil biota while conifers do not. However, there is evidence that decomposition rates and carbon and nitrogen mineralization rates in stands dominated by conifers may be the same or higher than those dominated by deciduous species (Reich et al., 1997, Giardina et al., 2001; Prescott et al., 2004; Mueller et al., 2012). Rapid initial decomposition rates of high quality deciduous tree litter may only be temporary as long-term litter decomposition studies have found that over time the amount of mass remaining of both high quality deciduous litter and low-quality conifer litter converge (Berg, 1986; Prescott et al., 2000, 2004; Giardina et al., 2001). Furthermore, differences in biogeochemical cycling in a forest stand attributed to tree species alone may be related to variation in forest stand management, stand age, site history, and soil type (Binkley, 1995, Reich et al., 1997, Mueller et al., 2012). Nonetheless, in regard to urbanization studies, it is important to account for the overstory composition of a forest stand as this may reflect differences in stand origin that then may impact biogeochemical cycling and the overall effect of urbanization.

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Table 3.1. Plant and forest floor characteristics for select land covers in western Florida. Standing crop biomass and stem density includes stems with diameter at breast height ≥ 10 cm. Standard errors of the means are in parentheses.

Land cover	**Stem density (stems ha^{-1})**		**Standing crop biomass (g m^{-2})**	**Foliar productivity (g m^{-2} yr^{-1})**	**Forest floor depth (cm)**	**Forest floor mass (g m^{-2})**	**Common overstory tree species**
	Total	**Hard-woods**					
Natural Forest Oak	500 (49)	450 (40)	22,993 (8,076)	523.7 (30.9)	3.45 (0.74)	3,981 (1,493)	*Quercus hemisphaerica* *Q. virginiana*
Urban Forest Oak	494 (61)	394 (21)	23,527 (4,256)	686.4 (81.9)	4.11 (1.38)	4,544 (1,677)	*Q. hemisphaerica* *Q. nigra*
Natural Forest Pine	675 (38)	292 (51)	35,559 (3,375)	638.4 (41.8)	8.80 (2.12)	10,230 (2,051)	*Pinus elliottii*
Urban Forest Pine	650 (109)	233 (110)	31,042 (9,516)	779.7 (78.7)	8.30 (1.26)	8,807 (2,376)	*P. elliottii*
Pine plantation	917 (183)	92 (92)	15,152 (4,703)	615.1 (66.6)	6.10 (1.20)	6,216 (1,150)	*P. elliottii*

Table 3.2. Relative proportion (%) by mass of each plant species contained in the litterbags. These species are major components of the foliar litterfall in the 17 forested sites in western Florida (columns sum to 100 %). In sites with less than three species in the litterbags, the remaining composition includes many species, none of which occupy a dominant proportion. Additional species are either not present in the foliar litterfall (o) or are present (+), but not in enough quantity to include in the litterbag. Drainage classes within land covers are moderately well drained (MWD), somewhat poorly drained (SPD), poorly drained (PD) and very poorly drained (VPD).

	Natural Forest Oak				**Urban Forest Oak**				**Natural Forest Pine**			**Urban Forest Pine**			**Pine Plantation**		
	MWD	**SPD**	**PD**	**VPD**	**MWD**	**SPD**	**PD**	**VPD**	**SPD**	**PD**	**VPD**	**SPD**	**PD**	**VPD**	**SPD**	**PD**	**VPD**
Cinnamomum camphora	o	o	o	o	o	**6**	**+**	**6**	o	o	o	o	o	**12**	o	o	o
Pinus echinata	**3**	o	o	o	**57**	o	o	o	o	o	o	o	o	o	**15**	o	o
P. elliottii	o	o	o	**+**	o	**+**	**51**	**62**	**78**	**91**	**76**	**89**	**100**	**68**	**85**	**100**	**100**
P. palustris	o	o	o	o	o	o	o	o	o	o	**22**	o	o	o	o	o	o
Quercus geminata	**23**	o	o	o	**40**	o	o	o	o	o	o	o	o	o	**+**	o	o
Q. hemisphaerica	**74**	**54**	**46**	o	**+**	**57**	**38**	o	**+**	**5**	o	o	o	o	o	o	o
Q. laurifolia	o	o	o	**80**	o	o	o	**+**	o	o	**+**	o	o	o	o	o	o

Q. myrtifolia	o	o	o	o	**3**	o	o	o	o	o	o	o	o	o	o	o	o
Q. nigra	o	o	o	**14**	o	**37**	**+**	**32**	**17**	**4**	o	**11**	**+**	**20**	**+**	o	**+**
Q. virginiana	o	**46**	**54**	**6**	o	**+**	**11**	o	**5**	**+**	**3**	o	o	o	o	o	o

Table 3.3. Seasonal mean, minimum and maximum forest floor temperatures (^oC) and the temperature range for the forested sites in the Florida panhandle. Temperatures were measured hourly over the 82 week period of the decomposition study and averaged to obtain seasonal maximum, mean, and minimum values. Daily temperature ranges were calculated from the difference of daily maximum and minimum temperatures and then averaged for each season. Summer = June, July, August; Fall = September, October, November; Winter=December, January, February; Spring=March, April, May. Standard errors of the means are in parentheses.

	Spring	**Summer**	**Fall**	**Winter**	**Spring**	**Summer**	**Fall**	**Winter**
	Mean temperature				Temperature Range			
Natural Forest Oak	21.7 (0.1)	26.3 (0.1)	21.8 (0.1)	14.9 (0.1)	12.7 (0.3)	8.4 (0.8)	9.0 (0.8)	8.3 (0.6)
Urban Forest Oak	21.5 (0.0)	26.5 (0.1)	22.2 (0.1)	15.4 (0.1)	5.8 (0.6)	4.2 (0.6)	4.8 (0.2)	5.5 (0.3)
Natural Forest Pine	21.4 (0.5)	26.0 (0.3)	21.6 (0.4)	14.8 (0.3)	13.0 (1.8)	7.9 (0.7)	7.8 (1.3)	7.8 (0.3)
Urban Forest Pine	21.6 (0.1)	26.5 (0.1)	21.9 (0.2)	15.4 (0.3)	9.1 (2.3)	6.7 (1.6)	6.2 (0.9)	6.3 (1.4)
Pine Plantation	21.5 (0.5)	26.4 (0.4)	21.6 (0.5)	14.6 (0.2)	15.1 (2.0)	10.8 (1.1)	10.3 (2.1)	9.0 (2.0)
	Maximum temperature				Minimum Temperature			
Natural Forest Oak	29.5 (0.3)	31.7 (0.7)	27.4 (0.6)	19.7 (0.5)	16.8 (0.2)	23.3 (0.2)	18.4 (0.3)	11.5 (0.2)

Urban Forest Oak	24.7 (0.4)	28.9 (0.5)	24.8 (0.1)	18.3 (0.2)	18.9 (0.2)	24.6 (0.1)	20.0 (0.1)	12.9 (0.2)
Natural Forest Pine	29.6 (2.1)	31.1 (0.9)	26.4 (1.5)	19.1 (0.6)	16.6 (0.3)	23.2 (0.2)	18.5 (0.3)	11.3 (0.4)
Urban Forest Pine	26.8 (1.5)	30.6 (1.1)	25.2 (0.3)	18.7 (0.5)	17.7 (0.9)	23.9 (0.4)	19.1 (0.5)	12.4 (0.9)
Pine Plantation	31.0 (2.1)	33.4 (1.37)	28.0 (2.1)	19.8 (1.7)	15.9 (0.3)	22.7 (0.3)	17.8 (0.1)	10.9 (0.3)

Table 3.4. Results of the contrast statements for differences among urban forest and rural forests and for pine forests and oak dominated forests. Contrasts with p-values less than 0.10 are reported for a subset of the data (n=16) with one of the urban forest oak sites removed as an outlier (due to unusual maximum temperatures). The results of the contrast statements using the entire data set (n=17) for the same set of contrasts as the subset of data are also reported.

	Subset of sites (n=16)				**All sites (n=17)**			
Response Variable	**Point Estimate**	**Margin of Error**	**p-value**	**Signif-icance**	**Point Estimate**	**Margin of Error**	**p-value**	**Signif-icance**
			Contrast: Urban forest – rural forest					
Mean temperature, Winter	0.64	0.50	0.0181	*	0.63	0.44	0.0101	*
Maximum temperature, Fall	-2.22	2.37	0.0631		-1.72	2.34	0.1307	
Maximum temperature, Spring	-4.30	2.75	0.0070	*	-3.34	3.21	0.0432	*
Maximum temperature, Summer	-2.26	1.91	0.0260	*	-1.57	2.25	0.1481	
Minimum temperature, Fall	1.26	0.70	0.0032	*	1.24	0.61	0.0013	*
Minimum temperature, Spring	1.76	0.97	0.0031	*	1.77	0.85	0.0011	*
Minimum temperature, Summer	1.17	0.57	0.0016	*	1.21	0.51	0.0005	*
Minimum temperature, Winter	1.34	1.13	0.0264	*	1.32	0.99	0.0153	*
Temperature range, Fall	-3.47	2.37	0.0096	*	-2.96	2.36	0.0196	*

Temperature range, Spring	-6.06	2.96	0.0015	*	-5.11	3.34	0.0072	*
Temperature range, Summer	-3.43	2.07	0.0051	*	-2.78	2.31	0.0235	*
Temperature range, Winter	-2.30	2.50	0.0668		-2.23	2.19	0.0473	*
Pine forest – Oak forest								
Minimum temperature, Fall	-0.84	0.70	0.0245	*	-0.84	0.65	0.0169	*
Minimum temperature, Spring	-1.20	0.97	0.0020	*	-1.19	0.90	0.0148	*
Minimum temperature, Summer	-0.74	0.57	0.0179	*	-0.74	0.54	0.0124	*
Temperature range, Spring	3.24	2.96	0.0358	*	3.24	3.52	0.0679	
Temperature range, Summer	2.28	2.07	0.0348	*	2.28	2.44	0.0641	

Table 3.5. Initial litter quality indices for the mixed species litter placed in litterbags within naturally regenerating forests, urban forests, and pine plantation sites in western Florida. Decomposition rates constants (k) of the mixed species foliar litter were calculated over an 82 week long litterbag study and the mass remaining (%), C to N and C to P ratios of the mixed species foliar litter at week-82 was measured. Standard errors of the means are in parentheses.

Land Use	**Week 0**						**Decay constant**	**Week 82**		
	Lignin ($g\ kg^{-1}$)	**N ($g\ kg^{-1}$)**	**P ($g\ kg^{-1}$)**	**Lignin to N**	**C to N**	**C to P**	**k**	**Mass remaining (%)**	**C to N**	**C to P**
Natural Forest Oak	178.4 (2.0)	8.40 (1.19)	1.19 (0.19)	19.2 (1.9)	64.8 (11.7)	472 (89)	0.615 (0.042)	25.4 (1.7)	28.3 (2.8)	318 (32)
Urban Forest Oak	147.0 (4.5)	7.33 (0.77)	1.20 (0.16)	20.7 (1.9)	71.2 (7.9)	440 (60)	0.555 (0.100)	34.5 (6.8)	44.9 (13.1)	339 (71)
Natural Forest Pine	157.5 (3.7)	3.43 (0.65)	0.47 (0.16)	49.5 (7.2)	161.9 (26.9)	1445 (404)	0.368 (0.052)	47.1 (6.1)	86.1 (4.3)	738 (96)
Urban Forest Pine	155.9 (3.4)	4.33 (1.18)	0.36 (0.09)	43.9 (12.2)	143.0 (39.1)	1627 (337)	0.341 (0.084)	49.5 (8.5)	81.0 (13.3)	1038 (364)
Pine Plantation	156.4 (5.5)	3.12 (0.15)	0.29 (0.04)	50.4 (3.8)	165.1 (7.3)	2051 (332)	0.293 (0.033)	51.2 (3.7)	95.9 (13.2)	795 (192)

Table 3.6. Results of the contrast statements for differences among land covers for initial litter quality parameters, decay constants (k) of mixed species litter and popsicle sticks over an 82 week long litterbag study, and mass and nutrients remaining in the litterbags at week 82. Only contrasts with a p-value ≤ 0.10 are reported. Natural forest oak = NFO; Natural forest pine = NFP; Pine plantation = PP; and Urban forest oak = UFO.

Response Variable	Contrast	Point Estimate	Margin of Error	p-value	Significance
Initial Litter Quality for litterbags (Week-0)					
Lignin (g kg^{-1})	UFO - NFO	-27.5	12.59	0.0010	*
N (g kg^{-1})	Pine forest – Oak forest	-4.92	T[a]	0.0003	*
P (g kg^{-1})		-0.831	0.361	0.0002	*
Lignin to N ratio		28.1	T	0.0010	*
C to N ratio		95.6	T	0.0005	*
C to P ratio		1247	T	0.0003	*
Foliar litter decomposition					
k	Pine forest – Oak forest	-0.308	0.134	0.0006	*
Mass remaining (%),	Pine forest – Oak forest	20.2	T	0.0216	*

week-82

Nutrients remaining in litterbags (Week-82)					
N remaining (%)	Urban forest-Rural forest	-25.5	T	0.0238	*
	Pine forest – Oak forest	33.8	T	0.0106	*
P remaining (%)	NFP - PP	-65.9	25.46	0.0005	*
	Pine forest – Oak forest	55.7	17.90	0.0002	*
C to N ratio	Pine forest – Oak forest	49.0	29.23	0.0054	*
C to P ratio		496	527.4	0.0617	
Popsicle decomposition					
k	Pine forest – Oak forest	-0.359	0.235	0.0073	*

[a]Data are transformed (T) and a 95% confidence interval is not symmetric around the mean. Confidence intervals are given as ratios in text.

Figure 3.1. Monthly mean precipitation (cm) for the study area, measured at Apalachicola Airport, Apalachicola, Florida. Data is reported for (a) the historical mean (1900-2010) and the study period (July 2010 to June 2012) precipitation and for (b) the departure of the monthly precipitation during the study period from the long term monthly mean. Numbers from 0 to 82 above monthly precipitation bars in (a) indicate what litterbag collection (week) took place in that month.

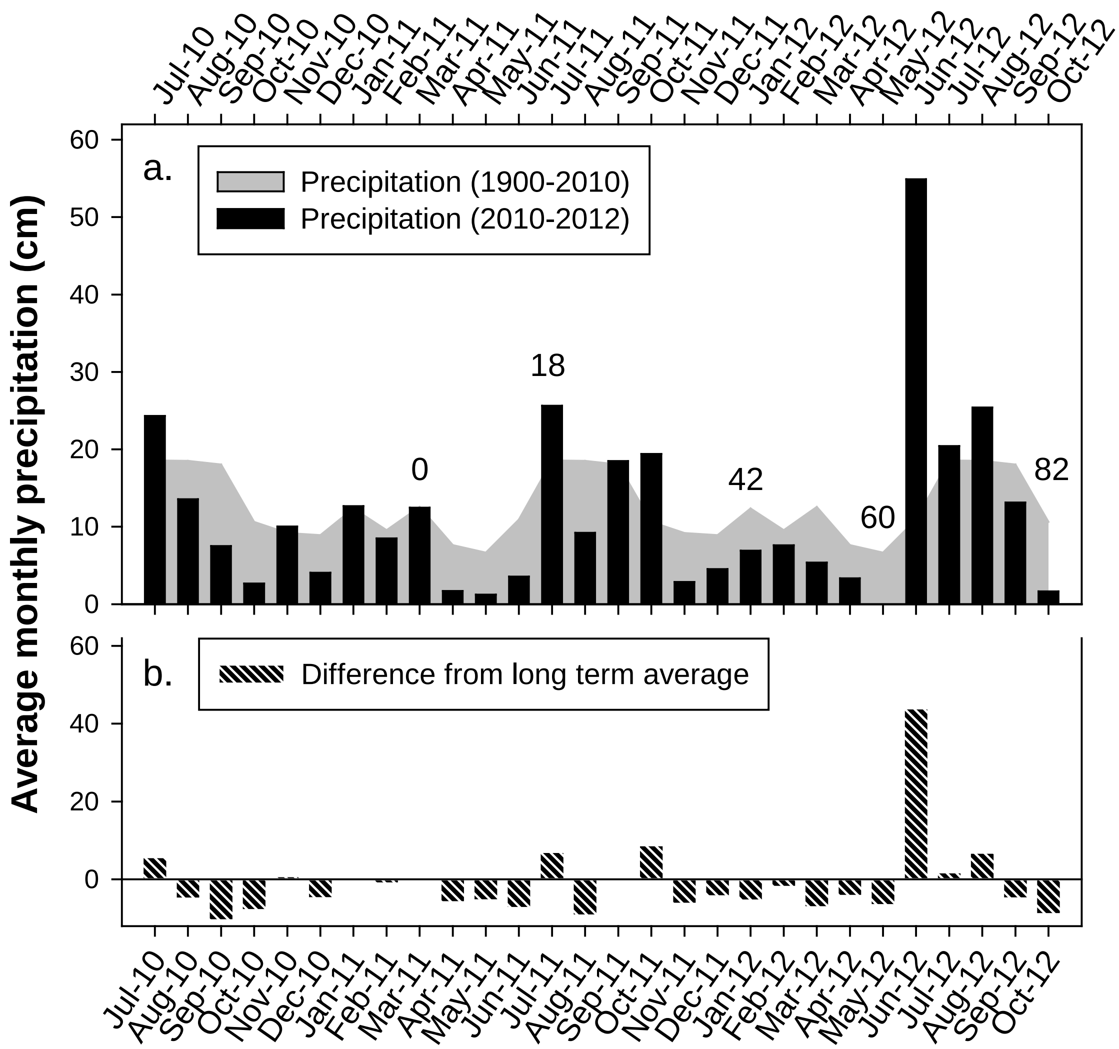


Figure 3.2. Monthly mean air temperature (^{o}C) for the study area, measured at Apalachicola Airport, Apalachicola, Florida. Air temperature is reported as a historic (1977-2000) mean and for the study period (2010-2012). Monthly mean forest floor temperatures are for all forested sites in this study. Standard error bars are present but are so narrow that they are not visible at the scale of this figure.

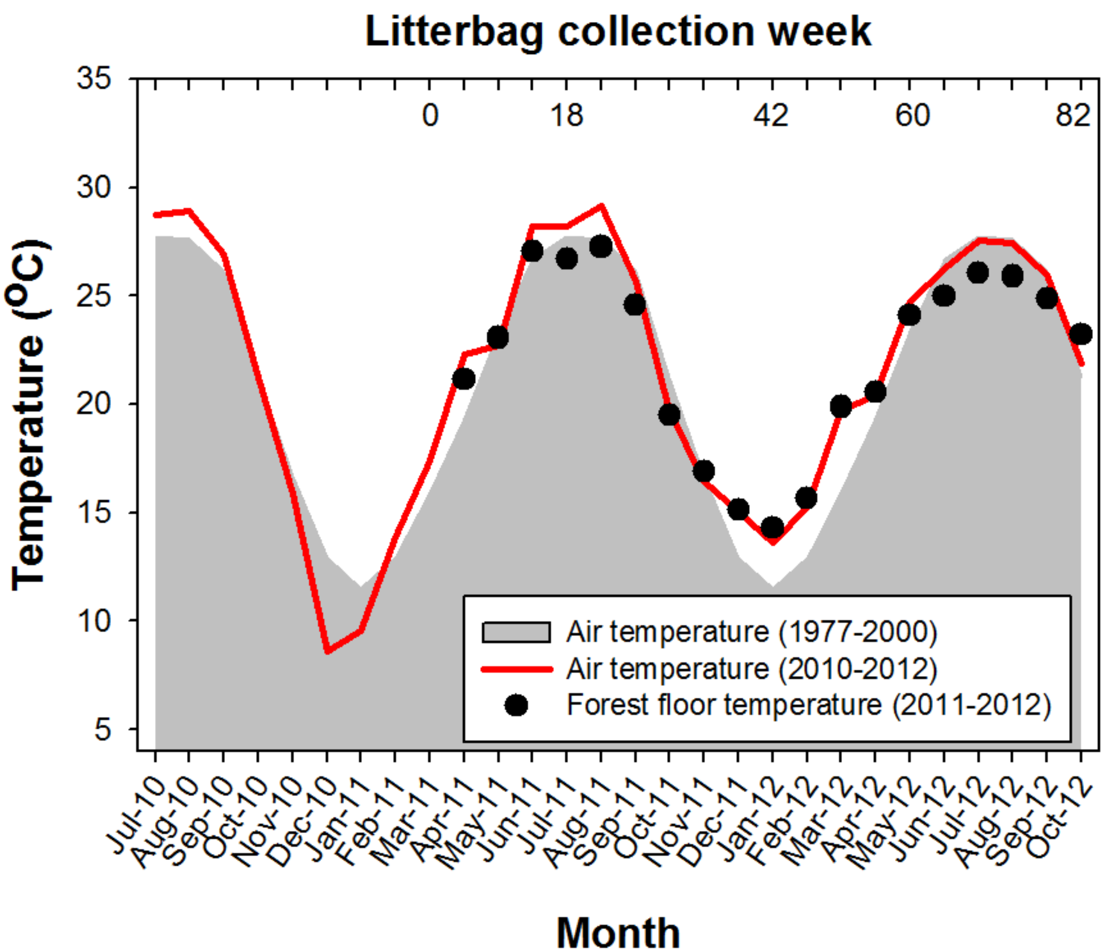


Figure 3.3. Forest floor temperatures (^{o}C) for the oak dominated and pine dominated forest sites in western Florida (n=16). Temperatures are reported for the oak sites as (a) maximum values, (b) minimum values, and (c) temperature range on a monthly basis. Temperatures are reported for the pine sites as (d) maximum values, (e) minimum values, and (f) temperature range on a monthly basis. Vertical bars represent one standard error from the mean.

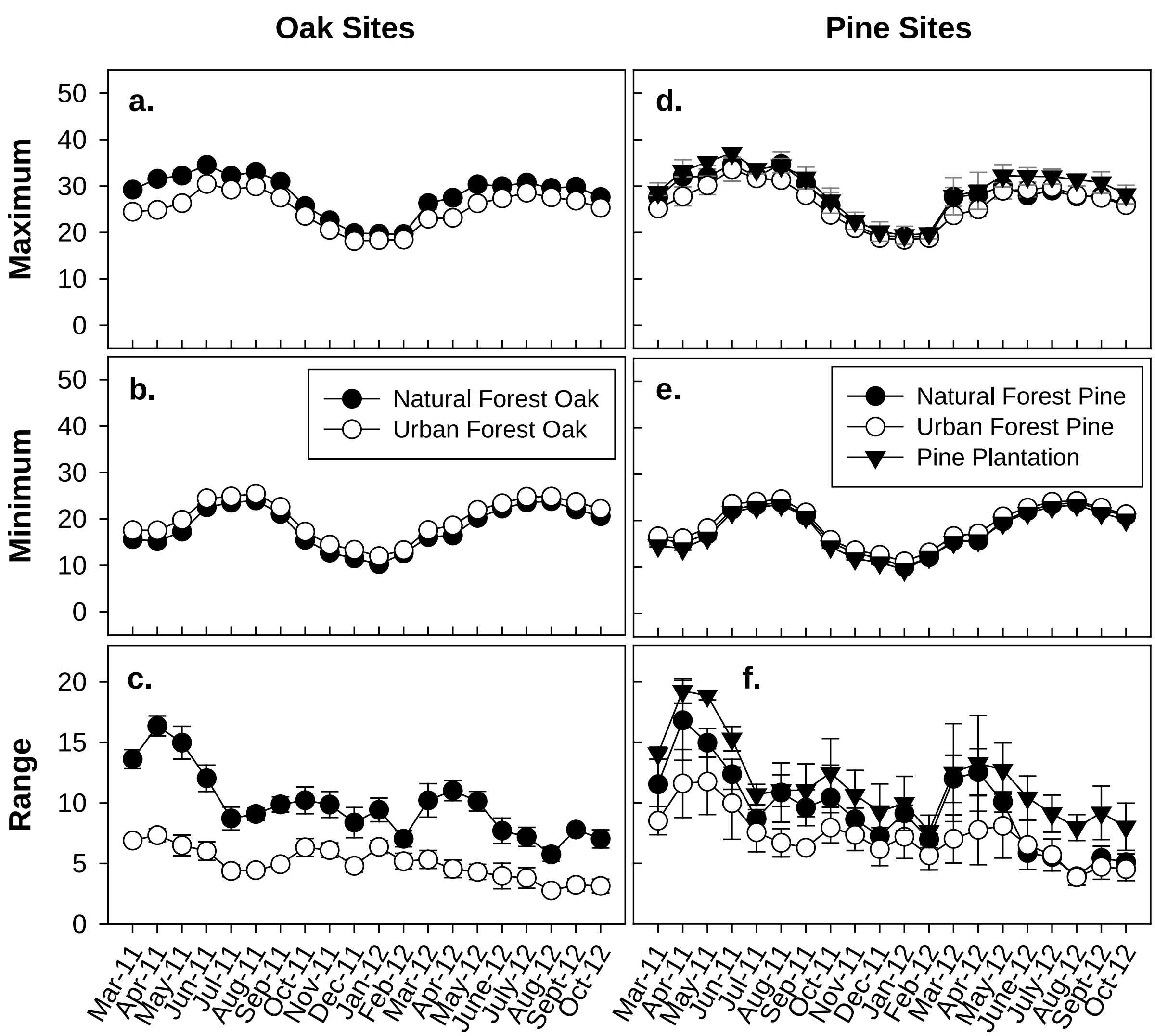


Figure 3.4. Percent mass remaining of foliar litter by (a) oak and (b) pine dominated land use during 82 weeks of decomposition (March 2010 – October 2012) in western Florida. Vertical bars represent one standard error from the mean.

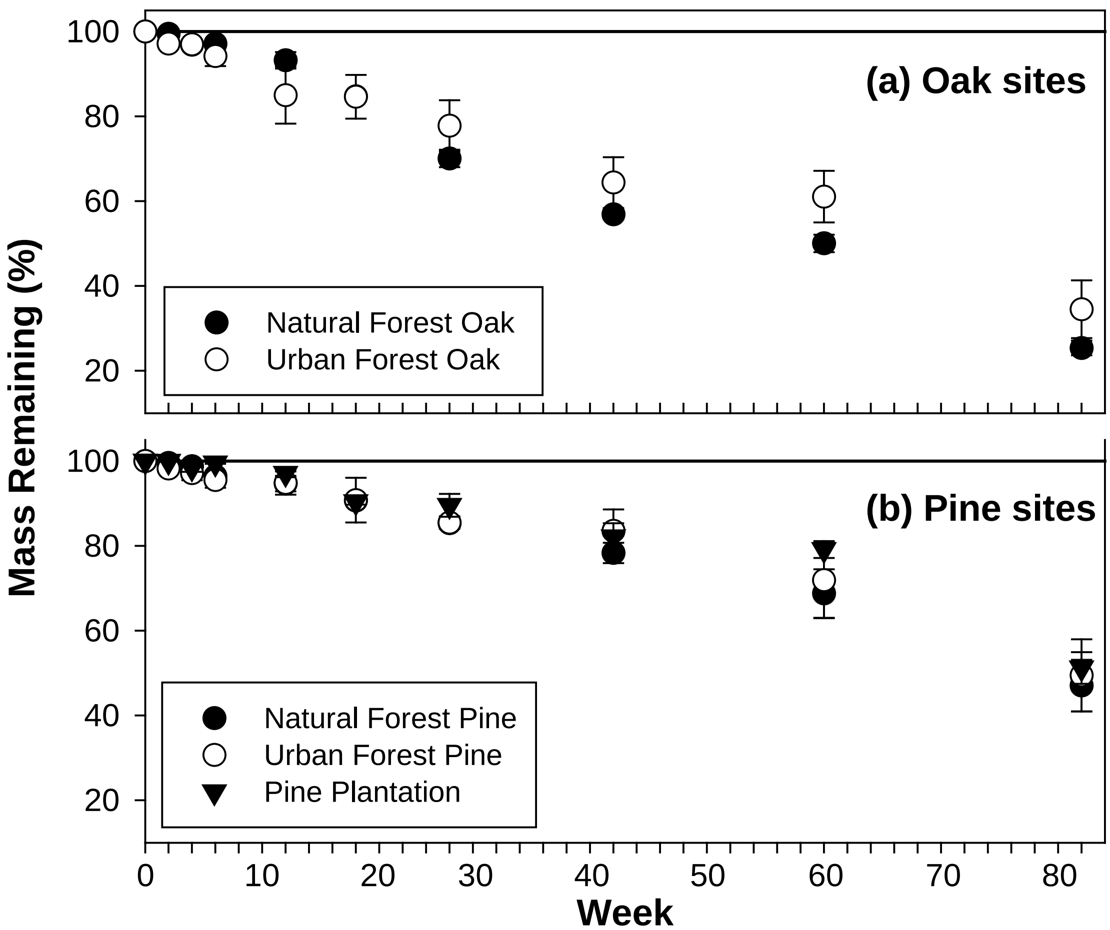


Figure 3.5. Percent mass remaining of a common substrate (popsicle sticks) by (a) oak and (b) pine dominated land use during 82 weeks of decomposition (March 2010 – October 2012) in western Florida. Vertical bars represent one standard error from the mean.

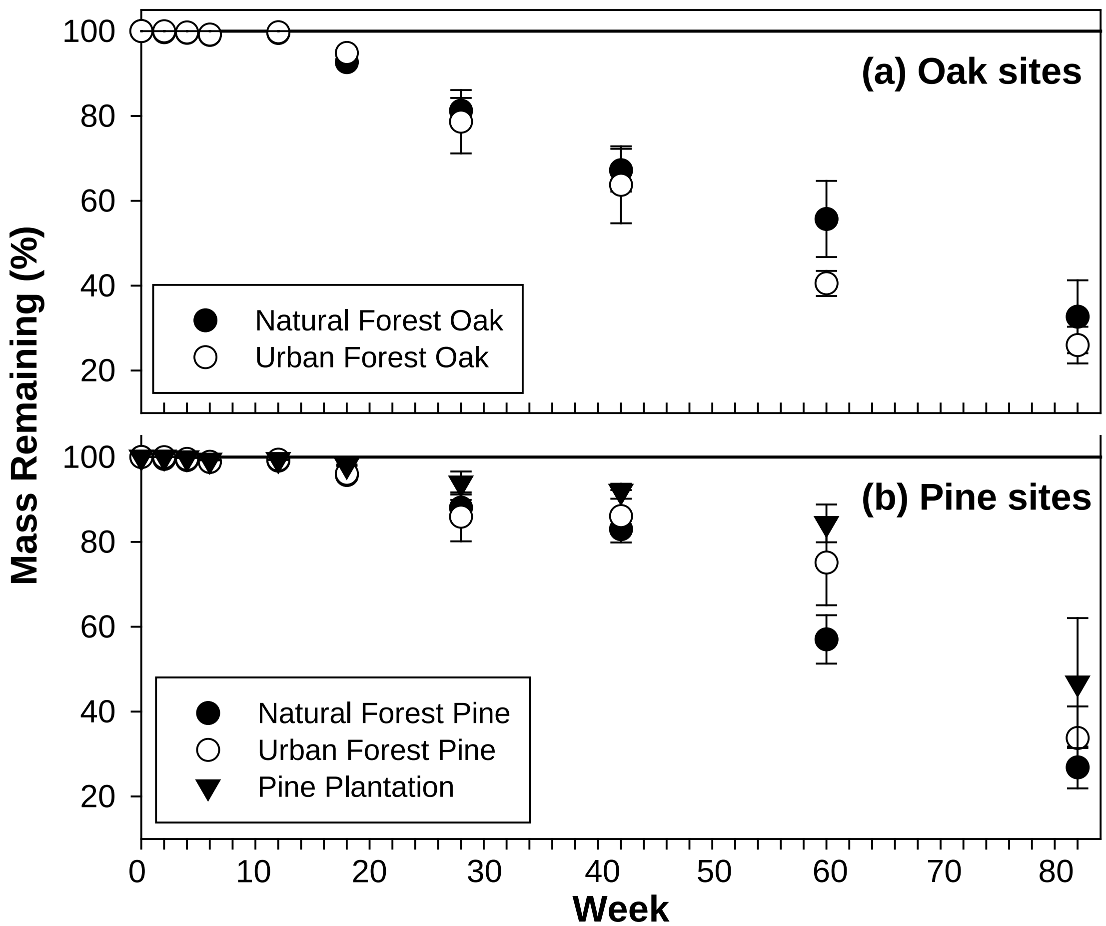


Figure 3.6. Pearson's correlation coefficients and associated p-values between the decomposition rate constant (k) and the initial litter quality parameters of (a) lignin to N, (b) C to N, (c) lignin content (g kg^{-1}) and (d) nitrogen content (g kg^{-1}) for the 17 forested sites in western Florida.

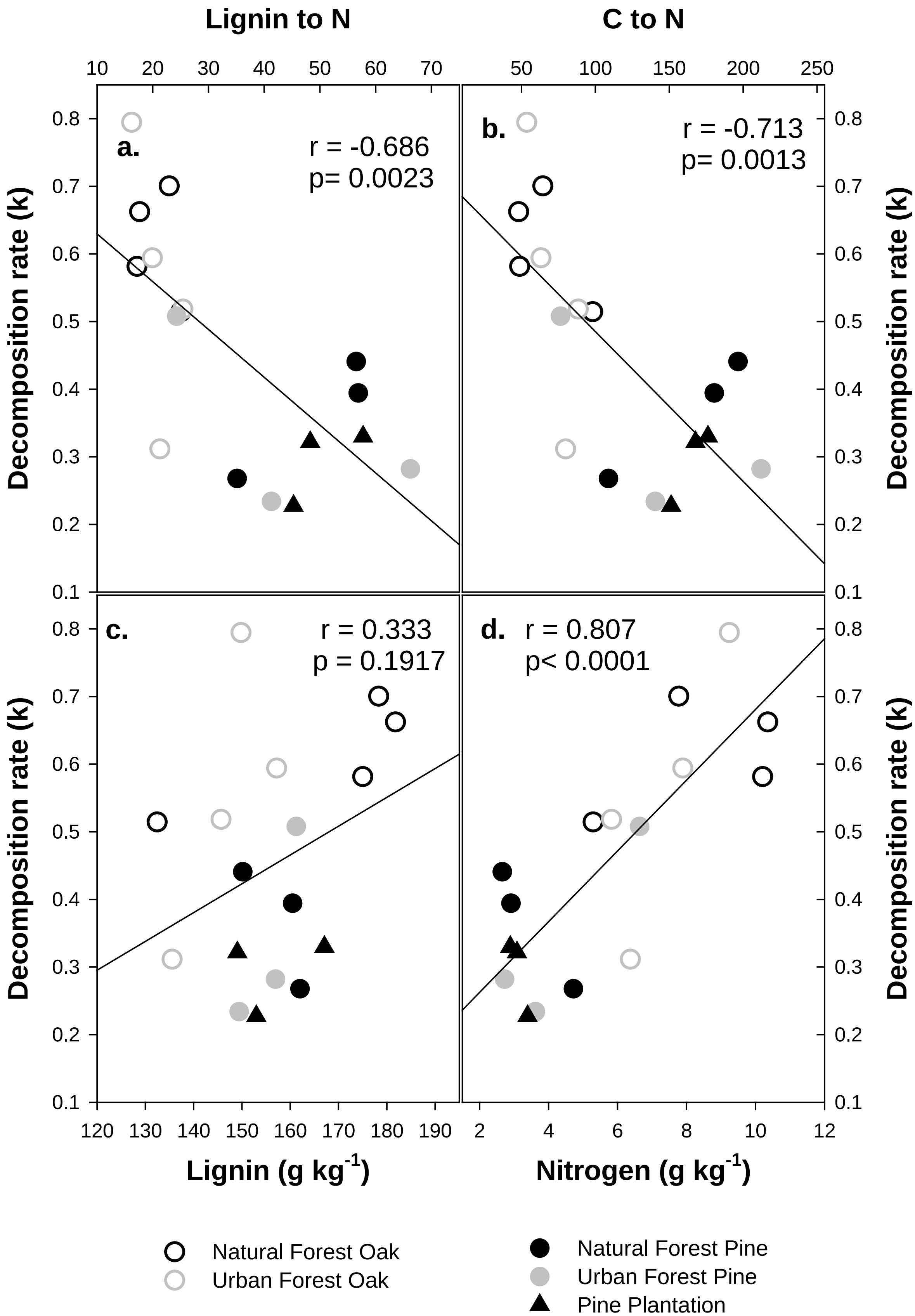

Lignin to N
C to N
a.
r = -0.686
p= 0.0023
b.
r = -0.713
p= 0.0013
c.
r = 0.333
p = 0.1917
d.
r = 0.807
p< 0.0001
Decomposition rate (k)
Lignin (g kg-1)
Nitrogen (g kg-1)
Natural Forest Oak
Urban Forest Oak
Natural Forest Pine
Urban Forest Pine
Pine Plantation

Figure 3.7. Foliar litter quality indices, as measured by lignin concentration, nitrogen concentration and lignin to nitrogen ratio, for three common overstory tree species in natural forests (NF) and urban forests (UF) in western Florida.

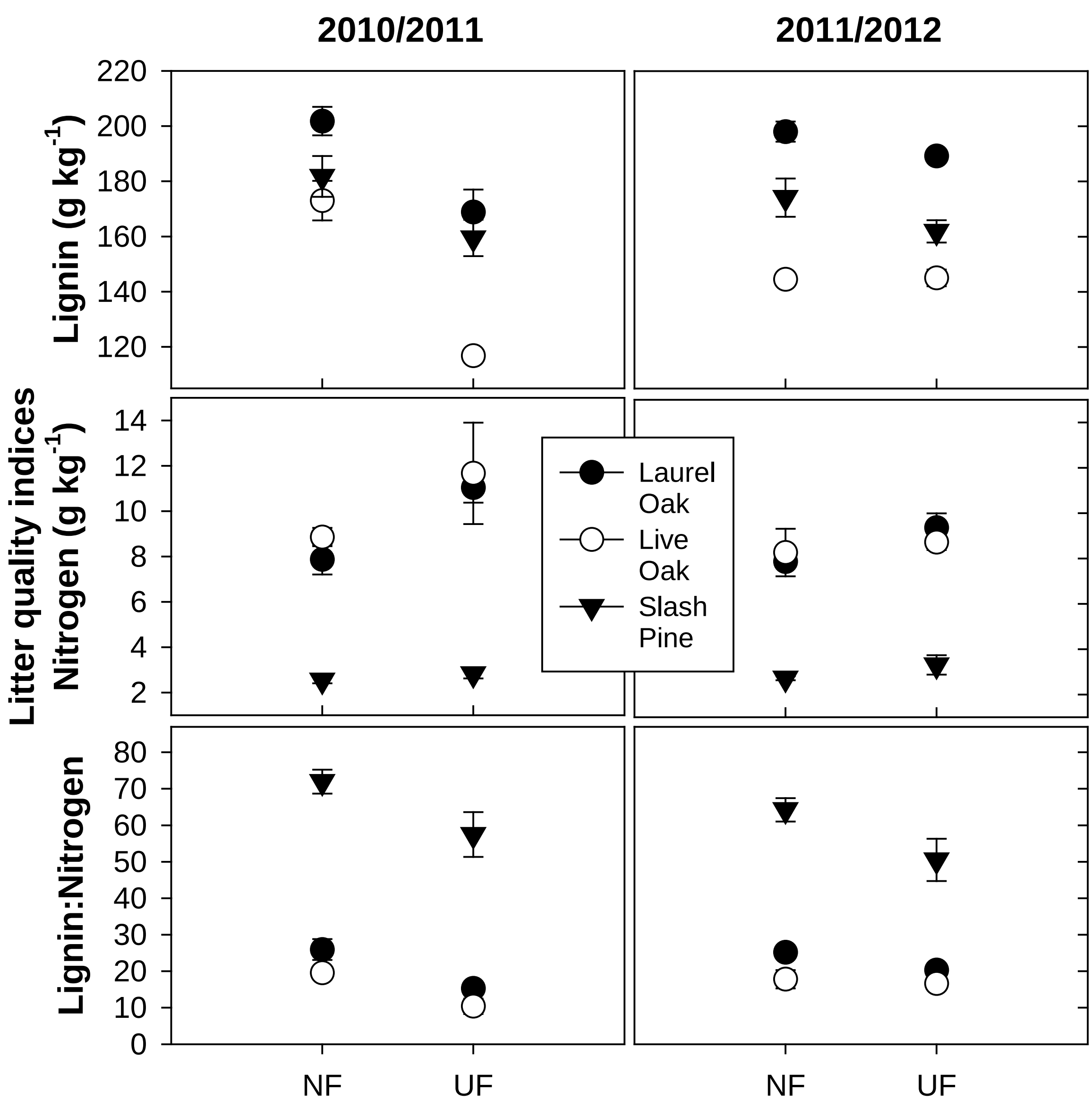


Figure 3.8. Percent nitrogen (N) remaining in the litterbags by (a) oak and (b) pine land use during decomposition (March 2010 – October 2012) in western Florida. Vertical bars represent one standard error from the mean.

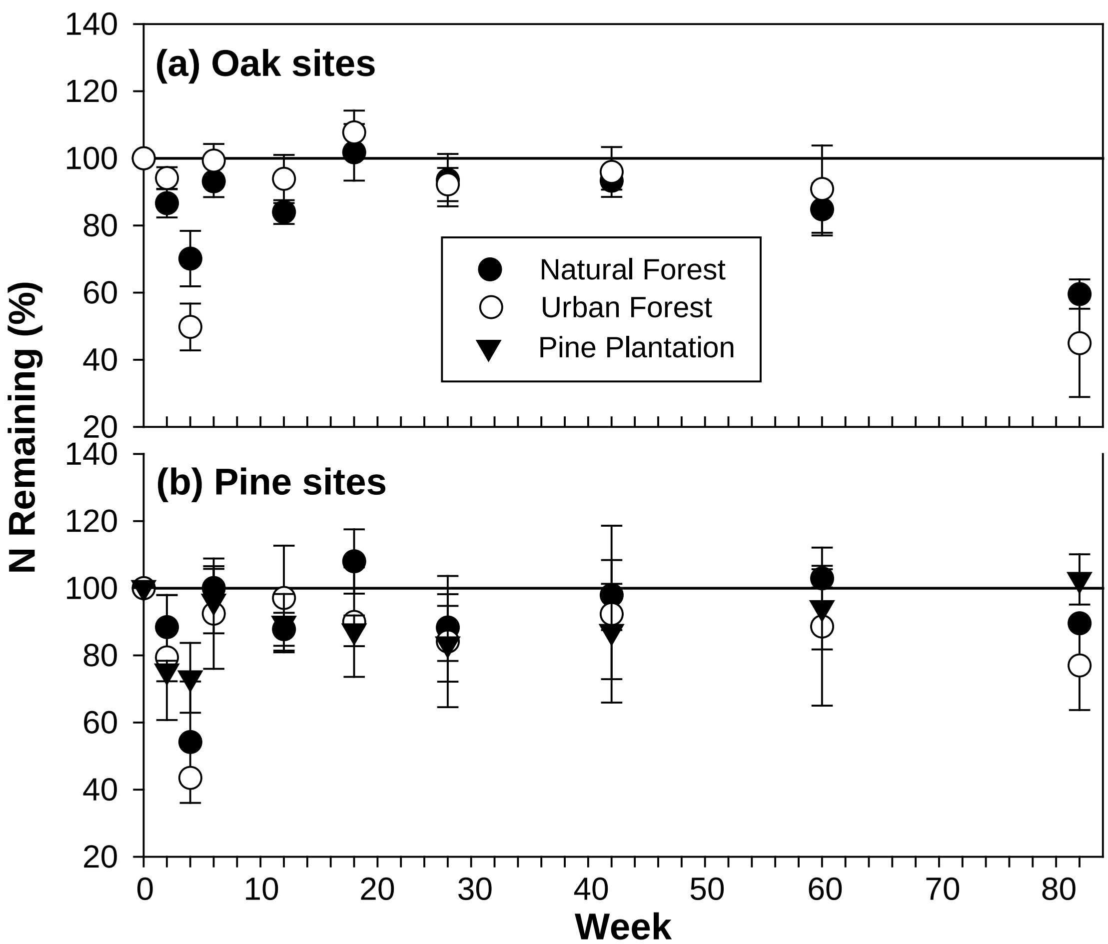
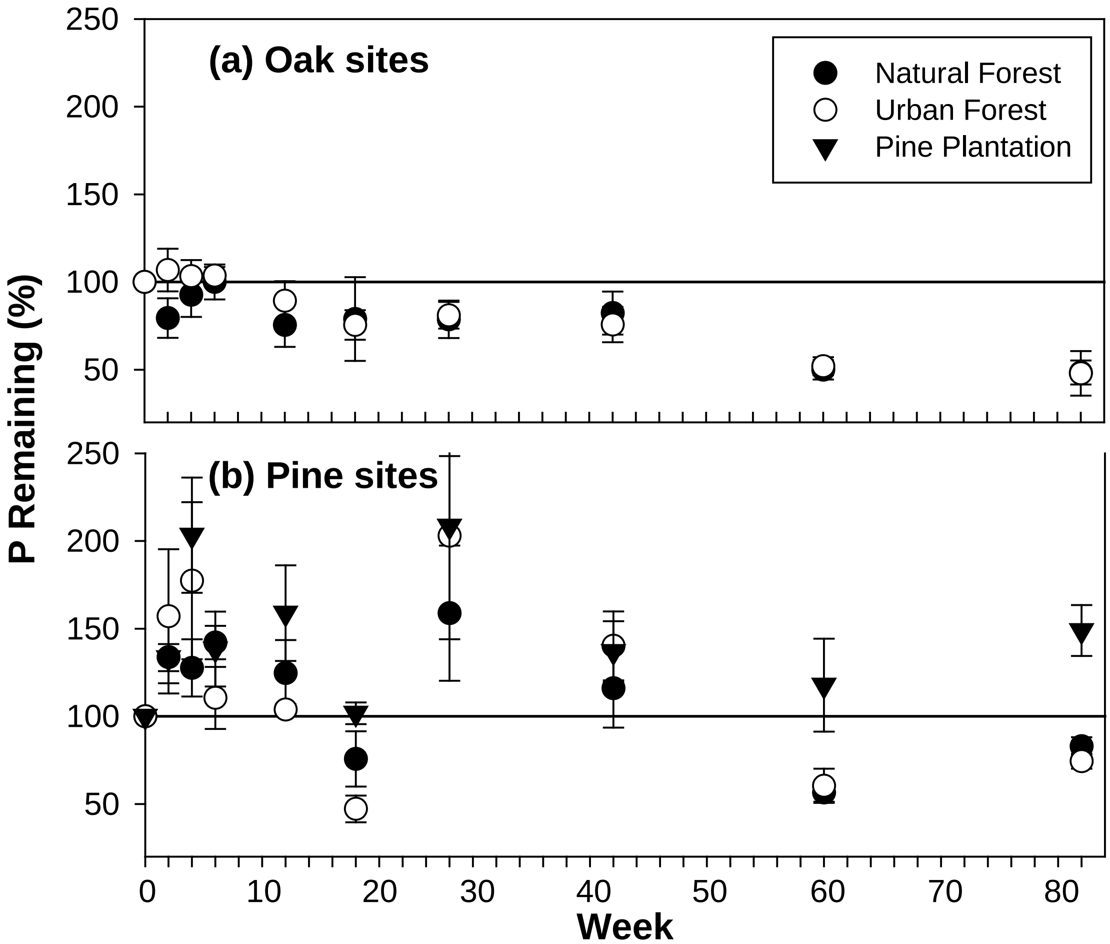


Figure 3.9. Percent phosphorus (P) remaining in the litterbags by (a) oak and (b) pine land use during decomposition (March 2010 – October 2012) in western Florida. Vertical bars represent one standard error from the mean.



4. Impact of urbanization on soil carbon and nitrogen dynamics

Abstract

Urbanization can involve multiple alterations to the soil environment and it is uncertain how this may alter soil nitrification rates and the availability of nitrogen to plants. This study investigated whether net soil nitrogen mineralization rates and soil microbial biomass are different between urban forests and rural forests (naturally regenerated or *Pinus ellottii* plantation) and between urban forests and urban lawns in a developing area of western Florida. Forest land covers are further categorized by their dominant overstory vegetation (oak or pine) in order to minimize the variation within a land cover designation. Urban forest pine dominated sites are found to have 2.5 times as much N mineralized on a daily basis as natural forest pine sites and urban forest oak sites are found to have 1.9 times as much N mineralized as natural forest oak sites based on the mean daily rates averaged over the two year study. The significant difference between urban versus rural sites in the mean net N mineralization rates is due to urban forest oak sites having a significantly higher rate of nitrification and to urban forest pine sites trending towards higher rates of ammonium mineralization compared to their natural forest counterparts. Urbanization appears to stimulate soil microbial biomass and activity (potential carbon mineralization rates) and this may be influencing the soil nitrogen mineralization rates in the forest sites. To include an urban lawn component in the study, one time measurements of soils from the aforementioned forest sites and from urban lawn sites (no fertilization, no irrigation) were collected and then incubated in a laboratory setting under constant temperature and moisture. Urban forest sites and urban lawns sites in the study area are not significantly different in their

potential carbon mineralization rates, potential net total N mineralization rates or microbial biomass carbon and nitrogen contents. Potential net nitrification rates in unfertilized lawns in this study are significantly higher than those in urban forests, with nitrification composing the entire potential net N mineralized in urban lawn soils.

Introduction

One of the most important microbial mediated processes in surface soils is nitrogen (N) mineralization and immobilization, as most of soil N is stored in organic matter and unavailable to plants. Soil net N mineralization rates indicate the availability of nitrogen to plants and are strongly correlated to plant productivity (Reich et al., 1997). Furthermore, the net production of nitrate can result in ecosystem loss of N since nitrate can be lost via leaching or denitrification. Microbial immobilization of N has been implicated in conserving this nutrient when plant growth is minimal during the spring in the northern United States (i.e. the “vernal dam” hypothesis, Zak et al., 1990) and after disturbance from timber harvesting (Vitosek and Matson, 1984) and hurricanes (Rice et al., 1997) in the southeast United States. There is a concern that alterations to the soil abiotic and biotic environment (i.e. higher temperatures, increased carbon dioxide, increased presence of exotic species, N deposition) from urbanization may alter the N conserving mechanisms of microbes and N mineralization rates in these soils.

Urbanization studies originating from an urban to rural gradient in the New York City Metropolitan area provide the highest number of published studies on nitrogen cycling in urban versus rural forests. Urban remnant forest soils have been found to have similar, lower and higher net N mineralization rates and similar to higher net nitrification rates compared to rural forest soils (Pouyat et al., 1997, Steinburg et al., 1997, Zhu and Carreiro, 1999, Pouyat and Turachek, 2001, Zhu and Carreiro et al., 2004). The results from these studies are inconsistent and this may be due to the high variability between urban sites, the units used to calculate N mineralization (on a soil or a soil organic matter basis), the experimental design (incubation study versus *in-situ* measurements) and the duration of the soil incubation over which N mineralization was measured. Any differences between urban and rural sites in these studies were likely due to the higher abundance of exotic earthworms in the urban sites.

Baltimore, Maryland, is another center of urbanization studies as it is one of the urban components of the United States National Science Foundation's Long-term Ecological Research network. Nitrogen cycling in forested stands in the Baltimore Metropolitan area was determined more by soil parent material and land use history than by proximity to the urban core (Groffman et al., 2006). Lastly, higher net nitrogen mineralization rates in urban forests in Asheville, North Carolina, compared to rural sites were attributed to warmer temperatures in the urban sites (Pavao-Zuckermann and Coleman, 2005). Net nitrification rates between the urban and rural forested sites were similar in the North Carolina study. There is a large amount of natural variation in soil properties across landscapes and urban land use change involves multiple alterations to the abiotic and biotic environment. Hence, additional research will clarify how unifying or divergent the effects of urbanization on nitrogen cycling in forest stands can be across cities in the eastern United States.

Another powerful driver of regional change in the nitrogen cycle is the conversion of land into turfgrass dominated systems. In the Baltimore Metropolitan area, the conversion of agricultural lands to urban lawns has likely decreased nitrate leaching over the past few decades in the region (Groffman et al., 2009). This has been countered to some degree by the conversion of forest lands to urban lawns. Urban lawns in Baltimore lose more N via nitrate leaching compared to urban forests (Groffman et al., 2009). Potential net nitrification in the surface soils of urban lawns has been found to be significantly higher than nearby urban remnant forests in studies from Baltimore (Groffman et al., 2009) and Pinehurst, North Carolina (Shi et al., 2006). Levels of nitrate in forest soils are typically low (Fisher and Binkley, 2000) and very little nitrate is leached from undisturbed forested stands (Montagnini et al., 1991; Dittman et al., 2007).

Urbanization in the southern United States has occurred to great extent on formerly forested areas. Between 1990 and 2000, 1.1 million hectares were converted to urban land use in the southeast (Nowak et al., 2010). The south is forecasted to lose between 4.4 and 9.3 million hectares of forest land to urban uses from 1997 to 2060 (Wear, 2013). It is uncertain how much of this new urban land will be dominated by turfgrass, but it is likely significant. Turfgrass land cover (residential, commercial, and institutional lawns, parks, golf courses and athletic fields) accounts for an estimated 1.9 % of the total continental United States area (Milesi et al., 2005), whereas urban lands account for an estimated 3.5 to 4.9% (Nowak et al., 2001). Very little research exists on how urban lawns and urban forests differ in regard to net nitrogen mineralization and nitrification rates in the southeastern United States.

The purpose of this study is to gain insight into how urbanization alters nitrogen cycling in sandy soils along the coast in western Florida. Specifically, we investigated whether net nitrogen mineralization rates and microbial biomass are different between urban forests and naturally regenerating forests and slash pine (*Pinus elliotti* Engelmann) plantations and between urban forests and urban lawns. As microbial biomass carbon and nitrogen can be sensitive to management and environmental change, we expected alterations in net nitrogen mineralization and nitrification to also be reflected in these microbial properties. Net soil nitrogen mineralization rates, net nitrification rates, and net ammonium production rates were measured *in-situ* over a two year period in the surface soils of seven naturally regenerating forest stands, three pine plantation stands and seven urban forest stands. Microbial biomass carbon and nitrogen content were also measured during the two year period. To include an urban lawn component in the study, one time measurements of soils from the aforementioned forested sites and from five urban lawn sites were collected in the summer of 2012 and then incubated in a laboratory setting under constant temperature and moisture. Potential N mineralization and nitrification rates were measured along with potential carbon mineralization rates (a metric of microbial activity) and microbial C and N biomass. In order to minimize the variation within a land cover designation, all forested sites and urban lawn sites are located on soils with similar parent material, sites in the natural and urban forest land covers are further categorized by their dominant overstory vegetation (oak or pine), and all sites within a land cover span across the same range in soil drainage classifications.

Urbanization studies typically take place in large cities, where direct effects of urbanization can dominate changes in soil biogeochemical cycling, such as coverage of soil by fill material or the incorporation of anthropic materials into the soil. This study examines one of the direct effects of urbanization (forest conversion to urban lawns) on nitrogen cycling. This study also examines the indirect effects related to urbanization that alters the abiotic and biotic environment of undisturbed soils (land conversion from natural forests to urban forests). Furthermore, this study investigates those effects on nitrogen cycling when urbanization occurs at low population densities. The effect of urbanization is often determined after major development has occurred and, with this study, we have an opportunity to establish baseline data as well as study how these data change as this coastal area transitions into a post-developed state.

Methods

Study site description

Soil samples were collected from seven urban forest sites and five urban lawns within Apalachicola and Eastpoint, Franklin County, Florida, and from seven naturally regenerated forests and three slash pine plantations in rural areas outside of these two cities. Apalachicola was considered an urban cluster by the United States Census Bureau in 2010 (U.S. Census Bureau, 2010) and watersheds in the Apalachicola and Eastpoint area had impervious surface percentages that did not exceed 15 % of the watershed land area (Nagy et al., 2012). This area has a subtropical climate, with average summer temperatures of 27.2 ^{o}C and average winter temperatures of 12.4 ^{o}C, and historically receives 143.5 cm of precipitation per year. Actual precipitation was 41.3 cm below the historical average during the first year of the study (July 2010 to June 2011) and was 29.9 cm below the historical average during the majority of the second year of the study (July 2011 to May 2012). A tropical storm in June 2012 added 31.3

cm of precipitation to the study area over a 48 hour period. This single storm event brought the study area out of a deficit and into a surplus of 14.3 cm above the historical average for July 2011 to June 2012 (National Climatic Data Center, 2012).

The urban forest sites in this study are unmanaged forest fragments within the urban matrix. Six of the seven urban forest sites are urban remnant forests and the seventh urban forest is emergent (Chapter 2). Remnant urban forests have never been cleared for urban use and emergent urban forests have been cleared for urban use, abandoned, and then reverted to a tree-covered site (Zipperer, 2002). Both urban and natural forest sites are defined as oak dominated or pine dominated forest sites. Urban lawns were dominated by a mix of turf and weedy species, contained at the most 6.3 stems ha^{-1}, and were not fertilized, limed or irrigated at the time of the study. Lawn sites were mowed approximately once a month. The urban lawns in this study had been in the current land cover since 1953 (earliest accessed aerial photograph of the area) and did not appear to have been disturbed since then (Chapter 2). Six of the seven urban forest sites appeared to be undisturbed and to be occupied by naturally regenerating forest in an urban setting since 1953. The seventh urban forest site had a small house on the site in 1953 and by 1969 the house was removed and the site exhibited complete forest canopy closure by 1984 (Chapter 2). Sites within a land cover are located across a range of drainage classifications (Table 4.1). Forested sites were selected in July 2010. Site selection, soil properties, plant community analysis, and aboveground productivity and standing crop biomass have been described in detail elsewhere (Chapter 2). Five additional sites were added in June 2012 to incorporate the urban lawn land use into the study.

Soil net nitrogen mineralization

Samples to measure soil nitrogen mineralization rates were processed over a two year period from July 2010 through May 2012 for the forested sites. Net N-mineralization and nitrification were determined using an *in-situ*, buried bag method (Hart et al., 1994b). Within each of the forested sites, two soil cores were extracted to a depth of 7.5 cm. Soil from each core was split into two polyethylene bags (approximately 300 mL of soil in each bag). One of these bags was immediately reburied and incubated for 25 to 30 days and the other bag was brought back to Auburn University. Post-incubation samples and pre-incubation samples were processed for ammonium and nitrate within three days of bringing the samples back from the field. Processing consisted of passing the soil through a 2 mm sieve and then extracting 10 g of the < 2 mm field moist soil in 100 mL of 2 mol L^{-1} KCl. The extracted KCl solutions were filtered and then frozen until the extracts were analyzed using standard colorimetric techniques, with the developed color analyzed using a microplate reader (Sims et al., 1995). Bulk density for the surface (0 to 7.5 cm) soil depth was calculated for each of the forested sites (Blake and Hartge, 1986) and used to convert concentrations to content values. Bulk density was calculated for the <2 mm soil fraction by accounting for the weight and volume of coarse fragments. Coarse fragments did not exceed 5 % of the volume of the bulk density samples.

Soil net nitrogen mineralization rates (g ha^{-1} day^{-1}) were calculated as the accumulation of NO_3-N and NH_4-N during the incubation and were reported in terms of nitrogen as ammonium (NH_4-N) mineralized, nitrogen as nitrate (NO_3-N) produced and total nitrogen mineralized (NH_4-N + NO_3-N). Net nitrogen mineralization rates were measured over 30 days and for two months out of each season. Monthly values for total nitrogen mineralization rates, nitrification rates and ammonium mineralization rates were averaged across both years. Additionally, monthly values were averaged for the winter and summer seasons over both years. Since the climate in the panhandle of Florida is subtropical, the months included in each season were defined as follows: Summer = June, July, August; Fall = September, October, November; and Winter=December, January, February; Spring=March, April, May.

Potential soil carbon and net nitrogen mineralization

Potential carbon and nitrogen mineralization rates were determined using procedures of Robertson et al. (1999) and Hopkins (2008). Two surface soil samples (0-7.5cm) were collected from forested and urban lawn sites in June 2012, a week before a tropical storm reached the study area. Soils were passed through a 2 mm sieve. In July 2012, field moist sub-samples (35 g on an oven dried basis) were moistened to field capacity and incubated in the dark at 25^{o}C for 30 days in 1-L canning jars that also contained a 20 mL vial of deionized water to maintain humidity. To estimate potential net nitrogen mineralization rates, soil ammonium and nitrate were measured on pre- and post-incubated samples using the same techniques for the monthly net nitrogen mineralization estimates. As for the C mineralization rates, 1 mol L^{-1} NaOH traps were added after a one week settling period had passed in the incubation chamber and potential C mineralization was measured over 23 days. The settling time was incorporated into this study in order to avoid the flush of carbon dioxide (CO_2) that occurs from sieving and moistening soil (Hopkins, 2008). Evolved CO_2 was determined by titration of the alkali trap with 1 mol L^{-1} HCl and potential C mineralization rates were calculated as the amount of C (as CO_2) evolved over the 23 days (g C-CO_2 kg^{-1} soil).

Soil microbial biomass carbon and nitrogen

Soil microbial biomass was estimated using the chloroform-fumigation method (Vance et al., 1987). Soil sub-samples were taken from the sieved, field moist pre-incubated soil samples collected for actual net nitrogen mineralization measurements (October 2010 through May 2012) of the forested sites and for the potential net nitrogen mineralization measurement (June 2012) of the forested and lawn sites. Fumigated samples were exposed to chloroform for 24 hours. Fumigated and unfumigated samples (18.5 g) were extracted with 100 mL 0.5 mol L^{-1} K_2SO_4. The extracts were filtered and then frozen. After thawing, the samples were analyzed for organic C and N using a Shimadzu TOC-V and total N combustion analyzer (Shimadzu Scientific Instruments, Columbia, MD). The differences between fumigated and unfumigated samples represented microbial N and C (in g g^{-1} soil). For the forested sites, soil microbial biomass C and N were measured once per season from October 2010 through June 2012 and were averaged across all sampling periods to obtain a two year average for each parameter. A one-time measurement of soil microbial biomass was taken for the urban lawn soils in June 2012.

Statistical analysis

All data were analyzed by SAS version 9.2 (SAS Institute, Cary, NC). Treatment effects on two year averages of soil N mineralization rates and annual microbial biomass estimates, and one-time measurements of microbial biomass and potential C and N mineralization rates were tested using analysis of variance with land use and soil drainage classification as the primary treatment variables. Contrast statements were used for the following planned comparisons for differences among the forest sites (a-e) and for differences among the forested and lawn sites (f):

a. Urban forest pine versus natural forest pine;

b. Urban forest oak versus natural forest oak;

c. Natural forest pine versus pine plantation;

d. Urban sites (oak + pine) versus rural sites (natural oak + natural pine + pine plantation);

e. Pine (urban +natural+plantation) versus oak (urban + natural); and

f. Urban forest (oak + pine) versus urban lawn.

Contrast f is the only contrast reported for comparisons of microbial biomass C and N among land covers for the one-time soil sampling in June 2012 that included the lawn soils. This is because microbial biomass data from the June 2012 collection are used in the calculation of the two year average microbial biomass content of the forested sites.

To discern whether actual net N mineralization varied seasonally, a two sample t-test was used to test for differences in net N mineralized (as total, ammonium and nitrate) between the winter and summer seasons within land use categories. Summer and winter net N mineralization values were compared within each land use, as differences in N mineralized between these two seasons were expected to be the highest of all potential seasonal comparisons.

A Pearson product-moment correlation was used to measure the association between potential carbon mineralization rates and both microbial biomass C and net N mineralization rates for the one-time soil sampling of forest sites and urban lawns in June 2012. A probability value for each Pearson correlation was calculated and considered statistically significant at $\alpha \leq 0.05$. For both the planned and unplanned comparisons, differences among means were considered statistically significant at $\alpha \leq 0.05$ and differences with $\alpha \leq 0.10$ were reported for informational purposes. The model assumptions of normality were not violated. In select circumstances, the data were logarithmically transformed to meet the assumptions of equality of variance.

Results

Urban and rural forests

Averaged across both study years (July 2010 to May 2012), the mean total net nitrogen mineralization rates in the upper 0 to 7.5 cm of soil range from 42.8 g hectare^{-1} day^{-1} (g ha^{-1} d^{-1}) for pine plantation sites up to 187.2 g ha^{-1} d^{-1} for urban forest oak dominated sites (Table 4.2, Figure 4.1). The mean net nitrogen mineralization rates for slash pine plantation sites in this study are similar to those rates estimated for surface soils in slash pine plantations in Louisiana using *in situ* soil incubations (44.2 g ha^{-1} d^{-1}, Discus and Dean, 2008). Net nitrogen mineralization rates did not differ between pine dominated sites and oak sites (contrast e). The total net nitrogen mineralization rates for urban forested sites are significantly larger than that for rural sites (p=0.0012, Table 4.3). Urban forest pine sites are found to have 2.5 times as much N mineralized on a daily basis as natural forest pine sites and urban forest oak sites are found to have 1.9 times as much N mineralized as natural forest oak sites based on the mean daily rates averaged over the two year study.

The difference between urban versus rural sites in the mean net N mineralization rates is due to urban forest oak sites having a significantly higher rate of nitrification (p=0.0027, Table 4.3) and to urban forest pine sites trending towards higher rates of ammonium mineralization (p=0.0639). Urban forest pine sites also have larger average nitrification rates compared to rural pine dominated forests, however the difference between the two land covers is not significant (p=0.2282). Nitrification is a main component of total N mineralized in the urban forest sites, with on average 79 % of the net nitrogen mineralized was nitrified in the urban forest oak sites and 31 % in the urban forest pine sites. In comparison, on average 21 % of the net nitrogen mineralized was nitrified in the natural forest sites. In regard to ammonium mineralization rates, the pine and the oak sites have opposing urbanization trends. The oak sites exhibit a significant decrease in ammonium mineralization rates in the urban forests compared to the natural forests (p=0.0319), whereas the urban forest pine sites are trending towards higher ammonium mineralization rates compared to the natural forest pine dominated sites (p=0.0639). Similar trends are observed among the forested land uses for the potential net N mineralization rates as those for the two year mean net N mineralization rates (Table 4.3, Figure 4.1).

Seasonal differences between summer and winter total net nitrogen mineralization rates and nitrification rates (g ha^{-1} d^{-1}) are present for the oak dominated sites, but not for the pine dominated sites (pine plantation data not shown, Figure 4.2, 4.3, 4.4). Natural forest oak sites have higher rates of nitrification by (20.0 ± 7.4 g ha^{-1} d^{-1}, 95 % confidence interval, p=0.0006) during the summer season compared to the winter, but this is not reflected in a different total net N mineralization rate between these two seasons (p=0.2760). Urban forest oak sites have both higher nitrification rates (261.2 ± 130.8 g ha^{-1} d^{-1}, p=0.0027) and higher total N mineralization rates (238.3 ± 106.0 g ha^{-1} d^{-1}, p=0.0015) in the summer compared to the winter.

Urban forested sites have higher microbial biomass carbon and nitrogen contents in the upper 7.5 cm of soil compared to the rural sites (p=0.0522 and p=0.0170, respectively) based on an analysis of the two year averages (Table 4.4, 4.5). Microbial biomass carbon to nitrogen ratios did not differ between urban and rural sites, but did differ between natural forest pine sites and pine plantation sites (p=0.0332). Pine plantation sites have a significantly higher ratio than natural forest pine sites.

Potential C mineralization rates in this study (5.8 to 10.2 mg C-CO_2 kg^{-1} soil d^{-1}, Table 4.4, 4.5) are similar to those found for surface soils in pine plantations (3.8 mg C-CO_2 kg^{-1} soil d^{-1}), upland forests (4.0 mg C-CO_2 kg^{-1} soil d^{-1}) and urban areas (5.8 mg C-CO_2 kg^{-1} soil d^{-1}) in northern Florida (Ahn et al., 2009). The one-time measurements of potential C mineralization rates indicate that urban forests are trending towards higher rates of activity compared to rural forests (p=0.0811, Table 4.5). No differences were observed for potential carbon mineralization rates between pine and oak dominated forested sites (p=0.5031). Potential C mineralization rates measured during the 30-day incubation for all of the sites (forested and lawn) are strongly and positively correlated with soil microbial biomass carbon content ($r=0.8443$, $p < 0.0001$) and potential net N mineralization rates ($r=0.8555$, $p < 0.0001$) of incubation soil samples.

Urban lawns

One time measurements of soils in June 2012 indicate that urban lawns do have higher microbial carbon to nitrogen ratios than the urban forests ($p=0.0120$, Table 4.4, 4.5), but did not differ significantly in microbial biomass carbon and nitrogen contents compared to urban forests ($p=0.5242$ and $p=0.7174$, respectively). Potential C and total net N mineralization rates measured during the 30-day incubation for the lawn and urban forest sites also are not statistically different between these land covers ($p=0.4281$ and $p=0.1201$, respectively, Table 4.2, 4.4). However, compared to urban forest sites, urban lawns have significantly lower rates of potential net ammonium production ($p=0.0051$) and higher rates of net nitrate production ($p=0.0274$). Urban lawns displayed little to no potential net ammonium production. On average, all of the net nitrogen mineralized came from the production of nitrate.

Discussion

Urban verses rural forests

Urbanization appears to have a distinct impact on net N-mineralization and nitrification in surface soils of forested areas in a developing region in western Florida. Based on the mean daily rates averaged over the two year study, urban forest pine sites are found to have on average 2.5 times and urban forest oak sites are found to have 1.9

times as much net N mineralized as their natural forest counterparts. This indicates that plant available N in the surface soils of urban forests is significantly higher than that of rural forest soils. Urbanization can alter the abiotic and biotic soil environment through several means, including nitrogen deposition, the urban heat island effect and the introduction of exotic species, which may have contrasting effects on microbial activity and function.

In this study, urbanization appears to stimulate soil microbial activity, which may lead to higher N mineralization rates. First, averaged over a two year period, surface soil microbial biomass carbon and nitrogen contents are higher in urban forests, especially for urban forest oak dominated sites, than those of rural forests. Second, urban forest sites are trending towards higher potential carbon mineralization rates compared to those of rural forested sites in an incubation study under similar moisture and temperature conditions. Third, urban forested sites in this study have been found to have higher winter temperatures and narrower temperature ranges in the forest floor compared to rural sites. The urban heat island effect has been thought to increase net N mineralization rates in urban forest soils compared to rural forest soils (Pavao-Zuckermann and Coleman, 2005; Chen et al., 2010). It is unclear whether urban sites in this study received additions of N from dry and wet deposition. However, increased inputs of N to soils have been found to reduce soil microbial biomass and activity (Fisk and Fahey, 2001; Ramirez et al., 2012).

Cause and effect interpretations regarding N mineralization rates and microbial activity and factors that can increase microbial activity (i.e. urban heat island) should be made with some caution as net N mineralization is a net process whereas carbon mineralization is a gross process. Net N mineralization rates do not account for the N that is released and then immobilized within the microbial biomass and therefore may not always correlate with C mineralization rates (Giardina et al., 2001; Hart et. al. 1994a) as gross N mineralization rates do (Hart et al., 1994a). Potential carbon mineralization and net N mineralization rates are strongly and positively correlated in this study ($r=0.8555$, $p < 0.0001$). Additional research would be needed to clarify how gross N mineralization rates and immobilization rates change due to urbanization.

Two year averages of daily net N mineralization rates and of microbial biomass carbon and nitrogen contents for the forest soils were measured during a drought (Chapter 3). Previous research of the forest sites in this study found that urbanization did not impact foliar decomposition rates at the soil surface. A possible explanation is that drought limited microbial activity in the forest floor across all sites, masking any effect of urbanization (Chapter 3). It is likely that the forest floor protected the surface soils from water lost through evaporation and ameliorated the low moisture stress of the drought within the surface soils.

Oak and pine dominated stands have distinct differences in biogeochemical cycling, regardless of whether they are in an urban or rural setting. Additional research of these sites has found that oak dominated forest sites have higher foliar N and P contents entering the forest floor on a yearly basis, smaller forest floor C contents, and higher surface soil pH values compared to pine dominated forest sites (Chapter 2). A litterbag study in the forest sites indicated that after 82 weeks of decomposition, the site specific, mixed species foliar litter in pine sites had on average 1.76 times ($p=0.0196$) more mass remaining than oak sites (Chapter 3). In this study, oak dominated forest sites, both natural and urban, exhibit a difference between summer and winter mean net nitrification rates, but pine sites do not. Furthermore, urbanization has different results on net ammonium production rates depending on whether the stand was pine or oak dominated. Net nitrification rates are higher in urban forest oak sites compared to natural forest oak sites, but not significantly different between urban forest pine and natural forest pine dominated sites.

The presence of an invasive, non-native tree (*Cinnamomum camphora* (L.) J. Presl) is more apparent in the urban oak versus the urban pine dominated forests and not present in rural forest sites (Chapter 2). Soil nitrification rates commonly increase after invasions by exotic plants (Ehrenfeld, 2003). Therefore, it is important to account for the overstory composition of a forest stand as it reflects differences in stand origin that then impact biogeochemical cycling and the effect of urbanization.

Higher litter quality indices and more suitable edaphic conditions likely caused higher short-term foliar litter mass loss in oak forest sites compared to rural forest sites (Chapter 3). However these factors did not lead to overstory specific differences in potential C mineralization rates and total net N mineralization rates. Several studies have found that hardwood and evergreen forest stands have similar surface soil net N mineralization rates (Reich et al., 1997; Giardina et al., 2001; Mueller et al., 2012). Giardina et al. (2001) concluded that their results were consistent with the decay filter hypothesis proposed by Melillo et al. (1989). The decay filter hypothesis predicts that as decomposition proceeds, the quality of the decomposing litter from diverse litter types converges. This hypothesis suggests that the initial advantages that higher litter quality has on faster decomposition rates and nutrient release at the forest floor are not conferred to the surface soil because material in the soil that originated from foliar decomposition is older, highly decomposed organic matter of low quality. Further investigation into root abundance and turnover in oak and pine dominated forest types in the study area would likely shed light on the similarities both forest types share in N mineralization and C mineralization in the surface soils.

Urban forests verses urban lawns

A change in vegetation cover from forest to lawn did not alter potential soil net total N mineralization rates, but it did strongly impact net nitrification rates and microbial carbon to nitrogen ratios. The lower carbon to nitrogen ratio of microbial biomass in the surface soils of the urban lawns compared to those in the urban forests may indicate a more active N-cycling soil microbial community in the urban lawns (Raciti et al., 2011b). Potential net ammonium production in urban lawns was significantly lower than those in urban forests ($p=0.0051$). However, potential net nitrification rates in unfertilized lawns in this study (based on current management) are significantly higher than those in urban forests ($p=0.0274$). Nitrification comprised the entire potential net N mineralized in urban lawn soils.

Urban lawn soils have been found to have higher rates of potential net nitrification compared to urban remnant forest soils in Baltimore, Maryland (Groffman et al., 2009; Raciti et al., 2011b). Soils collected from golf course fairways in Pinehurst, North Carolina, also had higher potential net nitrification rates than soils in nearby undisturbed pine (*Pinus palustris* Miller) stands (Shi et al., 2006). The urban lawns in these studies included sites that were fertilized with N. The form of the applied N (as nitrate, ammonium or both) is unclear from their methods. Additions of ammonium would likely stimulate nitrification in the lawn soils. However, fertilizer addition was not a significant predictor of available nitrate in lawn soils in Baltimore (Raciti et al., 2011b). The authors suggest this may be due to their experimental design or due to the transient availability of inorganic N after fertilizer application.

It is unclear if urban lawns soils in this study have higher rates of net nitrification compared to urban forests due to higher rates of gross nitrification or to lower rates of microbial nitrate immobilization or to a combination of both. There is some indication that the rates of gross nitrification may be higher in these sites, as urban lawn sites have surface soil pH of 5.8 ± 0.2 (mean ± standard error), which is on average 1.8 ± 0.5 (95 % confidence interval, $p<0.0001$) units higher than those found for urban forest soils (Chapter 2). Nitrification in soils is due to the activity of chemoautrophic nitrifiers and/or heterotrophic nitrifiers. Heterotropic nitrifiers are thought to dominate in acidic soils. However, research in acidic forest soils in the New York City metropolitan area has shown that chemoautotrophs are active in soil pH as low as 4.4 (Zhu and Carreiro, 1999). Nonetheless, nitrification rates are typically low in acidic forest soils and this is characteristic of the natural forest sites in this study. The population of chemoautrophic nitrifiers expands rapidly when acidity is corrected (Fisher and Binkley, 2000), and this may be the cause of the higher nitrification rates in the urban lawn sites compared to the urban forest sites. Urban lawn sites were not currently managed with lime or fertilizer and were not irrigated at the time of sampling. Therefore, we can only hypothesize that higher soil pH in the lawns soils is from historic lime applications, as lawn sites have been under the same land cover since 1953 (Chapter 2).

Urban lawns can have a considerable capacity for N retention (Raciti et al., 2008; Groffman et al., 2009). Nitrogen retention in lawn soils is dependent on high levels of soil organic matter and rates of carbon cycling, as indicated by potential carbon mineralization rates and microbial biomass (Groffman et al., 2009). Other research comparing soil carbon cycling of lawns (often fertilized) to remnant forests in the eastern United States suggests that urban lawns can strongly resemble urban forests in soil organic carbon and microbial carbon levels (Shi et al., 2006; Groffman et al., 2009; Raciti et al., 2011a; Raciti et al, 2011b). Urban forest sites and urban lawns sites in this study are not significantly different in their potential carbon mineralization rates or one-time measurements of microbial biomass carbon and nitrogen contents. Previous research of the sites in this study indicated that urban lawns have lower mean soil organic carbon and soil nitrogen contents in the surface soil (0 to 7.5 cm) and the mineral soil (0 to 90 cm) compared to urban forests, however the differences are not significant (Chapter 2). Urban lawns in this study may have a similar potential to that of urban forest soils to retain N, but their higher rates of nitrification compared to the urban forests combined with their sandy soil texture implies that urban lawn soils can be susceptible to nitrate leaching.

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Table 4.1. Location, plant community and soil information for each of the study sites within the oak dominated forest, pine dominated forest and lawn land covers in western Florida.

Land cover	Plant community	Soil drainage	Location UTM UPS NAD 1983 (meters)		
Natural Forest Oak	mesic hammock	very poorly drained	16 R	684434	3293733
	xeric hammock	poorly drained	16 R	684474	3293715
	mesic hammock	somewhat poorly drained	16 R	684631	3293865
	xeric hammock	moderately well drained	16 R	683428	3293155
Urban Forest Oak	urban forest	very poorly drained	16 R	693266	3289183
	urban forest	poorly drained	16 R	694288	3289516
	urban forest	somewhat poorly drained	16 R	694037	3289783
	urban forest	moderately well drained	16 R	703440	3291945
Urban Lawn	urban lawn	very poorly drained	16 R	694529	3290007
	urban lawn	poorly drained	16 R	694300	3289432
	urban lawn	somewhat poorly drained	16 R	694225	3289886
	urban lawn	somewhat poorly drained	16 R	694116	3290519
	urban lawn	moderately well drained	16 R	694845	3289967

Natural Forest Pine	wet flatwood	very poorly drained	16 R	684574	3293648
	wet flatwood	poorly drained	16 R	684613	3293696
	mesic flatwood	somewhat poorly drained	16 R	684675	3293950
Pine Plantation	pine plantation	very poorly drained	16 R	683627	3293600
	pine plantation	poorly drained	16 R	683657	3293509
	pine plantation	somewhat poorly drained	16 R	713654	3296434
Urban Forest Pine	urban forest	very poorly drained	16 R	693263	3289280
	urban forest	poorly drained	16 R	690576	3291264
	urban forest	somewhat poorly drained	16 R	691995	3290655

Table 4.2. Actual net nitrogen mineralization rates averaged over a two year period and potential nitrogen mineralization rates (measured during a 30-day laboratory incubation) for each land use located in western Florida. Standard errors of the means are in parentheses.

Land Use	**Actual N mineralized** (g N ha^{-1} d^{-1})			**Potential N mineralized** (g N ha^{-1} d^{-1})		
	N-total	$N\text{-}NH_4$	$N\text{-}NO_3$	N-total	$N\text{-}NH_4$	$N\text{-}NO_3$
Urban Lawns	NA[a]	NA	NA	572.9 (64.4)	-6.9 (19.4)	579.8 (72.9)
Natural Forest Oak	97.5 (21.2)	78.4 (18.60)	19.1 (3.38)	261.7 (62.0)	216.9 (50.8)	44.9 (12.45)
Urban Forest Oak	187.2 (38.1)	26.4 (14.6)	160.8 (44.0)	456.2 (134.7)	114.5 (38.3)	341.7 (164.7)
Natural Forest Pine	69.8 (10.77)	56.2 (12.8)	13.6 (3.09)	179.6 (55.6)	162.1 (59.6)	17.5 (12.1)
Urban Forest Pine	171.6 (51.0)	106.2 (8.2)	65.3 (43.4)	448.6 (149.1)	228.6 (37.8)	220.0 (185.9)
Pine Plantation	42.8 (16.4)	28.6 (7.7)	14.2 (9.4)	189.4 (60.5)	163.6 (43.2)	25.9 (22.3)

[a]Not applicable (NA) as lawn sites did not undergo repeated sampling.

Table 4.3. Results of the contrast statements for differences among land covers in western Florida for net nitrogen mineralization rates averaged over the study period (actual net N mineralization rates) and measured during a 30-day incubation study (potential net N mineralization rates). Only contrasts with a p-value ≤ 0.10 are reported. Natural forest oak = NFO; Natural forest pine = NFP; Pine plantation = PP; Urban forest oak = UFO; and Urban forest pine (UFP).

Response Variable	Contrast	Point Estimate	Margin of Error	p-value	Significance
Actual net N mineralization (g N ha^{-1} d^{-1})					
N-total	UFO-NFO	89.6	77.9	0.0286	*
	UFP-NFP	101.8	90.0	0.0307	*
	Urban forest - Rural forest	113.7	55.0	0.0012	*
$N\text{-}NH_4$	UFO-NFO	-52.0	46.4	0.0319	*
	UFP-NFP	50.0	53.6	0.0639	
$N\text{-}NO_3$	UFO-NFO	141.6	78.5	0.0027	*
Potential net N mineralization (g N ha^{-1} d^{-1})					
N-total	UFP-NFP	269.0	T[a]	0.0318	*
	Urban forest-Rural forest	245.2	T	0.0062	*
$N\text{-}NH_4$	Lawn-Urban forest	-179.8	113.3	0.0051	*

N-NO_3	Urban forest-Rural forest	254.3	T	0.0916	
	Lawn-Urban forest	339.8	T	0.0274	*

[a]Data are transformed (T) and a 95% confidence interval is not symmetric around the mean. Confidence intervals are given as ratios in text.

Table 4.4. Potential carbon mineralized (C-min) and soil microbial biomass carbon and nitrogen content in soils collected from land covers located in western Florida. Standard errors of the means are in parentheses.

	Incubation Study				Two Year Average		
Land Cover	Potential C-min (mg C kg^{-1} d^{-1})	Microbial Biomass (µg g^{-1} soil)		Microbial Biomass	Microbial Biomass (µg g^{-1} soil)		Microbial Biomass
	$C\text{-}CO_2$	C	N	C to N ratio	C	N	C to N ratio
Urban Lawns	9.99 (1.66)	254.03 (56.84)	49.91 (12.77)	5.13 (0.44)	NA[a]	NA	NA
Natural Forest Oak	5.78 (1.34)	160.34 (22.57)	23.76 (4.16)	7.14 (0.74)	135.30 (18.64)	16.85 (2.11)	9.29 (2.45)
Urban Forest Oak	9.32 (3.17)	334.29 (110.97)	58.64 (19.43)	6.21 (0.67)	238.56 (48.74)	42.34 (13.51)	6.41 (1.02)
Natural Forest Pine	5.47 (1.74)	189.96 (32.55)	29.04 (6.00)	6.72 (0.36)	277.94 (19.76)	38.14 (6.26)	7.56 (1.04)
Urban Forest Pine	10.18 (3.11)	411.03 (131.15)	51.72 (18.32)	8.13 (0.80)	473.99 (129.20)	54.73 (18.49)	8.70 (0.78)
Pine Plantation	6.70 (4.48)	253.86 (115.94)	35.73 (12.47)	6.66 (0.80)	275.97 (73.36)	24.83 (7.37)	13.57 (4.04)

[a]Not applicable (NA) as lawn sites did not undergo repeated sampling.

Table 4.5. Results of the contrast statements for differences among land covers in western Florida for microbial biomass carbon and nitrogen contents and potential carbon mineralized rates. Only contrasts with a p-value ≤ 0.10 are reported. Natural forest oak = NFO; Natural forest pine = NFP; Pine plantation = PP; Urban forest oak = UFO; and Urban forest pine (UFP).

Response Variable	Contrast	Point Estimate	Margin of Error	p-value	Significance
Potential C mineralization (mg C kg^{-1} d^{-1})					
	Urban forest-Rural forest	3.85	T[a]	0.0811	
Microbial biomass, incubation study (µg g^{-1} soil)					
C:N	Lawn-Urban forest	-2.08	1.54	0.0120	*
Microbial biomass, two year averages (µg g^{-1} soil)					
C	UFP-NFP	196.05	211.99	0.0660	
	Urban forest-Rural forest	128.00	129.52	0.0522	
	Pine forest – Oak forest	146.82	136.84	0.0382	*
N	UFO - NFO	25.49	T	0.0240	*
	Urban forest-Rural forest	22.86	T	0.0170	*

C:N	NFP-PP	-6.01	T	0.0332	*

[a]Data are transformed (T) and a 95% confidence interval is not symmetric around the mean. Confidence intervals are given as ratios in text.

Figure 4.1. Net nitrogen mineralized (g ha^{-1} day^{-1}) as total N mineralized, nitrogen as ammonium (NH_4-N) and nitrogen as nitrate (NO_3-N) for forest soils (a) averaged over two years and (b) during a laboratory incubation study. Data from lawn sites are not shown. Vertical bars represent one standard error from the mean.

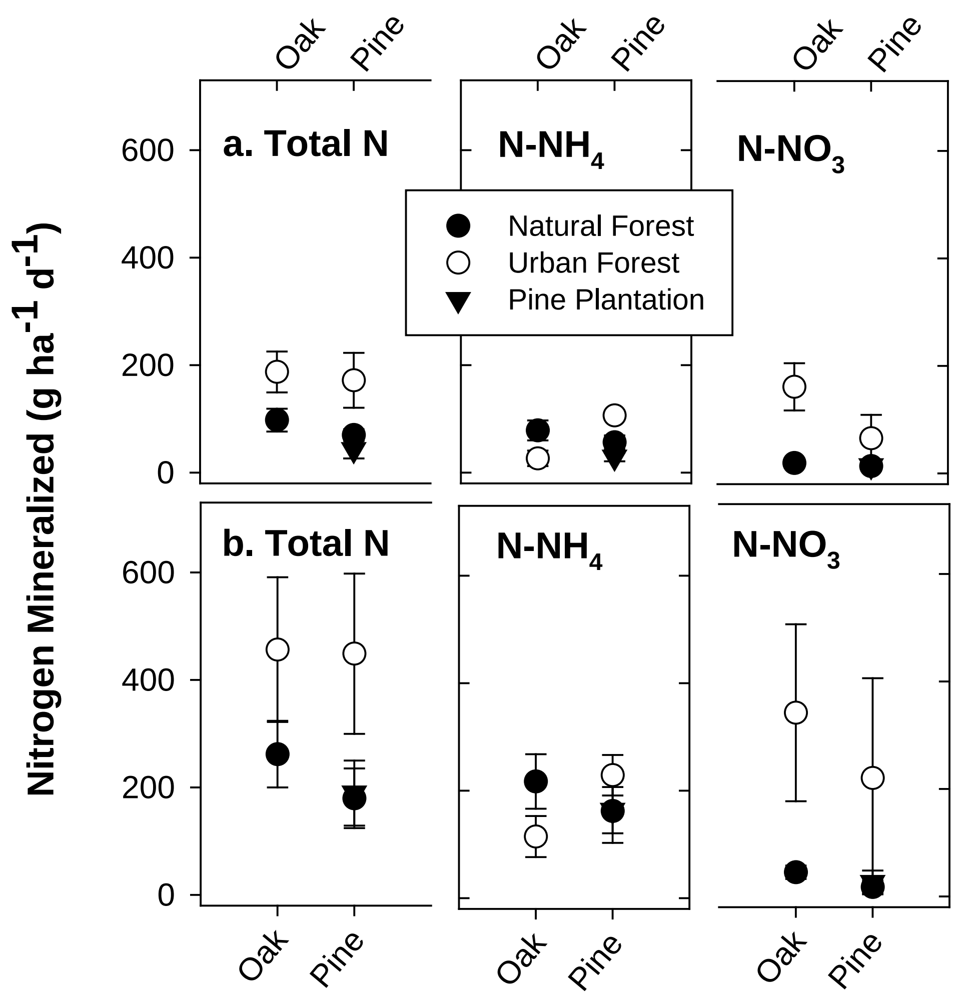


Figure 4.2. Seasonal averages for nitrogen mineralized (g ha^{-1} day^{-1}) over the two year study period for surface soils in the natural forest and urban forest land uses. Vertical bars represent one standard error from the mean. Winter=December, January, February; Spring=March, April, May; Summer = June, July, August; Fall = September, October, November.

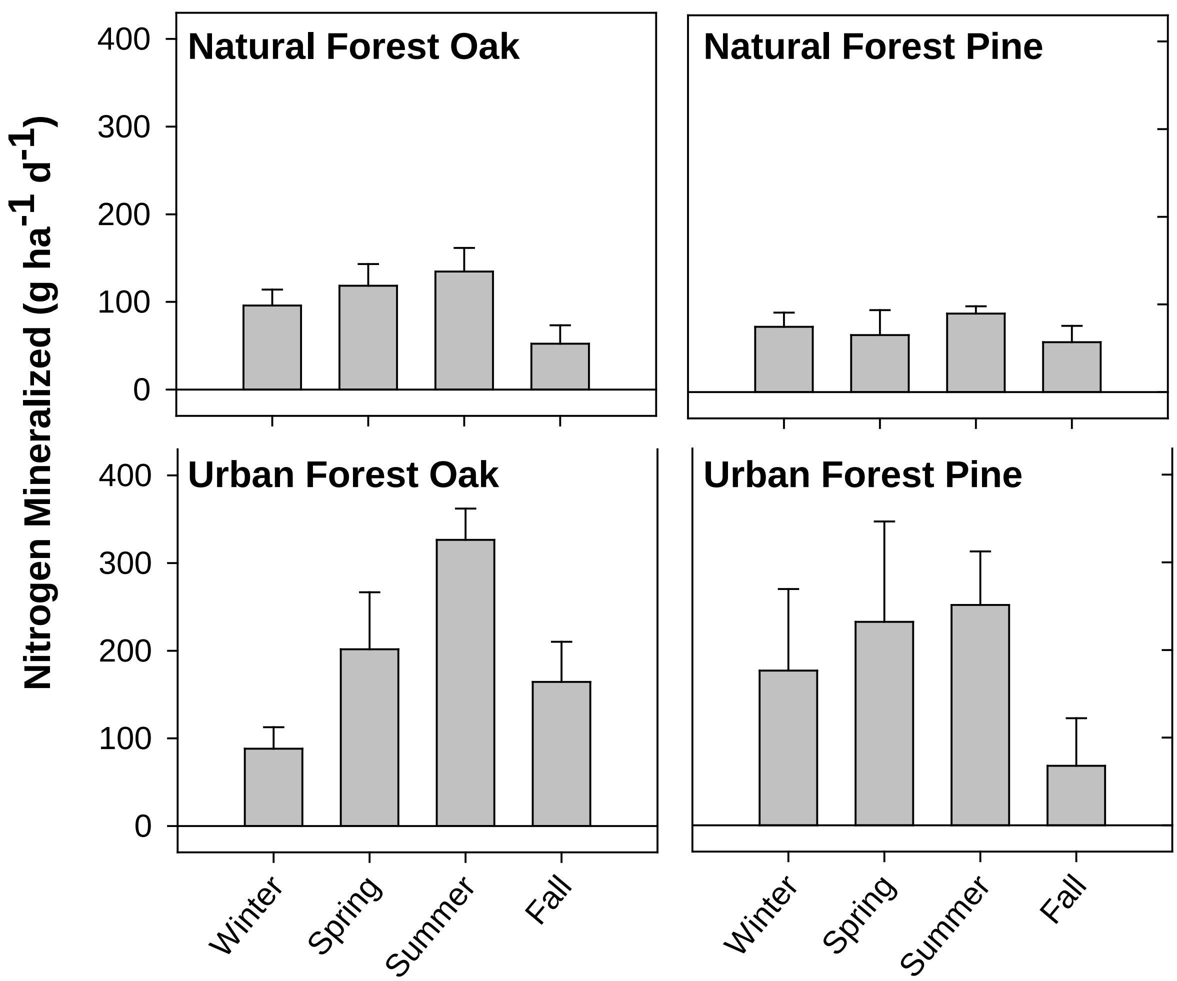


Figure 4.3. Seasonal averages for ammonium mineralized (g ha^{-1} day^{-1}) over the two year study period for surface soils in the natural forest and urban forest land uses. Vertical bars represent one standard error from the mean. Winter=December, January, February; Spring = March, April, May; Summer = June, July, August; Fall = September, October, November.

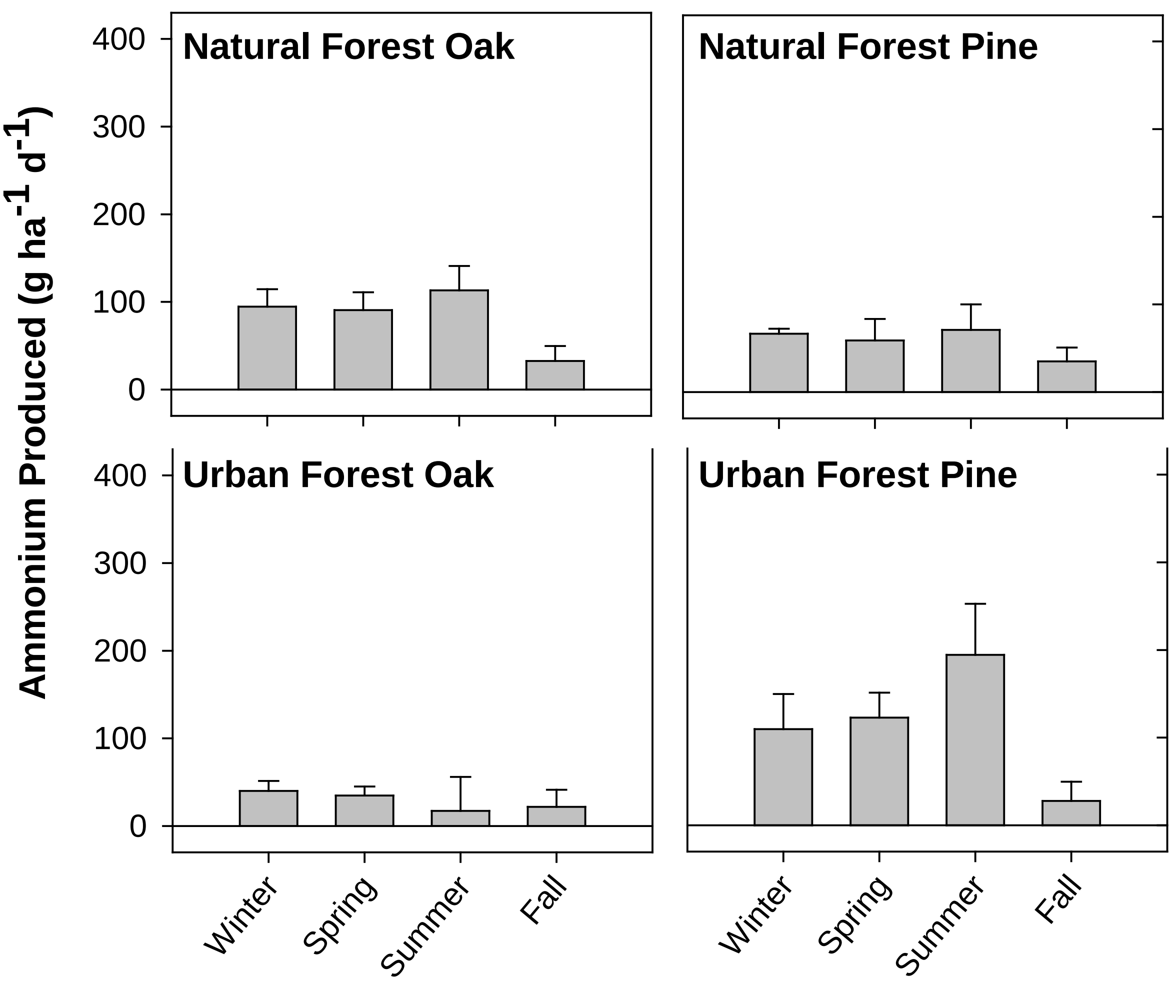
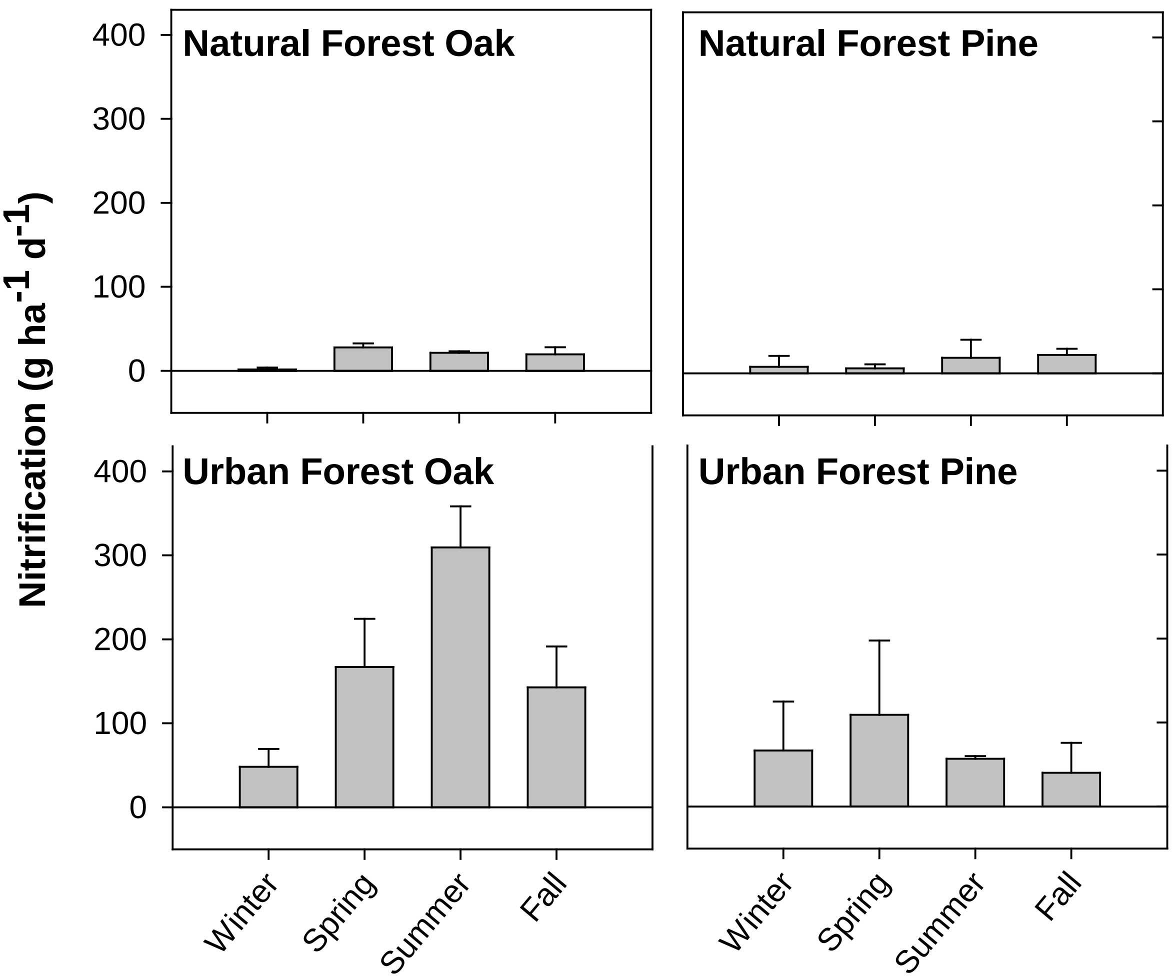


Figure 4.4. Seasonal averages for nitrate mineralized (g ha^{-1} day^{-1}) over the two year study period for surface soils in the natural forest and urban forest land uses. Vertical bars represent one standard error from the mean. Winter=December, January, February; Spring=March, April, May; Summer = June, July, August; Fall = September, October, November.



5. Summary and Conclusions

Nutrient cycling is a supporting process to ecosystem services. Urbanization has an uncertain impact on nutrient cycling within urban forests and urban lawns. This is partly due to the limited number of studies addressing this topic and due to the contrasting effects that urbanization can have on biogeochemical cycling. Key foci of urban soil research are in regards to how urbanization impacts: (1) carbon storage in soils, (2) decomposition of foliar litter on the soil surface and (3) soil nitrogen cycling. This research addressed each of these three topics in Chapters 2, 3 and 4, respectively, in a developing region in western Florida. In particular, we chose to examine the impacts of urbanization between rural forests (naturally regenerated and pine plantation) and urban forests and between urban forests and urban lawns. Briefly, the results of each of these three topics are as follows:

1. Urbanization increases soil carbon content in oak dominated forest systems compared to natural oak dominated forests, but does not increase soil carbon content between urban pine dominated forests and natural pine dominated forests. Urban lawns and urban forests did not vary significantly in soil carbon content or soil nitrogen content (Chapter 2).
2. In forested sites, urbanization did not have an impact on foliar litter decomposition rates over an 82 week period during a drought period. Urbanization did increase litter quality of individual species foliar litter, but did not significantly change the litter quality of mixed species foliar litter.

Forest floor temperatures appear to be impacted by the urban heat island effect (Chapter 3).

3. Urban forested sites have higher total net soil nitrogen mineralization rates compared to rural forested sites (based on two year averages of daily nitrogen mineralization rates measured *in situ*). This difference is driven by higher rates of nitrification in urban forest oak sites compared to natural forest oak sites and from a trend of higher ammonium production in urban forest pine dominated sites compared to natural forest pine dominated sites. Total potential net soil nitrogen mineralization rates did not differ between urban forests and urban lawns (based on a 30 day laboratory incubation study). However, urban lawns had significantly higher nitrification rates compared to urban forests. All of the net nitrogen released in urban lawns was from nitrification (Chapter 4).

To aid in explaining our results, we evaluated plant species composition and measured aboveground net primary productivity, standing crop biomass, and nutrient contents in litterfall at each forested site. We measured soil microbial biomass carbon and nitrogen contents and potential carbon mineralization rates for all sites (lawns included). A visual analysis of the sites across time since 1953 indicated that the natural forest sites and urban forest sites were of similar age. Pine and oak dominated urban forest sites have similar aboveground standing crop biomass, but higher aboveground net primary productivity rates than their natural forest counterparts. Specifically, the higher aboveground productivity in the urban forested sites is largely from increased (not statistically significant, $p=0.1000$) foliar production in the urban oak forest sites and higher woody production in the urban pine forest sites. It appears that the trend of urban forest oak dominated sites towards higher inputs of foliar C and N (kg m^{-2} yr^{-1}) into the forest floor may have then promoted higher mineral soil C and N contents compared to the natural forest oak sites, especially in the upper 30 cm of mineral soil (Chapter 2).

Soil organic carbon content is a balance between carbon inputs and decomposition rates. Decomposition rates between urban forest and natural forest sites are not different during a drought period. Urban forests do have higher forest floor temperatures in the winter and narrower temperature ranges throughout most of the year compared to rural forested sites (Chapter 3). This difference in microclimate between urban and rural forested sites may alter foliar litter decomposition during a normal precipitation year. Additional research in measuring decomposition rates is needed in the study area.

Like aboveground net primary productivity, microbial biomass and activity in surface soils of the forest sites appear to be stimulated due to urbanization. Potential C and net N mineralization rates were positively and strongly correlated in this study, however there is uncertainty in concluding that the higher microbial activity leads to higher net N mineralization rates (Chapter 4). Additional research would be needed to clarify how gross N mineralization rates and immobilization rates change due to urbanization.

The literature suggests that urban lawns soils have incredible potential to store C and N. Usually these lawns are highly managed (fertilized, irrigated, mowed). The lawns in this study are not currently fertilized, irrigated or limed. However the high soil pH of urban lawns suggests that they may have been limed in the past. Urban lawns resemble urban forests in regards to soil C and N contents (0 to 90 cm) and soil microbial C and N contents (0 to 7.5 cm). Urban lawns may have a similar potential to that of urban forest soils to retain N, but their higher rates of nitrification compared to the urban forests combined with their sandy soil texture implies that urban lawn soils can be susceptible to nitrate leaching (Chapter 4). Future research is needed to investigate *in situ* soil nitrification rates and leaching losses of nitrate from both fertilized and unfertilized lawn soils in the study area.

Oak dominated sites and pine dominated sites had distinct differences in their biogeochemical cycling, regardless of whether they were in an urban or rural setting. Pine dominated sites have a higher foliar productivity rate at a given content of N in litterfall (an indices of N availability) than oak dominated systems (Chapter 2). Oak dominated forest sites have higher foliar N and P contents entering the forest floor on a yearly basis, smaller forest floor C contents, and higher surface soil pH values compared to pine dominated forest sites (Chapter 2). A litterbag study in the forest sites indicated that after 82 weeks of decomposition, the site specific, mixed species foliar litter in pine sites had on average 1.76 times ($p=0.0196$) more mass remaining than oak sites (Chapter 3). Although N and P are largely immobilized over 60 weeks of decomposition for the oak sites, it appears that both nutrients after 82 weeks may be mineralized and plant accessible over time. It is likely for the pine sites in this study that N and P are stored in the forest floor and that release is very slow and plant access is limited.

It is uncertain if these differences between pine and oak sites came about due to inherent differences in soil fertility (as certain tree species are more successful on distinct soil types) or due to previous management practices (especially fire return frequency). Or if these differences in biogeochemical cycling developed because of the presence of certain tree species. For example, tree species can influence soil nitrogen mineralization rates through litter quality and decomposition rates through soil animal communities characteristic to that tree species. Nonetheless, oak dominated sites and pine dominated sites have distinct differences in their biogeochemical cycling that does appear to influence the effect of urbanization. The pine and the oak sites have opposing urbanization trends in regard to net ammonium mineralization rates. The oak sites exhibit a significant decrease in ammonium mineralization rates in the urban forests compared to the natural forests, whereas the urban forest pine sites are trending towards higher ammonium mineralization rates compared to the natural forest pine dominated sites (Chapter 4).

Oak dominated systems may be more sensitive or susceptible to the indirect effects of urbanization compared to pine dominated systems. For instance, the presence of camphortree is more apparent in the urban oak versus the urban pine dominated forests. Additionally, this sensitivity is observed in a shift in overstory species composition and different soil pH between urban forest oak sites and natural forest oak sites that are not seen between urban and rural pine dominated forested sites. The shift in overstory species may have caused an earlier leaf drop in the urban oak sites compared to the natural forest oak sites.

6. Appendix

Appendix A. Woody species inventory of 17 forested sites in the Florida panhandle, 2010-2012. Sites include natural forest oak (NFO), urban forest oak (UFO), natural forest pine (NFP), urban forest pine (UFP) and pine plantation (PP). Drainage classes within land uses are moderately well drained (MWD), somewhat poorly drained (SPD), poorly drained (PD) and very poorly drained (VPD).

Species	**Land use and drainage**															
	Natural forest oak				Urban forest oak				Natural forest pine			Urban forest pine			Pine plantation		
	MWD	SPD	PD	VPD	MWD	SPD	PD	VPD	SPD	PD	VPD	SPD	PD	VPD	SPD	PD	VPD
Acer rubrum L.									X			X					
Callicarpa americana L.								X									
Cinnamomum camphora (L.) J. Presl						X	X	X						X			
Clethra alnifolia L.								X		X				X			
Cliftonia monophylla (Lam.) Britton ex											X	X	X				X
Ilex coriacea (Pursh) Chapm.				X				X		X	X		X	X		X	
Ilex glabra (L.) A. Gray											X	X	X				
Ilex opaca Aiton		X	X	X					X	X							

Ilex vomitoria Aiton						X							X				
Lyonia ferruginea (Walter) Nutt.			X	X	X					X	X				X	X	X
Lyonia lucida (Lam.) K. Koch				X							X	X		X		X	
Magnolia grandiflora L.		X	X	X		X	X		X			X					
Magnolia virginiana L.								X						X			
Morella cerifera (L.) Small				X	X							X					
Osmanthus americanus (L.) Benth. & Hook. F. ex.		X	X	X					X	X	X						
Persea borbonia (L.) Spreng.								X				X					X
Persea palustris (Raf.) Sarg.														X			X
Pinus echinata Mill.	X				X					X					X		
Pinus elliottii Engelm.				X		X	X	X	X	X	X	X	X	X	X	X	X
Pinus palustris Mill.			X								X						
Prunus caroliniana Aiton						X	X										
Quercus chapmanii Sarg.															X	X	
Quercus geminata Small	X				X										X		
Quercus hemisphaerica Bartram ex Willd.	X	X	X		X	X	X		X	X							
Quercus laevis Walter	X																

Quercus laurifolia Michx.				X				X			X						X
Quercus margarettae (Ashe) Small	X																
Quercus myrtifolia Willd.					X										X	X	
Quercus nigra L.		X		X		X	X	X	X	X		X	X	X	X		X
Quercus virginiana Mill.		X	X	X		X	X		X	X	X						
Rhus copallinum L.						X								X			
Serona repens (Bartram) Small	X	X	X	X	X		X	X	X	X	X	X			X	X	
Vaccinium arboreum Marsh	X	X	X	X	X					X	X						
Vaccinium corymbosum L.			X	X	X												
Vaccinium stamineum L.			X	X					X								

Appendix B. Information on land cover change gathered from aerial photographs of each site in the study area from 1953 to 2004. Drainage classes within land uses are moderately well drained (MWD), somewhat poorly drained (SPD), poorly drained (PD) and very poorly drained (VPD).

Land Cover	Drainage	Time in current land cover	Canopy closure	Aerial photograph[a]
		year		
Natural forest oak	MWD	Since 1953	By 1953	1953, 1969, 1973, 2004
	SPD	Since 1953	By 1953	1953, 1969, 1973, 2004
	PD	Since 1953	By 1953	1953, 1969, 1973, 2004
	VPD	Since 1953	By 1953	1953, 1969, 1973, 2004
Urban forest oak	MWD	Since 1953	By 1984	1953, 1984, 2004
	SPD	Since 1969	By 1984	1953, 1969, 1984, 2004
	PD	Since 1953	By 1969	1953, 1969, 1984, 2004
	VPD	Since 1953	By 1969	1953, 1969, 1994, 2004
Natural forest pine	SPD	Since 1953	By 1953	1953, 1969, 1973, 2004
	PD	Since 1953	By 1953	1953, 1969, 1973, 2004
	VPD	Since 1953	By 1953	1953, 1969, 1973, 2004
Urban forest pine	SPD	Since 1953	By 1980	1953, 1969, 1973, 1980, 1993/94
	PD	Since 1953	By 1973	1953, 1969, 1973, 1980, 1993/94
	VPD	Since 1953	By 1953	1953, 1969, 1973, 1980, 1993/94
Pine plantation	SPD	Since 1984	By 2004	1953, 1984, 2004
	PD	Since 1984	By 2004	1953, 1969, 1973, 2004
	VPD	Since 1984	By 2004	1953, 1969, 1973, 2004

	MWD	Since 1953	NA[b]	1953, 1969, 1984, 2004
	SPD	Since 1953	NA	1953, 1969, 1984, 2004
Urban Lawn	SPD	Since 1953	NA	1953, 1969, 1984, 2004
	PD	Since 1953	NA	1953, 1969, 1984, 2004
	VPD	Since 1953	NA	1953, 1969, 1984, 2004

[a]Aerial photographs from 1953 were from the US Department of Agriculture (2013). Aerial photographs from 1969 to 1994 are from the Florida Department of Transportation (2013). Aerial photographs from 2004 are from the Florida Department of Environmental Protection (2004).

[b]Not applicable (NA) as sites are dominated by a mix of turf grass and broadleaved herbaceous plants and have at most 6.3 stems ha^{-1}.

Appendix C. Allometric regression equations used to estimate the total-tree (wood, bark and foliage) above ground biomass (TM), from diameter at breast height (DBH) for species and species groups. Total tree above ground biomass is in kilograms and DBH is in cm for Brantley 2008. Total tree above ground biomass is in pounds and DBH is in inches for Clark et al., 1985. Total tree above ground biomass is in grams and DBH is in cm for Swindel et al. (1979).

Target species for this study	n	Source species or species group	Regression equation coefficients				Regression equation	Source
			Trees < 27.9 cm DBH		Trees > 27.9 cm DBH			
			a	b	a	b		
Acer rubrum	3	*A. rubrum*	2.52363	1.19648			$TM=a*(DBH^2)^b$	Clark et al., 1985
Pinus echinata	31							
Pinus elliottii	184	*P. elliottii*	4.30	2.54	4.30	2.54	$TM = Exp(a+b*log_e(DBH))$	Swindel et al., 1979
Pinus palustris	4							
Quercus hemisphaerica	28							
Quercus laevis	2	*Q. laurifolia*	3.18283	1.19758	9.68515	0.96554	$TM=a*(DBH^2)^b$	Clark et al., 1985
Quercus laurifolia	*7*							
Quercus nigra	39	Q. nigra	3.47724	1.20469	6.13318	1.08636	$TM=a*(DBH^2)^b$	Clark et al., 1985

Quercus geminata		48	White oaks	2.39278	1.25903	1.64515	1.33715	$TM=a*(DBH^2)^b$	Clark et al., 1985
Quercus virginiana		13							
Cinnamomum camphora	20								
Magnolia grandiflora	15		Soft hardwoods	2.30894	1.20171	1.77809	1.25619	$TM=a*(DBH^2)^b$	Clark et al., 1985
Magnolia virginiana	5								
Persea spp.	3								
Cliftonia monophylla	19								
Ilex spp.	10								
Lyonia ferruginea	4		*Ligustrum sinense*	2.100	0.2749			$TM=a*e^{(b*DBH)}$	Brantley 2008
Osmanthus americanus	6								
Prunus caroliniana	3								
Vaccinium arboreum	5								

Appendix D. Allometric regression equations used to estimate the total-tree woody biomass (WM), i.e., wood and bark, and total-tree foliage mass (FM) from diameter at breast height (DBH) for species and species groups. Woody biomass was estimated for pine species by subtracting FM from total tree above ground biomass (see Appendix C). Total-tree woody biomass is in kilograms and DBH is in cm for Brantley 2008. Total-tree woody biomass is in pounds and DBH is in inches for Clark et al., 1985. Total tree total-tree foliage mass is in grams and DBH is in cm for Swindel et al. (1979).

Target species for this study	n	Source species or species group	Regression equation coefficients: Trees < 27.9 cm DBH		Regression equation coefficients: Trees > 27.9 cm DBH		Regression equation	Source
			a	b	a	b		
Acer rubrum	3	*A. rubrum*	2.39959	1.2003			$WM=a*(DBH^2)^b$	Clark et al., 1985
Pinus echinata	31							
Pinus elliottii	184	*P. elliottii*	1.56	2.44	1.56	2.44	$FM = Exp(a+b*log_e(DBH))$	Swindel et al., 1979
Pinus palustris	4							
Quercus hemisphaerica	28							
Quercus laevis	2	*Q. laurifolia*	2.89221	1.21296	10.22597	0.94962	$WM=a*(DBH^2)^b$	Clark et al., 1985
Quercus laurifolia	7							

Quercus nigra	39	*Q. nigra*	3.15067	1.21955	5.99898	1.08527	$WM=a*(DBH^2)^b$	Clark et al., 1985
Quercus geminata	48	White oaks	2.20767	1.26916	1.56965	1.34028	$WM=a*(DBH^2)^b$	Clark et al., 1985
Quercus virginiana	13							
Cinnamomum camphora	20							
Magnolia grandiflora	15	Soft hardwoods	2.2017	1.20751	1.69855	1.26161	$WM=a*(DBH^2)^b$	Clark et al., 1985
Magnolia virginiana	5							
Persea spp.	3							
Cliftonia monophylla	19							
Ilex spp.	10							
Lyonia ferruginea	4	*Ligustrum sinense*	0.1214	2.4919			$WM=a*(DBH)^b$	Brantley 2008
Osmanthus americanus	6							
Prunus caroliniana	3							
Vaccinium arboreum	5							

Appendix E. Characteristics and productivity of slash pine (*Pinus ellottii*) plantation stands from this study and from other studies in Florida. Some information is not available depending on the study. Means with standard errors in parenthesis (if given) or data ranges are reported.

Location	**Plantation age (years)**	**Basal area (m^2 ha^{-1})**	**Standing crop biomass (g m^{-2})**	**DBH (cm)**	**Stem density (stems ha^{-1})**	**Aboveground NPP (g m^{-2} yr^{-1})**	**Foliar productivity (g C m^{-2} yr^{-1})**	**Source**
western Florida	28 to 30	35.97 (6.67)	15,152 (4,703)	18.52 (0.64)	917 (183)	1,279 (258)	314 (29)	this study
northern Florida	17[a]		11,390			1,230		Shan et al. (2001)
western Florida	25[a]		7,940					Harding and Jokela (1994)
northern Florida	34	20.6 - 28.6	10,049 - 19,413	16.4 -18.6	928 - 1056			Gholz and Fisher (1982)
northern Florida	34		13,885			873		Gholz and Fisher (1982)
northern Florida	24 to 26	31.4 (1.4)			1301 (81)		288.1 (35.1)	Clark et al. (1999)

[a]Data is from the control sites.

Appendix F. Average annual woody growth rate and number of live trees in naturally regenerating forests, urban forests, urban lawns and pine plantation in western Florida. Values are reported for species with diameter at breast height greater than 10 cm and for species with more than one individual in a land cover. Standard errors are in parenthesis. UF = urban forest, NF = natural forest, and P = plantation.

Species	Land cover overstory dominance	Forest Type	Number of live trees	Annual woody growth rate (cm yr^{-1})
Cinnamomum camphora (L.) J. Presl	Oak	UF	12	0.69 (0.20)
	Pine	UF	8	0.99 (0.38)
Magnolia grandiflora L.	Oak	NF	6	0.17 (0.07)
		UF	5	0.43 (0.11)
	Pine	UF	3	0.25 (0.16)
Pinus echinata Mill.	Oak	NF	8	0.57 (0.09)
		UF	11	0.36 (0.07)
	Pine	P	11	0.31 (0.08)
Pinus elliottii Engelm.	Oak	UF	5	0.80 (0.35)
	Pine	NF	41	0.19 (0.03)
		UF	50	0.46 (0.07)
		P	88	0.33 (0.03)

Quercus geminata Small	Oak	NF	12	0.14 (0.03)
	Oak	UF	13	0.45 (0.07)
Quercus hemisphaerica Bartram ex Willd.	Oak	NF	18	0.20 (0.03)
		UF	4	0.28 (0.11)
	Pine	NF	5	0.07 (0.01)
		UF	1	0.10
Quercus laurifolia Michx.	Oak	UF	6	0.53 (0.14)
	Pine	P	1	0.51
Quercus nigra L.	Oak	NF	5	0.46 (0.14)
		UF	25	0.39 (0.07)
	Pine	NF	4	0.30 (0.09)
		UF	5	0.17 (0.07)
Quercus virginiana Mill.	Oak	NF	23	0.39 (0.09)
	Pine	NF	13	0.20 (0.04)

Appendix G. Regression relationships between mean N and P contents in litterfall, vegetation type, and foliar litterfall productivity rates in naturally regenerating forests, urban forests and pine plantation sites in western Florida. Vegetation type is oak (0) or pine (1).

Variable	Parameter Estimate	Margin of Error	p-value	Significant
Regression relationship between foliar litterfall productivity and foliar N content				
Intercept	213	141	0.0061	*
Nitrogen (g m^{-2} yr^{-1})	90.5	31.1	< 0.0001	*
Vegetation Type	191	178	0.0371	*
Vegetation Type*Nitrogen	22.2	51.8	0.3706	
Regression relationship between foliar litterfall productivity and foliar P content				
Intercept	323	227	0.0089	*
Phosphorus (g m^{-2} yr^{-1})	483	369	0.0142	*
Vegetation Type	104	310	0.4789	
Vegetation Type*Phosphorus	342	751	0.3436	

Appendix H. Species specific foliar litter collected in litter traps at each of the 17 forested sites in western Florida. For species that are major components of the foliar litter fall, the relative proportion (%) by mass of each species contained in the litterbags is provided. Additional species specific foliar litterfall information is indicated by a presence (+) or absence (o) symbol. Species that are present (+) contributed to the litterfall but did not occupy a dominant proportion of the litterfall. Drainage classes within land covers are moderately well drained (MWD), somewhat poorly drained (SPD), poorly drained (PD) and very poorly drained (VPD).

Species	Natural Forest Oak				Urban Forest Oak				Natural Forest Pine			Urban Forest Pine			Pine Plantation		
	MWD	**SPD**	**PD**	**VPD**	**MWD**	**SPD**	**PD**	**VPD**	**SPD**	**PD**	**VPD**	**SPD**	**PD**	**VPD**	**SPD**	**PD**	**VPD**
Acer rubrum	o	o	o	o	o	o	o	o	+	o	o	+	o	o	o	o	o
Cinnamomum camphora	o	o	o	o	o	**6**	+	**6**	o	o	o	o	o	**12**	o	o	o
Cliftonia monophylla	o	o	o	o	o	o	o	o	o	o	+	o	+	o	o	o	+
Ilex spp.	o	+	o	+	o	o	o	o	o	+	o	o	o	+	o	+	o
Lyonia spp.	o	o	o	o	o	o	o	o	o	+	+	o	+	o	o	+	o
Magnolia grandiflora	o	+	o	+	o	+	o	o	o	o	o	+	o	o	o	o	o
Magnolia virginiana	o	o	o	o	o	o	o	+	o	o	o	o	o	+	o	o	o

Osmanthus americanus	o	**+**	o	o	o	o	o	o	o	**+**	**+**	o	o	o	o	o	o
Pinus echinata	**3**	o	o	o	**57**	o	o	o	o	o	o	o	o	o	**15**	o	o
P. elliottii	o	o	o	**+**	o	**+**	**51**	**62**	**78**	**91**	**76**	**89**	**100**	**68**	**85**	**100**	**100**
P. palustris	o	o	o	o	o	o	o	o	o	o	**22**	o	o	o	o	o	o
Osmanthus americanus	o	**+**	o	o	o	o	o	o	o	**+**	**+**	o	o	o	o	o	o
Prunus carolinia	o	o	o	o	o	o	**+**	o	o	o	o	o	o	o	o	o	o
Quercus geminata	**23**	o	o	o	**40**	o	o	o	o	o	o	o	o	o	**+**	o	o
Q. hemisphaerica	**74**	**54**	**46**	o	**+**	**57**	**38**	o	**+**	**5**	o	o	o	o	o	o	o
Q. laurifolia	o	o	o	**80**	o	o	o	**+**	o	o	**+**	o	o	o	o	o	o
Q. myrtifolia	o	o	o	o	**3**	o	o	o	o	o	o	o	o	o	o	o	o
Q. nigra	o	o	o	**14**	o	**37**	**+**	**32**	**17**	**4**	o	**11**	**+**	**20**	**+**	o	**+**
Q. virginiana	o	**46**	**54**	**6**	o	**+**	**11**	o	**5**	**+**	**3**	o	o	o	o	o	o
Vaccinium arboreum	**+**	**+**	**+**	o	**+**	o	o	o	o	**+**	o	o	o	o	o	o	o
Vitus spp.	o	o	o	o	o	o	o	o	**+**	**+**	**+**	**+**	o	**+**	**+**	o	**+**

Appendix I. Percent mass remaining (on an ash free basis) of foliar litter during 82 weeks of decomposition (March 2010 – October 2012) within naturally regenerating forests, urban forests, and pine plantation sites in western Florida. Standard errors of the means are in parentheses.

Mass remaining (%)	**Week**									
	0	**2**	**4**	**6**	**12**	**18**	**28**	**42**	**60**	**82**
Natural Forest Oak	100 (0)	99.4 (0.3)	96.9 (0.7)	97.1 (0.4)	93.2 (1.9)	84.7 (0.4)	70.1 (2.01)	56.9 (1.2)	50.0 (2.1)	25.4 (1.7)
Urban Forest Oak	100 (0)	97.1 (1.2)	97.2 (1.2)	94.2 (2.4)	85.09 (6.7)	84.6 (5.2)	77.8 (6.08)	64.4 (6.0)	61.1 (6.1)	34.5 (6.8)
Natural Forest Pine	100 (0)	99.5 (0.5)	98.8 (0.2)	96.4 (0.5)	94.7 (1.8)	90.7 (0.6)	85.3 (1.1)	78.3 (2.4)	68.8 (5.7)	47.1 (6.1)
Urban Forest Pine	100 (0)	98.2 (0.9)	97.2 (1.7)	95.5 (1.8)	94.8 (2.7)	90.8 (5.3)	85.5 (1.5)	83.5 (5.1)	71.9 (8.9)	49.5 (8.5)
Pine Plantation	100 (0)	100.0 (0.0)	98.4 (1.0)	99.6 (0.2)	97.1 (0.9)	90.4 (0.7)	89.6 (2.7)	82.1 (3.3)	79.2 (2.0)	51.2 (3.7)

Appendix J. Percent N remaining and C to N ratios of foliar litter during 82 weeks of decomposition (March 2010 – October 2012) within naturally regenerating forests, urban forests, and pine plantation sites in western Florida. Standard errors of the means are in parentheses. Natural forest oak = NFO; Urban forest oak = UFO; Natural forest pine = NFP; Urban forest pine=UFP; and Pine plantation = PP.

	Week									
	0	**2**	**4**	**6**	**12**	**18**	**28**	**42**	**60**	**82**
					N remaining (%)					
NFO	100 (0)	86.6 (4.3)	70.1 (8.3)	93.1 (4.7)	84.0 (3.5)	101.8 (8.4)	93.5 (7.8)	93.3 (2.63)	84.8 (7.8)	59.5 (4.4)
UFO	100 (0)	94.1 (3.3)	49.7 (7.0)	99.3 (5.0)	93.8 (7.2)	107.7 (6.5)	92.2 (4.9)	95.9 (7.4)	90.8 (13.0)	44.9 (16.0)
NFP	100 (0)	88.3 (9.6)	54.1 (18.1)	100.0 (6.5)	87.7 (4.9)	107.9 (9.6)	88.3 (9.9)	97.9 (10.4)	102.9 (2.7)	89.5 (1.4)
UFP	100 (0)	79.3 (18.6)	43.5 (1.3)	92.4 (16.4)	97.0 (15.6)	89.9 (16.3)	84.1 (19.5)	92.3 (26.4)	88.6 (23.5)	76.9 (13.2)
PP	100 (0)	75.3 (3.0)	73.3 (10.4)	96.2 (9.6)	89.6 (8.7)	87.3 (4.6)	83.4 (11.3)	87.1 (14.2)	94.2 (12.5)	102.6 (7.5)
					C to N ratio					
NFO	65 (12)	74 (11)	87 (5)	56 (4)	71 (11)	46 (2)	48 (7)	34 (4)	38 (4)	28 (3)
UFO	71 (8)	73 (11)	155 (42)	69 (9)	65 (7)	57 (7)	61 (7)	48 (6)	49 (8)	50 (17)
NFP	162 (27)	177 (12)	355 (83)	150 (15)	168 (19)	132 (13)	152 (13)	128 (15)	102 (10)	86 (4)

UFP	143 (39)	177 (39)	252 (90)	141 (27)	139 (35)	123 (30)	140 (36)	132 (36)	117 (34)	79 (13)
PP	165 (7)	224 (12)	223 (28)	171 (13)	184 (11)	171 (3)	172 (16)	169 (31)	142 (14)	96 (13)

Appendix K. Percent P remaining (on an ash free basis) and C to P ratios of foliar litter during 82 weeks of decomposition (March 2010 – October 2012) within naturally regenerating forests, urban forests, and pine plantation sites in western Florida. Standard errors of the means are in parentheses. Natural forest oak = NFO; Urban forest oak = UFO; Natural forest pine = NFP; Urban forest pine=UFP; and Pine plantation = PP.

	Week									
	0	**2**	**4**	**6**	**12**	**18**	**28**	**42**	**60**	**82**
					P remaining (%)					
NFO	100 (0)	79.4 (11.3)	92.7 (12.5)	100.0 (10.0)	75.5 (12.5)	78.9 (23.9)	78.7 (10.7)	82.3 (12.2)	49.9 (5.6)	48.4 (6.9)
UFO	100 (0)	106.8 (12.2)	103.3 (9.2)	103.6 (4.9)	89.2 (11.2)	75.5 (8.4)	81.0 (7.6)	75.7 (10.1)	51.9 (5.1)	47.9 (12.8)
NFP	100 (0)	133.7 (20.7)	127.6 (16.3)	142.1 (9.5)	124.7 (18.8)	75.7 (15.8)	158.8 (38.5)	116.0 (22.5)	56.5 (5.3)	83.0 (5.1)
UFP	100 (0)	157.1 (38.2)	177.3 (44.8)	110.5 (17.7)	103.9 (3.0)	47.2 (7.6)	202.9 (45.5)	140.2 (19.7)	60.4 (9.8)	74.4 (4.3)
PP	100 (0)	133.5 (7.8)	203.3 (32.8)	138.3 (21.4)	158.8 (27.3)	101.7 (6.2)	208.5 (64.6)	136.0 (17.3)	117.8 (26.5)	148.9 (14.5)
					C to P ratio					
NFO	472 (90)	609 (119)	480 (56)	440 (77)	567 (24)	563 (21)	403 (26)	363 (78)	468 (46)	318 (32)
UFO	439 (60)	396 (33)	399 (52)	401 (43)	427 (11)	512 (72)	428 (50)	368 (23)	533 (83)	339 (71)
NFP	1445 (404)	1003 (294)	989 (174)	980 (297)	948 (123)	1573 (349)	705 (69)	861 (142)	1248 (91)	738 (96)
UFP	1627 (337)	1141 (372)	1104 (425)	1396 (299)	1459 (312)	2425 (953)	681 (49)	1014 (184)	1491 (284)	1026 (368)

PP	2051 (332)	1553 (333)	1035 (188)	1391 (4)	1187 (25)	1737 (229)	932 (284)	1207 (265)	1473 (417)	795 (192)