A Methodology for Inventorying Stored Carbon in An Urban Forest

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ABSTRACT

Trees in urban areas store carbon directly through photosynthesis, but they also provide the added benefit of reducing carbon emissions produced by fossil-fuel burning power plants, by means of energy conservation from strategically-planted trees near buildings, as well as by area-wide reductions in the urban heat island effect. Quantifying the role of urban forests is an important prerequisite to managing the vegetation to optimize benefits, and also serves to assign value to the important ecosystem services provided by urban trees. Decisions by policy-makers regarding the management and use of urban trees requires accurate and precise information about the state of the resource. This paper creates a methodology for conducting a carbon inventory in an urban forest in the Washington, DC area, one that requires a minimum of data gathering. The methodology could serve as a tool for other similar high-density urban areas to measure carbon resources in urban forests and to serve as the basis for further research. Carbon trading systems may provide opportunities for forest owners to sell carbon credits to entities that produce CO_2 in excess of national or international limits; quantifying urban forest carbon would be necessary as a baseline for future carbon offset projects.

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INTRODUCTION

This paper was inspired by the U.S. Geological Survey's (USGS) Urban Biodiversity Information Node (UrBIN), part of the National Biological Information Infrastructure (NBII), in cooperation with Virginia Tech and the Metropolitan Washington Council of Governments. The UrBIN Pilot project aims to provide communities with the information and tools needed to proactively manage urban natural resources. UrBIN's goal is to serve a coordinating role in the delivery of standards, tools and techniques necessary to find and make use of biological resources information. Stakeholders in UrBIN include resource managers, scientists, educators and the general public (Bryant et al. 2003).

The UrBIN Pilot Project focused on an urban watershed encompassing portions of Fairfax County and the cities of Falls Church and Alexandria in Northern Virginia, in the metropolitan Washington, DC area. This watershed is referenced by several different names on maps used by various political jurisdictions.

Although officially designated Hunting Creek Watershed by the USGS, the headwaters drainages in this watershed originate as Holmes Run in the eastern portion of Fairfax County, and Tripps Run in the city of Falls Church. These streams conjoin behind Lake Barcroft Dam, a Water Improvement District impoundment, and exit the spillway as Holmes Run. Four miles southeast of the lake, at the confluence of Backlick Run and Holmes Run, the stream's name changes to Cameron Run. Further downstream, the main stem name changes again, to Hunting Creek, where Hunting Creek Branch flows into Cameron Run. Hunting Creek discharges into the Potomac River southwest of the District of Columbia (DC). Figure 1 shows all the tributaries of the watershed.

Urban forests can play a critical role in helping to reduce increasing levels of atmospheric CO_2, as well as provide a wide variety of ecological services and amenities to communities. Trees store carbon (C) derived from CO_2—the major gas contributing to global climate change, reduce peak cooling and heating loads on power plants, thereby reducing C emissions. They can also reduce the higher ambient air temperatures that occur in urbanized areas due to large amounts of heat-absorbing materials.

The Hunting Creek Watershed provides an interesting case for urban forest analysis because the entire watershed is highly urbanized, and is representative of the built-out regions

immediately surrounding metropolitan Washington, DC. Population increases and residential and commercial development have dramatically altered the landscape in the last 40 years (Mid-Atlantic Regional Earth Science Applications Center 2002). In order to assess the state of forest resources in Hunting Creek, it is necessary to gather accurate information about them. In this case, the goal is to quantify the amount of C sequestered in the watershed's urban trees, to assign a value to the urban forest and the ecological services it provides. A secondary goal is to provide a baseline for future C trading systems.

Based on a literature survey, a methodology was devised for conducting a C inventory in urban forests in the Hunting Creek. The results of such an inventory would add to the available biological database on the watershed, and could serve as a basis for further research and inventory of other important environmental parameters. In addition, the inventory methodology would be a useful tool that is transferable to other similar high-density urban areas.
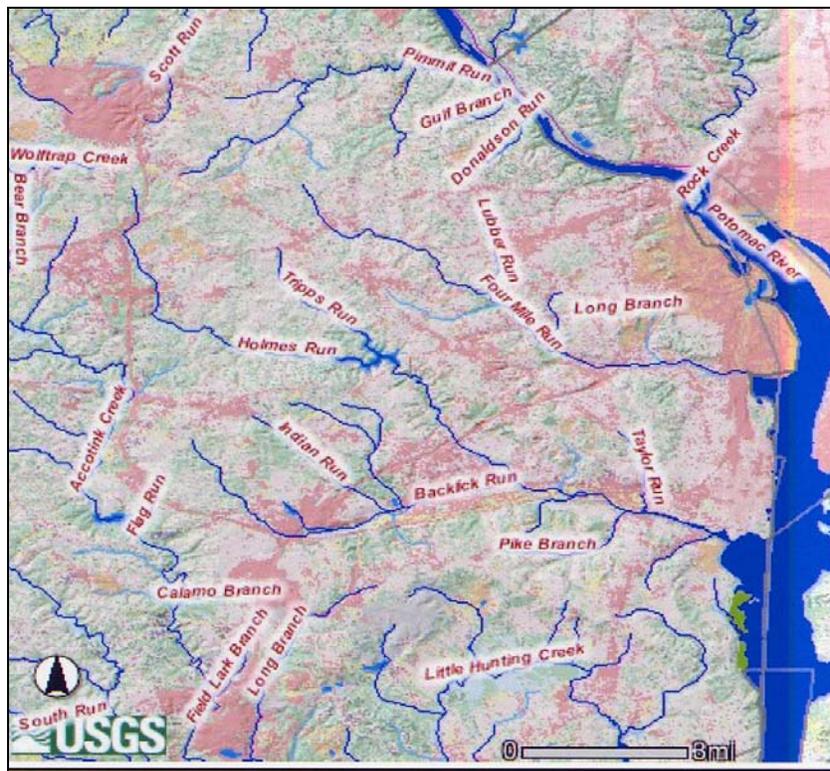
Figure 1. Hunting Creek Watershed Source: http://seamless.usgs.gov/viewer.htm

PROJECT AREA

The project area lies between latitudes 38° 54 min 00 sec. N (northern boundary) and 38° 47 min. 30 sec. N (southern boundary), and longitudes 77° 6 min. 00 sec. W (eastern boundary) and 77° 14 min. 00 sec. W (western boundary). Figure 2 shows a map of the watershed. Hunting Creek is part of the Potomac River Basin in the Washington, DC, metropolitan area. Major tributaries include Backlick Run, Indian Run, Turkeycock Run and Tripps Run. Most of the streams are encased. Lake Barcroft, located at the confluence of Holmes and Tripps Runs, is the major water body in the watershed.
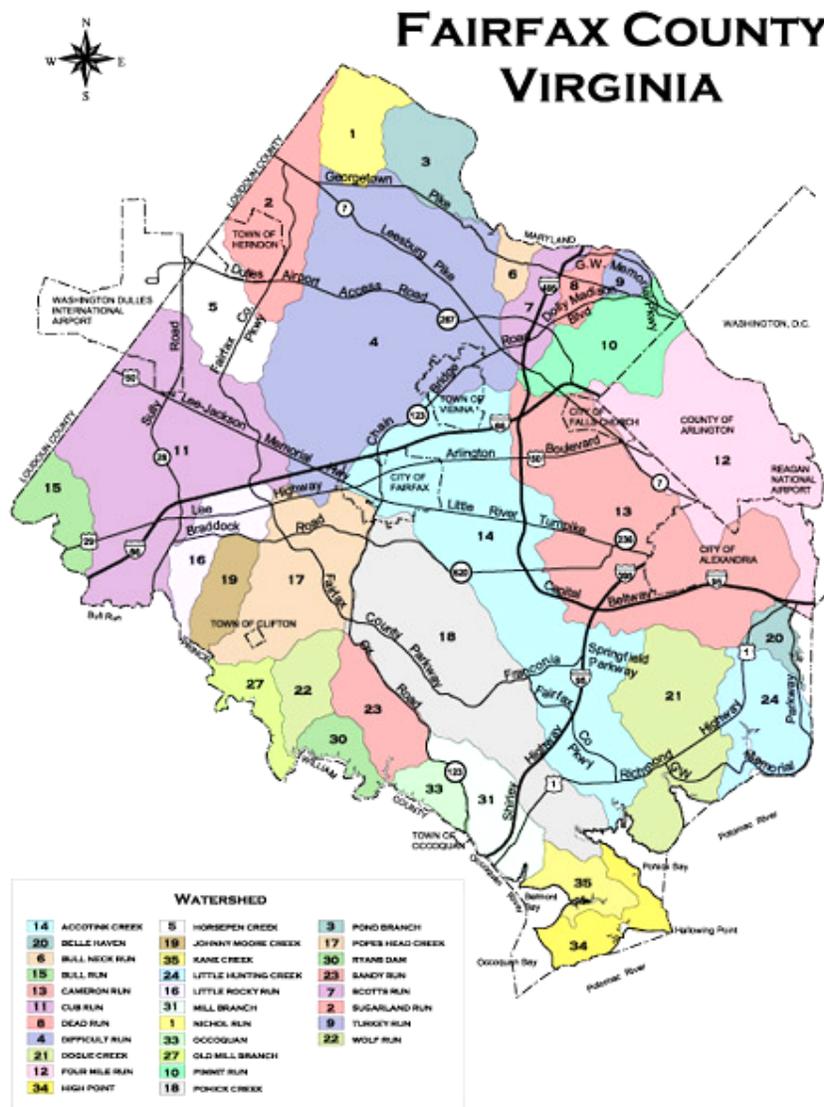
Figure 2. Watershed Map of Northern Virginia Source:
http://www.co.fairfax.va.us/maps/images/maps/handouts/watersheds_Tnail.jpg

The Hunting Creek watershed is approximately 42 square miles, or 116 square km.[1] The majority of the watershed is located in eastern Fairfax County, Virginia (31.5 square miles); the remaining, smaller portions are in Falls Church (2.2 square miles) and Alexandria (10 square miles). Hunting Creek is almost evenly divided between the Coastal Plain and Piedmont Plateau physiographic provinces. Landforms vary from rolling hills in the western part, to flat lands in the eastern sections (Soule 1976, Bryant et al. 2003).

Until the mid-20th century, the area economy was based on agriculture and forest products. After World War II, the growth of the Federal Government and suburbanization caused the watershed to evolve into residential communities for the District of Columbia. By 1998, almost 80% of the total area had been developed. Agriculture has disappeared completely, and less than 20% of the land remains forested. Two percent of the land area is in water bodies (Lake Barcroft, Fairview Lake [15 acres], regional ponds), or open urban land, including parks, golf courses, sports fields, stream buffer zones and utility rights-of-way (Mid-Atlantic Regional Earth Science Applications Center 2002). Today, the watershed is a densely populated, mixed-use land area, and resembles the kind of medium- to high-density developed areas common throughout the country (Bryant et al. 2003, Mid-Atlantic Regional Earth Science Applications Center 2002).

The natural environment has been highly altered. Before European settlement of the area, oak-hickory, American beech, and oak-pine forests, following demarcations of the physiographic provinces, formed habitats for a variety of wildlife. (See Appendix 1 for a list of animal and plant species in the watershed). Today the undeveloped land that remains is mostly in stream corridors. Remnant natural systems continue to function, with more generalist wildlife abundant, including possums, raccoons, whitetail deer, foxes, muskrats, bats, and recently, coyotes (Bryant et al. 2003). The flora are dominated by invasive exotic species.

Forest resources within the watershed area are generally small, fragmented parcels, often associated with riparian corridors, as is most open protected land and parks. Upland habitat has been especially impacted by development and impervious surfaces. Decades of development has transformed the watershed from a forested area into a highly urbanized space, with a total impervious surface greater than 40% (Bryant et al. 2003). Upland patches that do persist are small and fragmented—private yards and the urban forest canopy serve as the main upland

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[1] Source: Fairfax County GIS and Mapping Office, Fairfax, Virginia.

habitat elements. Generally, larger parks provide the only connection of upland to riparian habitats. Riparian habitats that provide remnant patches and corridors are often compromised by land development and roads. The watershed has been largely built-out for decades, and this limits the ability to assess land use change using recent geospatial data and recent land use management tools. There is little undeveloped land and limited opportunity for land and habitat conservation (Bryant et al. 2003).

The remaining natural forest patches are predominantly oak-hickory type forest mix, the largest class in Fairfax County and across Virginia (US Department of Agriculture Forest Service 2002b, 2002d). The oak-hickory type forest consists of upland oaks or hickories comprising the plurality of stocking. Common associates include yellow-poplar, elms, maples and black walnut.

UNIQUE FEATURES OF THE URBAN FOREST

Urban forests are ecosystems characterized by the presence of trees and other vegetation in association with people and their developments (Nowak et al. 2001). They may be defined as the sum of all woody and associated vegetation in and around dense human settlements, from small communities to large metropolitan cities. Urban forests include trees managed by municipalities and other public agencies, such as trees along streets and highways, and trees in parks and around public buildings. Publicly-owned urban trees typically represent only 10% of the urban forest within the US. The remaining 90% are found on private property such as residential yards, corporate parks, industrial sites and are managed by their owners, or not managed at all (Miller 1997).

Suburban landscapes are very different from landscapes with a lesser human presence. Suburban landscapes have high species richness due to the presence of non-native exotics; a predominance of generalist species and a scarcity of specialists; less interior forest area; high inputs of minerals and toxins in the soil; and high levels of soil erosion and nutrient and pollutant runoff (Mid-Atlantic Regional Earth Science Applications Center 2002). In addition, the people-to-trees ratio may be hundreds of times higher in an urban area than in a rural forest, so any management activity must be done with a high degree of sensitivity to social and political factors (Sampson et al. 1992).

In urban forests, street trees are most widely understood, because they are often planted and maintained by municipal governments. An official tree inventory is more likely to exist, which

usually includes data on species, size and condition. Unlike forest trees most urban trees are planted in rows or small groupings, with arborists or urban foresters selecting specific species, cultivars, sizes and shapes for each planting space (Sampson et al. 1992).

Street trees are forest trees and cultivars of forest trees, transplanted to an alien environment, thus demanding intensive management (Miller 1997). Street trees in city centers are generally characterized by small size, poor health and short life spans (Sampson et al. 1992). Growing space is much more limited in central city areas—trees are often planted close to concrete or asphalt surfaces, in too-small pits, and are routinely installed near established trees (Lipkis and Lipkis 1990).

Urban soils are typically a mixture of subsoil, bedrock and construction wastes, often compacted so that soil porosity is quite low (Lipkis and Lipkis 1990). Soil nutrient levels may be too low for normal tree growth, or too high in sodium (from de-icing salts put on roads), heavy metals, oils, or pesticides, making the soils toxic to trees. Urban soils also tend to be more alkaline than forest soils due to leaching of calcium from concrete sidewalks (Society of Municipal Arborists 2001). Urban soils likely contain less C per hectare than forest soils due to lower C inputs and increased soil decomposition rates due to warmer air and soil temperatures (Nowak and Crane 2002). Drainage is frequently so poor that routine irrigation leads to a waterlogged environment in which roots are unable to grow; also common root pathogens thrive under warm, moist conditions created when water is frequently applied to lawns and gardens during summer months (Society of Municipal Arborists 2001).

Analysis shows that as adjacent land-use becomes more urbanized, forest soil C pools can be affected even in stands not directly disturbed by urban land development (Pouyat et al. 2002). Direct effects include physical disturbances, burial or coverage of soil by fill material and impervious surfaces, and soil management inputs such as fertilization and irrigation. Indirect effects involve changes in the abiotic and biotic environment that can influence soil development in intact soils. Indirect effects include the urban heat island effect, soil hydrophobicity, introductions of exotic plant species, and atmospheric deposition of various pollutants (Pouyat et al. 2002).

Urban habitat can be just as harsh above ground as it is below. Overhead utility lines, buildings and traffic ways often occupy the space into which the tree's branches normally grow (Lipkis

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and Lipkis 1990). Unlike forest trees, many urban trees lack protection from wind and elevated temperatures that neighboring trees have to offer. They receive light from above as well as the sides, and are often exposed to artificial light at night. Increased light may raise growth rates, but usually results in increased air temperatures, which increase water loss from the leaves (Society of Municipal Arborists 2001).

Increased exposure to wind and sunlight alters growth form. Urban trees tend to be more broad-headed as their lower branches persist and grow horizontally (Society of Municipal Arborists 2001). Forest trees self-prune as a result of close spacing and shading of the lower branches, but city trees are open grown and do not self prune (Miller 1997). In natural forests, new trees are always being planted through natural seeding. There is a mix of tree ages. (Lipkis and Lipkis 1990). Concrete and lack of natural ecosystem can prevent trees from re-seeding themselves in urban forests.

Street trees in residential areas ordinarily have more room to grow, and are larger and longer-lived. Even there, however, trees planted on the tree lawn between streets and sidewalks are often characterized by one-sided root systems, growing under the sidewalk and into adjoining yards to find adequate space (Sampson et al. 1992). Street widening, curb installation or sidewalk projects that cut existing roots and block additional root formation can convert healthy, safe trees into trees that have dying branches and the inability to withstand strong winds, i.e., safety hazards (Sampson et al. 1992).

Yard trees provide great variety in urban forests, and reflect the preferences of individual landowners rather than the professional guidelines of arborists. Older residential areas tend to be dominated by large old trees, and yard trees tend to make up almost half of trees in the urban forest (Sampson et al. 1992). Parks and greenways offer great variety of space and growing conditions for trees within urban areas. Trees are often much more successful in suburban areas because conditions are less harsh, and more space is available to support root and tree growth (Miller 1997, Society of Municipal Arborists 2001).   Here trees tend to live longer, approximating life spans of the same species grown in forest conditions. Greenways such as stream corridors, abandoned rail lines, and hiking trails are a special case; they contain largely natural vegetation. These areas are important for recreation, but also urban wildlife corridors (Sampson et al. 1992).

Urban trees are characteristically surrounded by pavement, which increases the ambient air temperature around them. Pavement also transfers some of the heat gained by exposure to sunlight to the soil below, creating unfavorable conditions for roots and associated soil micro-organisms (Society of Municipal Arborists 2001).

CLIMATE CHANGE, CARBON DIOXIDE AND TREES

The general consensus among scientists is that increased levels of greenhouse gases (GHGs) from human activities—burning fossil fuels, and massive deforestation in many regions of the world—are changing the earth's climate. CO_2 plays the major role in absorbing outgoing terrestrial radiation and contributes about half of the total greenhouse effect. Between 1850 and 1990, around 100 gigatons (1 x 10^9) of C was released into the air just from land-use changes (Pandey 2002). Most of the increase has been since 1940 (Hair and Sampson 1992).

The atmospheric CO_2 concentration is currently rising by 4% per decade (Jo and McPherson 2001). This trend could double pre-industrial CO_2 concentrations within the next 50 to 100 years. Most authoritative sources estimate a doubling of atmospheric CO_2 could cause mean global temperature increases from 2.7 - 8.1º F (1.4 - 5.8º C) (Hair and Sampson 1992, Pandey 2002). Climate disruption may include changes in mean temperatures, melting of polar ice caps, ocean warming and expansion, and new, more erratic weather patterns.

Concentrations of CO_2, as well as other GHGs such as chlorofluorocarbons and NO_x, are persistent. Even in the absence of additional emissions, it would take decades to remove the build-up that has taken place since the mid-19th century. If existing predictions are correct, such changes may pose a serious threat to ecological and socio-economic systems (Jo and McPherson 2001).

Worldwide concern about global climate change has created increasing interest in trees to help reduce the level of atmospheric CO_2 (Dwyer et al. 1992). As urban forests both sequester C and affect the emission of CO_2 from urban areas, urban forests can play a critical role in helping combat increasing CO_2 levels (Nowak and Crane 2002). Trees in the conterminous United States currently store 700 million tons of C, with a gross C sequestration rate of 22.8 million tC/yr, worth an estimated US$460 million per year (Nowak and Crane 2002). Between 1952 to 1992, increases in biomass and organic matter on forestlands added an average of 0.3 petagrams (1 x 10^{15} g) per year of stored C to forest ecosystems, enough to offset 25% of US

emissions of CO_2 for the period (Murray et al. 2000). Currently, US forests offset about 11% of total US emissions, and have a greater effect on C emissions than any another sector except energy (US Environmental Protection Agency 2000).

Forests are the most critical for taking C out of circulation for long periods of time. Of the total amount of C tied up in earthbound forms, an estimated 90% is contained in the world's forests, including trees and forest soils. For each cubic foot of merchantable wood produced in a tree, about 33 lb. (14.9 kg) of C is stored in total tree biomass (Sampson et al. 1992). Forests sequester 1 Gt C annually through the combined effect of reforestation, regeneration and enhanced growth of existing forests (Pandey 2002). Urban trees' contribution may be even more significant if fast-growing species are given good growing space and favorable light, fertilization and moisture conditions (Sampson et al. 1992).

Trees remove C from the atmosphere through photosynthesis, and store excess C not used in the process as biomass. The C will remain bound until it is released again through respiration, burning or some other chemical transformation (Sampson 1989). The net long-term CO_2 source/sink dynamics of forests change through time as trees grow, die and decay. In addition, human influences on forests (e.g. management) can further affect CO_2 source/sink dynamics of forests through such factors as fossil fuel emissions and harvesting/utilization of biomass (Nowak and Crane 2002).

Urban forests, due to their relatively low tree cover, typically store less C per hectare in trees (25.1 tC/ha) than forest stands (53.5 tC/ha) (Nowak and Crane 2002). However, on a per unit tree cover basis, C storage by urban trees and gross sequestration may be greater than in forest stands. This is due to a greater proportion of large trees in urban environments and relatively fast growth rates due to the more open urban forest structure (Nowak 1994). Individual urban trees, on average, contain approximately four times more C than individual trees in forest stands. This difference is largely due to differences in tree diameter distributions between urban and forest areas (Nowak and Crane 2002).

Urban trees may have an indirect effect many times larger than their direct benefit of taking up CO_2 through photosynthesis and storing it as cellulose (Nowak 1993). This indirect effect comes from the energy-conserving value of trees' shade blocking solar radiation from reaching buildings, and the cooling effect of evapotranspiration. The lower energy consumption for air

conditioning reduces coal and oil burning, and less CO_2 is released into the atmosphere (Jo and McPherson 2001). Trees can also reduce wind speed, which in winter decreases the infiltration of cold outside air into building interiors, which decreases heat loss (Ebenreck 1989). A 10% increase in tree canopy is associated with wind speed reductions of 5 to 15% (McPherson 1994). The energy conservation effects of a single urban tree can prevent the release of 15 times more atmospheric C than the amount of C a tree can sequester (Sampson et al. 1992).

In urbanized areas, vegetation and soil have been replaced with radiation-absorbing concrete and asphalt. Because of the large amounts of impervious surfaces, and because cities use large amounts of energy and emit waste heat, urban areas experience higher temperatures than surrounding, less-developed areas. This heat island effect can cause temperatures to run 5.4º - 9º F (3 - 5º C) higher than adjacent rural areas (Miller 1997, Sampson et al. 1992). Trees in non-energy conserving sites can have an overall impact on reducing urban C emissions by reducing air temperatures 1 - 9º F (0.5 - 5º C) and consequent emissions associated with urban heat islands (Nowak et al. 2002).

Large numbers of trees and parks can reduce local air temperatures by the advection of cooler air and lessen city-scale cooling loads on power plants (McPherson 1994, Nowak 1994). Measurement of afternoon temperatures in Chicago on a hot summer day were 1 - 2º F (0.5 - 1º C) cooler in a city block with 59% tree cover than a nearby block with 36% tree cover (McPherson 1994). Evapotranspiration and wind reduction effects result from aggregate impacts of all neighborhood vegetation, not only the trees directly shading buildings. Benefits are shared by entire neighborhoods, they do not just accrue to those people whose houses are surrounded by trees (Jo and McPherson 2001).

URBAN FOREST INVENTORY METHODS

Quantifying the impact of urban trees is an important prerequisite to managing city vegetation for optimal beneficial effects (Nowak and Crane 2000). Decisions by policy-makers regarding the management and use of forests and trees require accurate and precise information on the state of the resource, and patterns and rates of change. The collection of data for use in an urban forest resource management strategy should be viewed as the most critical component of a strategic initiative (US Department of Agriculture Forest Service 2002a). The accuracy and reliability of the data collected will determine the ultimate value and usefulness of management tools that are developed for a community's urban forest.

Various strategies for the collection of field data for urban forest management can be employed, based on the ultimate intent of use. Urban tree inventories can include a collection of data on a large scale, such as canopy cover, forest type and condition, or can examine the specific condition of individual trees, based on field inspection and assessment. This wide range of scale represents problems and opportunities related to the level of management that is being employed in a particular community. Long-term strategic planning may utilize a more broad scale analysis of the forest, while short-term operational planning often uses a finer scale of analysis, based on day-to-day management of individual trees (US Department of Agriculture Forest Service 2002a). Data can be collected in a variety of ways, from fast and simple windshield surveys, to complete tallies of all trees. For large parcels, stratification and sampling survey procedures can be used; estimated parameters allow inferences to be made about the population as a whole, without having to intensively survey all stands (Holmström et al. 2001, Miller 1997).

Most urban street and park tree inventories are complete enumerations, wherein each and every tree is inventoried, and a number of parameters is recorded. These parameters can include diameter breast height (dbh), species, cultivar, height, location, condition, proximity of utilities, et al. Tree data is usually entered into a commercial software package, such as Davey Treekeeper, and used as a basis for maintenance activities such as pruning, watering and fertilization. Community tree inventory data is operational, that is, it is intended to be used in the context of municipal management activities, such as public works, planning and safety (Goodwin 1996, Miller 1997).

Foresters involved in stand-based production forestry have been sampling and measuring forests for merchantable volume and tree growth for decades; their techniques are well-developed and accepted. Carbon inventories can readily take advantage of these mensuration techniques (Brown 1999a). Two major types of forest inventories are continuous and in-place or stand-based. Continuous forest inventories (CFIs), such as the US Department of Agriculture Forest Service's Forest Inventory and Analysis (FIA) program and the Swiss National Forest Inventory, use periodic re-measurement of permanent sample plots to assess changes in forest stands over time. Permanent plots are particularly useful in assessing changes due to growth, mortality, species composition and catastrophic events. However, a CFI system is not effective without adequate financial support and trained personnel (Bell 2000). Stand-based inventories

don't measure changes as efficiently as a CFI system, but they do give better estimates of stand conditions. Plots that are not permanently marked for locating again are less costly (Bell 2000).

Stratified random sampling generally yields more precise estimates for a fixed cost than other options, and the sampling error tends to be smaller when compared to a simple random sample of the same size (Arvanitis and Portier 1997, Kohl 2001). The process requires stratification, or dividing the population into homogenous, non-overlapping subpopulations. Each stratum can be identified by vegetation type, soil type, et al. For C inventory strata would most logically be defined by estimated C pool weight e.g., aboveground biomass, roots, soil, soil litter, and underbrush.

In two-phased sampling for stratification, the first step is to obtain land cover data from satellites or specially-equipped airplanes. Data are generally acquired during the growing season, when leaves are on the trees. Digital data for all plots is analyzed to classify images into useable data, i.e., stratifications or land cover classes, and to estimate area. Because solar radiation reflected by tree crowns may be radiometrically different from its surrounding environment, it is possible (given scale, film type and resolution of data) to differentiate forested areas (Haara and Haarala 2002). Different substances such as asphalt and trees, for example, have different reflectance spectra, and it is the spectral differences between materials that enable a user to classify the image into different land cover types (Brown 1996). Scale will determine photographic resolution and this determines how accurately tree locations can be defined (Goodwin 1996). Subsequently, the size of the individual stratum is calculated by adding the parcels together (Kohl 2001).

Once the level of precision has been decided upon, sample sizes must be determined for each stratum in the project area (MacDicken 1997). In this phase of the process, a random sample of individuals from the first phase group is selected for another measurement, usually a more expensive measurement involving field visits to plots to gather data. Ground-based data will always be required of any remote sensing technique, no matter the resolution or source (Brown 1996).

The results are two measurements on a fraction of the total sample, and the less expensive measurement on the whole sample (Arvanitis and Portier 1997). In many cases, the more expensive method measures the specific parameter of interest, while the cheaper measures a

characteristic of the individual known to be related to the parameter of interest. A ratio of the two measurements, or a regression to find the correlation between the variables, is often performed. This is then applied to measurements from many plots where ground truthing was not performed, to obtain a sample at a relatively low cost.

INVENTORY PHASES

There are several generic phases of an inventory: planning, training, surveying, analyzing, interpreting and reporting. The first is the planning phase, in which the scope of the project is decided upon; remotely-sensed data and maps are gathered; the number and skill set of the personnel needed to gather and analyze data is determined; needed training is arranged; equipment is obtained; and a timeframe for completion of each step is devised, given budget constraints. Another important consideration is doing some title searching with local jurisdictions, to obtain contact information for owners of potential plots; it is necessary to obtain permission from private landowners to enter their property to measure their trees.

Training is a key component in a forest inventory; training the survey team forms the basis for the quality of data gathered. The Swiss National Forest Inventory devoted almost 10% of its entire working time on training (Zinggeler 2001). While this proposed methodology covers a much smaller land area than the Swiss inventory, and would have a much smaller budget, it remains that data precision, completeness and plausibility will depend on the people gathering that data knowing how to operate equipment and take tree measurements (Stierlin 2001).

The survey phase involves the actual photo image interpretation, followed by field inventory, in which parameter values and plot boundaries are determined (Holmström et al. 2001). The field inventory can be broken down into the following tasks:

- **Official trip**: driving from place of residency, university, etc., to the survey area and back.
- **Drive and preparation**: driving from one sample plot to another, plus the time expenditure required for the daily work plan and other preparatory work.
- **Walking**: all of the time needed to walk from the vehicle to the sample plot area and return.
- **Measurement**: measuring sample plot coordinates, determined in the aerial photo.
- **Inventory**: data gathering on the sample plot
- **Enquiry**: data gathering at the local government offices or on-line.
- **Other work time** : preparation to survey the sample plot, transfer data, and maintain equipment (adapted from Stierlin and Zinggeler 2001).

Analysis of the data will include the photo interpretation to determine forest cover, as well as the use of a spreadsheet or statistical program to perform the equations for estimating biomass and C, based on forest type, and perform statistical tests to measure central tendencies and variance. Next, the results of the data analysis would be used to make inferences about the population, and a report on the outcome would be produced.

CALCULATION OF STORED CARBON

Carbon is calculated by measuring a sample of trees and using a mathematical equation to determine the amount of biomass in each tree. The biomass amount is multiplied by a conversion factor that varies with species and region, to yield the amount of stored C in the tree. Most equations used in production forestry produce the merchantable volume, but to derive the C content, the total volume is necessary to produce an outcome that includes all parts of a tree, including leaves, twigs and smaller branches, and roots.

For C inventory, stratified random sampling generally yields more precise estimates for a fixed costs than do other options (Arvanitis and Portier 1997, Kohl 2001). The methodology proposed in this paper is for a two-phase stratified selection, with a random start.

Equipment for an inventory should be precise, robust and easy to operate (Stierlin and Zinggeler 2001). In addition to survey equipment and software, a vehicle and mobile phone would be key items for transportation and coordination with other people gathering field data. The equipment needed for the proposed inventory includes:

- Compass combination for navigation, plotting on the map, taking bearings.
- Hypsometer/clinometer for tree height and slope measurements, with a monopod or tripod and ball joint adapter.
- Diameter tape for measuring dbh.
- Statistical software for running analysis on data.
- GPS with differential correction capability and remote antenna to georeference the location of the plots.
- A personal digital assistant (PDA) for database use and for generating maps. Using a PDA eliminates need for manual data entry following field data collection activities, thereby reducing inventory costs (Boniarz et al. 2001).
- ArcPad or other Palm OS data collection program software for PDA.
- Laminated maps or photos, with plot locations and coordinates
- Ribbon (flagging) or paint for marking plots.
- Caliper for trees less than 3.5" in diameter.

▪ Supplementary tools: wedge prism (to measure basal area), increment bore to establish tree age.

The first step in undertaking a C inventory is to obtain remote sensing coverage for the entire watershed. Multi-Resolution Land Characteristics (MRLC) data is available from the (USGS). The latest data currently available is for 2000, which costs $45.00 per scene to order on cd-rom. The 2000 data was taken by satellite, the Landsat 7 Enhanced Thematic Mapper Plus (EMP+). MRLC data has a 30 meter resolution, and a scene covers 185 km^2. The scale is equivalent to 1:1,000,000, the standard scale for Landsat data. When processed into imagery, the photo scale can be altered up to a scale of 1:250.

The scale of photos will determine photographic resolution and this determines how accurately tree locations can be defined (Goodwin 1996). The scale of maps for use in data gathering on the ground is important; trees are small features on most maps. The scale of the base map needs to be large enough to distinguish one tree from another. A map at 1:24,000 (1"=1,000') is most likely too small scale for this type of mapping. Maps of building sites at 1:240 (1"=20') are most likely too big in scale. A map in the 1:1200 (1"=100') or 1:3000 (1"=250') is optimal for urban tree mapping (Goodwin 1996). It is large enough to show a neighborhood and small enough to show individual tree locations.

The next step in the process is to distribute a regular grid of plots (30 m^2) over the satellite photographs of the study area. A plot size of 30 m^2 was chosen because it corresponds to the pixel size of the MRLC data. The watershed contains 11,724 hectares, so the number of potential plots based on 30 m^2 is 130,266. Cover types that are represented must be determined using remote sensing images. The Landsat Thematic Mapper digital satellite imagery separates images into different spectral classes.

In this case, the top level strata would consist of forested and non-forested; if a plot is determined to be forested, it would then be classified into a second tier of finer landcover classes, for example, based on percentage canopy cover and the amount of impervious surface. Criteria for each substrata would need to be defined. For the methodology described here, three sub-strata are proposed: downtown areas (street trees), suburban, and remnant patches (of natural forest).

Downtown or city center areas tend to have the least percentage of tree canopy cover; trees are commonly planted individually in pots or holes in sidewalks. Downtown areas are characterized by high density development for commercial use, and can also include high intensity residential areas of multifamily housing and/or row houses. According to the USGS land cover class definitions, in high density areas vegetation accounts for less than 20% of landcover, with constructed materials making up for 80 to 100% of the cover (US Geological Survey 2001). Hill et al. (2002) define "high density" as multifamily housing or single-family housing over 8 units per acre, with a high percentage of impervious surface.

Suburban areas usually have more room for tree growth, above and below ground, and trees are larger and longer lived (Sampson et al. 1992). Older, well-established suburbs such as Falls Church and Alexandria have many areas of relatively low-density, single family development, plus long-established parks, and there are many large, old trees. Vegetation may account for 20 to 70% of the landcover, with a smaller proportion of concrete and asphalt than in downtown areas. (US Geological Survey 2001).

In the Hunting Creek Watershed, most remnant patches of natural forest are located in parks, stream corridors, and along hiking and biking trails (Bryant et al. 2003). Here, trees live longer than street trees, often approximating the lifespans of the same species grown in forest conditions (Sampson et al. 1992). These areas have the highest percentage of tree cover among land uses (Nowak et al. 1996), and lower proportions of impervious surface.

The discrimination of forest and non-forest areas in remotely-sensed data requires unambiguous, reproducible definitions of what "forested" means (Keller 2001), as well as unambiguous criteria to define substrata such as "downtown," "suburban" or "remnant." Unfortunately, there is no standard classification scheme used to stratify urban areas based on tree or vegetation cover. Land-cover data are analyzed using several methods, and there is little consistency between them (Hill et al. 2000, Homer et al. 2002). For the most current MRLC data (from 2000), there has been an effort on the part of the USGS to improve regional stratification methods. The goal is to produce a land cover database with nationally standardized land-cover classifications (Homer et al. 2002). There is also an international standardized system of land cover definitions being developed by the United Nations Food and Agriculture Organization (FAO). The Global Land Cover 2000 project end product, called GLC2000, will create a single database of classified land cover for the entire globe. All legend classes will be directly

comparable (Ahlchrona et al. 2002). While there are more urban land-use types represented under the GLC2000 scheme, there are still not integrated urban forest land cover types.

Hill et al. (2000) have developed a methodology for assessing urban watershed conditions, one which uses Landsat satellite imagery to produce land-cover characterization. The index characterizes the magnitude of urban development in a watershed. There are several urban land cover classes, based on impervious surface area, tree cover percentage, grass cover and development density. Land classification methodologies such as this one would be more useful to urban foresters than many classification schemes currently in use, which are not urban-oriented.

The number of plots that need to be sampled to reach an acceptable precision interval must be determined; the interval usually used is ± 10% with α=.05 (Boscolo et al. 2000, Cowardin et al. 1981, MacDicken 1999). Higher levels of precision require a more intensive inventory. Attaining precision levels higher than 10% increases the costs significantly (Boscolo et al. 2000). Thus aiming for a precision level of 5% almost doubles the cost of inventorying at the precision level of 10% (Bell 2000). The number of plots sampled should be proportionate to the total number of plots in each landcover class. For example, if it were determined that 100 total plots needed to be sampled—the number used to develop the budget for this project—the number of sample plots for each substrata would need to reflect the proportion of landcover in each forested substrata (downtown, suburban and remnant).

A subset of plots to be sampled should next be randomly chosen. It will be necessary to find out which plots are on public land, and which ore located on privately-owned land. Permission must be obtained from landowners to enter their property to conduct the inventory (Lipkis and Lipkis 1990). Public land will be more straightforward to inventory. For municipally-owned land that is not parks or recreation areas open to the public, permission should also be obtained from the respective city or county. If plots are in Fairfax County, there is an on-line Department of Tax Assessment (DTA) parcel finder to find detailed information about property and view that parcel. The database will produce owner information from a street address. For the cities of Falls Church and Alexandria, a title search at the county or city tax assessor's office must be performed, in person or via email, to find out owner information, then proceed with contacting owners.

Sample plots must be visited to take dbh measurements. A GPS unit can be used to geo-reference boundaries, to ensure finding the right location. It will be easier to perform on-the-ground data gathering (tree measurements) during the winter, when there is little underbrush, especially in remnant plots of natural (unmanaged) forest. All trees within each test plot that are greater than 2.5 cm dbh should be measured. Every tree in the plot should also be marked, after it is measured; this shows that the plot inventory was complete, and also provides an opportunity to double check that every tree has been measured. It might be useful to measure height, basal area and site quality for some plots, for later analysis. The dbh (plus any other data collected) should be entered into the PDA, for later uploading.

To estimate live tree biomass, diameters of all trees are measured and converted to biomass and C estimates using allometric biomass regression equations. The advantage of using generic equations, stratified by forest type, is that they tend to be based on a large number of trees and span a wider range of diameters; this increases the accuracy and precision of the equations. By pooling general species groups, highly significant regressions equations are produced between dbh only and biomass per tree, with r^2 of 0.98 or more (Brown 2002).

Applying equations developed via dimensional analysis is the only reasonable method to estimate tree biomass without destructive sampling (Jenkins et al. 2003). The disadvantage is that the generic equations may not accurately reflect the true biomass of the trees in the project. Jenkins et al. (2003) general biomass equations, based solely on dbh, are not precise on a tree-by-tree basis, but are carbon-oriented, and suitable for larger scale sequestration analysis. The equations are based on compilations of all available diameter-based allometric regression equations for estimating total aboveground biomass, defined in dry weight terms for US tree species. Plot-level biomass estimates are typically expressed on a per-unit-area basis. They are made by summing the biomass values for the individual trees on a plot, then standardizing for the land area covered by that plot. (Jenkins et al. 2003).

The **formula for total aboveground biomass** is:

$Bm = Exp(\beta_0 + \beta_1 \ln dbh)$ where

Bm = total aboveground biomass (kg. dry weight) for trees 2.5 cm dbh or larger (1)
Dbh = diameter breast height (cm)
Exp = exponential function
ln = log base e(2.718282)
β_0 –2.0127 β_1 2.4342 = parameters for hard maple/oak/hickory/beech species group

For example, if a tree dbh is 20 cm, the total (aboveground) biomass calculation would be:

Bm = Exp(-2.0127 + 2.4342 * ln(20))
 or Bm = Exp(-2.0127 + 2.4342 * 2.9957 = Exp(-2.0127+7.2922)
Bm = Exp (5.2795)
Bm = 196.274 kg

As the majority of the study area is in the Piedmont Plateau, where oak-hickory is the predominant natural forest type (with some American beech), the parameters for the "hard maple/oak/hickory/beech" species group were used. This species group contains 44 species in 3 families, including sugar maple, several hickory species, American beech, and various oaks (See Appendix 1 for the complete list of species included). For other regions of the US, the appropriate species group should be used, to ensure the biomass calculations are the most accurate.

To **calculate root biomass**, the allometric equation derived by Cairns et al. (1997) can be used:

RBD = exp(-1.085+0.926 ln(ABD)) where   (2)

RBD = root biomass density
ABD = aboveground biomass density (outcome of Jenkins et al. [2003] equation)
Exp = exponential function
ln = log base e(2.718292)
ß_0 −1.085 ß_1 0.926 = parameters

For the sample tree with a 20 cm dbh, the total root biomass would be 44.87 kg. Adding together the ABD and RBD, the total tree biomass would be 241.14 kg. Jenkins et al. (2003) and Cairns et al. (1997) equations are for individual tree biomass, so each tree's dbh is used. For each tree sampled, the aboveground and root biomass numbers are calculated for total individual tree biomass, then the numbers aggregated per plot.

Next, using the factors for oak/hickory forest type, convert the tree biomass to C. The average percentage of C for hardwoods is 49.1%, with some slight regional variations (Birdsey 1992b). For the sample tree of 20 cm dbh, at 241.14 kg of biomass, the C amount would be 118.399 kg. The easiest way to calculate C is to multiply aggregated biomass amounts of plots by the conversion factor to yield the C estimate.

The mean and standard deviation of data need to be determined. The next step is to perform statistical tests to determine data central tendencies and variance, using an appropriate statistical software package, such as Statistical Package for the Social Sciences (SPSS). Mean values, along with confidence limits ($\alpha=0.05$) would be reported.

The total percentage of hectares or acreage in forest (and non-forest) needs to be extrapolated, to produce the C estimate for strata in the entire watershed. Totals for each forested substratum also need to be calculated. This can be done by multiplying the mean C stored per plot by the number of acres or hectares that are in forest, to find an estimate of the total stored C in tree biomass in the watershed. To determine the stored biomass in downtown, suburban and remnant forested areas, the mean C stored in each plot in a particular substratum can be multiplied by the total acres or hectares classified in that stratum, for example. The last step is to report the results.

There is a caveat for using allometric equations on urban trees. Nowak (1994) says that biomass equations derived from forest stands overestimate biomass from open-grown urban trees by a factor of 1.25. At the same time, some urban trees, given optimal care, may contain more biomass than trees in forest stands. Open-grown trees are typically shorter, but often have larger more branchy crowns than forest-grown trees. However, urban tree crowns are often pruned, which removes stored C. More research is needed to further test the applicability of existing biomass equations to urban trees, and on how biomass estimates vary by land-use type and maintenance practices (Nowak 1994).

ESTIMATED COSTS AND BUDGET

How much does it cost to inventory C? Several factors influencing the cost are related to site-specific characteristics, monitoring scope, and the inventory and monitoring methods considered. Furthermore, inventorying costs depend on the local costs of labor, equipment, transportation, and logistics. The previous experience of personnel and the availability of quality data also influence the total budget (Boscolo et al. 2000).

Time and financial expenditures of the field work for large-scale inventories are usually not known, or only inadequately known. For the planning and execution of inventories, knowledge about the required time for important working stages is indispensable. If these basics are not available, it is difficult to employ personnel and material in a cost efficient way (Zinggeler 2001).

From reviewing the available literature on large-scale forest and urban tree inventories, the following costs must be included : purchasing of equipment, acquiring remote sensing data, the per hour or per diem wage for people doing data collection, based on their status (students, volunteers), transportation costs (per mile for gas, etc.), and costs for analysis of data—both analysis of remotely sensed data and of survey results. There will be additional costs for training, if using volunteers or inexperienced people, and for performing quality checks after data collection.

For this methodology, the assumption is that there would be two people who are student interns, therefore the wage per hour or per diem need not be as high as that paid to a government agency employee or person employed by a private company. Therefore the estimated per-hour wage is $20, with the assumption that 100 plots will need to be measured, and that an average of 2 hours will be needed to survey each plot. Furthermore, it is assumed that most equipment and software (SPSS, ArcInfo, ArcPad) will already be available to borrow from Virginia Tech's Northern Virginia Campus. For transportation, the assumption is that private cars will be used, therefore calculations use the Internal Revenue Service's standard mileage rate of 36.5¢ per mile to estimate transport costs.

Equipment costs include the purchase of low-cost GPS units, PDAs and a digital camera. GPS units (such as Garmin) are available for around $200.00 (US Department of Agriculture Forest Service 2002a). PDAs such as Palm Pilots can be purchased for approximately $150.00 each. Inventory software is available for free or from a variety of sources at low cost (Boniarz et al. 2001). A digital camera with a resolution of 4.0 megapixels can be obtained for less than $400.00. Because the equipment can be used for other work, the cost can be spread among this proposed inventory and other projects. Ten percent of the equipment purchase price was used as the equipment cost in calculating the total costs for this inventory. Other forest measuring equipment could be borrowed from Virginia Tech, such as hypsometers, logger's tape, increment bores, and wedge prisms. If the newest available MRLC data from 2000 is used, the cost would be $45.00 to purchase one view on a CD-Rom from the USGS. Several views may need to be obtained to adequately cover the total project area.

The cost of collecting data on individual trees is directly related to the amount of information that is obtained on each tree and the expertise of the data collector (US Department of Agriculture Forest Service 2002a). Using the Jenkins et al. (2003) equation necessitates gathering only one

tree measurement, dbh, so the time to measure and record this information should be minimal. However, it may take time to locate and verify the coordinates of the plot. Based on other inventory data, it is estimated that professional analysis of remote sensing data will cost $450.00 per view, based on averaging the analysis costs of several other studies (Bauer et al. 2003, Boscolo et al. 2000, Schreuder et al. 1999). Table 1 shows the proposed budget for the project:

Table 1. Estimated Costs for Forest inventory of Hunting Creek

FIXED COSTS	
Purchase of 2 GPS units and 2 PDAs ($200 x 2) + ($150 x 2) x .10	70.00
Purchase of digital camera (4.0 MP resolution) $400 x .10	40.00
Purchase of MRLC Data ($45 @ view x 3 views)	135.00
TOTAL FIXED COSTS	*$245.00*
VARIABLE COSTS	
Analysis of remote sensing data ($450 x 3 views)	1,350.00
Ground survey (2 people @ $20/hour, 2 hrs/plot @ 100 plots)	8,000.00
Transportation (1,000 miles @ 0.365 per mile [IRS Standard Mileage allowance])	365.00
Training (2 people @ $20/hour, 8 hours training)	320.00
Data Compilation, Analysis and Reporting (1 person @ $20/hour x 25 hours)	500.00
TOTAL VARIABLE COSTS	*$10,535.00*
TOTAL COSTS	**$10,780.00**

(Bauer et al. 2003, Boscolo et al. 2000, US Department of Agriculture Forest Service 2002a, MacDicken 1999, Schreuder et al. 1999, Zinggeler 2001)

The cost of training personnel is variable based on experience and expertise, e.g., in-house staff vs. contracted services, or paid interns vs. community volunteers. The utilization of completely inexperienced volunteers will require a minimum of 8-12 hours of training, while training of experienced personnel is normally completed in 3-4 hours of instruction (US Department of Agriculture Forest Service 2002a). If it is assumed that each person requires a minimum of eight hours for training, at the rate of $20.00 per hour, the total per person training expenditure would be $160.00. To compile data, analyze and report on inventory results, it was estimated to take 25 hours, again at the rate of $20.00 per hour.

The cost for performing this inventory would be approximately $108 per plot. As a comparison, the US Forest Service's FIA program cost approximately $1231 per plot in FY2001, the latest data available (US Department of Agriculture Forest Service 2002c). Large-scale inventories in Switzerland, Bolivia, and Minnesota ranged from $423, to $560, to $700 per plot, respectively (Boscolo et al. 2000, Schreuder et al. 1999, Zinggeler 2001).

BARRIERS AND PROBLEMS

Data Gathering
Remote sensing is a cost-efficient source of information for forest inventory and monitoring purposes, and has been widely applied in different forest inventory and monitoring tasks, which include estimating forest parameters. However, errors arising from coarse spatial resolution may occur, leading to a mismatch between spectral and informational classes (Pekkarinen 2002). This could cause mistakes in the preliminary identification of land cover, i.e., whether it is forest or non-forest.

Despite the many positive qualities inherent in using a GPS unit, accuracy may not be sufficient with a general-grade receiver. In a heavily urbanized community it may be difficult to receive sufficient signals to accurately locate plots (Goodwin 1996). Also steep terrain and heavy canopy cover can make it difficult to get consistent fixes due to loss of satellite signals (Leech 2000, MacDicken 1999). A surveyor-grade unit may be necessary to insure accurate plot placement, which could cost up to $5,000, significantly increasing project costs (Goodwin 1996, US Department of Agriculture Forest Service 2002a). In some cases, a skilled orienteer with an accurate map on the ground may be a better investment than an expensive GPS unit (Brown 1996).

Regardless of the inventory system used, accurate measurements are critical. Personnel should be highly trained and conscientious. In addition to training, accurate field measurements depend on quality instruments and their careful use. It is important to check instrument accuracy before going into the field. Generally it is better to make fewer accurate measurements rather than a greater number of estimates. It is also important to develop written procedures that ensure consistent measurements (Bell 2000).

Biopolitical Issues

A problem encountered in some inventories in the literature was the inability to obtain permission from property owners for data collection (Cowardin et al. 1981, Jo and McPherson 1995, 2001). Limited cooperation by property owners, and inability of people to enter property to gather data, could affect sample size. This could increase the time needed to complete data gathering, as replacement plots would need to be chosen. Ownership of these replacement plots would have to be researched; the owners' permission would have to be granted before the trees on those plots could be measured.

Data Analysis

Because urban areas are unique, they are difficult to measure in terms of C storage. Normal soil C dynamics are interrupted by compacted soil and impervious surfaces that prevent normal C contributions from aboveground sources and may also prevent C oxidation from the soil to the air (Person 1992). Urban trees branch out and have a different canopy cover than forest trees; because of this, some formulas used to estimate biomass and volume of forest-grown trees do not apply to open-grown urban trees (Nowak 1994, Jo and McPherson 1995, Person 1992). On the one hand, an urban tree with the same dbh or height as a forest tree could have a wider crown due to less competition, or more irrigation, and fertilization, and have a higher biomass. On the other hand, poor rooting conditions, air pollution, heat and severe pruning might lower biomass accumulation in an urban tree (Jo and McPherson 1995). Research is needed to further test the applicability of existing biomass equations to urban trees, and on how biomass estimates vary by land-use type and associated maintenance practices (Nowak 1994).

Annual tree mortality rates are needed to calculate biomass decomposition. Annual consumption of gasoline and diesel fuel over the course of a year by municipalities' urban forest departments needs to be compiled, and subtracted from the total amount of C sequestered. Tree care practices release C back to the atmosphere by fossil-fuel emissions from maintenance equipment (e.g., chain saws, trucks, chippers). Thus some of the C gains from tree growth are offset by C losses to the atmosphere via fossil fuels used in maintenance activities (Nowak, et al. 2002).

Since urban yard and street trees are often introduced species or cultivars (i.e., not part of the native forest mix of species) the final estimates of total C may be over or underestimated. If there are no models or conversion factors for a particular species, the actual biomass and C may be very different from what is estimated with allometric equations based on general forest

types. For jurisdictions with complete tree inventories, such as the City of Falls Church, the specific tree data for the actual species can be used to develop more precise C estimates, and to correct the over- or underestimation of C.

SYNTHESIS AND NEED

Decisions by policy-makers regarding urban tree management requires accurate information about the state of the resource. Quantifying C stored in the forest helps to assign value to an important ecosystem service—C sequestration—and estimates tree cover for the study area. Cover data can be used to determine the benefits provided by city trees and provide a basis to assess variations in tree cover across the landscape. To fully account for the impacts of forest and land-use activities, it would first be necessary to assess all relevant C pools (Murray et al. 2000). Performing a C inventory also verifies the C baseline which would be required to estimate offsets under a C trading system, or tree ordinances, planning initiatives or utility efforts to promote tree planting as a new kind of "weatherization" to increase and retain biomass and reduce CO_2.

Urban vegetation has the potential to make an important contribution to the reduction of atmospheric C, although it is only part of a solution to minimize risks of climate change. Carbon emissions reduction from shading is likely to be substantial in regions where the cooling season is long and coal is the main source for cooling energy (Jo and McPherson 2001). This information could be useful for evaluating the cost effectiveness of a utility-sponsored retrofit program that plants trees in strategic locations to obtain energy usage reductions.

Emissions trading involves moving a reduction in emissions output from one party to another. The reduction is then subtracted from the seller's total output. The buyer uses this reduction to achieve a legislative or voluntary emissions cap. The goal of such a scheme is to reduce costs by allowing a field of players to achieve emissions reductions using market mechanisms. It is perceived that over time, these mechanisms will drive emissions down and finance a shift to cleaner energy.

Carbon trading systems exist on many scales, from the mechanisms set out under international instruments such as the Kyoto Protocol, to regional and national schemes. The US has withdrawn support for the Kyoto treaty; however the anticipation of other C regulatory systems, such as the European Union's trading scheme which will come on-line in 2005, has prompted a

voluntary GHG trading market to emerge, based in part on the successful SO_2 trading program in the US (Miura 2003). There is also a proposed bill in the US Senate, the Lieberman-McCain bill, to establish a market-based cap and trade scheme, requiring US CO_2 emissions of year 2000 levels by 2010, and 1990 levels by 2016.

Trading GHG credits is a way for businesses and utilities to meet emissions standards, whether they be international standards or government standards at the federal, state, regional or local level. Even if US CO_2 trading system legislation is not passed, the US remains a potential market for companies from Europe or elsewhere that want to buy C credits. A company that produces excess CO_2 could pay a landowner, farmer or other entity to sequester C.

There are some potential limitations to C trading in the context of urban forests in general, and to the Hunting Creek area in particular. Given that the watershed is mostly built out, tree planting opportunities are fairly limited in the watershed. There is little available land on which to plant significant numbers of additional trees, except in parks and along stream corridors. The most common change in the study area is the removal of trees for infill development. It may be hard to produce a net increase in total biomass that is very large; since it would be net changes in the C pool that trigger C credits, the small net increases that could be achieved may not generate many funds. The small amount of income derived from increased C stored may be offset by the expense needed to monitor the resource.

Another problem is that the urban forest is owned by many entities. To be able to sell C credits, a property owner would need to have clear title to those credits, and a buyer would need such title to be able to claim any offsets. As there are thousands of entities who own small parts of the urban forest, the individual C sequestration potential may be too small for most land parcels. In addition, trading with numerous small landowners may be too difficult logistically to be viable in a C market.

However, there are some possibilities for creating a workable system for trading C credits. First, county or city governments involved own significant aggregate amounts of park land, recreational areas, rights-of-way and stream corridors. Jurisdictions such as Fairfax County, that are large enough, may possess enough forested land to be able to sell offsets on their own. Another alternative would be to promote strategic tree planting to reduce energy costs

and prevent emissions by reducing peak power loads. This would decrease individual property owners' heating and cooling bills, but also reduce power loads at generating plants, thereby preventing some C emissions.

If governments could use tax incentives or some other reward to landowners to persuade them to plant trees, jurisdictions could claim the increased C sequestration potential as part of their aggregate C pools, thus increasing the amount of offset they could sell. Returns from offset sales could be returned to participating home- and business-owners as a property tax refund or other incentive.

CONCLUSIONS

Few people know how many and what kind of trees are found in urban areas, or how these trees affect a city's environment and the health and well-being of its inhabitants. Measuring the urban forest is one of the first steps toward understanding this resource and developing appropriate management plans. The inventory process yields baseline reports that serve as benchmarks on which future changes in C pool size would be calculated.

Millions of metric tons of C currently stored by urban trees is a strong argument for at least maintaining the present urban forest structure. The loss of urban trees without replacement will act as a net C source to the atmosphere, both directly and indirectly, due to the loss of energy conservation around buildings. Establishing more properly chosen and located urban trees, in addition to maintaining the present structure, can make urban forests a larger sink for atmospheric C, along with producing other urban forest benefits.

Most forest sector actions that promote C conservation and sequestration make good social, economic and ecological sense even in the absence of climate change considerations (Brown 1999b). Lowering summer temperatures reduces the urban heat island effect, and can reduce the production of tropospheric ozone and smog (US Environmental Protection Agency 1992). Increased tree cover can improve water quality, reduce storm water runoff, improve habitat for wildlife, including enhancing wildlife corridors, and raise property values. The C conserved and sequestered from managing for these objectives will be an added benefit. The "happy side effect of such actions is they can't go wrong—if global warming doesn't occur or is less bad than predicted, we've done the right thing anyway"(Sampson 1989).

Future research is needed to analyze the C budget of urban trees (Nowak 1993). A better understanding of C cycling in urban forests is necessary to quantify growth and mortality rates of urban trees, determine which species are the best for C sequestration, and to analyze through time the C production (i.e. through planting and maintenance) and reduction (i.e. through sequestration and avoidance) by urban trees in an urban forest C budget (Nowak 1993). More research is also needed on quantifying how tree pruning ultimately affects longevity, and how it can be quantified to assess its affects on the urban forest's C budget (Nowak et al. 2002).

Urban forests likely will have a greater impact per area of tree canopy cover than non-urban forests due to faster growth rates, increased proportions of large trees, and the possible secondary effects of reduced building energy use and consequent C emissions from power plants. More field measurements are needed in urban areas to help improve C accounting and other functions of urban forest ecosystems. Quantitative monitoring of C sequestration over time, and to a lesser extent verification of estimates, requires a series of C inventories (MacDicken 1997). Long-term permanent plot data are needed to assess urban forest growth, regeneration and mortality; research needs to develop better urban tree biomass equations, improve estimates of tree decomposition and maintenance emissions, and investigate the effect of urban soils on C storage and flux in cities (Nowak and Crane 2002).

The methodology proposed here is only a starting point for calculating the C stored in an urban forest. To represent a true calculation of the total urban forest C pool, inventories need to be taken of all C sinks in Hunting Creek (as well as in other regions); these pools include mineral soils, underbrush, and dead plant mass such as fine litter and wood (Brown 1997). More research is needed to add to this basic methodology to make it inclusive of all C sinks in urban forests.

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APPENDIX 1. TREE AND MAMMAL SPECIES

Species in Hunting Creek Run Watershed (Source: Bryant et al. 2003)

TREES		**MAMMALS**	
		Canis latrans	*Coyote*
		Didelphis virginiana	*Possum*
Acer spp.	*Maples*	*Eptesicus fuscus*	*Big Brown Bat*
Carya spp.	*Hickories*	*Lasiurus cinereus*	*Hoary Bat*
Fagus grandifolia	*American Beech*	*Myotis lucifugus*	*Little Brown Bat*
Juglans nigra	*Black walnut*	*Odocoileus virginianus*	*White tail deer*
Liriodendron tulipifera	*Yellow poplar*	*Ondatra zibethicus*	*Muskrat*
Quercus spp.	*Oaks*	*Procyon lotor*	*Raccoon*
Ulmus spp.	*Elms*	*Urocyon cinereoargenteus*	*Grey fox*
		Vulpes vulpes	*Red fox*

Hard Maple/Oak/Hickory/Beech Forest Type Species (Source: Jenkins et al. 2003)

Acer	*nigrum*	Black maple	*ilicifolia*	Bear oak
	saccharum	Sugar maple	*imbricaria*	Shingle oak
Carya	*aquatica*	Water hickory	*incana*	Bluejack oak
	cordiformis	Bitternut hickory	*kelloggii*	California black oak
	glabra	Pignut hickory	*laevis*	Turkey oak
	illinoensis	Pecan	*laurifolia*	Laurel oak
	laciniosa	Shellbark hickory	*lobata*	Overcup oak
	ovata	Shagbark hickory	*macrocarpa*	Bur oak
	texana	Black hickory	*marilandica*	Blackjack oak
	tomentosa	Mockernut	*michauxii*	Swamp chestnut oak
		hickory	*muehlenbergii*	Chinkapin oak
Fagus	*grandifolia*	American beech	*nigra*	Water oak
Quercus	*agrifolia*	California live oak	*nuttali*	Nuttall oak
	alba	White oak	*palustris*	Pin oak
	bicolor	Swamp white oak	*phellos*	Willow oak
	chrysolepsis	Canyon live oak	*prinus*	Chestnut oak
	coccinea	Scarlet oak	*rubra*	Northern red oak
	douglasii	Blue oak	*shumardii*	Shumard oak
	durandii	Durand oak	*stellata*	Post oak
	ellipsoidalis	Northern pin oak	*velutina*	Black oak
	engelmannii	Engelmann oak	*virgininiana*	Live oak
	falcata	Southern red oak	*wislizensi*	Interior live oak
	garryana	Oregon white oak		

VITA

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While a student, she worked for the Missouri Department of Natural Resources each summer from 1977 through 1981, before moving to the Washington, DC area. Sara worked for the Information Services Group of the International Bank for Reconstruction and Development (World Bank) from 1985 through 1995. She has also worked as a consultant at the World Bank, and at Aon Risk Services, in Washington. Since 1998 she has been a researcher for the International Finance Corporation, Washington, DC.