ABSTRACT

Title of Document:	ECOLOGICAL VALUES AND ECOSYSTEM SERVICES OF NATURAL FORESTS: A STUDY OF PRINCE WILLIAM FOREST PARK, VIRGINIA
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The Urban Forest Effects (UFORE) model developed by the USDA Forest Service quantifies the ecological benefits of urban forests. UFORE has been used to analyze many urban areas, including National Park land in Washington, D.C., but has not been applied to natural forests. We conducted a UFORE analysis of Prince William Forest Park for species composition and individual tree characteristics including tree height, DBH, canopy architecture, and general tree health, collecting data during the 2007 field season. The results show that the park contains over 6,287,000 trees and these trees store 394,000 tons of carbon with an annual net sequestration rate of 12,300 tons. This forest also abates 414 tons of air pollution annually. These results quantify and affirm to policymakers and the public the value and ecological importance of the forests managed by the National Park Service surrounding metropolitan Washington, D.C.

ECOLOGICAL VALUES AND ECOSYSTEM SERVICES OF NATURAL FORESTS: A STUDY OF PRINCE WILLIAM FOREST PARK, VIRGINIA

By

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Chapter 1: Introduction

Introduction

Forests account for 642 million of the 2.3 billion acres of the United States (USDA Economic Research Service 2002). Not included in this total are 84 million acres of National Parks that are considered "special use" rather than forest by the Economic Research Service (ERS). The value of natural forests is commonly considered from the perspectives of timber production and recreational usage. However, much of the current research in natural forests focuses on the effects of air pollution and greenhouse gases on plant growth and forest health rather than on the value of forest products (e.g. Runion *et al.* 2006, Nemani *et al.* 2003, Norby *et al.* 2002, and Wellburn 1998). These values include such disparate ecosystem functions as pollination, food production, wastewater management, air quality improvement, and carbon storage. These benefits that humans derive from ecosystem functions are considered ecosystem services (Costanza *et al.* 1997).

Analysis of Forest Ecosystems in the U.S. Piedmont

Due to their obvious interactions with human populations, urban forests have been one of the first forest types studied extensively by the USDA Forest Service for the value of the ecosystem services they provide (Nowak *et al.* 2006a, 2006b, 2002b, Nowak and Crane 2002). Natural forests—defined here as forests that exist on land that does not meet the Census Bureau definition of urban land as 5000 people per square mile in residency—have not received the same degree of scrutiny for the ecosystem services provided as individual sites. On the other hand, the ecology of the natural forests of the Piedmont geographic region has been studied since European settlement (e.g. Godfrey 1980, Braun 1950, and Oosting 1942).

Some of the seminal studies on forests of the Piedmont include those of Henry Oosting (1942) working on the Piedmont Plateau of North Carolina. Oosting documented changes in species composition in plant communities following abandonment of agricultural fields through to the establishment of climax communities. He observed ecological succession of abandoned fields transforming into forests of loblolly pine (*Pinus taeda* L.), mixed with shortleaf pine (*Pinus echinata* Mill.) and/or Virginia pine (*Pinus virginiana* Mill.). He also documented that bottomland forests varied in the region based on disturbance history and moisture content of soils, with pine species becoming increasingly rare and hardwoods such as birch (*Betula sp.*), American sycamore (*Platanus occidentalis* L.), and sweetgum (named redgum by Oosting, *Liquidambar styraciflua* L.) the common dominants (Oosting 1942). Regardless of the community described, Oosting took pains to identify as many species as possible in order to classify the community he studied as well as the age of the community. Hence, abandoned fields include detailed descriptions of the existing species that include mention of many grasses and common weeds.

Subsequently, E. Lucy Braun published the book *Deciduous Forests of Eastern North America* (1950) within a decade of the publication of Oosting’s work. Instead of meticulously examining every species found within a given geographic region, Braun looked at the dominant tree species of entire regions and developed association data over the United States east of the Mississippi River. She categorized the forests of the east into major groups such as mixed mesophytic, oak-hickory, oak-chestnut, and oak-pine before further breaking them down into associations within these larger groups and then detailing the deviations that could be found within each association. She classified northeastern Virginia primarily as an oak-chestnut association but noted the loss of American chestnut (*Castanea dentata* (Marsh.) Borkh.). She also included mixed mesophytic forest and oak-pine forest as common forest types in the region (Braun 1950). The typical forest of the region consisted of various oak species such as white (*Quercus alba* L.), black (*Quercus velutina* Lam.), chestnut (*Quercus prinus* L.), post (*Quercus stellata* Wangenh.), and blackjack (*Quercus marilandica* Münchh.) oaks. Common codominant species or accessory trees in the region included tulip-poplar (*Liriodendron tulipifera* L.), red maple (*Acer rubrum* L.), black tupelo (*Nyssa sylvatica* Marsh.), and dogwood (*Cornus spp.*). Less attention was paid to the herbaceous cover found in the forest and more emphasis was placed by Braun on the tree and shrub species found in a given community. Like Oosting, the primary objective of this work was to describe the vegetation of the region and document its succession as well as the environmental limitations on species composition and distribution.

By the 1970s many scientists had begun to examine physiological aspects of the forest community in an effort to understand interactions between plants and their surroundings. Kinerson (1975) examined the relationship in loblolly pine (*Pinus* *taeda* L.) between the surface area of stem and needles and respiration by analyzing the exchange of CO_2 between the atmosphere and the surface of loblolly pine trees growing in a plantation in the North Carolina Piedmont. Annual CO_2 flux measurements of the bole, root, leaf and branch portions of the tree were made and compared with the quantity of heartwood and sapwood in multiple pines. These data were determined by carefully removing each branch, trunk, and root of a given tree and establishing the total quantities of living and dead material within a given tree. Kinerson (1975) found that respiration rates increased with the onset of spring due to cambial growth and with increases in temperature and concluded with annual CO_2 flux measurements of the bole, root, leaf, and branch portions of the tree. Studies such as this began to expand our understanding of the forest beyond the more descriptive studies of Oosting and Braun and focused attention on the biological processes of the plants within the forest.

Numerous studies have also evaluated the impacts of acid precipitation and air pollutants such as ozone on eastern forests. For example, Kohut (2007) evaluated the impacts of ozone damage on National Parks in the Vital Signs Monitoring Network. The Network was initiated to assist parks in inventory and monitoring of biota that are at risk of change or degradation (Kohut 2007). Kohut (2007) evaluated the risk of future ozone damage by examining ozone and soil moisture records over five years and compiling a list of ozone-sensitive species within each park. Parks that exceeded specific levels of ozone exposure and possessed specific soil moisture characteristics were given higher danger ratings (Kohut 2007). Parks that were considered particularly susceptible to damage included parks in the National Capital Region, such as Prince William Forest Park. These ratings were not quantitative estimates of potential damage, but rather qualitative predictions (Kohut 2007). That study did not attempt to evaluate the total costs of damages to these parks, so the ecosystem services of the parks were not given a specific monetary value. However, one could speculate that parks at high risk of damage could lose some of their value as a result of ozone exposure.

A number of other ecological studies were conducted on the forests of the Piedmont region in the twentieth century (Abrams and Copenheaver 1999, Cole and Ware 1997, Godfrey 1980, Braun 1950, Oosting 1942). For example, Abrams and Copenheaver (1999) examined the forest community structure at Great Falls National Park in northern Virginia. The forest overstory at Great Falls National Park was dominated by white oak, northern red oak (*Quercus rubra* L.), tulip poplar (*Liriodendron tulipifera* L.), American beech (*Fagus grandifolia* Ehrh.), and pignut hickory (*Carya glabra* (Mill.) Sweet). The oldest trees in the park were white oak trees that showed a noticeable difference in growth rates in different centuries that could be attributed to factors as diverse as chestnut blight and reduced quantity and intensity of forest fires (Abrams and Copenheaver 1999). Oak regeneration during the twentieth century was quite low in the park, a pattern that they identified as common among mixed-oak forests in the eastern United States.

Descriptive studies such as these provide documentation on the composition of forests of the Coastal Plain and the Piedmont in the eastern United States. They also provide a base to our understanding of how these forests change over time and respond to disturbance. This understanding is necessary in order to look forward at the future regional forests, their ecological roles, and the ecosystem services they provide.

Ecosystem services

Recently a number of studies have begun to assess the ecosystem services provided by forests. For example, an attempt at approaching this problem was made by Ehrlich and Mooney (1983) who reviewed the inherent difficulties in evaluating the changes in ecosystem services that are the result of extinctions. They point out the wide range of effects possible from the loss of a single species and suggest that the removal of one species could cause the collapse of an entire ecosystem. For example, they examined the establishment of plantations of Monterey pine (*Pinus radiata* D. Don) in Australia in the early twentieth century to replace forests lost to deforestation. The replacement stands of Monterey pine were found to provide reduced energy flow within the ecosystem and slower nutrient cycling that resulted in a loss of soil nutrients under management conditions of the time (Ehrlich and Mooney 1983).

One of the seminal papers on ecosystem service valuation is the valuation of global ecosystem services by Costanza *et al.* (1997) published in *Nature*. They defined ecosystem functions as “the habitat, biological, or system properties or processes of ecosystems” and ecosystem goods and services as “the benefits human populations derive, directly or indirectly, from ecosystem functions”. They further discussed how changes in the availability of these services affect human well-being and noted that changes occur at large scales and small. An example of a large change at a small scale is a dramatic change in local forest composition (Costanza *et al.* 1997). Their global estimate of ecosystem service values is derived as a synthesis of values for different biomes based on the 'willingness-to-pay' of individuals for given services. The sum total of the value of 17 different ecosystem services across all biomes was $33 trillion. Ecosystem services included pollination, gas regulation, erosion control and sediment regulation, climate regulation, nutrient cycling, and food production, among others. Costanza *et al.* (1997) were unable to calculate a value for some ecosystem services due to lack of data. In particular, two services were not assessed for forests: pollination and gas regulation services, which are defined as "regulation of atmospheric chemical composition" and includes the CO_2/O_2 balance. The missing services add uncertainty in the Costanza *et al.* (1997) model and suggest that the estimate is likely low. Although this study failed to solve all issues varying from economic valuation methodologies to errors derived from extrapolation of data from smaller scales to larger, this article has stood as a foundation for assessment of ecosystem service values covering a variety of scales and different types of ecosystem services.

There are also studies in the existing literature that have evaluated the ecosystem services provided over large yet discrete land areas. One such study was the assessment of the ecosystem services provided by the U.S. National Wildlife Refuge System, undertaken by Ingraham and Foster (2008). Through use of GIS technology, they estimated the non-use ecosystem services attributable to these lands.

The 13.3 million acres of land in the Refuge System were estimated to accrue $26.9 billion in value annually primarily from wetlands. Ingraham and Foster (2008) suggested that the total ecosystem service value of the Refuge System could be attained by combining their results with the results of others that might look into the recreational value and use value of ecosystem services for the same region.

Ecosystem services have also been assessed on the basis of undisturbed versus restored ecosystems. Dodds *et al.* (2008) examined the ecosystem services of the five largest coterminous ecoregions of the United States and compared the benefits of lands that were unaffected by anthropogenic disturbances to the benefits of lands that had been restored to their original use. They used a variation of the ecosystem service categories employed in the Costanza study to facilitate comparison (Dodds *et al.* 2008). The analysis was limited to restored lands created within ten years of the study's initiation, which reflects the timeframe of many projects (Dodds *et al.* 2008). Native and restored wetlands supplied the greatest value of ecosystem services overall, while the Great Plains supplied the highest biodiversity benefits in both native and restored ecosystems. The benefits derived from ecosystems in the Dodds study are greater than those of the Costanza study. The Dodds study obtained data for several ecosystem functions that were unavailable for the Costanza study, such as gas regulation (Dodds *et al.* 2008). Focus on the coterminous United States by Dodds versus the global evaluation of the Costanza study is also cited as a possible factor for the higher estimate of ecosystem services (Dodds *et al.* 2008). One cautionary note by the authors was that their particular methodology is susceptible to double-counting of ecosystem services, moreso than accounting for each individual service (Dodds *et al.* 2008, Costanza *et al.* 1997).

Several studies over the last several decades have assessed how the various components of global climate change may impact eastern Piedmont forests (Finzi *et al.* 2006, Moore *et al.* 2006, Hamilton *et al.* 2004, DeLucia *et al.* 1999). Albani *et al.* (2006) studied the ecosystem services of carbon storage and sequestration on a regional scale. Ecosystem demography models—models employing individual-based vegetation data, abiotic factors, and other biotic factors to predict future ecosystem structure and function—were used to predict that the East Coast will shift from a carbon sink to a carbon source by 2100 (Albani *et al.* 2006). The basis for these predicted changes is the current understanding of plant natural history, land-use changes, and ecosystem development. Land-use by the end of the twenty-first century is predicted to change to permit forests to remain unharvested instead of the modern cycle of harvest and regrowth (Albani *et al.* 2006). Trees growing today in the mid-Atlantic region are young and growing quickly, but mature trees have reduced net annual CO_2 uptake. The younger tree population will mature by the end of the century and may no longer serve as significant carbon sinks. This shift in land-use would result in a shift in forest CO_2 uptake from sink to source (Albani *et al.* 2006).

Large-scale ecosystem service valuations are more common in the literature than smaller-scale valuations for single forests. One example of a local scale study was recently completed in China by Chen *et al.* (2009). They used GIS technology to map Tiantai County in southeast China and calculated the direct use value of ecosystem services in three categories: agricultural products (based on yield per acre), forest products (derived from stumpage per acre), and tourism (from annual tourism income). Chen determined the agricultural, forest, and tourism values of each cell, and summed the values from each cell for each category to estimate a value for the ecosystem services in the county. Agricultural products were the greatest source of ecosystem services, representing 65% of the benefits, followed by forest products (30%) and tourism (5%) (Chen *et al.* 2009). Management recommendations based on the study included maintenance of more lucrative cash crops in specific areas, establishment of riparian buffers to increase the nutrient uptake of agricultural lands, and an increase in tourism (Chen *et al.* 2009).

Models exist to assess the ecosystem services for urban forests as well as natural forests. One commonly-used model for urban ecosystem service assessment is the Urban Forest Effects (UFORE) model (USDA UFORE 2007). The UFORE model uses weather and pollution data combined with field sampling to estimate the ecosystem services provided by an urban forest. UFORE estimates the carbon storage, carbon sequestration, and air pollution abatement of the urban forest as well as evaluating the vulnerability of the forest to several specific pests and pathogens. Several other models that can quantify the ecosystem services provided by forests include CITYgreen, STRATUM, and the Forest Inventory and Analysis (FIA) model used by the USDA Forest Service. Each model has different strengths based on the design of the application that emphasize specific ecosystem services that are of value to the intended clients.

CITYgreen uses a landcover dataset provided by the user to estimate ecological and economic benefits of urban forests (American Forests 2009). This model can assess stormwater runoff, carbon storage and sequestration, air pollution removal, and also estimate the results of management activities under different scenarios (American Forests 2009). CITYgreen functions at fine scale and can assess the benefits that a single tree provides (American Forests 2009).

STRATUM is similar to the UFORE model and also is used by the USDA Forest Service (US Forest Service 2009). This model takes tree inventory data and data regarding the existing community such as the budget available for management, community electrical usage, and community population to calculate the forest's structure, function, value, and management needs (US Forest Service 2009). The model is not designed to assess the entire urban forest but just street trees (US Forest Service 2009). The outputs of this particular model can be used to assist project planning and management decisions by presenting the results of different scenarios.

The Forest Inventory and Analysis (FIA) program thoroughly details forests through establishment of semi-permanent plots and subplots (USDA Forest Service 2007). Plots have been established across the United States to characterize the benefits of the nation's forests (USDA Forest Service 2007). Once plots are established, metrics that are measured include crown class, number of stems, tree damage, year of mortality, and many others (USDA Forest Service 2006). The primary focus of the FIA is management of forests as a source of timber, which is a direct-use ecosystem service. Data regarding woody debris and other ecological values are also collected (USDA Forest Service 2006).

Carbon sequestration is an ecosystem service that has received increased scrutiny in the literature over the past decade. Many plants respond to rising atmospheric CO_2 levels with an increase in net primary or ecosystem production due to CO_2 fertilization (Mohan *et al.* 2007, Runion *et al.* 2006, LaDeau *et al.* 2001, Norby *et al.* 1999). However, models and experiments conducted globally do not always agree that an increase in net primary or ecosystem production is occurring or will occur (Hansen *et al.* 2001). For example, in a simple community designed by Runion *et al.* (2006) to model the effects of CO_2 enrichment on plants, plants with C_4 metabolism attained only modest growth. Those species expected to perform well (e.g. C_3 plants) varied in overall performance (Runion *et al.* 2006).

The sequestration of carbon within plant tissue is affected by various other environmental parameters. Nitrogen and water availability are key determinants of the ability of plants to effectively use elevated levels of CO_2 to enhance carbon sequestration and storage (Heimann and Reichstein 2008, Bytnerowicz *et al.* 2007, Beedlow *et al.* 2004). Environmental stresses such as ozone or UV-B radiation may act as a potential offset to the predicted increases in net primary productivity by damaging plant tissues sufficiently to offset potential gains by increased growth (Zak *et al.* 2007, Kohut 2007, Bytnerowicz *et al.* 2007, King *et al.* 2005, Coulston *et al.* 2003, Teramura and Sullivan 1994). These other environmental factors should be included in models in order to predict the services provided by natural forests under various management scenarios.

Goals

The extensive ecological studies of the Piedmont region make this region particularly suitable for an ecosystem service assessment. With increasing population pressure and widespread urbanization, the natural forests of the Piedmont face exposure to increasing quantities of urban air pollution and fragmentation. The effectiveness of natural forests as providers of ecosystem services must be determined to permit accurate assessment of their value in the future plans of policymakers nationwide. The widespread existence of ecological data on the region can then be used to make management decisions based on the ecosystem services provided and the goals of the organization responsible for a given forest.

Prince William Forest Park (PWFP) lies in the margins of the heavily urbanized metropolitan Washington DC area. Given that the human population within the region continues to grow, pressure from the citizenry on lawmakers to utilize the existing natural forests for resources to drive economic growth will likely increase. The ecosystem services provided by natural forests will be lost if lawmakers succumb to this pressure. Urban sprawl has already increased the fragmentation of the forests of the mid-Atlantic region. Influenced by global climate change, Prince William Forest Park could change dramatically over the course of the twenty-first century. With this in mind, it is important to evaluate the ecosystem services of this Park in order to establish its current ecological value and to provide a benchmark for other parks in the National Capital Parks region.

Therefore, the purposes of this study were twofold. First, I attempted to use

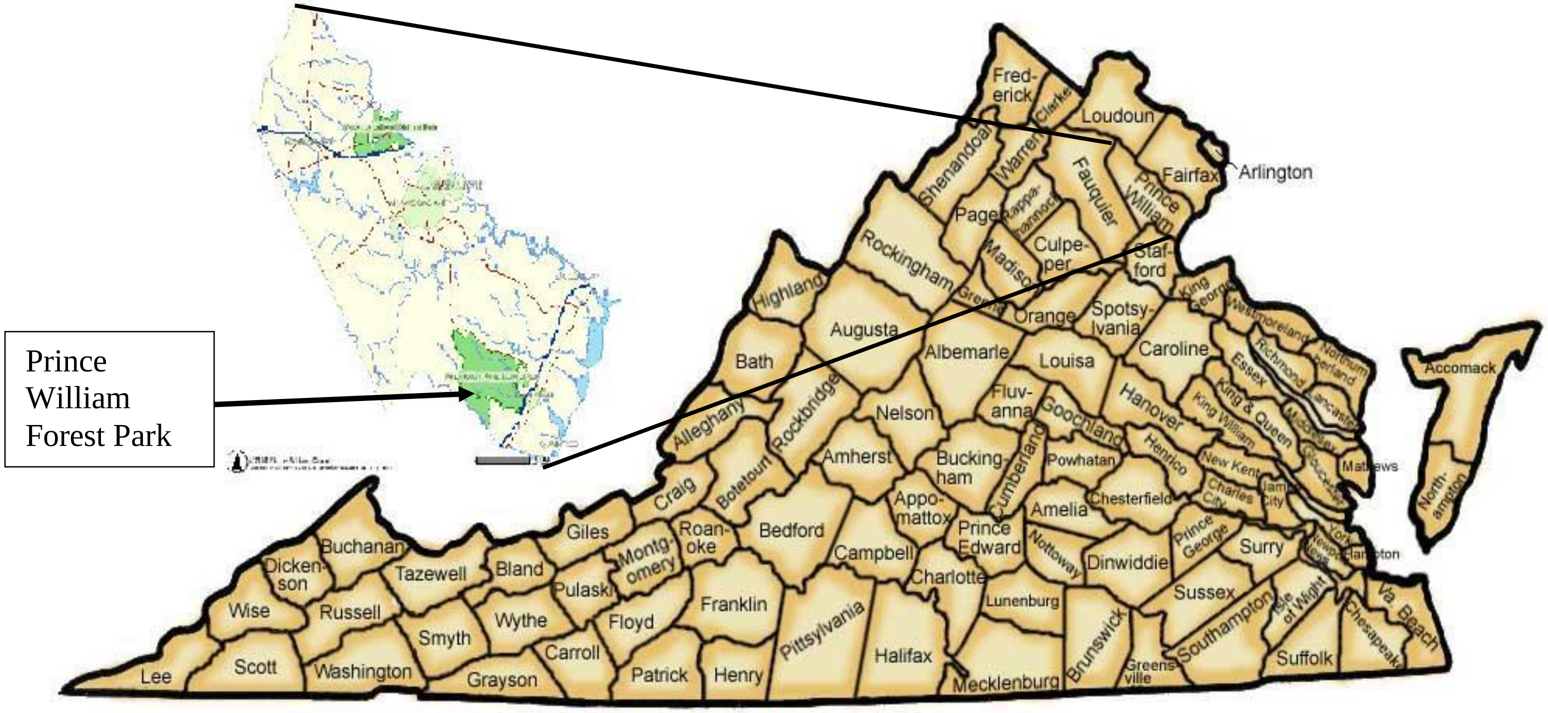
Prince William Forest Park as a model system for the application of the UFORE model to a natural ecosystem and assess the ecosystem services provided by the natural forest. Carbon storage and sequestration, tropospheric ozone reduction, sulfur and nitrous oxide abatement, as well as particulate matter levels (specifically particulate matter at the ten-micron size or less) were calculated by the UFORE model in order to place an estimated value on the ecosystem services provided by this forest. Upon conclusion of this portion of the study, I then attempted to determine the future species composition of Prince William Forest Park by looking at model predictions of the future climate of the region. I also consider the impact of pest and pathogen outbreak on the future services that will be provided by the park.

Chapter 2: Methods

Study site: Prince William Forest Park

This study was conducted in Prince William Forest Park (PWFP), located in Prince William County, Virginia (Figure 1), approximately forty-eight kilometers south of Washington D.C. along Interstate 95. Prince William Forest Park varied in land usage from forest to farmland and housing to pyrite mines and social services before establishment as a National Park on November 14, 1936 with the establishment of the Chopawamsic Recreation Demonstration Area (National Park Service 2008). The modern park is primarily forest with a limited number of trails and roads connecting campgrounds and buildings. Quantico Creek divides the forest before emptying into the Potomac River at Possum Point. We surveyed trees in 5,090 hectares of second-growth forest within the park. Prince William County contains 39,817 hectares of forest within its borders, excluding Prince William Forest Park and the independent cities of Manassas and Manassas Park.

Figure 1: The location of Prince William Forest Park in Prince William County, Virginia.



Prince William Forest Park has served in a variety of roles prior to its establishment as a National Park. The park served an important role as a training ground for the Civilian Conservation Service (CCS) during the Great Depression. During World War II, the park also served as a training ground for spies under the Office of Strategic Services (National Park Service 2008). Historically, much of the park was previously farmland or pyrite mine. Remnants of these diverse activities are visible in such locations as the Farm-to-Forest trail, the historic cabins located at various campgrounds, and the Cabin Branch Pyrite Mine. Today, Prince William Forest Park provides campgrounds, hiking trails, and opportunities for canoeing along the North Quantico Creek.

Prince William Forest Park contains two geographic provinces within its boundaries. The eastern third of the park is part of the Atlantic Coastal Plain province, while the western two-thirds are part of the Piedmont province (National Park Service 2008). This park is the largest example of eastern Piedmont forest in the National Parks system (National Park Service 2008). The Coastal Plain region of Prince William Forest Park is characterized by soils only a few million years old (Bailey 1999). The Piedmont region consists of rolling hills and valleys, ranging in elevation from 400 feet to 1000 feet above sea level (Bailey 1999). Soils within the Piedmont region are older and demonstrate greater development than soils of the Atlantic Coastal Plain, based on parent materials of variable age extending back 500 million years. The parent materials of the soils within Prince William Forest Park trace their origins to the Cambrian and Ordovician periods (Bailey 1999) although a region exists to the northwest of the park with Triassic sediments (Farrell and Ware

1991). According to Braun (1950), this region is part of the historic oak-chestnut association, though the American chestnut (*Castanea dentata* L.) has been replaced by hickory (*Carya spp.*) as the co-dominant species (Godfrey 1980). More recent research debates the importance of hickory as a co-dominant, however (Ware 1992), and emphasizes the change from oak-hickory forest to a maple-beech association (Abrams 1999).

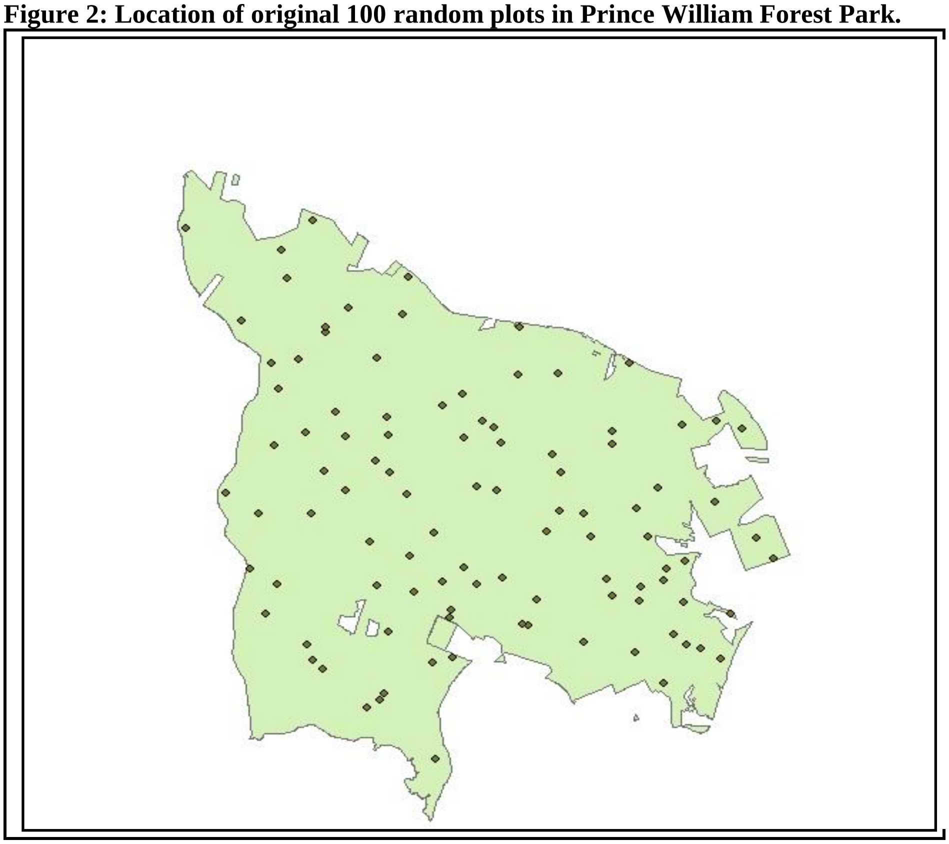
Urban Forest Effects (UFORE) model

The primary tool for this study was the UFORE model. Formulas from the existing literature were used to calculate the structural and functional values of the forest (USDA UFORE 2008). In addition to field data, the UFORE model required hourly air pollution data and meteorological data for the region surveyed as well as boundary layer height measurements (Nowak *et al.* 2005). The air pollution and meteorological data for the Prince William Forest study came from Manassas, a city located 20 kilometers northwest of the park. The U.S. Environmental Protection Agency (EPA) actively monitors pollutants throughout the U.S. that can harm the environment, injure humans, and cause property damage (U.S. Environmental Protection Agency Air Pollution Monitoring 2008). Air pollution abatement for five pollutants was assessed in the UFORE model: carbon monoxide, ozone, sulfur dioxide, nitrogen dioxide, and particulate matter.

Field techniques

One hundred random points were selected by the USDA Forest Service on a map of the park (Figure 2). Plot distribution was limited to the undisputed park property, a region 5,090 hectares in size. The research team used a Trimble GPS Pathfinder Pro XRS Receiver™ (Trimble Navigation Limited—Sunnyvale, CA) to locate the plots in the field via GPS coordinates. Data were recorded manually and then entered into the *i-Tree* program provided by the USDA Forest Service as well as into an Excel spreadsheet.

Figure 2: Location of original 100 random plots in Prince William Forest Park.



Plots were 0.04 hectares in size. One or two reference objects were chosen whenever possible within each plot for use as future reference points for quality control. For the purposes of this study, any woody plant with a diameter at breast height (DBH) of 2.5 cm or greater was considered a tree and evaluated accordingly. Data collection included DBH, tree height, crown heights, and crown width in two dimensions. The percent of canopy missing was estimated by observing the crown and visually estimating the quantity of foliage missing based on the existing crown architecture. Percent dieback was estimated visually as well and defined as any portion of the crown that was devoid of branches for any reason except natural self-pruning. The percentages of impervious ground cover and percent shrub cover beneath the canopy of a given tree were both visually estimated. Crown light exposure values were estimated visually by walking around the tree to determine how many cardinal directions the tree would receive full sunlight for a significant portion of the day. Trees were rated from 0 (completely boxed-in) to 4 (open-on all sides). An additional "side" was included in the total if the tree was not overtopped by adjacent trees, resulting in a total scale of 0 to 5. Each tree was identified to species on-site if possible using a field guide published by National Audubon Society (1980) or the Peterson Field Guide series (1972). If identification on-site was not possible, samples were collected of living trees for later identification. Follow-up visits to the park were made for species that were improperly identified or suspect. Upon conclusion of data collection within each plot all evidence of the research team's presence was removed at the National Park Service's request.

Once data collection was complete, the assembled data were sent to the Forest Service for processing. Model outputs included ecological data such as species richness, leaf area, Shannon-Wiener index values, Menhinick's diversity index, species evenness, Simpson's diversity index, percentage of the forest consisting of exotic species, and tree density of each species and the forest as a whole. Leaf area/area (as m^2/ha) of each species was calculated with the UFORE model. These values were transformed into leaf area index (LAI) by converting units from $m^2\ ha^{-1}$ to $m^2\ m^{-2}$ for each species and for the sum of all species. Relative LAI was computed as the LAI of a species divided by the sum of LAI for all species and expressed as a percentage. Importance values (I.V.) of species that made up > 2% of all samples were computed as the sums of relative frequency, relative density, and relative dominance in this study. Relative frequency was calculated as the number of plots a species was sampled in divided by the total number of plots multiplied by 100. Relative density was computed in two steps. Species density was taken directly from UFORE model outputs in the first step. In the second step, the density of a species was divided by the total density of all trees and multiplied by 100. Relative dominance was based on basal area calculations. Basal area of all trees of a species was computed from DBH values taken from the field. If a sampled tree in the field had more than one stem, basal area for each stem was computed and summed. Finally, the basal area of a species was divided by the basal area of all sampled trees and multiplied by 100 to compute relative dominance.

In addition to ecological outputs, the UFORE model estimated the total quantity of carbon stored in the forest and the individual species' contributions to this quantity. Carbon sequestration was estimated by the model for the forest and on a per-species basis as well. Both carbon storage and carbon sequestration were estimated across size classes in addition to the species-based estimates. Size classes ranged from 2.5-7.6 cm for the smallest class and increased by 7.6 cm intervals. No trees were sampled within two of the largest size classes but one tree was found in the largest class (99.1-106.7 cm). Additionally, the amount of air pollution abated by the forest was estimated. Carbon storage, carbon sequestration, and air pollution abatement estimates were all derived from empirically-derived formulae in the literature.

The UFORE model provided a simple assessment of the health of each species in each size category. Potential damage estimates for four pest species were calculated based on the species composition of the forest and measured in terms of percent of the forest susceptible to attack and the total damage possible by the species in U.S. dollars. The pest species in question were Asian long-horned beetle (*Anoplophora glabripennis* Motschulsky) (ALB), emerald ash borer (*Agrilus planipennis* Fairmaire) (EAB), Dutch elm disease (*Ophiostoma ulmi* (Buisman) Nannf.) (DED), and gypsy moths (*Lymantria dispar* L.).

Carbon sequestration is measured empirically through eddy-covariance techniques or chamber studies. Values from the UFORE study are derived from estimated carbon storage in the next year, probability of tree death based on health at the time of the study, and an estimate of the decomposition rate of wood in order to predict the net carbon sequestration of the forest. The decomposition rate of wood is highly variable based on lignin content, ambient moisture levels, presence of suitable decomposers, and other site characteristics. The UFORE model assumed that standing dead trees decomposed over the course of 20 years. Trees that have fallen to the ground were not counted in the analysis.

The UFORE field manual procedures served as the guidelines in the field for the performance of this study (USDA Forest Service UFORE Field Manual, 2008).

Data analysis

After fieldwork was completed, the plot locations were placed onto a vegetation map provided by the Park Service. Using ArcMap 9.2 (ESRI—Redlands, CA) land cover was measured for the relative contributions of each land cover type to the park's land area. The plot locations were also sorted to determine the number of plots surveyed within each land cover type. This was done to check that the random samples represented the landscape accurately. The plots were found in six different land cover types: floodplain, grassland, mixed hardwood, mesic, Virginia pine, and oak-Virginia pine. Comparison of the plot locations to the actual proportions of the park in each land cover type showed that the random plot locations reflected the actual land cover composition of the park with acceptable accuracy (Table 1).

Table 1: Comparison of sample plot land cover locations to actual park land cover.

Land cover type	Percent of all surveyed plots in land cover type	Percent of actual park land area in land cover type
Floodplain	1	3.8
Grassland	1	0.6
Oak-Virginia pine	12	12.0
Mesic	16	15.2
Mixed hardwood	47	43.6
Virginia pine	23	23.9

Statistical analysis of leaf area index, stem count, and relative frequency was done using Statistical Analysis Software (SAS) version 9.2 (SAS Institute Inc.—Cary, NC). Regression models of species' leaf area index contributions to stem count and to relative frequency were graphed using Microsoft Office Excel 2003 (Microsoft Corporation—Redmond, WA). Reported r^2 values and probability were taken from SAS outputs.

Future forest predictions

Climate model data and plant ranges were accessed through the Canadian Forest Service (CFS) website (Canadian Forest Service 2009 Yang 2009). Climate envelopes for over 130 North American tree species were outlined by McKenney in 2007. Species within Prince William Forest Park were examined using the estimated climate envelopes of the current climate and then compared to future projections of climate over three time periods: 2010-2040, 2040-2070, and 2070-2100. The Intergovernmental Panel on Climate Change (IPCC) defines different global emission scenarios for the future, of which two were considered here: A2 and B2. The A2 scenario assumes a world under high GHG emissions, while the B2 scenario assumes lower global GHG emissions (Intergovernmental Panel on Climate Change 2007). Maps of North America for each species were examined to determine if that species' range would include Prince William Forest Park in the given time period and under the chosen scenario. The climate model used for estimates of future climate conditions was the Hadley Center Couple Model, version 3.0 model (HadleyCM3) developed in the UK (Martin *et al.* 2004). Model scenarios included a future with high greenhouse gas emissions and a scenario with lower greenhouse gas emissions, named the A2 and B2 scenarios respectively. These scenarios were described in the Intergovernmental Panel on Climate Change Synthesis Report (2007). Three categories were created to define the prevalence of a species in a given scenario and time period: core, fringe, and not present. The climate envelope of a species is based on known locations where that species has been found. Climate data for each location were pooled. Data included mean annual temperature, mean temperature in the warmest month, mean temperature in the coolest month, mean annual precipitation, mean precipitation in the warmest month, and mean precipitation in the coolest month. If all six factors for a species were within the 5^{th}-95^{th} percentile, the species was considered within its core range. Fringe species were defined as species found within the region but with one or more climate factors lying below the 5^{th} percentile or above the 95^{th} percentile of all known locations within the species' range—near the minimum or maximum tolerance limits of a given species for one or more climate factors. Species with climate envelopes no longer located in northern Virginia were considered absent and listed as 'none'.

Chapter 3: Results

Vegetation Analysis

Over five thousand trees were sampled in the field (Table 1). The UFORE model estimated that the park contained more than 6 million trees at a density of 1126 trees ha^{-1} (Table 1). Tree density varied within sample plots from one tree in a streamside plot (24.7 trees ha^{-1}) to an early successional Virginia pine (*Pinus virginiana* Mill.) plot containing 111 trees (2700 trees ha^{-1}). The average number of trees per plot was 51, and the mean number of live trees sampled per plot was 46 (Table 1).

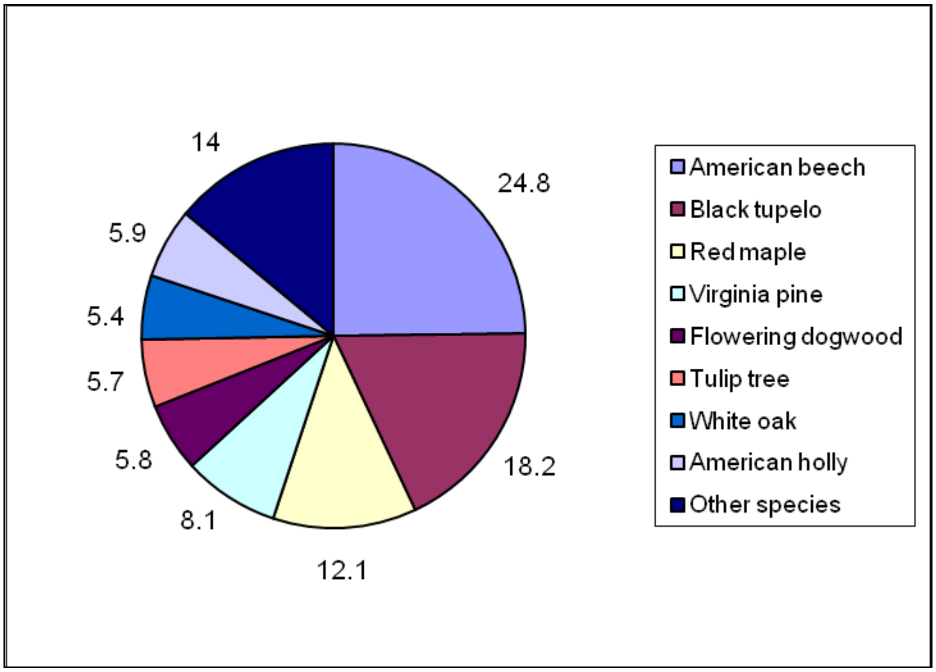
Table 2: Summary outputs from a UFORE analysis of 100 random plots sampled within the Prince William Forest Park.

Total number of trees surveyed	5,099
Survey area (hectares)	5,090
Mean trees/hectare	1,126
Total number of trees in park (estimated)	6,287,267
Mean trees per plot (total)	51
Mean trees per plot (live)	46
Number of tree species surveyed	39
Mean tree DBH (cm)	11.8
Shannon-Weiner Diversity Index	2.45
Leaf Area Index (LAI)	3.36

American beech, black tupelo and red maple accounted for over 50% of the trees samples with beech being the most common tree in the park (Fig. 3). Other canopy species that were common included Virginia pine (*Pinus virginiana* Mill.), tulip-poplar (*Liriodendron tulipifera* L.), and white oak (*Quercus alba* L.). Common understory species included American beech (*Fagus grandifolia* Ehrh.), black tupelo

(*Nyssa sylvatica* Marsh.), and red maple (*Acer rubrum* L.) as well as typical understory species flowering dogwood (*Cornus florida* L.) and American holly (*Ilex opaca* Aiton) (Figure 3).

Figure 3. Proportion of all trees sampled by species in 100 random plots within Prince William Forest Park.



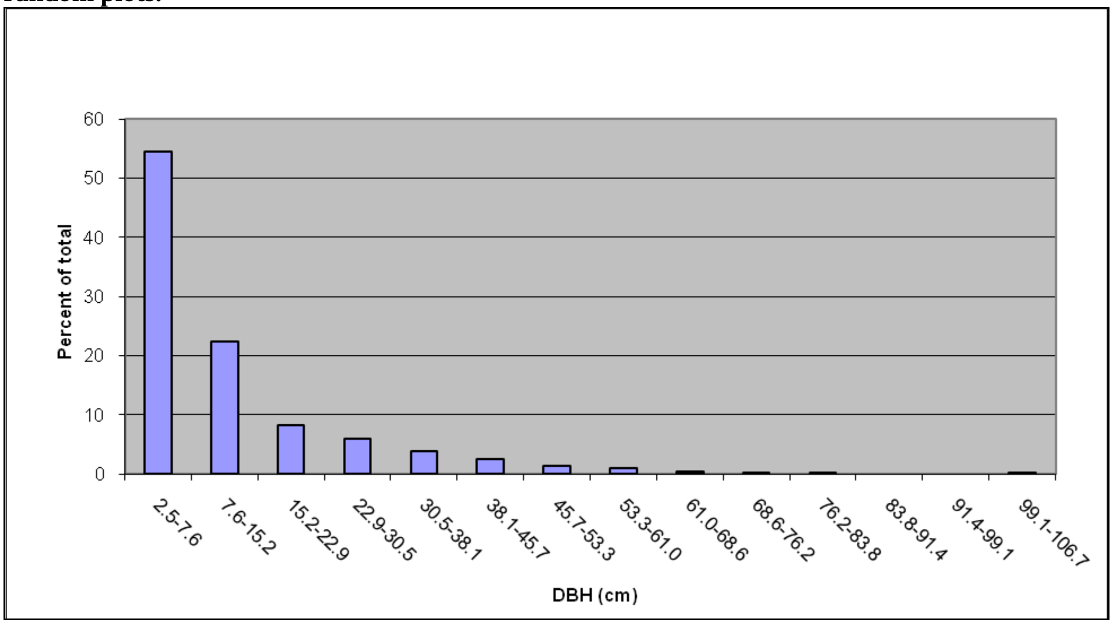
The DBH of sampled trees varied from 2.5 cm to 103 cm. There were more than 4500 live trees sampled, of which nearly 1700 trees were greater than 10 cm DBH and 231 live trees were greater than 20 cm DBH. American beech, black tupelo, and red maple were the most common species sampled (Table 2). When smaller living trees of less than 10 cm DBH were excluded, the most abundant species were beech, red maple, tulip-poplar, Virginia pine and white oak (Table 2).The most common species found that were greater than 20 cm DBH in size were tulip-poplar and white oak, which together accounted for over 40% of the trees in that size class (Table 2). Other notable species in the largest size class were black oak (*Quercus* *velutina* Lam.), Virginia pine, and scarlet oak (*Quercus coccinea* Münchh) (Table 2). The oak genus alone made up nearly half of the number of all trees > 20 cm DBH.

Table 3: Size class contributions of common tree species in Prince William Forest Park.

Species	All trees (number)	Percent of all trees	Trees ≥ 10cm DBH	Percent of all trees ≥ 10 cm DBH	Trees ≥ 20 cm DBH	Percent of all trees ≥ 20 cm DBH
Fagus grandifolia	1,116	24.8	239	14.2	14	6.1
Nyssa sylvatica	817	18.1	115	6.8	4	1.7
Acer rubrum	536	11.9	266	15.8	14	6.1
Pinus virginiana	372	8.3	254	15.0	22	9.5
Cornus florida	275	6.1	36	2.1	1	0.4
Ilex opaca	261	5.8	39	2.3	0	0
Liriodendron tulipifera	259	5.7	206	12.2	51	22.1
Quercus alba	240	5.3	203	12.0	51	22.1
Carya tomentosa	99	2.2	66	3.9	10	4.3
Quercus falcata	82	1.8	14	0.8	2	0.9
Quercus rubra	77	1.7	59	3.5	10	4.3
Carpinus caroliniana	64	1.4	14	0.8	0	0
Quercus velutina	54	1.2	54	3.2	24	10.4
Sassafras albidum	52	1.2	8	0.5	0	0
Liquidambar styraciflua	41	0.9	10	0.6	1	0.4
Juniperus virginiana	28	0.6	8	0.5	0	0
Quercus prinus	27	0.6	20	1.2	5	2.2
Quercus coccinea	19	0.4	19	1.1	15	6.5
Quercus phellos	18	0.4	14	0.8	0	0
Betula nigra	12	0.3	11	0.7	0	0
Quercus bicolor	12	0.3	8	0.5	3	1.3

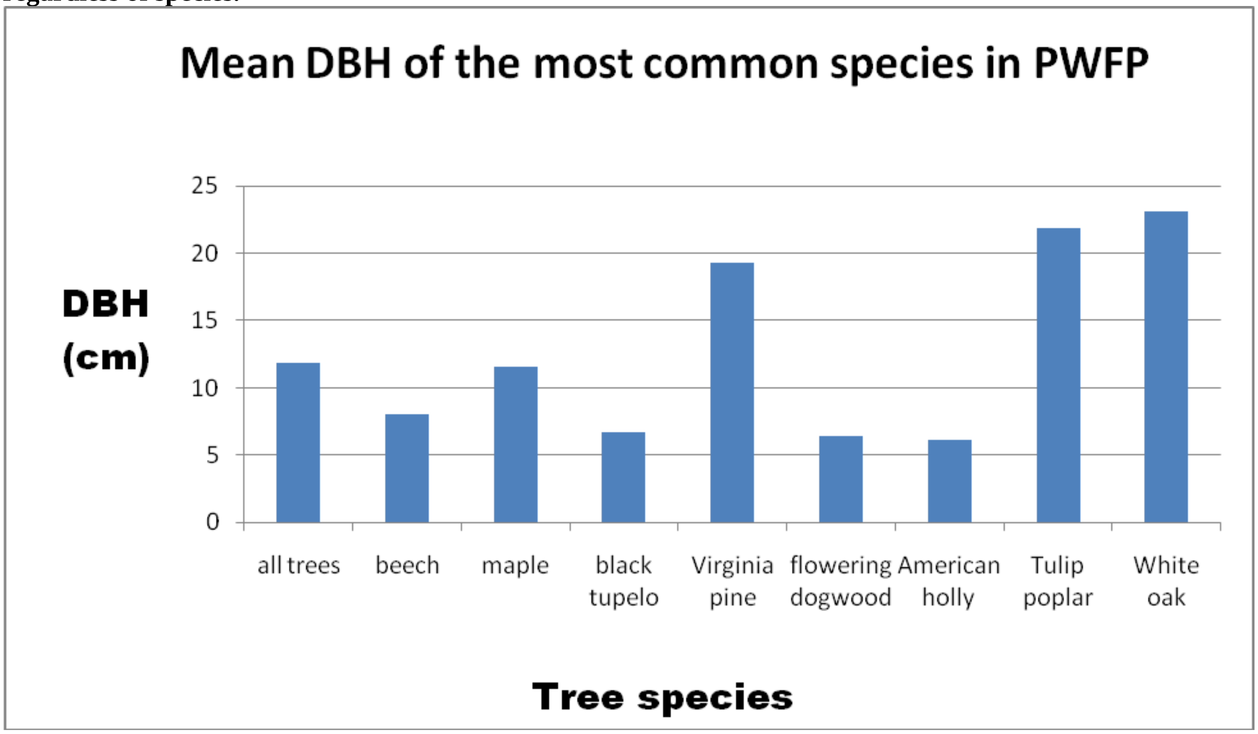
Three-quarters of the sampled trees were less than 15.2 cm in diameter at breast height (Figure 4). Many of these trees were young American beech, red maple, or black tupelo, as well as American holly and flowering dogwood (Table 3). Other species commonly found in the smallest size classes included mockernut hickory (*Carya tomentosa* (L.) Nutt.), several oak species, American hornbeam *(Carpinus caroliniana* Walter), and sassafras (*Sassafras albidum* (Nutt.) Nees).

Figure 4: Size distribution and percentage of trees in each size class of trees sampled in 100 random plots.



The mean DBH of all trees sampled within the park was 11.8 cm (Figure 5). American beech, black tupelo, red maple, American holly, and flowering dogwood all averaged less than 11.8 cm DBH (Figure 5). Common canopy dominants white oak, tulip-poplar, and Virginia pine possessed average DBH values greater than the mean DBH value of all trees (Figure 5).

Figure 5: Comparison of mean DBH of eight most common species sampled in Prince William Forest Park. The first column represents the mean DBH of all trees sampled in the park regardless of species.



The estimated leaf area index (LAI) of the entire park was 3.36 (Table 2). American beech provided the greatest proportion to the LAI within the forest as relative LAI. Species frequency was a fair indicator of the relative contribution to total LAI of a given species in the park ($p<0.0001$, $r^2=0.57$) (Figure 56. However, there was variation probably due to tree size. The total number of individuals or stems was not as effective in predicting the relative LAI of a given species ($p<0.0001$, $r^2=0.49$) (Figure 7). More common species generally provided greater relative LAI to the park's total leaf area.

Figure 6: Relationship between relative leaf area index and species' relative frequency of species sampled in Prince William Forest Park.

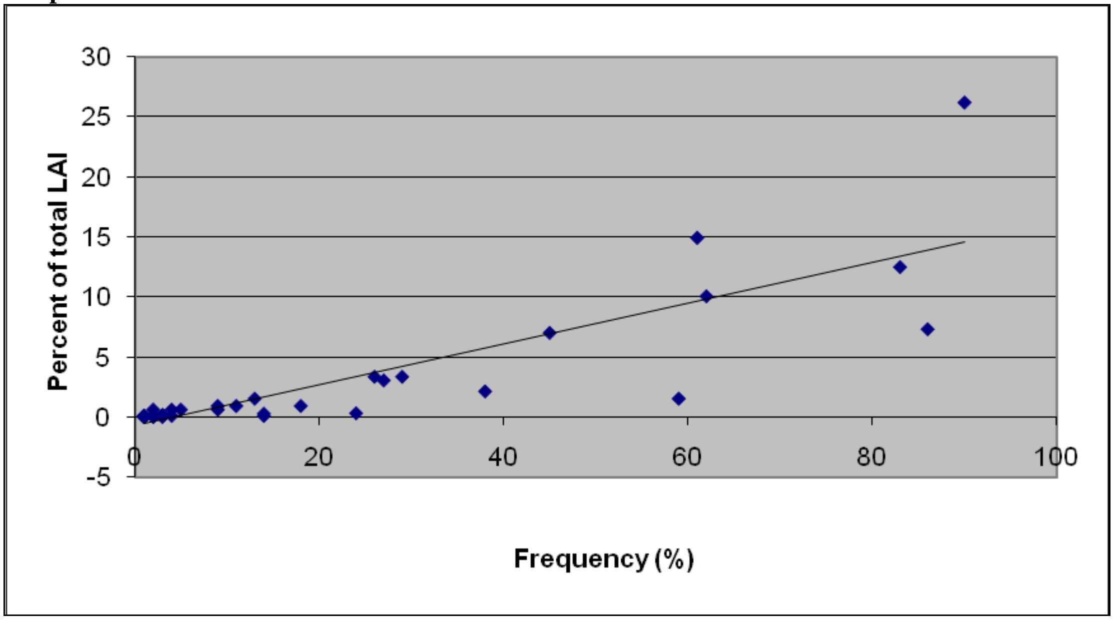
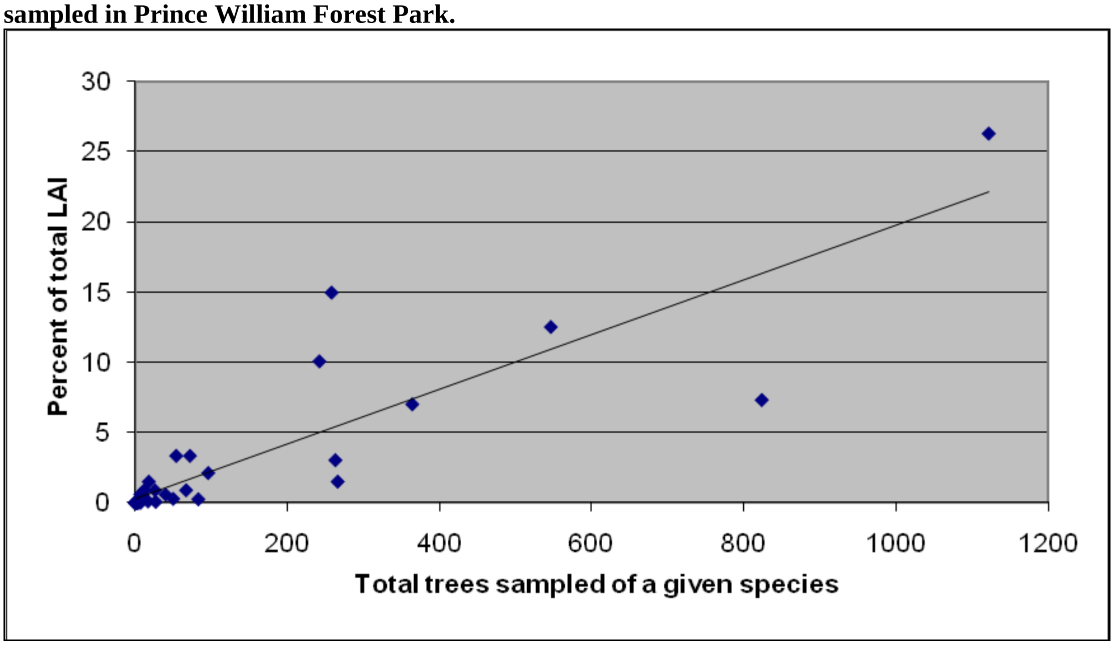


Figure 7: Relationship of relative leaf area index to total number of stems of a given species sampled in Prince William Forest Park.



Importance values have been used as an indicator of the ecological value of a given species, higher values indicating greater dominance of a species within an ecosystem (Sewanee: The University of the South 2010). I found that American beech, black tupelo, and red maple had the highest importance values in the Park. (Table 3).

Table 4: Importance values (I.V.) and leaf area index (LAI) of common tree species in Prince William Forest Park. Importance values were calculated as the sum of relative frequency, relative density, and relative basal area according to USDA Forest Service methods (USDA Forest Service 2007).

Species	Relative Frequency	Relative density (% of total)	Relative Basal Area (% of total)	Leaf Area Index (m^2 m^{-2})	Importance Value (I.V.)
American beech	90	24.6	11.0	0.86	125.6
Black tupelo	86	18.0	4.6	0.24	108.6
Red maple	83	11.8	9.3	0.41	104.1
White oak	62	5.3	15.1	0.32	82.4
Tulip poplar	61	5.7	16.5	0.49	73.2
American holly	59	5.8	1.4	0.05	66.2
Virginia pine	45	8.2	16.5	0.23	69.7
Flowering dogwood	27	6.1	1.4	0.1	34.5

Ecosystem Services

Carbon sequestration and storage

Net carbon sequestration for the entire park was estimated to be 12,346 metric tons (S.E. =1,093) year^{-1}. This averaged to 2.43 ± 0.21 tons of net carbon sequestration per hectare annually. The model calculated that trees of the smallest size classes sequester the greatest quantities of carbon annually (Figure 8). American beech was estimated to sequester the most carbon annually at a rate of 603 kg yr^{-1} ha^{-1} (Figure 9). This single species accounted for nearly 25% of the net annual carbon sequestration within the park. Eight species account for 88% of the park's net annual carbon sequestration (Figure 9).

Figure 8: Estimated net annual carbon sequestration by size class for all trees in Prince William Forest Park.

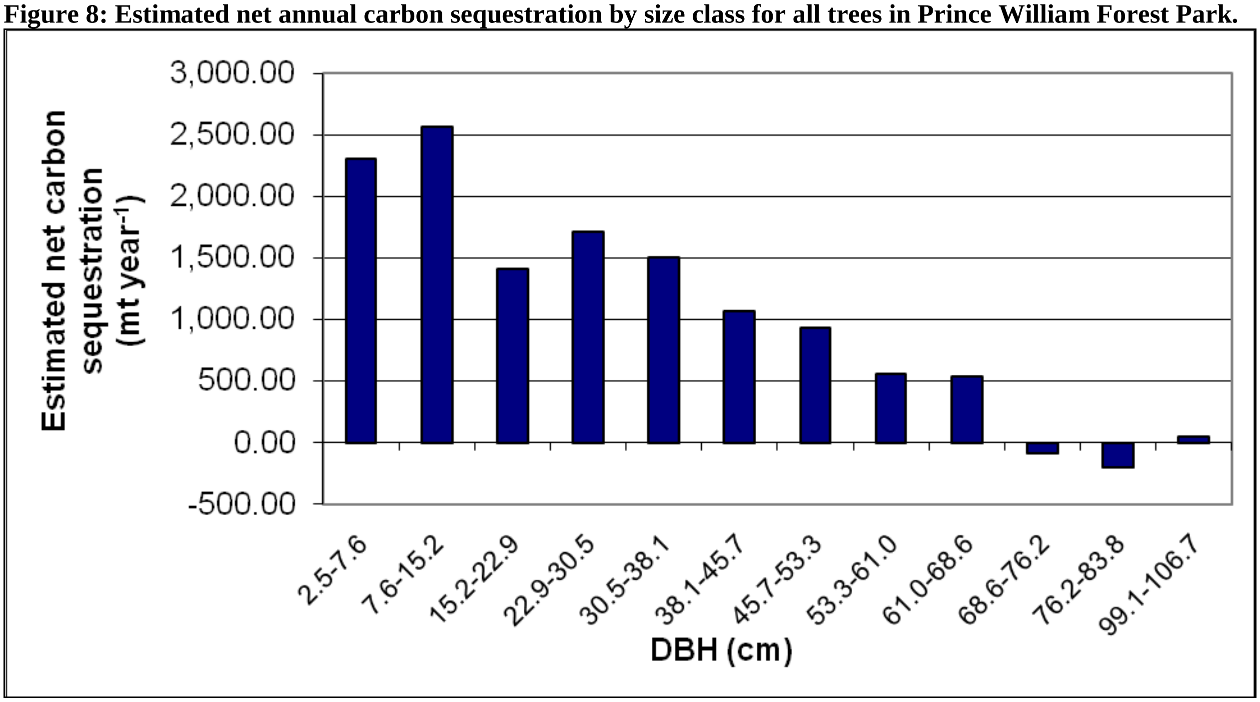
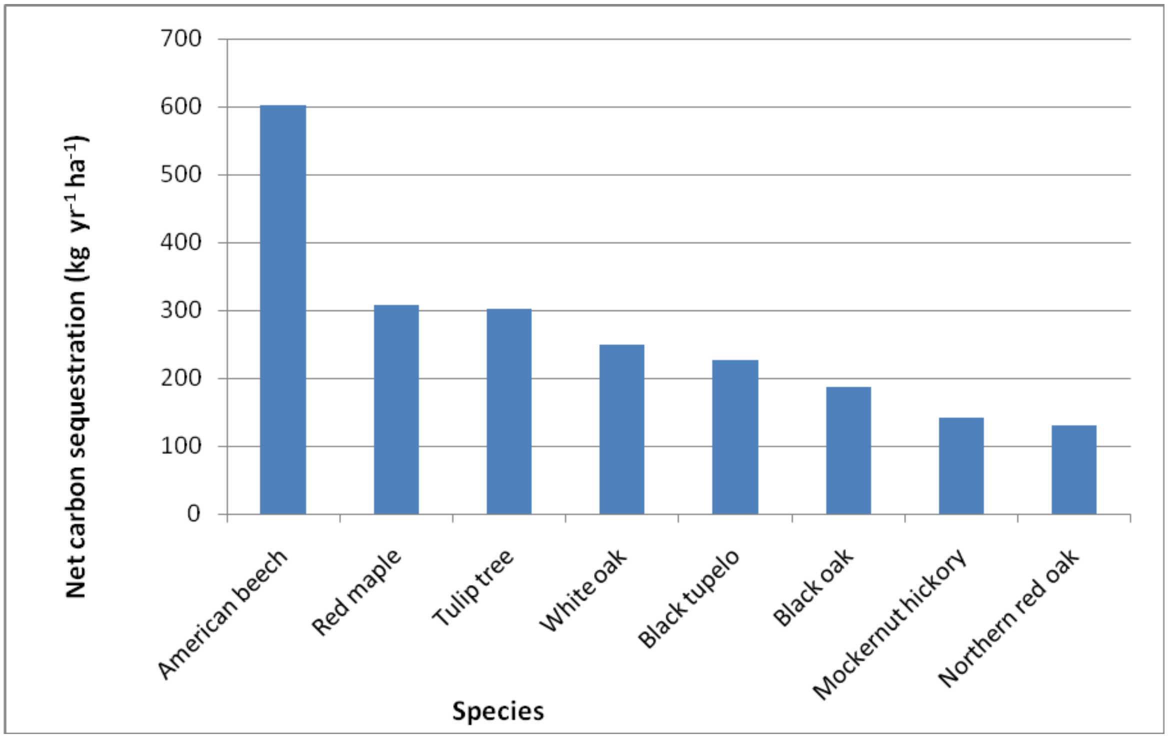


Figure 9: Species with highest estimated net annual carbon sequestration rates in Prince William Forest Park.



Trees can trap carbon for decades or centuries. However, annual carbon storage by growing trees is offset by the death and decomposition of trees within the forest. According to the UFORE model, the carbon stored in trees is the potential quantity of carbon that could be released into the atmosphere if the trees were removed and destroyed. Prince William Forest was assessed to store 394,241 metric tons of carbon (S.E. =18,698 tons). On average, a hectare of forest within Prince William Forest Park held 77.45 ± 3.68 tons of carbon as aboveground biomass. Carbon storage followed a bell curve that peaked in the 38.1 cm DBH group (Figure 10). This is likely due to tradeoffs between density and basal area. Trees that typically grow to larger size naturally stored the most carbon per hectare of forest. The ten species with the greatest carbon storage accounted for more than 350,000 metric tons of the park's total carbon storage (Figure 11). White oak stored the greatest amount of carbon, retaining nearly 19% of the total carbon stored within the park (Figure 11).

Figure 10: Total aboveground biomass of trees by DBH class for Prince William Forest Park.

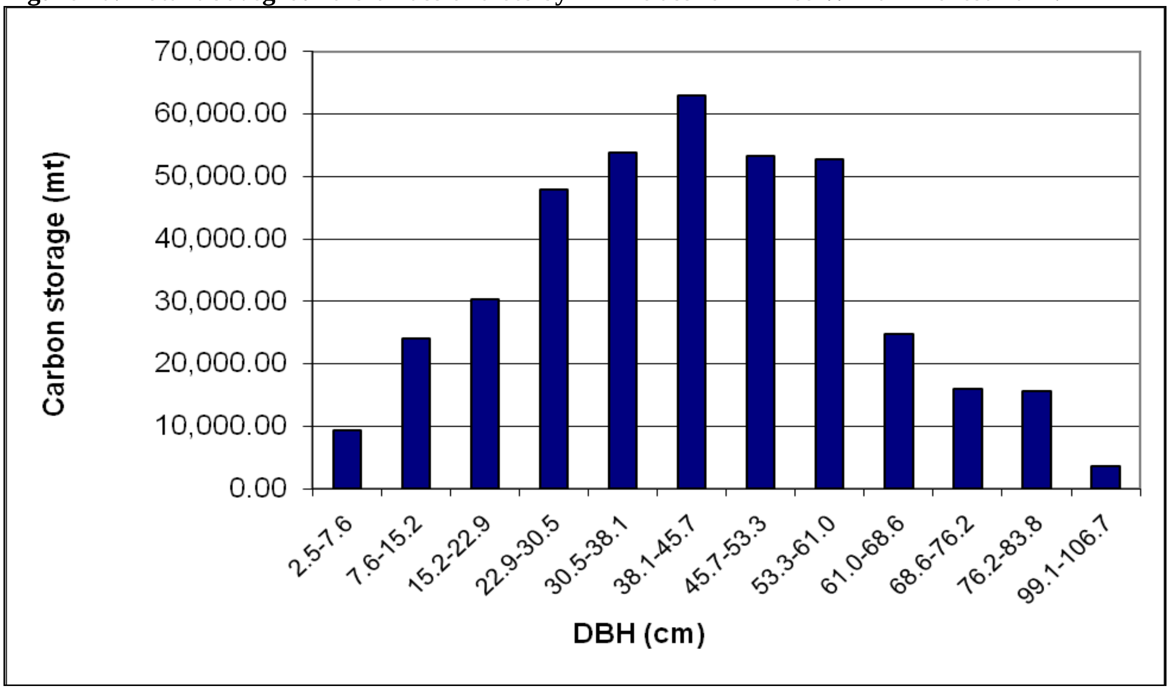
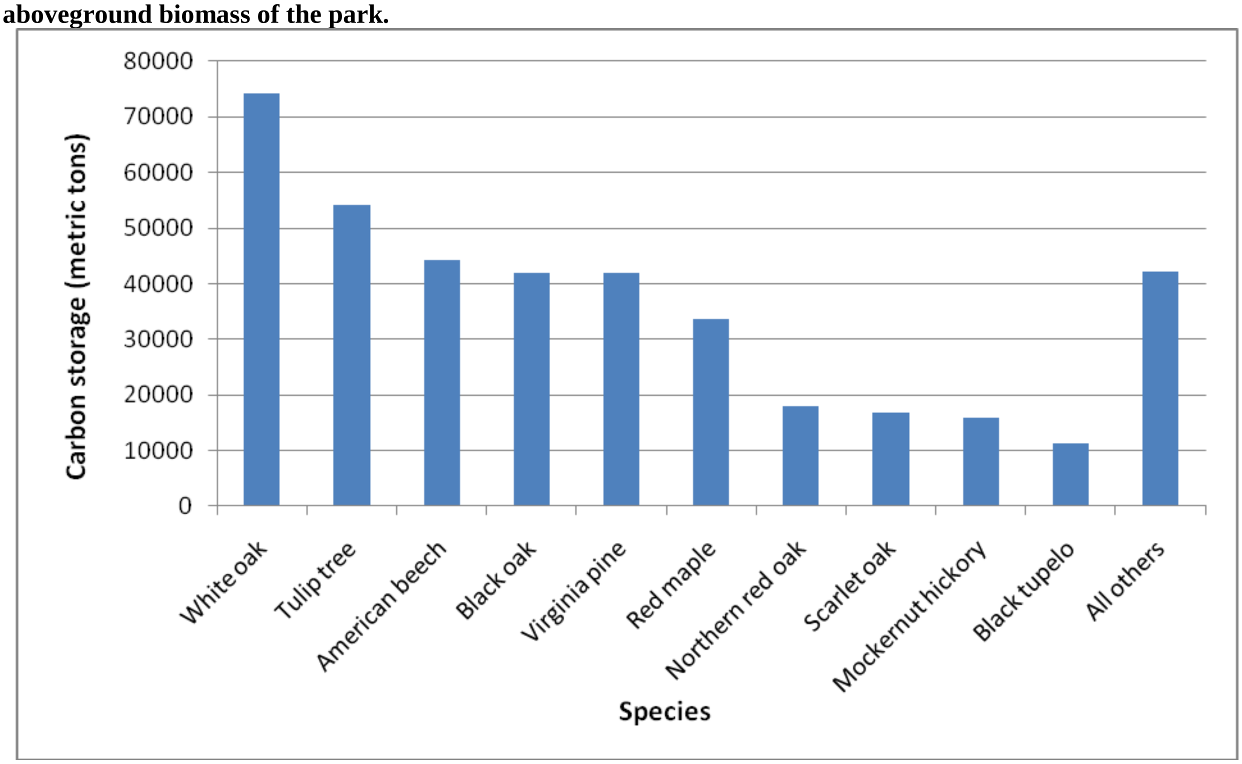


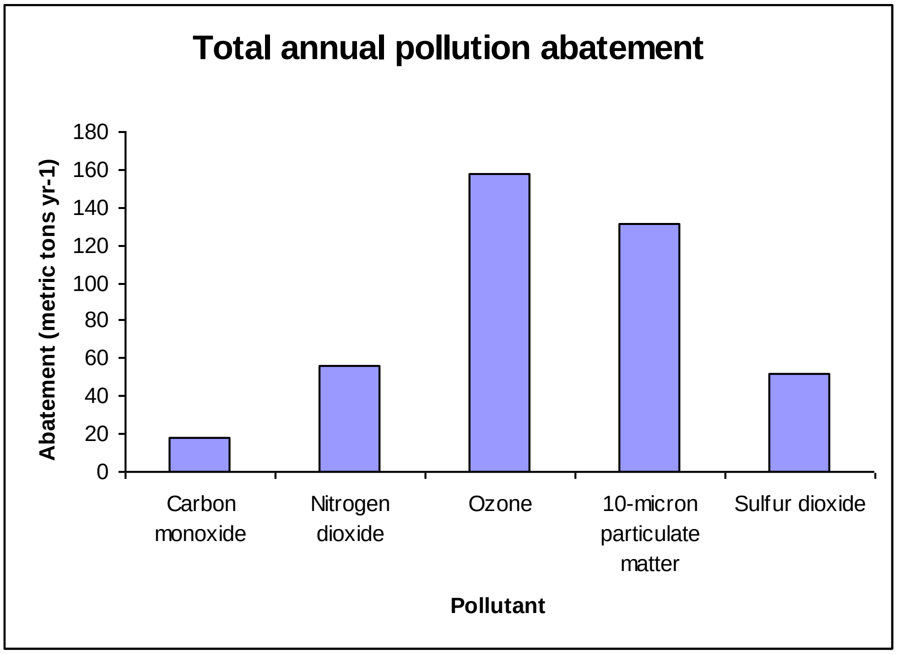
Figure 11: Biomass of ten species with greatest biomass in Prince William Forest Park. Species found in the park that are not listed are considered part of the 'all others' category and make up < 10% of total aboveground biomass of the park.



Air pollution abatement

Removal of key air pollutants (CO, NO_2, O_3, PM_{10}, and SO_2) was calculated by the UFORE model using field data and hourly pollution and meteorological data. UFORE used a combination of big-leaf models and canopy-deposition models hybridized together to calculate air pollution removal (Baldocchi *et al.* 1988; Baldocchi 1987). Trees and shrubs in Prince William Forest Park removed 414 metric tons of five criteria air pollutants annually (Figure 12). The greatest quantity of air pollution removed annually was ozone, of which 157.8 metric tons per year were removed by the forest according to the UFORE estimations (Figure 12). Particulate matter (under 10 microns in size), nitrogen dioxide, sulfur dioxide, and carbon monoxide follow are the other key pollutants estimated by the model. Particulate matter and nitrogen dioxide react within the atmosphere through a complicated series of reactions to form tropospheric ozone. Therefore, reduction in the quantities of these compounds improves air quality both through the reduction of respiratory irritants directly and a decrease in ozone production at ground level.

Figure 12: Pollution removed by trees and shrubs in Prince William Forest Park.



<u>*Vulnerability to pest outbreak*</u>

Four pests are considered in the UFORE model for the potential consequences of an outbreak. These species are Asian long-horned beetle (*Anoplophora glabripennis* Motschulsky), Dutch elm disease (*Ophiostoma sp.*), emerald ash borer (*Agrilus planipennis* Fairmaire), and gypsy moth (*Lymantria dispar* L.). Of the four, an outbreak of gypsy moth is potentially the most devastating to Prince William Forest Park. The current understanding of the gypsy moth diet suggests that nearly one-fourth of the leaf area of the park is at risk if an outbreak occurs and more than half of the park could be potentially affected if the preferred host species are unavailable or already consumed (Table 4). Dutch elm disease presents no threat to the forest since no elm trees (*Ulmus spp.*) were found within the park. Few ash trees were found, limiting the damage potential of emerald ash borer despite sightings in nearby counties over the past decade. The lack of knowledge of the dietary preferences of the Asian long-horned beetle made estimation of the effects of an invasion by this species particularly unreliable, since more than half of the leaf area of the park consists of species whose value as a food source for the beetle is poorly understood (Table 4).

Table 5: Potential damage to leaf area by pests and pathogens to trees in Prince William Forest Park. Primary host species are preferred hosts for a pest or pathogen. Secondary hosts are hosts known to be consumed or attacked by the pest but not as frequently as primary hosts. Immune species are those species not vulnerable to the pest or pathogen. Unknown represents species with no known interactions with the pest or pathogen in the literature.

Pest species	Primary host (% of park leaf area at risk)	Secondary host (% of park leaf area at risk)	Immune (% of leaf area unaffected)	Unknown (%)
Asian long horned beetle	13.1	22.3	6.9	57.7
Emerald ash borer	0.1	-	99.9	0
Dutch elm disease	0	0	100	0
Gypsy moth	23.5	52.7	23.6	0.2

Future range projections

Species ranges were defined by known locations of a given species by McKenney *et al.* (2007). They define a core range for each species by a suite of climate conditions including mean annual temperature, mean annual precipitation, mean temperature in the warmest month, mean temperature in the coolest month, mean precipitation in the wettest month, and mean precipitation in the driest month. The core range is then defined as any location where all of these values fall within the 5^{th}-95^{th} percentiles of the range of values for each factor. The fringe is delimited as any location the species is known to occur where one or more of these factors falls outside the 5^{th}-95^{th} percentile of all values. Of all species surveyed within Prince William Forest Park, only three were considered to be currently outside of their core ranges: northern catalpa (*Catalpa speciosa* (Warder) Warder ex Engelm.), southern red oak (*Quercus falcata*

Michx.), and willow oak (*Quercus phellos* L.). Northern catalpa is on the southern fringe of its range, while the two oak species are on the northern fringes of their optimal ranges (Table 5).

Table 6: Plant range predictions for the 21st century using the Hadley CM3 model under B2 scenario. No data were available for *Aralia spinosa* due to insufficient observations of the range of this species.

Scientific Name	1970-2000	2010-2140	2140-2170	2170-2100
Acer rubrum	core	core	fringe	fringe
Asimina triloba	core	core	fringe	none
Betula nigra	core	core	core	fringe
Carpinus caroliniana	core	core	core	fringe
Carya cordiformis	core	core	fringe	fringe
Carya glabra	core	core	fringe	fringe
Carya tomentosa	core	core	core	fringe
Castanea pumila	core	core	core	none
Catalpa speciosa	fringe	fringe	fringe	fringe
Celtis occidentalis	core	core	fringe	fringe
Cercis canadensis	core	core	fringe	fringe
Cornus florida	core	core	fringe	fringe
Diospyros virginiana	core	core	core	fringe
Fagus grandifolia	core	core	fringe	none
Fraxinus americana	core	core	fringe	fringe
Fraxinus pennsylvanica	core	core	fringe	fringe
Hamamelis virginiana	core	core	none	none
Ilex opaca	core	core	fringe	fringe
Juglans nigra	core	core	fringe	fringe
Juniperus virginiana	core	core	core	fringe
Liquidambar styraciflua	core	core	core	fringe
Liriodendron tulipifera	core	core	fringe	fringe
Nyssa sylvatica	core	core	core	fringe
Pinus echinata	core	core	core	fringe
Pinus virginiana	core	core	fringe	none
Platanus occidentalis	core	core	fringe	fringe
Prunus serotina	core	core	fringe	fringe
Quercus alba	core	core	fringe	fringe
Quercus bicolor	core	core	fringe	none
Quercus coccinea	core	core	fringe	none
Quercus falcata	fringe	fringe	fringe	fringe
Quercus palustris	core	core	fringe	none
Quercus phellos	fringe	fringe	fringe	fringe
Quercus prinus	core	core	fringe	none
Quercus rubra	core	core	fringe	none
Quercus stellata	core	core	core	fringe
Quercus velutina	core	core	fringe	fringe
Sassafras albidum	core	core	fringe	fringe

The low-emission B2 scenario outlined by the Intergovernmental Panel on Climate Change predicts moderate increases in mean annual global temperature by the end of the twenty-first century. This scenario estimates that in the next 30 years, 35 out of 39 tree species in the park will continue to experience optimal growth conditions (Table 5). By 2070, however, the model predicts that the number of species that would find the Park optimal would decrease to ten, while one species, witch-hazel (*Hamamelis virginiana* L.), would experience a climate entirely unsuitable for growth. This model also predicts that 10 species would be extirpated from the park by the end of this century and that all remaining tree species would be outside of optimal conditions for growth (Table 5).

Under the high-emission A2 scenario, mean annual global temperatures may rise by up to 5 degrees Centigrade. Six species (American holly, sweetgum, swamp white oak (*Quercus bicolor* Willd.), southern red oak, willow oak, and chestnut oak (*Quercus prinus* L.)) would already be outside of their optimal range by 2040 (Table 6). By 2070, none of the species currently found in the park would lie within its core range and five species—paw-paw (*Asimina triloba* (L.) Dunal), witch-hazel, tulip-poplar, scarlet oak, and chestnut oak— would be removed from the park. According to the model, only swamp white oak is anticipated to be within its fringe range under the warmer climate predicted at the end of the century (Table 6).

Table 7: Predicted plant ranges in the 21st century under A2 scenario using Hadley CM3 model. No data were available for *Aralia spinosa* due to insufficient observations of the range of this species.

Scientific Name	1970-2000	2010-2140	2140-2170	2170-2100
Acer rubrum	core	core	fringe	none
Asimina triloba	core	core	none	none
Betula nigra	core	core	fringe	none
Carpinus caroliniana	core	core	fringe	none
Carya cordiformis	core	core	fringe	none
Carya glabra	core	core	fringe	none
Carya tomentosa	core	core	fringe	none
Castanea pumila	core	core	fringe	none
Catalpa speciosa	fringe	core	fringe	none
Celtis occidentalis	core	core	fringe	none
Cercis canadensis	core	core	fringe	none
Cornus florida	core	core	fringe	none
Diospyros virginiana	core	core	fringe	none
Fagus grandifolia	core	core	fringe	none
Fraxinus americana	core	core	fringe	none
Fraxinus pennsylvanica	core	core	fringe	none
Hamamelis virginiana	core	core	none	none
Ilex opaca	core	fringe	fringe	none
Juglans nigra	core	core	fringe	none
Juniperus virginiana	core	core	fringe	none
Liquidambar styraciflua	core	fringe	fringe	none
Liriodendron tulipifera	core	core	none	none
Nyssa sylvatica	core	core	fringe	none
Pinus echinata	core	core	fringe	none
Pinus virginiana	core	core	fringe	none
Platanus occidentalis	core	core	fringe	none
Prunus serotina	core	core	fringe	none
Quercus alba	core	core	fringe	none
Quercus bicolor	core	fringe	fringe	none
Quercus coccinea	core	core	none	none
Quercus falcata	core	fringe	fringe	none
Quercus palustris	fringe	core	fringe	none
Quercus phellos	core	fringe	fringe	none
Quercus prinus	fringe	fringe	none	none
Quercus rubra	core	core	fringe	none
Quercus stellata	nd	core	fringe	none
Quercus velutina	core	core	fringe	none
Sassafras albidum	core	core	fringe	none

Chapter 4: Discussion

Current forest composition

The species composition of Prince William Forest Park (PWFP) reflects different forest associations found within northern Virginia. The presence of American beech and red maple within the park is similar to that documented for other mid-Atlantic forests (Abrams 1992). Virginia pine, common in PWFP, is also common throughout the Coastal Plain in Virginia (Monette and Ware 1983). Given the history of the region as oak-chestnut forest (Godfrey 1980, Braun 1950, Oosting 1942), the abundance of various oak species is also expected.

The National Park Service (NPS) conducts vegetation surveys annually within the National Capital Parks region, including Prince William Forest Park. With the assistance of National Park Service personnel, I obtained a draft of the 2009 vegetation survey data of the park. Comparison of this UFORE study with the NPS study reveals similar species composition between the two studies. In both studies, American beech was the most abundant species, although beech was more prevalent in this study than the NPS study while Virginia pine was more prevalent in the NPS plots than in this UFORE study. Red maple maintained a similar degree of importance in both studies, making up approximately ten percent of all trees sampled.

The abundance of several species differed between the two studies however. For example, scarlet oak made up 3% of all trees sampled in the NPS study but was only occasionally observed in the plots surveyed for the UFORE study. This difference may be attributed to random differences in sampling location between the two studies. Scarlet oak is shade-intolerant and commonly grows on upland sites with low canopy cover (Trees of Alabama and the Southeast 2009) and few of the UFORE plots were in such locations. Whether or not this difference is significant is unknown.

Even more noteworthy is the difference between the two studies in the prevalence of black tupelo and American holly. In this study I found black tupelo was second only to American beech in total number of trees sampled, making up 18% of all trees in the survey area. In comparison, the 2009 NPS survey found black tupelo to be half as common, representing almost 10% of all trees sampled. Given that black tupelo can survive under xeric or hydric conditions, endure fire, and persist in nutrient-poor sites (Abrams 2007), the reduced numbers of black tupelo trees found in the 2009 survey is a point of curiosity. Without further information regarding topography, aspect, and soil conditions of the NPS and UFORE survey plots no explanation for this difference can be named. Finally, American holly was found to be an important understory species in the NPS survey, outnumbering flowering dogwood two-to-one. In contrast, the two were approximately equal in prevalence in the UFORE data. The relative rankings of the remaining species otherwise were in general agreement between the two surveys.

One way to possibly correct for successional stage differences or tree age and to look at population demographics is to look at species composition within size classes. For example, removing trees <10 cm DBH will in general remove younger trees from the analysis. When all trees less than 10 cm DBH were excluded American beech, the most common tree in both studies, maintains nearly identical percentages between studies: 15% in the NPS study vs. 14.2% in the UFORE study. Similarly, two common understory species American holly and flowering dogwood were found in similar small numbers as large trees in both studies. American holly made up 2% of the larger trees in both studies and flowering dogwood made up 1% of the larger trees in the NPS study and 2% of trees in the UFORE study.

The number of individuals of other species however differed somewhat between the studies. For example, two-thirds of all Virginia pines sampled in this study fall into the >10 cm

DBH while nearly all Virginia pines sampled in the NPS study were larger than 10 cm DBH. This difference in Virginia pine size classes can be explained by two UFORE plots that contained a large number of young Virginia pines. One of these two sites was an early-successional roadside plot that contained 46 Virginia pine saplings. The second site was found slightly further back from the roadside in a small patch of grassland that had been invaded by Virginia pine and attained a density of 2,717 stems ha^{-1}, the highest density of any site in this study. It is likely that the density of pine saplings will rapidly decline in the coming years as competition for light thins the stand. Despite the difference in quantities of larger Virginia pines, the two studies nearly agreed on the percentage of the forest that was made up of larger pines (18.1% vs. 15.0%). More data are needed to determine if these differences are statistically significant.

Tulip-poplar and white oak were two of the more common species in the >10 cm size class and were found in similar amounts between the two studies. Tulip-poplar made up 15% of all trees > 10 cm DBH in the NPS study and 12.2% of trees > 10 cm DBH in this study. White oak made up 12% of all trees in the 10 cm or greater size class in both the NPS study and the UFORE study. These findings in both studies are consistent with common forest classifications of the region (Abrams and Copenheaver 1999).

Red maple made up a somewhat smaller portion of trees in the 10 cm DBH size class in the NPS vegetation surveys than in the UFORE study. Eight percent of all trees in the >10 cm DBH class were red maple according to the NPS study, but in this study nearly 16% of the larger trees were red maple. This difference in dominance may also be attributable to differences in soil moisture between the two studies. Red maple can grow in mildly xeric conditions through hydric conditions but favors mesic sites (Abrams 1998), while scarlet oak has little tolerance of anaerobic conditions (USDA Natural Resources Conservation Service Plants Database 2010, Trees of Alabama and the Southeast 2009). Scarlet oak abundance suggests that the NPS study favors upland sites which are likely to be more xeric in nature and are thus less favorable locations for red maple. Accordingly, topographic and soil moisture data are necessary to determine if this is the key difference between NPS and UFORE sites.

Future species composition

The differences in species composition of the forest between size classes provide some suggestions as to the composition of the future forests in PWFP. Many of the oaks in the larger size category are likely to experience increased mortality during the coming decades. The expected lifetime of white oak is approximately 300 years, though older specimens have been occasionally found (Abrams 2003, Abrams and Copenheaver 1999). At the time of the park's establishment in 1936, the large trees that were present were primarily oaks that had been left uncut due to inconvenient location for removal or to provide shade for existing dwellings (Bedell 2004), suggesting that existing oaks are now well into their second century of life. If mortality does in fact increase dramatically it is likely that they will be replaced with red maple and American beech from the understory and midstory of the forest. At the time of this study, both red maple and American beech were more common than white oak within the forest. This applied both in terms of the entire pool of sampled trees and for larger and presumably older trees of 10 cm DBH or greater size. Nearly 50% of all red maples sampled were less than 10 cm DBH, while only 10% of white oaks fell into this same size category, possibly indicating a limitation to oak regeneration within the park. Reduced oak regeneration has been observed in different forests within the region (Abrams 1992, Lorimer 1984) and has been attributed to multiple factors including fire suppression and white-tailed deer (*Odocoileus virginianus*) population explosions (Abrams 1998). In historically-disturbed forest stands in Pennsylvania, red maple frequently became one of the dominant canopy species (Abrams and Nowacki 1992). In addition, American beech has already been found as a canopy species in mid-Atlantic forests (Abrams 1992).

The continued succession of the park into beech-maple forest will alter the canopy structure and understory of the future forest. American beech typically grows into a dense canopy that prevents other trees or shrubs from growing beneath it (Gilman and Watson 1993). This particular architecture may alter forest regeneration and understory composition. For example, lack of recruitment of oak species beneath a closed canopy of later successional species such as beech has already been observed in the mid-Atlantic region (Abrams 1992). On the other hand, highly shade tolerant species such as American holly may be favored under this scenario and could increase in importance in the park. Therefore this alteration in light reaching the understory may lead to shifts in the dominance of understory trees and shrubs. Understory species may shift the timing of leaf-flush in the spring to compensate for the decrease in photosynthetically-active radiation (PAR) that occurs after canopy species leaf-out (Graves 1990). The phenological changes in leaf-flush could make a difference in the structure and growth of understory trees. If understory trees initiate growth earlier in the season, the forest floor would remain shaded longer into the growing season, potentially slowing germination from the seed bank.

In addition to autogenic successional factors, several allogenic factors such as climate change and management decisions will undoubtedly affect the future vegetation of the park. Increases in mean annual temperature can influence timing of various events such as budburst, leading to longer growing seasons (Fitter and Fitter 2002, Schwartz and Reiter 2000). Canopy species influence the reproductive success of understory species by reducing the light available to on the forest floor (Maeno and Hiura 2000), potentially shifting phenology in understory species. Understory species that germinate early in the year have higher survivorship than those emerging or germinating later (Jones *et al.* 1997).

In a different vein, management decisions can shift the direction of succession in the park. Anthropogenic impacts such as fire suppression and introduction of pathogens have altered the eastern deciduous forests over the course of the 20th century (Abrams 2003). The efforts to remove invasive species have likely altered the succession of the park as well. If invasive species removal is reduced or discontinued altogether, portions of the park could lose biodiversity as invasive species such as tree-of-heaven (*Ailanthus altissima*) or wisteria (*Wisteria spp.*) out-compete native species or even establish monoculture stands. Hutchinson and Vankat (1997) observed that species richness was reduced in the presence of Amur honeysuckle (*Lonicera maackii* (Rupr.) Herder) and that tree basal area were inversely proportional to the degree of cover that the honeysuckle attained.

Fire suppression and management will also play a key role in shaping the future vegetation of the park. Given the architecture of American beech and the fire suppression regime of the Park, a canopy-replacing fire within the park could easily result in a region that is temporarily barren of cover. One plot during my study had been burned a year prior to sampling and showed signs of regrowth, but the only living trees within the plot were mature black tupelo and white oaks that had not burned. Mature beech trees retain scaffolds close to the ground (Gilman and Watson 1993) which may be vulnerable to a ground fire and in turn permit fires to access to the crowns of midstory trees. With the accumulation of leaf litter and detritus in the Park as a factor, a serious fire could leave burned areas completely open for recruitment. A portion of burned land was found that displayed this pattern adjacent to the South Valley Trail within the northwestern region of the park, where a fire the previous year had killed all existing vegetation in the burned land and left fallen trees and standing deadfall of mountain laurel (*Kalmia latifolia* L.) throughout the area. Oaks are more resistant to fire than many other species due to their thicker bark, an adaptation postulated to permit survival through the occasional fire in the wild (Abrams 1998). Oak species may return to a burned site at a faster rate than other species that are more vulnerable to fire. The relative growth rate of oak was unaffected by fire that slowed the regeneration of both ash and maple in Wisconsin (Kruger and Reich 1997). A disturbance such as a major fire may also open the forest to invasion by non-indigenous plants which can alter the succession of a given ecosystem (Boyce 2009).

Continued natural succession and disturbance in the park may also impact consumers as well as the trees. For example, a reduction in acorn availability as oak becomes less prevalent could result in a shift in diet for white-tailed deer. Body mass of white-tailed deer fawns is correlated with acorn yield (Feldhamer *et al.* 1989) so a loss of oaks could impact future deer populations. Changes to the forest structure will require deer to shift to alternate food sources such as beech nuts and hickory nuts or to increase browsing. If browsing increases, regeneration of many tree species may be affected. Virginia is experiencing deer populations as great as those seen in colonial times (Virginia Department of Game and Inland Fisheries 2010). A study in northwestern Pennsylvania found that high population densities of white-tailed deer can prevent regeneration of many species, leaving only those species that are most unpalatable to local dear populations (Tilghman 1989).

In addition to impacts on deer, other organisms will clearly be impacted by future changes in forest composition. Black bears (*Ursus americanus*) also consume acorns in the autumn and winter, where they can make up more than 15% of the volume of food consumed (Benson and Chamberlain 2006). The effects of a shift in acorn availability on wildlife populations is complicated by the plethora of species that utilize acorns as a food source, such as deer, black bears, squirrels (*Sciuridae*), domestic animals, bobwhite quail (*Colinus virginianus*), crows *(Cortus brachyrhynchos),* jays (*Cyanocittac ristata*), woodpeckers (*Melanerpes sp., Dendrocopos sp.*), raccoons (*Procyon lotor*), rabbits (*Sylvilugus spp.*), and gray foxes *(Urocyon cinereoargenteth*) (Goodrum *et al.* 1971).

Another important change across trophic levels could be impacts on pests and pathogens. Possible temperature increases of the future due to global warming could lead to increased winter survival in overwintering pest species, shorter gestation periods, and increases in the number of generations that breed each year (Harvell *et al.* 2002). Increased mean annual temperatures may lead to greater insect diversity as well (Dale *et al.* 2001). The faster migration rates of insects compared to plants suggests that temperate tree species may become exposed to generalist insect herbivores previously native to the subtropics (Dale *et al.* 2001). Several plant pathogens have been shown to increase in overall severity after milder winters and/or during warmer summers (Harvell *et al.* 2002). Temperate species have shown gains in maximum net photosynthesis due to temperature change (Cunningham and Read 2003) and CO_2 enrichment (DeLucia *et al.* 1999). Whether the increased severity and frequency of pathogen outbreaks could potentially negate the potential advantages to the forest of increased mean annual temperature and CO_2 enrichment over long time periods is unknown.

Potential effects of pest and pathogens

The UFORE model also assessed the potential damage of four different notable pathogens and pests to the forest: Asian long-horned beetle (ALB), emerald ash borer (EAB), Dutch elm disease (DED), and gypsy moths. Asian long-horned beetle primarily attacks various maple species, although birch (*Betula spp*.) and ash (*Fraxinus spp.*) are also targeted (USDA Forest Service Northeastern Area Forest Health Protection 2009). An attack by ALB could reduce the living aboveground biomass of the park by as much as 46,000 metric tons and the leaf area by 12% in the event of red maple extirpation in the park. In the park's wetlands loss of red maple, ash, and birch would open some wetlands such as those found on the floodplain of the Quantico River to recruitment by other species favoring hydric conditions.

Dutch elm disease is not a significant threat to the park since no elm trees were found in this study, although one American elm (*Ulmus americana* L.) was located in the 2009 NPS vegetation survey. Dutch elm disease struck native elm populations in the area in the 1930s (Smith 2009) and probably eliminated most of the elms from the Park at that time. New cultivars of American elm are currently undergoing testing for resistance to the fungi responsible for Dutch elm disease (Colorado State University 2009, Smith 2009). If these cultivars can spread naturally, American elms may be recruited into Prince William Forest Park in years to come from local planting sites.

The Emerald ash borer (*Agrilus planipennis* Fairmaire) attacks *Fraxinus* species, of which there are two within the region: *Fraxinus americana* L, white ash, and *Fraxinus pennsylvanica* Marsh., green ash. White ash favors moist upland sites while green ash favors bottomlands, though there is some overlap in the range of these two species (Emerald Ash Borer 2009). This pest was found in neighboring Fairfax County, Virginia in 2003 and 2008 (Emerald Ash Borer 2009). An epidemic is unlikely to strongly influence the ecosystem services of the park due to the small numbers of ash trees present.

A gypsy moth outbreak could potentially devastate the forest. Oak species are a preferred food of these pests, but sweetgum and birch are also strongly favored (USDA Forest Service 2010). At high population densities most hardwoods and many softwoods are vulnerable to attack by gypsy moths, but certain species found within the park are seldom attacked if at all, including American holly, tulip-poplar, catalpa (*Catalpa spp.*), ash, flowering dogwood, black walnut (*Juglans nigra* L.), mountain laurel, American sycamore, and eastern redcedar (*Juniperus virginiana* L.) (USDA Forest Service 2010). A gypsy moth outbreak could cause high mortality in stressed trees (Lovett *et al.* 2006), such as those experiencing stress from heat or drought. The outbreak in turn would reduce long-term carbon storage by killing weakened trees which in turn decompose and release carbon dioxide back to the atmosphere over years or decades. Short-term carbon sequestration of the forest would also be reduced after the outbreak. In the New Jersey Pine Barrens, a gypsy moth outbreak in 2007 reduced carbon sequestration to 44% of the anticipated amount in oak-dominated forests (Clark *et al.* 2009). Succession of oak-dominated forests to maple-dominated forests during the twentieth century has been in part attributed to the alkaloid content of red maple foliage deterring gypsy moth defoliation (Abrams 1998). Evidence of this shift in successional pattern has been found in Michigan, where red maple and black cherry (*Prunus serotina* Ehrh.) were found to have grown more rapidly in a mixed oak-maple forest after gypsy moths defoliated the oaks forming the canopy (Jedlicka *et al.* 2004).

Climate change

Future changes in climate may affect the vegetation of the park in terms of species richness, community structure, and ecosystem services. Historically, forests in North America have expanded northward as temperatures have risen (Delcourt and Delcourt 1987). A similar response was seen in the HadleyCM3 model results of this study. The impact of this on the park is that southern species will migrate northward into the park over time. The rate at which new tree species immigrate into Prince William Forest Park will vary based on the individual species' methods of dispersal, soil development, cold hardiness, and climate (McKenney *et al.* 2007, Delcourt and Delcourt 1987). With increased mean annual temperatures expected to increase by as much as 5° C but mean annual precipitation expected to increase by less predictable quantities (IPCC 2007), species may begin to experience heat and drought stress sufficient to kill them and cause shifts in community structure (Thornley and Cannell 1996). White and others (2000) found that C_4 grasses increased in biomass while C_3 grasses declined in an experimental community exposed to increased temperatures, giving the warm-season grasses the opportunity to monopolize available light. Net primary production may increase as temperatures rise if precipitation is sufficient to maintain higher levels of metabolism (Cunningham and Read 2003), resulting in greater biomass accumulation by the natural forest.

Species differences in migration rates provide both opportunities for rapidly dispersing species to colonize new geographic areas and perhaps permit species with slower migration rates (e.g. black tupelo) the opportunity to increase in importance locally (Delcourt and Delcourt 1997). This has happened at least once since the most recent interglacial period began, as hornbeam (both *Carpinus caroliniana* Walter, American hornbeam, and *Ostrya virginiana* (Mill.) K. Koch, eastern hophornbeam) grew in importance in parts of the Midwest from 12,000 BP until 6,000 BP (Delcourt and Delcourt 1987). Hornbeam made up as much as 30% of the pollen found in the fossil records from 12,000 BP to 6,000 BP (Delcourt and Delcourt 1987). After 6,000 BP hornbeam pollen fell in quantity to much lower levels and never attained the same degree of importance that the species held during the 6,000 year period prior to 6,000 BP (Delcourt and Delcourt 1987). This particular phenomenon, if duplicated within Prince William

Forest Park by an opportunistic species, could shift the successional process from the current trajectory toward beech-maple forest onto new paths, such as tupelo-dominated forests on bottomland sites. Tupelo is already an important component of several forest associations in the region, though not all of these associations are bottomland forests (Abrams 2007).

The Hadley CM3 model predicts climate change based on atmospheric and oceanic interactions (Canadian Forest Service 2009). Biotic interactions (e.g. associations with soil mycorrhizae) and topography were not included as functions of the model. Any predictions on the range of a given species in the future, particularly those species established over a wide area, should be considered cautiously. American beech, for example, is found throughout North America from Florida to Canada (Canadian Forest Service 2009, Loehle and LeBlanc 1996). The HadleyCM3 model predicted that the climate in Prince William Forest Park by 2100 will be unsuitable for beech under the A2 and B2 scenarios. In the modern era, beech exists on mesic sites with infrequent fires (Loehle and LeBlanc 1996). In Florida, beech is found in ravines and stream valleys and is accordingly rare (Loehle and LeBlanc 1996) but still present. Such sites will likely continue to exist in future climates, in particular in the Appalachian Mountains at higher elevations as well as in the topographically varied regions such as the Piedmont (Loehle and LeBlanc 1996).

Other models have been created to examine global climate change. The HadleyCM3 has been used by the Intergovernmental Panel on Climate Change (IPCC) (IPCC 2010), the USDA Forest Service (USDA Forest Service 2009), and for independent research (Yang 2009). Another climate model is the Coupled Global Climate Model, version 2 (CGCM2). This model has seen use by the Canadian Forest Service (2009) and by Yang (2009) as well. Like the HadleyCM3 model, the CGCM2 couples atmospheric and oceanic data to predict climate. The predictions regarding the potential range of species based on this model are similar to those of the HadleyCM3 model. Under the A2 high emission scenario, no species from the UFORE study was predicted by the CGCM2 model to remain viable within the park by the year 2100. Under the B2 scenario, the CGCM2 contrasts with the HadleyCM3 model not in regards to the number of species that will find the park inhospitable but instead differs regarding which species will be unable to survive. The Hadley model predicted a climate regime to which American beech and Virginia pine could not survive, while the CGCM2 establishes these species as fringe species. Tulip poplar and black tupelo were expected to survive as fringe species in the park according to the Hadley model but the CGCM2 places these two species outside of their climate envelopes. These differences in the fates of four species of particular importance to the park's forests emphasize uncertainties in the reliability of using climate science to predicte the future range of a species.

Yang (2009) looked at the urban forest of Philadelphia, Pennsylvania and attempted to determine which species would be likely to survive in the city in the future. He utilized the models of McKenney *et al.* (2007) to determine the current range of each of the tree species found in Philadelphia and then determined whether or not a given species would remain in its optimal climate range over the next century by averaging scenarios predicted by various climate models. My study was limited to the HadleyCM3 model only. Averaging several models likely smoothed out any extremes within his results. Common knowledge suggests that species will migrate northward as mean annual temperatures rise. Changes in the range of a given species push that species northward into the northeastern United States. Philadelphia may have a disproportionate number of species that find the climate of the city falls within the middle of their range, as the city lies firmly within the mid-Atlantic region. The park may have many species that are at the extremes of their ranges due to its location in the transitional region between northern and southern forests. This difference in location may account for the different numbers of species expected to remain in the city (32 of 73 species) when compared to the park. Populations at the extremes of their range are vulnerable and may be limited in their ability to adapt (Kirkpatrick and Barton 1997).

<u>Ecosystem services of Prince William Forest</u>

Ecosystem services include all benefits humans derive from ecosystem functions, which in turn includes the habitat, biological, and system properties of an ecosystem (Costanza *et al.* 1997). These services include (but are not limited to) nutrient cycling, gas regulation, climate regulation, waste treatment, and provision of raw materials. Forests provide all of these services and are receiving increasing scrutiny with an eye toward the quantities of these services provided (e.g. Keith *et al.* 2009, Chen 2009, Dodds *et al.* 2008, Kirby and Potvin 2007, and Costanza *et al.* 1997). Two services that have garnered attention in the literature are carbon storage and carbon sequestration. These two services are of importance due to their links to global climate change (IPCC 2007) and are primary outputs of multiple models such as the UFORE model.

Several dozen cities have assessed the ecosystem services of the urban forests with the UFORE model, such as Tampa (Andreu *et al.* 2008), San Francisco (Nowak *et al.* 2007), Minneapolis (Nowak *et al.* 2006b), Oakville (McNeil and Vava 2006), and the borough Brooklyn in New York (Nowak *et al.* 2002a). Locally the ecosystem services provided by the urban forests of both Baltimore and Washington DC have been assessed with the UFORE model (Mead 2009, Nowak *et al.* 2006a). However, Prince William Forest Park is the first natural forest that has been surveyed with the UFORE model.

Carbon storage and sequestration

Carbon storage is measured as aboveground biomass by the UFORE model. White oak accounted for the most carbon stored within the park, representing approximately 19% of park's carbon as aboveground biomass. The white oak population consisted of larger (and ostensibly older) trees which were probably not growing as quickly as other species and therefore were not estimated to be sequestering carbon at the rate of other species. White oak also stored the most carbon in Washington DC, though the contribution to the total carbon storage of the DC urban forest was smaller (13.6%) (Nowak *et al.* 2006a).

Washington DC covers 159 km^2 of land with a canopy cover of 28.6%, or 45.5 km^2 (Nowak *et al.* 2006a). The urban forest proper of DC stores 105.0 tons C ha^{-1} (Nowak *et al.* 2006a) while the carbon storage averaged over the entire political boundary of DC is 30.0 tons C ha^{-1}. The carbon storage per unit area of PWFP was estimated at 77.1 tons C ha^{-1} of carbon as aboveground biomass. The model estimated that the total carbon stored in PWFP exceeds 390,000 metric tons over 51 km^2 land area compared to 470,000 metric tons in DC. Although no direct statistical comparisons of the two forests can be made, it appears that carbon storage per unit forested area of DC equals or exceed that of PWFP. However the natural forest of PWFP stores over twice as much carbon on a total land area basis.

American beech was the most abundant species in PWFP and was estimated to be the most important species in terms of carbon sequestration. In contrast, tulip-poplar was found to sequester the largest percentage of carbon annually in the DC urban forest (29%) even though it was only the 4th most abundant species encountered in the city. This suggests that the most abundant species may not the one that is most actively sequestering carbon. However, American beech outnumbered all tulip-poplar by nearly five-to-one in PWFP and tulip-poplar in PWFP had barely half the leaf area of American beech (14.5% vs. 25.7%). Tulip-poplar accounted for a higher percentage of leaf area (17%) for the DC urban forest (Nowak *et al.* 2006a). It is likely that the rapid growth and large size of tulip-poplar (Busing 1995) account for its estimated prominence in carbon sequestration in DC.

The net annual carbon sequestration for the entire District of Columbia was 7.0 tons C ha^{-1} yr^{-1} (Nowak *et al.* 2006a). However, if sequestration is calculated only on the land area under forest cover then the rate of sequestration was 26.0 tons ha^{-1} yr^{-1}. This compares favorably with the sequestration rate of 24.0 tons C ha^{-1} yr^{-1} of the park. This suggests that for these two examples, urban and natural forests are functioning at similar levels when carbon assimilation is considered.

On a total per unit area basis, PWFP sequesters approximately equal amounts of carbon yet uses one-third the land area to do so. This highlights the importance of natural areas surrounding urban environments to regional carbon C storage and sequestration. The UFORE model predicts that the urban forest sequesters carbon as well as a natural forest when only forested areas are include despite the urban backdrop of impervious surfaces and air pollution. Urban development permits generation of equivalent quantities of ecosystem services but only when larger land areas are used, demonstrating the value of natural forests as compact ecosystem service providers.

Air pollution removal

Air pollution removal is another key ecological service provided by forests. Pollution removal by a given forest is determined by the amount of pollution in the region, assessed using information gathered from air-pollution monitoring stations scattered across the United States. The Washington DC UFORE project used data from sources located within the city itself, the ideal source of air pollution data. Prince William Forest Park used data from the city of Manassas, located northwest of the park in central Prince William County. Accordingly, the actual quantity of air pollution abated by the park may be greater or less than that reported, as the exact amount of each pollutant within the park is not precisely known.

The UFORE model estimated that PWFP removed 414 tons of air pollution of five different types: ozone, carbon monoxide, sulfur oxides, nitrogen dioxide, and particulate matter. The urban forest removed 540 tons of these pollutants (Nowak *et al.* 2006a). A possibility for the different quantities of pollutants removed is the difference in pollution data sources. Washington DC used sources within the District of Columbia for pollution data, while this study used pollution data from a location outside the surveyed area. The effects of this difference on the pollution removal are unknown, but the data source for this study was not as heavily urbanized as Washington DC. Comparison of the quantities of pollution removed is therefore meaningless. Percentages of each pollutant removed by the forest, whether natural or urban, remained similar for all five pollutants despite differences in the species composition, land area, and species dominance of each forest. This may be a result of the model itself, as species-specific data regarding pollutant-plant interactions are not available for all species, requiring use of genus-level data or values for hardwoods or softwoods (Nowak, personal communication). Equations within the model may have estimated air pollution removal for both sites with a similar range of values as inputs.

Management recommendations

If Prince William Forest Park is managed for long-term carbon storage as well as preservation of the forest as an example of eastern deciduous forest in the Piedmont, changes to management strategies need to be implemented to maximize ecosystem services provided by the natural forest. Reversal of the oak decline trend in the mid-Atlantic region is one possible means to increase total carbon storage capacity. Oak trees within the park possess both great aboveground biomass stores and potential longevity; oaks over 300 years old have been found within the mid-Atlantic region (Abrams and Copenheaver 1999). Efforts to increase the population of the larger oak species (such as white oak, black oak, northern red oak, chestnut oak, and southern red oak) are necessary, possibly including controlled burns and more stringent control of existing deer populations. Results will not be quick, as the typical rotation of oak forest is > 65 years (Guyette *et al.* 2007). If carbon sequestration is also taken into account in an effort to maximize the benefits derived from ecosystem services within the park, then species known to grow rapidly and to large size are required. Tulip-poplar is the most common tree within the park that meets these criteria, growing to large size and quickly while also living for several centuries (Busing 1995). An additional consideration is the park's mandate to preserve the eastern deciduous forest. Depending upon the interpretation of this mandate, the forest might be managed to alter the community structure and composition to a pre-European settlement condition, to a post-Revolution condition, or any other time-point, each with differing effects on the ecosystem services the forest provides.

A recent study conducted by Fall and others (2009) examined the change in mean annual temperature of different land-uses over the past several decades. Forests within the United States experienced a rise in mean annual temperature of 0.031° C yr^{-1} from 1992-2001, while urban environments experienced an increase of 0.058° C yr^{-1} during this time (Fall *et al.*, 2009). Furthermore, sites converted from forest to urban land-use experienced increases up to 0.066° C yr^{-1} over the course of the decade, while sites that converted from urban to forest land use decreased in mean annual temperature by 0.019 ° C yr^{-1} (Fall *et al.* 2009). Prince William Forest

Park may also act as a regional heat sink if the mid-Atlantic region follows these trends. The IPCC report (2007) details the effects of an increase in mean annual temperature on human health to date, including increases in heat-related mortality in Europe. A reduction in the rate of temperature increase in the region could therefore generate additional ecosystem services for the population of the Washington D.C. Metropolitan area in the form of health benefits.

Conclusions

Multiple avenues of research lie ahead of this study. The outputs of the UFORE model estimated the ecosystem services of the entire forest. No distinctions were made between the various land-cover types and forest associations found within the forest. The four primary forest associations within the park vary from one another in terms of species dominance, soil conditions, canopy structure, and other factors that make each association unique. Stratification of the park into different forest types would have provided interesting contrasts between distinct forest communities. In turn, this information could have served multiple purposes. Detailed ecosystem service provision data could be used to improve understanding of the North American forests and also establish the value of ecosystem services provided by different forest types. This has potential applications in landscape management, permitting managers to plan the optimal use of their forests based on an improved understanding of the true value of the services provided by different forest associations. For example, if a road were to be cut through Prince William Forest Park, knowing which forest association(s) were the most and least valuable providers of ecosystem services could facilitate planning of the road to minimize the impact of the construction on those services. If the potential effects of various pest and pathogen species are also assessed based on forest association, the vulnerability of different forest types can be taken into account in such decisions as well.

On a larger scale, UFORE evaluations of the natural forest should continue. The logical first step would be evaluation of the other parks in the National Capital Park region. These parks are located in similar climate and physiographic locations to Prince William Forest Park but each has been managed differently historically. A complete evaluation of the National Capital Parks allows for management of ecosystem services of each park as an individual unit yet with an eye toward the services provided to the entire region. Beyond this first step, evaluation of the ecosystem services of the surrounding region is a possibility. According to the 2001 National Land Cover Dataset (NLCD), Prince William County contains 460 km^2 of forest within its bounds and covers 910 km^2 of area. Prince William Forest Park makes up only 51 km^2 of the land area within the county. As of 1992, the northern Piedmont region of Virginia contained 10,430 km^2 of forest land throughout 18 counties totaling 17,800 km^2, or 58% of the total land area (Thompson 1992). Given that Prince William Forest Park stored nearly 400,000 metric tons of carbon at the time of this study on 51 km^2 of land, it is reasonable to believe that northern Virginia retains millions of tons of carbon as aboveground biomass and sequesters as much carbon annually as the park stored. Detailed information regarding these ecosystem services would lead to more informed strategies that could maximilze the ecosystem services provided by the region's forests.

Appendix One: Carbon Storage in Tree Species sampled in Prince William Forest Park

Listed below are the tree species found in the surveyed plots, listed in order of their species-specific contribution to the total carbon storage of the forest. This list is not an exhaustive list of tree species within the park, as other species were spotted by the survey teams or National Park Service personnel but not found within the surveyed areas.

Scientific name	**Common name**	**Carbon stored (% of total)**	**Net Carbon sequestered (% of total)**	**Leaf area (% of total)**	**Tree value (% of total)**
Fagus grandifolia	*American beech*	22.2	18.3	25.7	12.1
Nyssa sylvatica	*Black tupelo*	17.8	6.9	7.2	5.1
Acer rubrum	*Red maple*	11.7	11.4	12.3	9.3
Pinus virginiana	*Virginia pine*	10.3	8.2	6.7	12.4
Cornus florida	*Flowering dogwood*	6.7	2.3	3.1	1.1
Liriodendron tulipifera	*Tulip tree*	5.6	10.6	14.5	15.7
Quercus alba	*White oak*	5.6	14.1	9.5	18
Ilex opaca	*American holly*	5.4	2	1.5	1
Carya tomentosa	*Mockernut hickory*	1.9	4.2	2.5	3.7
Quercus rubra	*Northern red oak*	1.5	4	3.4	4.4
Carpinus caroliniana	*American hornbeam*	1.4	0.6	0.9	0.3
Other species	*Other species*	1.2	0.3	0.5	0.3
Sassafras albidum	*Sassafras*	1.2	0.4	0.3	0.1
Quercus velutina	*Black oak*	1.1	7.5	3.3	6.8
Liquidambar styraciflua	*Sweetgum*	0.8	0.5	0.7	0.6
Juniperus virginiana	*Eastern redcedar*	0.7	0.1	0.1	0.1
Quercus prinus	*Chestnut oak*	0.6	1.9	1	1.6
Quercus phellos	*Willow oak*	0.4	0.1	0.1	0.1
Quercus coccinea	*Scarlet oak*	0.4	2.1	1.4	2.6
Betula nigra	*River birch*	0.3	0.4	0.5	0.3
Quercus bicolor	*Swamp white oak*	0.3	0.5	0.8	0.6
Aralia spinosa	*Devils walking stick*	0.2	0	0	0
Asimina triloba	*Pawpaw*	0.2	0	0.6	0
Fraxinus americana	*White ash*	0.2	0.1	0.1	0.1
Quercus palustris	*Pin oak*	0.1	0.1	0.2	0.1
Castanea pumila	*Chinquapin*	0.1	0	0	0
Quercus sp.	*Oak*	0.1	0.3	0.1	0.3
Quercus falcata	*Southern red oak*	1.2	1.1	1.3	1.2
Fraxinus pennsylvanica	*Green ash*	0.1	0	0	0
Hamamelis virginiana	*Witch hazel*	0.1	0	0	0
Platanus occidentalis	*American sycamore*	0.1	0.1	0.1	0.1
Juglans nigra	*Black walnut*	0.1	0.1	0.2	0.1
Carya cordiformis	*Bitternut hickory*	0	0	0	0
Carya glabra	*Pignut hickory*	0.1	0.3	0.3	0.3

Appendix I Cont.

Scientific name	**Common name**	**Carbon stored (% of total)**	**Carbon sequestered (% of total)**	**Leaf area (% of total)**	**Tree Value (% of total)**
Cercis canadensis	*Eastern redbud*	< 0.1	< 0.1	< 0.1	< 0.1
Pinus echinata	*Shortleaf pine*	< 0.1	0.1	0.1	0.1
Quercus stellata	*Post oak*	0.3	0.3	0.2	0.2
Catalpa speciosa	*Northern catalpa*	< 0.1	< 0.1	< 0.1	< 0.1
Celtis occidentalis	*Northern hackberry*	< 0.1	< 0.1	< 0.1	< 0.1
Diospyros virginiana	*Common persimmon*	< 0.1	< 0.1	< 0.1	< 0.1
Prunus serotina	*Black cherry*	< 0.1	< 0.1	< 0.1	< 0.1

Appendix Two: Tree Frequency and Density Data for Prince William Forest Park

Tree frequency and density data for trees sampled in Prince William Forest Park. Species are listed alphabetically by scientific name.

Scientific name	Common name	Frequency (%)	Total live trees sampled	Density (trees/ha)
Acer rubrum	Red maple	83	547	135.1
Aralia spinosa	Devil's walking-stick	2	8	2
Asimina triloba	Paw-paw	4	8	2
Betula nigra	River birch	2	11	2.7
Carpinus caroliniana	American hornbeam	18	68	16.8
Carya cordiformis	Bitternut hickory	1	2	0.5
Carya glabra	Pignut hickory	5	8	2
Carya sp.	Hickory species	2	2	0.5
Carya tomentosa	Mockernut hickory	38	97	24
Castanea pumila	Chinquapin	1	3	0.7
Catalpa speciosa	Northern catalpa	1	1	0.2
Celtis occidentalis	Common hackberry	1	1	0.2
Cercis canadensis	Redbud	1	2	0.5
Cornus florida	Flowering dogwood	27	264	65.1
Diospyros virginiana	Common persimmon	1	1	0.2
Fagus grandifolia	American beech	90	1122	277
Fraxinus americana	White ash	4	8	2
Fraxinus pennsylvanica	Green ash	3	5	1.2
Hamamelis virginiana	Witch-hazel	3	4	1
Ilex opaca	American holly	59	267	66
Juglans nigra	Black walnut	3	3	0.7
Juniperus virginiana	Eastern redcedar	14	28	6.9
Liquidambar styraciflua	Sweetgum	9	41	10.1
Liriodendron tulipifera	Tulip poplar	61	259	64
Nyssa sylvatica	Black tupelo	86	824	203.5
Pinus echinata	Shortleaf pine	1	2	0.5
Pinus virginiana	Virginia pine	45	365	90.1
Platanus occidentalis	American sycamore	3	4	1
Prunus serotina	Black cherry	1	1	0.2
Quercus alba	White oak	62	243	60
Quercus bicolor	Swamp oak	9	13	3.2
Quercus coccinea	Scarlet oak	13	19	4.7
Quercus falcata	Southern red oak	14	84	20.7
Quercus palustris	Pin oak	1	7	1.7
Quercus phellos	Willow oak	2	18	4.4
Quercus prinus	Chestnut oak	11	27	6.7
Quercus rubra	Northern red oak	29	73	18
Quercus sp.	Oak species	3	6	1.5
Quercus stellata	Post oak	3	5	1.2

Appendix Two cont.

Scientific name	Common name	Frequency (%)	Total live trees sampled	Density (trees/ha)
Quercus velutina	Black oak	26	55	13.6
Sassafras albidum	Sassafras	24	51	12.6

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