United States
Department of
Agriculture

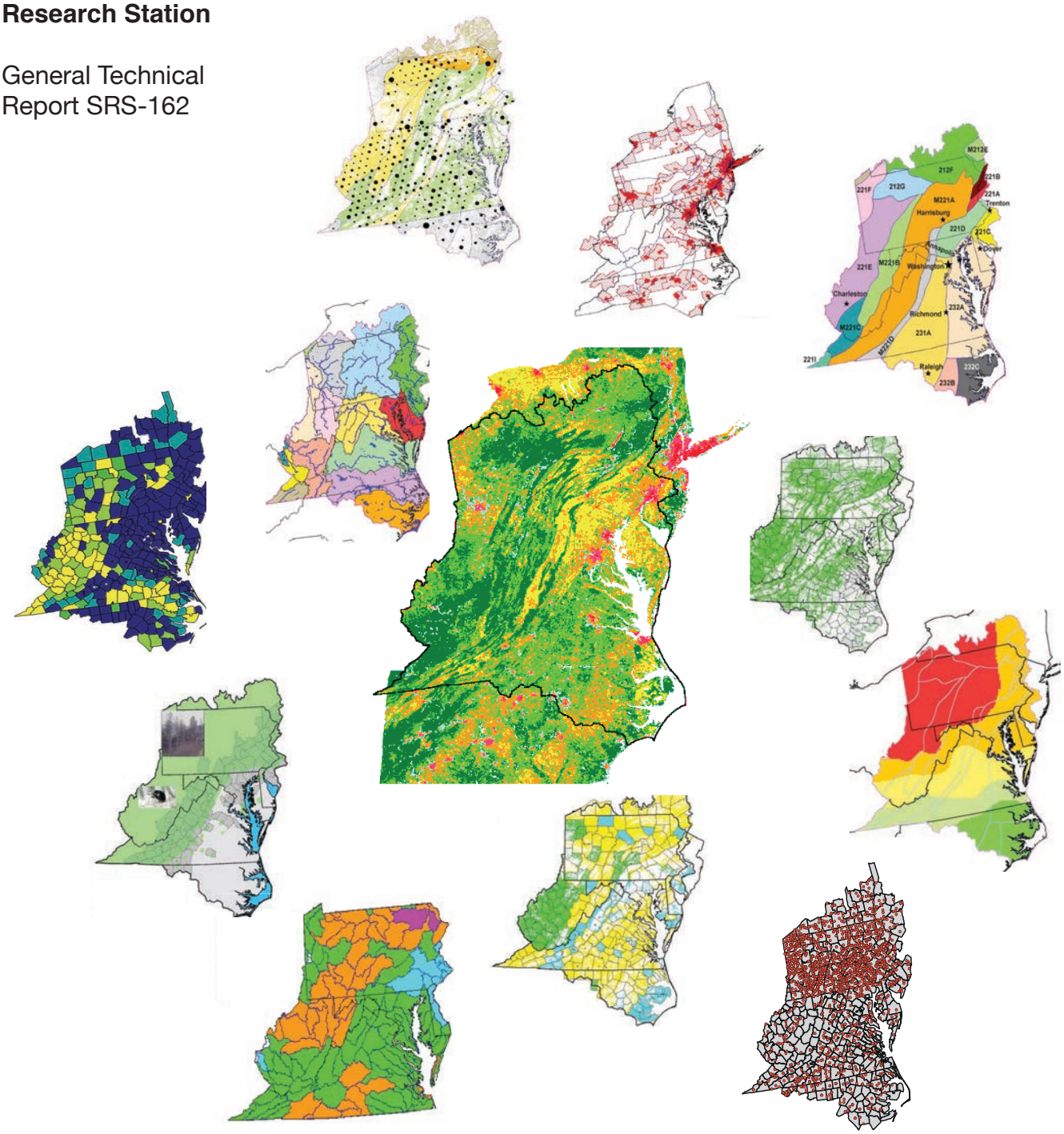
Forest Service

**Southern
Research Station**

General Technical
Report SRS-162

State of Mid-Atlantic Region Forests in 2000

Kenneth W. Stolte, Barbara L. Conkling, Stephanie Fulton and M. Patricia Bradley



The Editors:

Kenneth W. Stolte, Research Ecologist, USDA Forest Service's National Forest Health Monitoring Team, under the Southern Research Station's Eastern Environmental Forests Environmental Threats Assessment Center, Forestry Sciences Laboratory in Research Triangle Park, NC 27709; **Barbara L. Conkling,** Research Assistant Professor, North Carolina State University, Department of Forestry and Environmental Resources, Raleigh NC 27695; **Stephanie Fulton,** Program Support for Surface Coal Mining, Environmental Protection Agency, Region 4, Atlanta GA 30303-8960; **M. Patricia Bradley,** Research, U.S. Environmental Protection Agency, Environmental Effects Research Laboratory, Office of Research and Development, Atlantic Ecology Division, Narragansett, RI 02882.

September 2012

Southern Research Station
200 W. T. Weaver Blvd.
Ashville, NC 28804

www.srs.fs.usda.gov

State of Mid-Atlantic Region Forests in 2000

Kenneth W. Stolte, Barbara L. Conkling, Stephanie Fulton, and M. Patricia Bradley

Acknowledgments

This report was produced under an Interagency Agreement between the U.S. Forest Service (FS), U.S. Department of Agriculture, and the U.S. Environmental Protection Agency (EPA). We gratefully acknowledge the principle architects and engineers of this report, whose direction, initial collaborations, and total dedication made this document possible. Thomas DeMoss[1] and Stanley Laskowski[2] (EPA Region 3), Patricia Bradley[3], Ronald Landy[3], Richard Kutz[4] (EPA Office of Research and Development), Kenneth Stolte[5] and William Burkman[6] (Southern Research Station), and Andrew Gillespie[7] (EPA Cincinnati OH) invested considerable time, energy, and resources to design, execute, and finalize this report.

The thorough and thoughtful edits provided by Paul E. Smith[8] brought continuity and cohesiveness to the final document. Rod Kindlund[8] and Gary Kuhlmann[8] produced the final layout for this report.

Addresses in 2006:
[1]Canaan Valley Institute, Annapolis, Maryland 21401
[2]U.S. Environmental Protection Agency, Region III, 1650 Arch Street, Philadelphia, Pennsylvania 19103
[3]U.S. Environmental Protection Agency, USEPA Environmental Effects Research Laboratory
Atlantic Ecology Division/ORD, 27 Tarzwell Drive, Narragansett, Rhode Island 02882
[4]Consultant in Environmental Science, 4967 Moonfall Way, Columbia, Maryland 21044
[5]USDA Forest Service, Southern Research Station, Forestry Sciences Laboratory, 3041 Cornwallis Road, Research Triangle Park, North Carolina 27709
[6]USDA Forest Service, Southern Research Station, Forest Inventory and Analysis, Knoxville, Tennessee 37919
[7]U.S. Environmental Protection Agency, USEPA Facilities, 26 West Martin Luther King Drive,
Cincinnati, Ohio 45268
[8]USDA Forest Serivce Southern Research Station, 200 W.T. Weaver Boulevard, Asheville, North Carolina 28804

The planners and editors are especially grateful to the numerous authors who contributed the technical data, analyses, and synthesis of information found in this document, and completed numerous revisions to create a useful and informative report. Current addresses are provided for each, and the Table of Contents lists specific contributions.

Mark J. Ambrose[1]
North Carolina State University, USDA Forest Service, Southern Research Station, Forestry Sciences Laboratory, 3041 Cornwallis Road, Research Triangle Park, North Carolina 27709

M. Patricia Bradley[2]
U.S. Environmental Protection Agency, Environmental Effects Research Laboratory
Office of Research and Development, Environmental Science Center, Narragansett, Rhode Island 02882

William G. Burkman[3]
USDA Forest Service, Southern Research Station, Forest Inventory and Analysis, 4700 Old Kingston Pike
Knoxville, Tennessee 37919

Emmanuel Cam[4]
Laboratoire Evolution et Diversité Biologique, Batiment 4R3, 2e Etage Université P. Sabatier - Toulouse III 118, route de Narbonne 31062 TOULOUSE Cedex 04, France.

Barbara L. Conkling[5]
North Carolina State University, USDA Forest Service, Southern Research Station, Forestry Sciences Laboratory, 3041 Cornwallis Road, Research Triangle Park, North Carolina 27709

John Coulston[6]
USDA Forest Service, Southern Research Station, Forest Inventory and Analysis, 4700 Old Kingston Pike, Knoxville, Tennessee 37919

Robert Eaton[7]
USDA Forest Service, Southern Research Station, Forestry Sciences Laboratory, 3041 Cornwallis Road, Research Triangle Park, North Carolina 27709

Stephanie Fulton[8]
U.S. Environmental Protection Agency, Region IV, Atlanta, Georgia 30303

Laura E. Jackson[9]
U.S. Environmental Protection Agency, Office of Research and Development, B205-01, Research Triangle Park, North Carolina 27711

Evan Mercer[10]
USDA Forest Service, Southern Research Station, Forestry Sciences Laboratory, 3041 Cornwallis Road, Research Triangle Park, North Carolina 27709

A. Murthy[11]
Manager Economist, US-Global Financial Services, American Express, 1691 W. Winchester Way, Chandler, Arizona 85248

William D. Smith[12]
USDA Forest Service, Southern Research Station, Forestry Sciences Laboratory, 3041 Cornwallis Road, Research Triangle Park, North Carolina 27709

James Steinman[13]
USDA Forest Service, Northern Research Station, 11 Campus blvd., Suite 200, Newtown Square, Pennsylvania 19073

Kenneth W. Stolte[14]
USDA Forest Service, Southern Research Station, Forestry Sciences Laboratory, 3041 Cornwallis Road, Research Triangle Park, North Carolina 27709

Kent Thornton[15]
FTN Associates, 3 Innwood Circle, Suite 220, Little Rock, Arkansas 72211

Daniel Twardus[16]
USDA Forest Service, Forest Health Protection, Morgantown, West Virginia 26501

Henry A. Walker[17]
U.S. Environmental Protection Agency, Office of Research and Development, 27 Tarzwell Drive, Narragansett, Rhode Island 02882

Edward Yockey[18]
USDA Forest Service, Region 8, Forest Health Protection, 200 W.T. Weaver Boulevard,
Asheville, North Carolina 28804

Preface

This report is a comprehensive overview of the condition, use, and forces affecting the forests of the MAIA region through the year 2000. In some cases the data presented was the most recent possible through 2000 because of the differences between data collection methods in MAIA States. This was particularly true for much of the tree productivity data that was collected in different FIA inventories in different States and summarized in the 1992 Eastwide Database. In 2000 FIA began a nationally-standardized method of data collection using annual inventories, and the results of these improved, standardized timber inventories have only recently become available. Additionally there is a much greater coverage, and hence forest health data, in the MAIA region by the FIA program then was available in 2000. Since the report is based primarily on standardized data from state and national long-term monitoring programs that continue today, many issues in this report can now be updated and determinations of changes over time made possible. Thus information on forest productivity and economics, air pollution, insects and pathogens, tree and soil condition, urbanization and land use, game species, and other topics addressed in this report come from programs that are standardized and have continued to add new data after 2000. This does in no way negate the value of the information presented, but rather highlights the value as an excellent template for any new assessments of the forests in this region. Any new assessment should consider that a number of the databases are best aggregated and summarized at decade intervals to average the yearly variations that occur among some state-level data.

This publication presents information from a far-reaching study of the forest resources and surrounding lands of our Nation's Capital, and includes all States or parts of States whose watersheds feed into the Chesapeake Bay. A widely varied group of research scientists have studied the mid-Atlantic region and compiled information that is urgently needed today. We believe the reader will find this volume useful not only because it brings comprehensive information that may enhance the lives of millions of citizens who depend on healthy, sustainable forest ecosystems in the States surrounding the Chesapeake, but also because many decision-makers from a host of jurisdictions are facing challenges that require sound information to make decisions in the coming decades.

Data availability from the beginning varied considerably. Many datasets are from the late 1990s, while some of the most current datasets available for some topics were compiled from data in the 1970s, 1980s, and 1990s. In order to present the most accurate information and to fully cite the sources from which it was gathered, we have used a few research results published since 2000, but only as those results relate to resource conditions prior to the end of the period that the researchers and authors were assigned to assess.

This report was a joint effort of the U.S. Environmental Protection Agency (EPA) Office of Research and Development (ORD) in Region III, and the U.S. Department

of Agriculture (USDA) Forest Service Research program, the venture involved Region 3 of the EPA-ORD's Mid-Atlantic Integrated Assessment (MAIA) program and the Forest Health Monitoring (FHM) and Forest Inventory and Analysis (FIA) programs of the USDA Forest Service. All groups contributed considerable time, effort and mutual support to this project: in some cases contacting authors from other agencies to address specific topics; by conducting stakeholder workshops to ensure that local, practical, and relevant data were integrated into the report; and by conducting sufficient peer reviews, necessary revisions and, ultimately, publication of far-ranging and important information of the forest resources in the MAIA region.

CONTENTS

State of Mid-Atlantic Region Forests in 2000

Kenneth W. Stolte, Barbara L. Conkling, Stephanie Fulton and M. Patricia Bradley

ABSTRACT

Wet and warm climate, mountainous topography, and deep rich soils produced one of the most magnificent and diverse temperate forests in the world. In 1650 the Mid-Atlantic forests covered 95 percent of the region, but were greatly reduced in 1900 by extensive tree harvesting, and conversion to farms and pastures. Settlement of forests also led to severe wildfires, soil erosion, and destruction of wildlife. Recovery began in the early 1900s, and later improvements in agricultural allowed millions of acres to return to forest cover. Suppression of catastrophic wildfires reduced flooding and watershed degradation, and wildlife management returned native animal and fish populations. Forest management improvements led again to productive and diverse forests in more mature stages of development. By the end of the 20th century, the Mid-Atlantic forests covered 61 percent of the land area and produced numerous products that brought social and economic benefits to people. Continuing pressures from urbanization and fragmentation; selective species harvests; air pollution; exotic invasive species; wildlife habitat loss; historic fire regime changes; stream degradation; and climate change still affect and threaten these forests, and require enlightened management and policy decisions to ensure sustainability of healthy, diverse, and productive forests.

Keywords: forest health, forest economics, indicators, stressors, sustainability, Mid-Atlantic forests, Montreal Process Criteria and Indicators

SUMMARY

The forest in the Mid-Atlantic Integrated Assessment (MAIA) region have been highly productive and diverse since the glaciers retreated from lower North America about 10,000 BCE. Long term fluctuations in climate have altered the composition and abundance of tree species through the ages. At the beginning of the 17th century the forests were in various successional stages, often the result of natural or intentional burning to open up the understory. The mature forests were magnificent with oaks, tulip poplars, chestnuts, hickories, and other species reaching large stature and heights.

The colonization of the MAIA region started in the mid-1600s, and accelerated greatly in the 18th and 19th centuries. Like many newly colonized areas, the local forests were used for building, fuel, food, fencing, and transportation. The extent and degree of this use was overwhelming, and by the end of the 19th century the forests were largely depleted of any large trees, soils were eroding and silting rivers and harbors, wildlife was hunted to the point of extinction, and catastrophic wildfires were followed by massive flooding and mud flows.

The restoration of these forest lands began in the early 1900s with the establishment of land managing agencies created to stop the ruinous exploitation of the forests and begin more sustainable land management practices. Logging and landuse methods were improved, most wildlife protected enough to recover, improvements in agriculture, shifting from wood to oil as the primary energy source, and improvements in preservation of wood for railroads and other uses resulted in less forestland being needed to sustain a rapidly growing population. Needing less land for agriculture, shifting to oil as a primary source of energy, improved silvicultural methods and tree plantation, and the invention of the horseless carriage were major factors that reduced the enormous pressures on forests in the MAIA region.

By the end of the 20th century the forests had greatly improved. Abandoned agriculture and pasturelands became reforested, and forests in the western MAIA region were able to mature into remnants of pre-European forests. But stresses that had decimated the forests for over 200 hundred years were replaced with new stresses inherent in new technology and a swelling population. The deliberate or unintentional importation of non-native insects and pathogens greatly changed the forest landscape, particularly the loss of American chestnut due to the chestnut blight.

Air pollution from automobiles and industry created acidic deposition that was particularly harmful to forests in mountainous regions with shallow, poorly-buffered soils, and tropospheric ozone reduced the growth and vigor of plants susceptible to this strong oxidant. Urbanization of the coastal areas greatly reduced and fragmented the recovering forests, and pushed many larger wildlife species into the western half of the region. The large rampant fires that consumed eastern forests in the 19th century were so suppressed in the 20th century that many forests are now out of sync with normal fire regimes that causes many problems with natural succession processes, and sets a stage for catastrophic crown fires. Non-native vegetation and animals displace native species, and an increasing number of non-native insects and pathogens seriously threaten some keystone tree species like hemlock. Exacerbated populations of native insects and fauna are expanding beyond historical ranges, and some have exceeded the carrying capacity of the forests. The impending specter of climate change looms on the horizon.

Despite the effects of serious stressors, the MAIA region forests are highly productive, diverse, and resilient. Although the last 300 years of intense use have reduced the amount and diversity of forests in the region, some of the

remaining fragments are beginning to return to the former grandeur found in the early 1600s. Much of the western MAIA region is still highly forested, and some areas in western MAIA region are maturing ecologically (greater tree size, density, volume) and becoming more stable than any time in the last 300 years. Yet the forests will take another 100 years or more of ecological maturation to approach the magnificent forests found in the early 1600s.

The inherent tree-growing qualities of the MAIA region are seen in the high diversity of forest types, tree genera, and tree species. The annual growth rates and stocking in undisturbed forests, and the high per capita forest cover in the urban and metropolitan areas of the largest population densities, show these forests are still robust. The headwaters of the MAIA fresh-water systems (first and second order streams) appear to be minimally impacted at the present, although other threats like mountain top mining, not addressed in this report, are threatening these headwaters. It is encouraging that in the face of ever-increasing populations, enlightened management can protect the headwaters of this region and increase the probability of quality water supplies for the future.

INTRODUCTION

Chapter 1.

Forests and People in the Mid-Atlantic Region

Stephanie Fulton, Evan Mercer, and M.Patricia Bradley
Region 4, U.S. Environmental Protection Agency;
Southern Research Station, USDA Forest Service;
Region 3, U.S. Environmental Protection Agency

Human populations in the Mid-Atlantic region over the last 250 years have increased nearly 100-fold, from an estimated few hundred thousand people to over 30 million people (Mercer and Murthy 2000). Increased population growth usually results in the conversion of forestland to nonforest uses, particularly agriculture, pastureland, and urban development. Not only is the quantity of forestland reduced, but forest habitat quality also suffers as harvesting and forest conversion patterns chop the forest landscape into smaller, more isolated patches of forest—a process referred to as fragmentation.

Increased urbanization since European settlement significantly affected the historic extent, distribution, and composition of forests in the Mid Atlantic Integrated Assessment (MAIA) region. Prior to colonization, this multi-State area was a near-continuous cover of forest. During early periods of rapid population growth and colonial expansion, it was almost completely deforested by timber harvests that cleared the land for homes, cities, commercial agriculture, fuel, pastures, fences, railroads, and other uses. Due to improvements in agricultural techniques over the last century, much of the region has recovered from early agricultural land clearing, and forests once again have been dominating the landscape. Yet only isolated fragments of pre-settlement forests have remained. Over time the landscape came to represent a legacy of almost 400 years of settlement and intense use that made the eastern MAIA region a mosaic of forest patches of varying sizes and shapes found within a matrix of mixed urban, agriculture, and pasture land uses.

Both the rate and pattern of forest conversion and forest fragmentation were strongly influenced by regional socioeconomic patterns of land ownership, land use, and economic trade. Although occupying a relatively small portion of the watersheds' total land area, the Mid-Atlantic region's highly urbanized and growing population, coupled with diverse land use and ownership patterns, continued to threaten forest resources. Urban sprawl, with its associated impacts on forest resources, remains one of the region's most pressing forest health issues.

The purpose of this report is to describe the forested landscape based on data available through the year 2000, to show how land-use change and other stressors had affected the Mid-Atlantic region's forests. The large size of the region sometimes prohibited structuring any one complete data set that describes forest characteristics for the region as a whole. Therefore, we sometimes used different data sets to compile various landscape-scale indicators of forest health and sustainability. Our purpose was to quantify the impacts of land use change on critical forest ecosystem components and processes, and highlight where future risk to ecological stability may be the greatest.

Ancient Forests in the Mid-Atlantic Region

Prior to colonists settling in the region, Eastern forests were estimated to have covered 95 percent of the land surface. Fossil pollen records from the last ice age and beyond indicate that ancient forests in the region were composed primarily of spruce, pine, and fir, with some birch and alder, indicative of a cooler boreal climate, about 3º F to 8º F cooler than late in the last century (Brush 1986). Approximately 10,000 years ago temperatures rose, and oaks became more abundant, quickly followed by increased numbers of hemlock and hickory. Forests of the MAIA region 5,000 years ago resembled present-day forests in species composition, but abundances of species have fluctuated in response to climatic variations. Between 2750 BCE and 1450 BCE, forests were characterized by black gum and sweet gum, components of a wetter climate. Abundances of these species were greatly decreased after 1450 BCE. By 400 CE, holly, chestnut, and ericaceous shrubs, indicative of drier climates, dominated the landscape and remained dominant until European settlement approximately 1200 years later (Brush 1986).

Pre-European Settlement—Forest Extent and Composition

Tree species in relative dominance patterns commonly found together are referred to as forest types. According to Küchler (1964), the primary potential natural vegetation types in the Middle Atlantic Coastal Plain and Southeastern Plains pre-settlement were oak–hickory–pine forest (beech, sweet gum, magnolia, pine, and oak). In the southern flood plain forest (Omernik 1977; McNab and Avers 1994) and in the Piedmont areas oak–hickory–pine forest and southern mixed forest were found. Orwig and Abrams (1994) found that pre-settlement vegetation (prior to 1721) in Piedmont and Coastal Plains forests was a mixture of oak, primarily red and white oak, although black oak was of minor importance as were hickory species.

The Northern Piedmont region has been classified by Küchler (1964) as oak–hickory and by Braun (1950) as oak–chestnut. Before the early 17th century, native vegetation was composed mainly of oak and hickory; chestnut, yellow poplar, ash, walnut, and elm were associated species; and maple was dominant on wet bottomlands of the Piedmont (Loeb 1987).

Küchler (1964) described vegetation types in the North Central Appalachians as northern hardwoods forest (alternately referred to as maple-beech-birch forests) and Appalachian oak forest. Whitney (1990) wrote that the region lies in Braun's (1950) hemlock–white pine–northern hardwood region and Küchler's (1964) hemlock–northern hardwood forest type, and included extensions of more Southern Appalachian oak forests up into major river valleys of the region. Early surveys suggested that vast forests of white pine and hemlock once covered higher portions of the plateau in northern Pennsylvania (Sargent 1884; Whitney 1990). Early survey notes (1814 to 1815) for the region indicated a preponderance of beech and hemlock (> 60 percent) as witness trees—trees to which distances and azimuths from a particular point on the ground or object were recorded—and are distributed widely throughout the High Plateau, though neither species was common off the plateau (Whitney 1990). Sugar maple was associated with the flat top of the Plateau, birch occupied moister soils of foot slopes, and oak species preferentially occupied drier upper-slope positions.

The Blue Ridge Mountains region was classified by Küchler (1964) as a mixture of Appalachian oak, southeastern spruce-fir, and northern hardwoods forests. The Central Appalachian Ridge and Valleys regions were mapped by Küchler as Appalachian oak forest, oak–hickory–pine forest, and some northern hardwoods forest. Braun (1950) classified much of the area as oak-chestnut, but later revised the area to oak–hickory or mixed-oak forest (Nowacki and Abrams 1992). American chestnut was common in the region until chestnut blight decimated the species in the 1930s (Anagnostakis 1995, Brush 1986; Schlarbaum and others 1997). Nowacki and Abrams (1992) identified four forest types common in the region prior to European settlement (all dominated by oak species): (1) sweet birch–chestnut oak–northern red oak; (2) chestnut oak–northern red oak on forested ridges; (3) mixed oak species on mesic valley floors and transitional ridge/valley zones; and (4) white oak on gently sloping, low elevation sites. Oak species, pine species, and American chestnut were common on ridges; eastern hemlock increased while chestnut oak decreased in coves; and oak–pine–hickory (dominated by white oak) species dominated valley floors (Nowacki and Abrams 1992).

The Central Appalachians were mapped as northeastern spruce-fir, northern hardwoods, mixed-mesophytic Appalachian oak, and oak–hickory–pine forests. The Western Allegheny Plateau included beech–maple, Appalachian oak, northern hardwood, mixed mesophytic forests, and a small amount of oak–hickory types (Küchler 1964).

Post-European Settlement and Land Use

One of the most explosive periods of growth in the Chesapeake Bay region occurred between the initial colonization and the Revolutionary War. Early settlements were concentrated in the "Tidewater" areas of the Chesapeake Bay during the 17th century (Miller 1986), where colonists lived on isolated plantations scattered along creeks and river tributaries of the Chesapeake Bay. According to Miller (1986), early settlers showed a preference for waterfront property: Ninety-seven percent of 17th century archaeological sites were found within one mile of the water (75 percent were within 1,000 feet of the shore). Miller (1986) suggested this preference was a result of: (1) readily available land, (2) a primarily agricultural economy, (3) a market system dependent upon water transportation, and (4) a desire to live near water for easier travel and exploitation of estuarine resources.

Soil erosion was minimal during the 17th and early 18th centuries, due to the extensive use of slash-and-burn agricultural practices taught to the settlers by Native Americans (Miller 1986). Cleared forest was farmed using axe-and-hoe planting, but after 6 to 8 years the soil was exhausted and the land was abandoned to reforestation. After laying fallow for 20 years or so, the land was brought back into production. These practices created a patchwork of agricultural clearings buffered by riparian vegetation along streams, and interspersed with patches of forest in various stages of reforestation. While only a small amount of land was planted, a large area of land was needed to sustain the system over time.

Brush (1986) compared sediment accumulation rates before and after European settlement and estimated the extent and rates of forest clearing. Sedimentation rates were primarily controlled by climatic events (storms) and anthropogenic activity (land clearing, intensive agriculture). Brush (1984a, 1984b; 1986) found that although sediment accumulation

rates were highly variable before European settlement, they were consistently higher after. Sedimentation rates were greatest during periods (circa 1830 to 1930) of commercial agriculture (Cooper 1995), and did not begin to decrease until the late 1930s when soil conservation practices began.

During this early period of rapid growth and expansion, the Mid-Atlantic region experienced near-complete deforestation. Rapidly increasing populations, coupled with changing agricultural practices and the settlers' land tenure, soon led to the demise of the slash-and-burn techniques of the early colonial agricultural system (Miller 1986). For example, Maryland, established as a State in 1634, had 34,000 colonists by 1700, 100,000 by 1740, and 300,000 by the end of the colonial period. Population densities around Annapolis, MD, increased from 18 people per square mile in 1705, to 42 people per square mile at the beginning of the Revolutionary War. Land in southern Maryland under agricultural production rose from 2 percent in 1720 to nearly 40 percent in the early 1800s (Miller 1986). Planters essentially ran out of space to continue the long-term slash-and-burn, then lie-fallow system. In addition, land tenure changed from long-term leases at low annual rents to short-term leases at high annual rents.

Agriculture was widespread throughout the region by 1760 (Cooper 1995). All of the Tidewater and most of the Piedmont region of Maryland and Virginia were occupied, or in the process of being settled by the end of the Revolutionary War. Planters stopped rotating-cultivation, and shifted to intensive plow agriculture associated with grain production. This change in farm practice was hastened by instability of grain and tobacco markets after the Revolution, and became widespread in the Tidewater area during the last quarter of the 17th century. Development continued to expand into interior and Piedmont regions during the 18th century, and soil erosion and sediment runoff from the hilly Piedmont lands increased dramatically. The city of Baltimore's port had to be dredged regularly by 1780, and by 1807 small ports along navigable rivers draining into Chesapeake Bay had to be abandoned because of silt.

Increased Chenopodiaceous pollen (from plants that grow in marshes and in newly cleared fields) (Brush 1986) and the highest average sedimentation rates recorded (Cooper 1995) indicate landscape disturbance from agricultural was most intense from about 1830 to 1930. Not coincidentally, the population growth rate was also at its peak in the mid-to-late 19th century. According to census data, 40 to 50 percent of the land had been cleared for agriculture by 1840 (Cooper 1995), and by the end of the 19th century, land cleared for agriculture had increased to 80 percent (Brush 1986).

Chapter 2.

The Mid-Atlantic Integrated Assessment (MAIA) Region and Report

Kenneth W Stolte,[1] Stephanie Fulton,[2] and Evan Mercer[1]
[1]Southern Research Station, USDA Forest Service
[2]Region 4, U.S. Environmental Protection Agency

As part of the U.S. commitment to sustainable forest management and conservation principles, the USDA Forest Service (FS) and the Environmental Protection Agency (EPA) joined (circa 1997) in a collaborative effort to evaluate the condition of forests in the MAIA region. The MAIA project—one of several large-scale regional ecosystem-health assessments by EPA in Region III (http://www.epa.gov/epahome/aboutepa.htm)—was designed to: (a) balance ecological, economic, and social concerns; (b) build partnerships and involve stakeholders in the identification of problems, setting goals, and developing solutions that were the key phases of the assessment; and (c) use partnership and stakeholder interests to integrate actions by Federal, State, Tribal, and local agencies; between government and private enterprises; and between government and local communities. The MAIA program was organized along the lines of the National Environmental Monitoring Initiative led by the Committee on the Environment and Natural Resources (http://www.gcrio.org/USGCRP/CENR/) in the White House Office of Science and Technology. The MAIA project focused on policy and management issues of critical importance to resource managers and environmental decision-makers in the region.

The MAIA region encompassed the major watersheds within an eight-State region, which included the greater Chesapeake Bay watershed. The region included five States in their entirety: Delaware (3 counties), Maryland (23 counties), Pennsylvania (66 counties), Virginia (95 counties), and West Virginia (55 counties), and portions of the States of New Jersey (all or part of 12 counties), New York (all or part of 25 counties) and North Carolina (all or part of 47 counties) (fig. 1).

A highly diverse landscape characterized the Mid-Atlantic region. Ecoregion sections (Bailey 1995) were used to give an overview of the Mid-Atlantic region. Beginning in the east is the low-lying, flat Coastal Plain that contains the Chesapeake Bay, the Nation's largest estuary, an area characterized by dune fields, beaches, lagoons, and barrier islands. Farther inland are irregular or smooth plains and

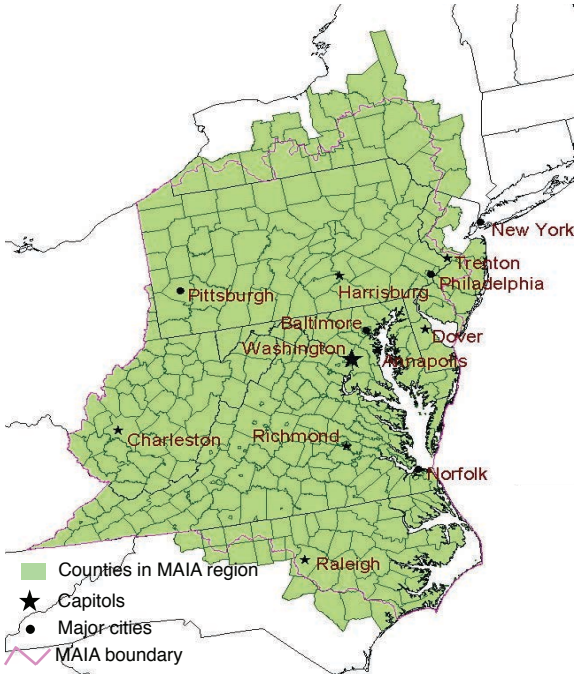


Figure 1—The Mid-Atlantic Integrated Assessment (MAIA) region includes all of Maryland, Virginia, West Virginia, Pennsylvania, Delaware, District of Columbia, and whole or partial counties in northeast North Carolina (~47), western New Jersey (~12), and southeastern New York (~25).Source: Environmental Protection Agency Region III; (http://www.epa.gov/region03/index.htm).

open hills. The soils were either well drained (inland) or poorly-drained (coastal) because of the higher water tables found near estuaries and bays.

Moving west from the coast we first encounter the low, rolling hills of the Piedmont, where a moderate density of perennial streams and associated rivers are found. Soils in the Piedmont tend to be well-drained and moderately-deep to very-deep. Further west are the higher Blue Ridge Mountains, a region with a high density of perennial streams and associated rivers. The soils are moderately-deep and medium-textured. Beyond the Blue Ridge Mountains are the Appalachian Mountains, which constitute a region characterized by narrow valleys and high ridges. The eastern boundary of the Appalachian Mountains is the Great Valley Lowland (Shenandoah Valley), and the western boundary is the steep, high ridge of the Allegheny Front. The Allegheny Mountains and the Allegheny Plateau are dominant features in the most western parts of the Mid-Atlantic region. The Allegheny Mountains are characterized by high, sharp ridges

and narrow valleys. Streams there are generally more acidic than those in the Appalachian Mountains, and soils are similarly classified.

The Allegheny Plateau contains both glaciated and unglaciated areas. In the north, glaciated areas are marked by broadly rolling high hills and steep valleys; in the south, the valleys are broad with round hills. Glaciated parts of the region have small natural lakes as well as perennial streams. Unglaciated portions of the region tend to have sharper ridge tops and narrower valleys than the glaciated portions, with many rapidly moving streams and rivers.

Major cities have risen along the larger rivers of both the northeastern and southeastern parts of the Mid-Atlantic region, including: Philadelphia, PA, on the Delaware River; Harrisburg, PA, on the Susquehanna River; Pittsburgh, PA, on the Ohio, Allegheny, and Monongahela Rivers; Baltimore, MD, on the Patapsaco River; Washington, DC, on the Potomac River; Richmond, VA, on the James River; and Norfolk, VA, on the Chesapeake Bay (fig. 1).

The Forest Health Monitoring Program

During the late 1980s, the EPA's Office of Research and Development began planning a multi-resource, interagency, cooperative program called the Environmental Monitoring and Assessment Program (EMAP). Forested lands, irrespective of ownership, were identified as a resource of importance and a special effort was made to develop the EMAP-Forests program in cooperation with similar efforts already underway by the USDA Forest Service, the National Vegetation Survey, a component of the first National Acid Precipitation Assessment Program (gcmd.nasa.gov/records/GCMD_EPA0141.html). Additional key collaboration was the involvement of State forestry and agriculture agencies and the National Association of State Foresters (http://www.stateforesters.org/). The resulting assembly was a multi-agency, cooperative program primarily managed by the USDA Forest Service and the EPA's EMAP-Forests, with essential support, guidance, and assistance provided by State agencies.

The Forest Health Monitoring (FHM) Program (www.fhm.fs.fed.us) is composed of four interrelated activities (Stolte 1997):

- The Detection Monitoring (DM) component constituted a nationally standardized network of permanent, fixed-area ground plots (1 plot per 160,000 acres) where each year measurements of key environmental indicators are taken. Aerial and ground surveys of damage or mortality from insects, pathogens, and other stressors were made independently from the plot network. Satellite monitoring provided expanded coverage of changes in forest extent and fragmentation, indicated how forests related spatially to other terrestrial and aquatic ecosystems, and in conjunction with aerial mapping it was used to expand plot-level data to population-levels. In 2000, the plot component of DM was integrated with the national ground monitoring system of the USDA Forest Service's Forest Inventory and Analysis (FIA) program, and the FHM-EMAP grid was intensified to 1 plot per 6,000 acres for tree-based evaluations, and 1 plot per 96,000 acres for trees and a broader suite of ecological indicators.
- Evaluation Monitoring (EM) examined the extent, severity, and probable causes of undesirable changes in forest condition that could not be determined by DM. Techniques included intensifying ground plot monitoring, taking additional plot-based measurements, ground surveys, aerial and satellite monitoring, experimental studies, as well as more detailed and innovative data analyses.
- Intensive Site Monitoring (ISM) identified key components and processes of forest ecosystems, how these components and processes were integrated, and relationships of these components and processes to the environmental indicators in DM. It facilitated determinations of expected consequences of perturbations, improved interpretation of DM and EM data, and facilitated the development of risk assessments.
- Research on Monitoring Techniques (ROMT) was research specifically directed to improve DM, EM, and ISM monitoring activities. ROMT activities included developing new indicators, improving existing indicators, development of new analytical techniques, and other fine-tuning of monitoring activities.

The health of forest ecosystems can be evaluated using measurements of key ecosystem processes and components with known variability, understanding how stressors affect processes and components, and how forest ecosystems are changed because key components and processes are changed. The Montreal Process Criteria and Indicators (MPCI) (Anon. 1995a) were developed from a series of international meetings and workshops of scientists, land managers, and policy makers from 12 nations representing over 90 percent of the world's temperate and boreal forests. These meetings resulted in the Santiago Declaration (Anon. 1995b), a document that listed a set of basic ecological, socioeconomic, and institutional Criteria and Indicators to evaluate forest health and sustainability. To evaluate

the condition of the forests in the MAIA region, the FHM program developed an initial set of assessment issues based on this common set of ecological and socio-economic criteria and indicators (table 1).

Purpose of the MAIA Report

A MAIA steering team, composed of representatives from USFS, EPA, and other groups, was formed and emphasized participation by interested individuals, organizations, and agencies in obtaining information and writing the report. Subsequently this steering team identified and organized a group of stakeholders that constituted approximately 100 members representing various Federal and State agencies and organizations, universities, industry, environmental interest groups, public, and other groups in the MAIA region. A 2-day workshop was held in April 1997 to gather and share information needs, available data sources, and useful ways of reporting information. As part of the workshop, stakeholders were asked to prioritize a list of initial assessment issues—both ecological and socioeconomic—and to provide any additional input about the most important topics to address in a report on the forests of the MAIA region (table 1).

The first issues presented to the stakeholders were based on five ecological Criteria from the Santiago Declaration–Productivity (Criterion 1), Biological Diversity (Criterion 2), Vitality (Criterion 3), Conservation of Soil and Water (Criterion 4), and forests contribution to global Carbon Cycling (Criterion 5) (Anon. 1995b). The stakeholders identified a need to expand on these issues and provide more detailed analyses in three additional areas: landscape ecology, wildlife, and aquatic systems. They also recommended that ecological criteria and indicators for the MAIA assessment be translated into broader themes of Forest Stressors (biotic and abiotic), Forest Responses, and Forest Condition—terms more compatible with themes used by other resource groups working in the MAIA region. The FHM-MAIA team reorganized the MPCI ecological criteria into Forest Stressors (Criterion 3, Indicators 1 and 2), Forest Responses (Criterion 3, Indicator 3) and Forest Condition (Criterion 2-Productivity; Criterion 4-Conservation of Soil and Water (Aquatic) Systems; Criterion 5-Carbon Cycling; and Criterion 1- Biological Diversity) (table 1).

The team developed ecological and socioeconomic assessment questions around these themes to guide data collection, analyses, interpretation, and reporting. A numerical scale (0 to 10, with 10 highest) was used to quantify the relative importance of each issue to the stakeholders. The results are presented in table 1.

Three expert teams were organized to address data gaps identified in the initial stakeholder meetings, which included landscape ecology, wildlife, and aquatic systems. A 1-day workshop was held in July 1999, where team leaders distributed the first full draft outline of the technical support document for review by a cross-section of the larger stakeholder group. Members agreed upon four levels of reports, and reached a consensus to focus on publication of the technical support document first. The four types of reports about forested lands in the MAIA region initially envisioned are:

- Peer-reviewed scientific papers and technical manuscripts, prepared by individual topic authors, describing the methodology and detailed results on each major report theme.
- Geo-referenced, quality-assured data available on a Website.
- A public report for lay readers and senior administrators similar in format to the other public reports of other MAIA resource groups.
- A technical support document that synthesizes information in the scientific papers and manuscripts and supported the public document.

This Report is the technical support document that presents all the information that has been collected and will be the basis for any public report that may be prepared hereafter.

Assessing Health and Sustainability of Forested Ecosystems

Increasing world populations threaten forest ecosystems on almost every continent. Humankind has created tools that enable us to exert tremendous pressures on natural ecosystems, to the extent that ecosystems can be completely changed, and sometimes lost forever. As a result, we must address issues of forest health and sustainability because we are the biggest collective threat, and our policies and management practices are the only solution.

Scientists will never completely agree on what constitutes "healthy and sustainable forests," what should be measured, and how to analyze and interpret data. Yet sound ecological management of forest resources will be essential, if we are to maintain healthy and sustainable forest ecosystems that are vital to the present and future social, economic, and spiritual needs of all nations. Biological sustainability also should be considered in the context of socio-economic uses that have shaped forest ecosystems into what they are today. To maintain forests in some desirable condition, it is necessary to balance desirable ecological attributes with

Table 1—Rank of relative Importance of forest health stressors, response, and condition topics from two MAIA Stakeholder Workshops in Annapolis, Maryland in 1997

Category	Component	Assessment issue[a]	Stakeholder priority[b]
Stressors	Abiotic	*Air pollution*	10.0
		Storms (hurricanes, ice)	6.3
		Fire (lightning strikes)	6.3
	Biotic	*Insects and diseases*	9.5
		Animal damage	
		Exotic plants and animals	8.3
	Land use	*Urban expansion*	
		Fire[e] and *fire suppression*	6.3
		Mining activities	
		Timber harvest	7.6
		Road building	6.3
Response[c]	Tree vitality	*Crown dieback*	5.9
		Tree damage	6.3
		Tree mortality	8.5
		Tree regeneration	10.0
Condition[d]	Productivity	*Timber productivity*	7.6
		Non-timber productivity	5.0
		Game species productivity	7.5
	Soils systems	Erosion	8.3
		Accumulation of toxins	8.8
		Nutrient pools and cycling	8.3
		Compaction	5.5
	Aquatic systems	*Sedimentation*	10.0
		Chemical Contamination	10.0
		Riparian buffers	
	Carbon sequestration	*Soil carbon*	8.3
		Above-ground trees	
		Above-ground plants	
	Biological diversity	*Plant species richness*	7.5
		Non-game wildlife species richness	9.0
		Forest birds species richness	9.0
		Habitat suitability for T & E species	9.0

[a] Issues in italics were addressed in MAIA report.
[b] Based on average of low(3), medium (6), and high (10) ratings.
[c] Addressed Criterion 3 of Montreal Process Criteria and Indicators (Anon. 1995b).
[d] Addressed Criteria 1, 2, 4, and 5 of Montreal Process Criteria and Indicators (Anon. 1995b).
[e] Human-caused ignitions.

desired socioeconomic use. Policy and management strategies to maintain healthy and sustainable forest ecosystems must include inventory and monitoring programs to evaluate current condition, change over time, and causal agents of change.

Establishment and maintenance of stable, healthy, and sustainable forest ecosystems does not imply static, unchanging conditions or the absence of ecological disturbances. Forest ecosystems have co-evolved with cyclic natural disturbance patterns (e.g., fires, severe storms, and native insects and pathogens) and natural forces (e.g., climate, geologic, and topographic changes). Natural disturbance patterns have maintained these systems within relatively stable, long-term patterns of endless cycles of seral development and succession. As a result of such pattern maintenance a forest may evolve into a climax state, where the major tree species composition is self-replicating, and no further major changes in tree species composition occur without a major disturbance event (e.g., severe fires, clear-cut harvests, etc.).

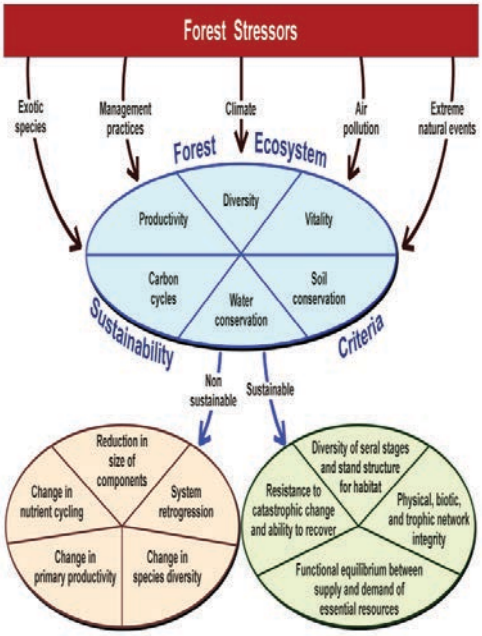


Figure 2—Conceptual model of relationships among stressors, forest ecosystems, and healthy and sustainable forests. New stressors, or exacerbated endemic stressors (top bar), interact with forest ecosystem components and processes (middle oval) in one or more primary functional areas organized by ecological criteria of the Montreal Process Criteria and Indicators. Forest ecosystems respond to new stressors in a sustainable or non-sustainable manner (bottom two ovals) depending on above factors. Source: Stolte 1997.

Forest ecosystems with little or no anthropogenic disturbances can be used to evaluate responses of healthy and sustainable forests to natural forces and disturbances. The relationships among natural causes and effects can then be used to determine if human disturbances on similar forest ecosystems are affecting such systems. Thus if indicators of forest condition in human-disturbed forests are similar to those found in a healthy, undisturbed *reference* forest system in a similar ecological unit, the human activities observed have a low probability of negatively affecting the health and sustainability of that forest system.

Our conceptual model to evaluate forest health and sustainability considers a variety of natural forces and human-induced stressors, the inherent vulnerability of different forest ecosystems to stressors, and whether single or combined stressors acting on a forest system are causing measurable and significant changes to a forest that threaten long-term health and sustainability (fig. 2).

If the stressor(s) are of a sufficient magnitude (type, severity, and duration) and a forest system is sufficiently vulnerable, then indicators of forest condition, aggregated according to the MPCI, may show changes that suggest a movement towards a non-sustainable condition. Alternately, a forest ecosystem may not seem to be affected by human-induced stressors, and the system remains within historic or reference variance patterns, and is continuing in a healthy and sustainable direction.

A healthy and sustainable forest ecosystem would have the following attributes and the quality of those attributes can be determined by comparisons with historic or relevant reference conditions:

- large and not overly fragmented
- biologically and structurally diverse
- productive for wood, fruits, and other commodities
- low air, ground, and water pollution
- balanced distribution of age classes
- free from exotic, invasive species
- adapted to endemic forces (native insects and pathogens; storms; droughts,)
- expected soil chemistry, physical structure, and biological components
- expected water flow, chemistry, and biology
- expected carbon cycling patterns

Only historic (pre 1650 CE) forests, or forests in remote and undisturbed areas of the U.S. today, might have all

or most of these attributes, but they are useful attributes to compare to current forest conditions found in similar ecological strata. That is, forest stands that are relatively undisturbed by human activities provide reference conditions to evaluate the health and sustainability of forest stands that have had one or more substantial human disturbances.

There is general agreement that forest health and sustainability should be evaluated at different spatial and temporal scales. Spatial scales range from landscape-size (composition and arrangement of forest fragments and condition of forests over large geographical areas) to the landtype or fragment scale (e.g., small watersheds). Which is to say it is necessary that we account for the condition of forests at large regional scales such as domain, division, province, section, and subsection units (Bailey's (1995), as well as at smaller ecological unit-size (landtype associations, landtypes, landtype phase, and site). The landscape scale addresses fragmentation of forest ecosystems using criteria such as patch size, landscape pattern types, and connectivity. Thus consideration of diversity, productivity, vitality, conservation of soil and water, and carbon cycling are evaluated at small and large ecological scales, using techniques that are appropriate for each.

This Report identifies many of the major forest health issues in the Mid-Atlantic region. It evaluates the health and sustainability of forests in this region—up to the year 2000—by identifying key forest health issues, relevant indicators of forest ecosystem stress, responses to stressors, and indicators of forest ecosystem condition. We discuss the FHM program's conceptual approach to forest sustainability, evaluates the temporal and spatial condition of forest ecosystems in the MAIA region using the most relevant and available information prior to the year 2001, and establish baseline or reference conditions for future evaluations of forest health and sustainability in this region.

We addressed three major categories (table 1) essential for determining the health and sustainability of forests in the MAIA region:

- forest stressors (native and exotic insects and pathogens, exotic invasive plants, urbanization and fragmentation, altered fire regimes, air pollution, and climate change)
- forest responses (tree crown condition, damage, and mortality)
- forest condition (productivity, market benefits, game species, soil and water systems, carbon sequestration, and biological diversity)

Evaluation of Forests in the MAIA Region

Typically we classify the amount and type of forests using one or more or the following approaches that are spatially and temporarily different depending on the intended purpose. One method is to use satellite data to 'classify' lands as forest, agriculture, bare land, water, and other type groupings. Depending on the kind of sensor in the satellite, other levels of differentiation are possible, e.g., hardwood or softwood forests types, vegetation activity, and foliage discoloration. The advantage of this approach is that total surface coverage *(wall-to-wall)* is possible. The disadvantage is that the resolution of the type, condition, and other aspects of the forest, as well as our ability to discern forests conditions below the tree canopies, is often limited.

A second method is to use aerial surveys (typically taken from fixed-wing aircraft or helicopter) to photograph or videograph forest condition. This method gives broad coverage, but not wall-to-wall; and differences between forest, land, and water are very pronounced. Still, we can differentiate type (different tree species) and condition (mortality, storm damage, and other attributes) of forests to a better, but still limited, resolution.

A third method is using ground surveys or long-term ground monitoring plots (fixed or variable area), points, or transects. In this approach crews of foresters and technicians sample the forest at ground level; and they derive estimates of forest extent, and population estimates by combining information from ground monitoring with satellite and aerial survey data. This method provides a great detail of information about forest condition, an inventory of all plant species, condition of trees (tree crowns and damages), forest structure and wildlife habitat, soil condition, fuel loading, detailed information on insect and pathogen damage, and other conditions that may be present. Obviously, as the amount and resolution of the data increases, the cost of data collection increases also.

Different methods are used by different agencies or research groups for different purposes. For example, within the USDA, the Natural Resources and Conservation Service (NRCS) conducts a National Resource Inventory (NRI) to categorize land types and uses, change in land use, soil conditions, and other data on all non-Federal land. The NRI category of *developed land* varies from that used by some other data collection entities. The NRI's intent is to identify lands that have been permanently removed from the rural land base. Therefore, the NRI-developed land category includes: (a) large tracts of urban and developed land; (b) small tracts of developed land totaling less than

10 acres; and (c) land outside of these developed areas that is in roads, railroads, and associated rights-of-way (USDA 2000). Additionally, NRI defines *forest land* as an acre or more (at least 100 feet wide) that is 10 percent stocked (areal canopy cover of 25 percent or greater) with single-stemmed woody plants that attain heights of 4 meters (~13 feet). For general purposes, NRI's definition of 'forest' closely approximates the USDA Forest Service's definition of *timberlands,* which is a subset of forests (table 2). Smaller multi-stemmed trees, like some juniper species in desert lands), large shrubs, and other such plants are classified by NRI as rangelands.

Also within USDA, the Forest Service's FHP and FHM programs use aerial and ground surveys to quantify insect and pathogenic damage, windfall, senescence and mortality, storm damage, and other damaging agents. In addition the FIA and FHM programs use ground monitoring surveys and fixed-area plots to quantify the type, characteristics, condition, and changes in condition of forest ecosystems. The Forest Service defines forests as land that is at least 10 percent stocked with trees that are or will be of commercial value, with a 20 percent areal canopy coverage by forest tree species of any size ≥1 acres, and strips of forest of that size and at least 120 feet wide, or land formerly having such tree cover, which is not currently developed for a nonforest use (http://www.fia.fs.fed.us/). Strips of forests include stream-side and windbreak strips of trees that have a composite crown cover of at least 120 feet.

Forest land is divided into timberland, reserved timberland, and woodland. The Forest Service defines woodlands as forests containing woody species that produce less than 20 cubic feet of wood per acre per year at the end of a growing season. FIA and FHM's concept of woodlands closely approximate NRI's definition of rangelands. Rarely are there discrepancies, however small, in estimates of the amount of forest land and other determinants, particularly when comparing data on the same attribute that is collected or computed by different agencies. The primary cause for any discrepancies is that the agencies have slightly different definitions of forests, as well as methods of data collection, aggregation, and analyses.

We used the information from the Forest Service FIA and FHM programs often when describing forest composition, condition and change, responses of forests to stressors, and other details. On the other hand, we used NRI data when describing ownership and change in land-use of forests, particularly involving change from forests to other resource-use categories, e.g., urban, agriculture, or vice-versa. For all tables and figures in this report, the source of data presented is intended to facilitate understanding of any discrepancies that might occur between or among different methods of data collection.

Land area that has a significant component of trees can be subdivided into four groups or categories of forests:

Table 2—Land cover types in MAIA states circa 2000

States	Woodland	Reserved	Total forest[a]	Non-Forest	Total land[b]	Forest	Non-forest
	thousand acres					*percent*	
Delaware	10.2	2.0	388.2	847.2	1,235.2	31.4	68.6
Maryland	126.4	152.9	2,703.4	3,592.2	6,295.4	42.9	57.1
New Jersey[c]	5.6	86.2	1,717.6	2,040.5	3,758.6	45.7	54.3
New York[d]	47.9	799.9	8,425.0	5,729.1	14,154.9	59.5	40.5
North Carolina[e]	321.0	87.0	8,386.0	6,475.0	14,861.0	56.4	43.6
Pennsylvania	285.0	834.9	16,992.9	11,735.6	28,728.5	59.2	40.8
Virginia	47.0	532.0	16,026.0	9,382.0	25,408.0	63.1	36.9
West Virginia	27.7	181.1	12,126.8	3,309.5	15,436.3	78.6	21.4
MAIA total	581.8	2,676.9	66,766.9	43,111.1	109,877.0	60.7	39.3

[a]Total forest = woodland + reserved + timberland.
[b]Total land = total forest + non-forest.
[c]Includes all or part of 12 counties only.
[d]ncludes all or part of 25 counties only.
[e]Includes all or part of 47 counties only.
Source: USDA Forest Service's Forest Inventory and Analysis Program; (http://www.fia.fs.fed.us).

woodlands (areas of land covered by trees where the intended management, but the lands are relatively unproductive for timber production); reserved forest lands (tree covered areas where the intended use is forest management, the lands are suitable for timber production, but the land is reserved from being used for timber production by long-term protection laws or regulations); timberlands are tree-covered land that is intended for forest use, not reserved from timber use, and relatively productive in tree growth (growth is capable of 20 or more cubic feet of wood per acre per year); and urban forests, where the intended use is not forest management, and census tracts contain 500 or more people per square mile. FIA estimates of forest cover in three categories (woodland, reserved, and timberland), total forest, non-forest land, and total land area are given in table 2.

NRI estimates of forest land cover from aerial surveys most closely approximate FIA estimates of timberlands in the MAIA region, because most reserved forest lands are Federal (not measured by NRI); woodlands do not meet NRI's definitions of forests and, as stated above, are measured by NRI as rangelands; and other factors. For example, the FIA-estimated 389.2 million acres of timberlands in Delaware (table 2) compared to the NRI-estimated 365 million acres total (table 3).

Almost all forestlands in the MAIA region were timberlands (63.51 million acres), only 2.68 million acres were reserved forests, and 0.58 million acres were woodlands (fig. 3). Delaware had the lowest percentage of forestlands (31.5 percent) and West Virginia the highest (78.6 percent) (table 2).

Table 3—Forestlanda in MAIA region by ownership class circa 2000

State	National Forest	Other federal	State	Local	Forest Industry	Other private	Total all ownerships
	thousand acres						
Delaware	0	0	13	0	31	320	365
Maryland	0	25	188	33	131	2,048	2,424
New Jersey	0	19	322	57	0	1,228	1,627
New York	8	23	474	104	150	6,821	7,580
North Carolina	118	282	169	220	1,273	6,373	8,435
Pennsylvania	465	38	2,642	233	613	11,883	15,874
Virginia	1468	221	211	83	1,537	11,909	15,429
West Virginia	922	37	209	5	803	9,942	11,918
Total	2,980	646	4,228	734	4,538	50,524	63,651

State	Federal government	State government	Local government	Forest industry	Other private
	percent				
Delaware	0	4	0.01	9	88
Maryland	1	8	1	5	84
New Jersey	1	20	3	0	75
New York	0.4	6	1	2	90
N. Carolina	5	2	3	15	76
Penn	3	17	1	4	75
Virginia	11	1	0.5	10	77
W. Virginia	8	2	0.04	7	83
Total	6	7	1	7	79

[a] NRI definition of forestlands most closely approximate U.S. Forest Service's Forest Inventory and Analysis (FIA) program designation of timberlands (a subset of forestlands that designate areas producing, or capable of producing, 20 cubic feet (0.57 cubic meter) or more acre (0.4 ha) per year of wood at the end of the growing season.
Source: Natural Resource Conservation Service's National Resource Inventory 1992 Database; USDA 2000.

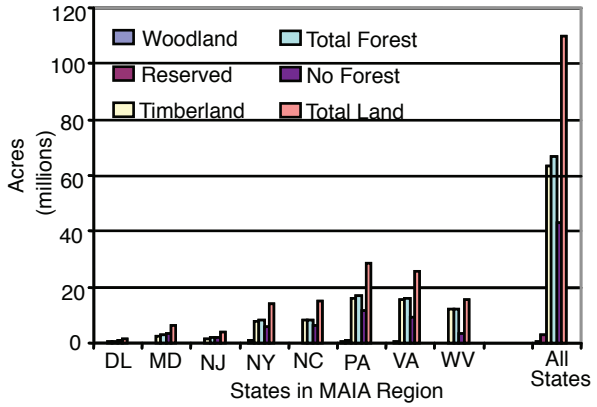


Figure 3—Classifications of forestland and non-forestland types and areas by MAIA region states in 1990s. Entire Washington D.C. area is considered urban and therefore not forestland.
Source: USDA 2000; (http://www.nrcs.usda.gov/technical/NRI/).

Forests in the MAIA region were composed of woodlands, reserved lands, and timberlands which, when combined, are considered the total forested lands (forestlands). Total forestlands in 2000 covered over 66.8 million acres (60.8 percent) of the 109.9 million acres of total land area in the MAIA region (table 2). The amount and percent of forested lands, non-forest land, and total land differs greatly by State in the MAIA region (fig. 3), with Pennsylvania and Virginia both having over 16 million acres of forestlands, covering 59.2 and 63.1 percent of the total land areas, respectively (table 2).

SocioEconomic Issues in the MAIA Region

The purpose of the socioeconomic component of the MAIA forest assessment was to develop approaches for understanding and monitoring the relationship between and among changes in forest ecosystems and the quality of life in the region. Our main objective for the socioeconomic assessment was to describe and analyze resource and land use variables that influence human well-being, as well as the mix of market and non-market benefits produced by the resources and land use systems. For future analyses of the sociological aspects, the development of meta-indicators to measure how aggregate human welfare is influenced by changes in forest ecosystems will be necessary. In addition, monitoring and evaluating the legal, institutional, and policy frameworks associated with forest conservation and management, as well as the resulting impacts on resource conditions and human welfare, is needed. These information needs are further enumerated in Criteria 6 and 7 of the Montreal Process Criteria and Indicators (Anon. 1995a).

We organized our analyses using several major themes: land use, resource use (including market commodities), population and demographics, and forest management and investments. Much of the socioeconomic context of forest ecosystems in the MAIA region is provided in this information.

To establish background for the social and economic relationships to forest ecosystems in the MAIA region, we started with information about forest ownership therein. Private landowners held 86 percent of forested lands. Private forest industry owned 7 percent of this land, while non-industrial private landowners owned 79 percent. The remaining 14 percent was owned by States (7 percent), Federal agencies (6 percent), and municipal and county governments (1 percent) (fig. 4).

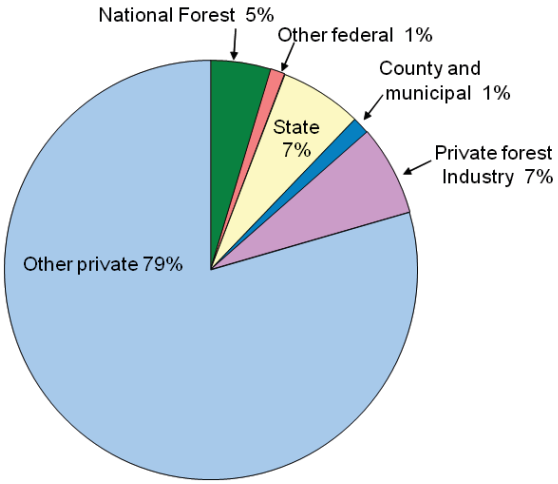


Figure 4—Ownership of forestlands in the MAIA region. Forestlands in Natural Resource Inventory are similar to Forest Inventory and Analysis's timberland classification of forestlands. Source: USDA 2000; (http://www.nrcs.usda.gov/technical/NRI/). Birch 1996a.

The 2.1 million non-industrial, private forest landowners in the MAIA region comprised 21 percent of all non-industrial, private forest landowners in the U.S., and owned 13 percent of all such land in the entire U.S. (Birch, 1996a). Thus, non-industrial, private forestlands were held by a diverse mix of private individuals, farmers, and other corporate and non-corporate entities (Birch 1996a, 1996b, 1996c).

Table 3 shows total number of acres and percentage of all forest land (USDA 2000) by ownership class for individual States in the region. Ownership varied widely across the MAIA States. Federal forests ranged from 0 percent of forests in Delaware to 11 percent in Virginia, while State

and local government ownership ranged from 1.5 percent in Virginia to 23 percent in New Jersey. Forest industry owned 9 to 15 percent of all forests in Delaware, Virginia, and North Carolina, compared to 0 to 2 percent in New Jersey and New York, respectively. In contrast, non-industry private ownership (other private) accounted for 75 percent or more for all States, ranging from 75 percent of all forests in New Jersey and Pennsylvania to 90 percent of all forests in New York.

DESCRIPTION OF FORESTS

Chapter 3.

Biological and Physical Attributes

Mark J. Ambrose and Barbara L. Conkling
Department of Forestry, North Carolina State University

Ecoregion Sections

The MAIA region contains all or portions of different forest ecosystem sections. For analysis purposes forests can be grouped or classified as ecological units of various shapes and sizes; and selection of spatial scales for analyses often depends on data availability for species, forest types, ecological processes, stressors of interest, and other factors. Ecological units identify, at different levels of resolution, land areas that have similar biotic and environmental attributes, and hence capabilities and potentials for management. Generally, ecological units of various sizes are similar in: (1) potential natural plant communities; (2) soils; (3) hydrologic function; (4) landform and topography; (5) geologic origin; (6) climate; and/or (7) natural processes such as nutrient cycling, productivity, succession, and natural disturbance regimes associated with flooding, wind, or fire (Cleland and others 1997).

We analyzed data provided by the USDA Forest Service, Forest Health Monitoring (FHM) and Forest Inventory and Analysis (FIA) programs, as well as data from a number of other programs, to identify ecological units and help facilitate ecologically sound approaches to resource planning, management, and research. The Forest Service uses a national hierarchical system of ecological units developed by Bailey (1995) that classifies the entire U.S. into ecoregion domains, divisions, provinces, sections, subsections, landtype associations, and landtypes (McNab and Avers 1994). Areas within the (MAIA) ecological units similarly have the geology, lithology, regional climate, soils (examined to the levels of orders, suborders, or great groups), and potential natural vegetation, and/or potential natural communities that Bailey described (Cleland and others 1997). We analyzed data at the level of ecoregion section, because a section constitutes land areas on the order of thousands of square miles, and because this is the finest scale that much of the FHM and FIA ground plot data can adequately represent in our analyses (Stolte 1997).

Boundaries of Bailey's ecoregion sections, which are derived from geology, climate, and topography, generally do not correspond with the MAIA boundary, which is based primarily on watershed landforms, management units, and other units of measure. Many ecoregion sections extend beyond the northern and southern borders of the MAIA region. Figure 5 shows the location of ecoregion sections represented in the MAIA, and table 4 provides information on the main attributes of each section. The MAIA region comprises a wide variety of forest communities, each of which is home to numerous plant and animal species. By way of illustration, the MAIA region contains portions of six different ecoregion provinces (Laurentian Mixed Forest; Eastern Broadleaf Forest; Southeastern Mixed Forest; Outer Coastal Plain Mixed Forest; Adirondack-New England Mixed Forest-Coniferous Forest-Alpine Meadow; and Central Appalachian Broadleaf Forest-Coniferous Forest-Meadow) (Bailey 1995) and nineteen different sections (table 4).

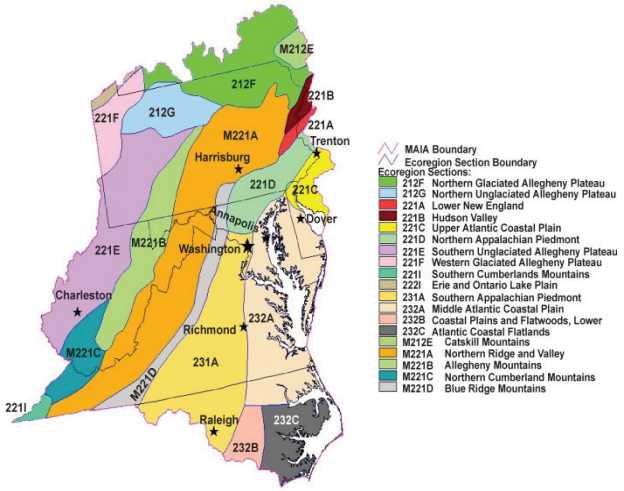


Figure 5—Bailey's ecoregion sections within the MAIA region. The U.S. Forest Service's Forest Health Monitoring, Forest Inventory and Analyses, and other program data were frequently averaged at these spatial scales for comparison of forests within the MAIA region. Source: Bailey 1995; USDA Forest Service; (http://www.fs.fed.us/land/ecosysmgmt/index.html).

Table 4—Features of ecoregion sections in the MAIA region

Ecoregion[a]	Soil Taxa	Potential Forest Vegetation	Elevation (feet)	Growing Season (days)	Land Use
----------212 Laurentian Mixed Forest Province----------					
212F	Inceptisols	N. hardwoods, App. oak	650-1970	100-160	Agriculture, forestry
212G	Ultisols, Inceptisols, Entisols, Alfisols	Hemlock-N. hardwoods, App. oak-pine	1000-2000	120-150	Forestry, oil, agriculture
----------221 Eastern Broadleaf Forest Province----------					
221A	Inceptisols, Entisols, Histosols	N. hardwood, App. oak, NE oak-pine	200-1000	120-180	Forestry, agriculture, urban
221B	Inceptisols, Alfisols	N. hardwood, App. oak	200-1000	160-180	Forestry, urban, agriculture
221C	Ultisols, Entisols	NE oak-pine forest, oak-pine-hickory	0-300	240-250	Forestry, agriculture, pasture, urban
221D	Ultisols, Alfisols, Inceptisols, Entisols	App. oak forest	80-1650	160-250	Agriculture, forestry, urban
221E	Alfisols, Ultisols, Inceptisols	Mixed mesophytic App. oak	650-1300	120-180	Agriculture, urban, coal, oil
221F	Alfisols, Ultisols	Beech-maple, App. oak, N. hardwood, mixed mesophytic	650-1000	160 (aver.)	Agriculture, forestry
221I	Ultisols, Inceptisols, Entisols	App. oak, mixed mesophytic	800-1000	175 (aver.)	Fire; forestry
----------231 Southeastern Mixed Forest Province----------					
231A	Ultisols, Inceptisols, Alfisols, Entisols	Oak-hickory-pine, Southern mixed	330-1300	205-235	Agriculture
----------232 Outer Coastal Plain Mixed Forest Province----------					
232A	Ultisols, Entisols, Alfisols, Spodosols	Oak-hickory-pine, Southern floodplain	0-80	185-220	Reserved; Agriculture
232B	Ultisols, Alfisols	Southern mixed forest, oak-hickory-pine	80-660	200-280	Agriculture
232C	Ultisols, Spodosols, Entisols, Histosols	Southern mixed forest, oak-hickory-pine	0-80	185-220	Agriculture
----------M212 Adirondack-New England Mixed Forest-Coniferous Forest-Alpine Meadow----------					
M212E	Inceptisols	N. hardwood, N. hardwood-spruce	900-4200	120-160	Forestry
----------M221 Central Appalachian Broadleaf Forest – Coniferous Forest – Meadow Province----------					
M221A	Inceptisols, Ultisols, Alfisols	App. oak, oak-hickory, pine	300-4000	120-180	Forestry, urban
M221B	Inceptisols, Ultisols, Alfisols	Mixed hardwoods, spruce-fir	1000-4500	140-160	Forestry, mining
M221C	Inceptisols, Ultisols	Mixed mesophytic, App. oak	2000-2600	140-160	Agriculture, forestry, mining

[a]See fig.5 for map of ecoregions in MAIA region.
Source: Bailey 1995; (http://www.fs.fed.us/land/pubs/ecoregions/).

Major Watersheds

EPA chose to delimit the MAIA region—primarily focused on the Chesapeake Bay watershed—in order to include other watersheds that are of particular interest to land and resource managers, elected officials and their constituents, and all who depend on healthy and sustainable ecosystems within the MAIA boundary. The region therefore includes all of Pennsylvania, Delaware, Maryland, the District of Columbia, Virginia, and West Virginia. Parts of New Jersey, New York, and North Carolina also are within the MAIA boundary so that several other major watersheds could be considered in their entirety.

The United States Geological Survey (USGS 1974-1987) defined four levels of water systems of the U.S. that comprises four levels of successively smaller hydrologic units: regions, sub-regions, accounting units, and cataloging units. USGS hydrologists devised a system whereby the units were assigned a hydrologic unit code (HUC), a two-to-eight digit number that they use to assign codes to hydrologic units (http://water.usgs.gov/GIS/huc.html). The first level (2-digit HUC) classification describes the U.S. as composed of major regions, each of which contains the drainage of a major river or series of rivers. The second level (4-digit HUC) classification, or sub-region, describes "the area drained by a river system, a reach of a river and its tributaries in that reach, a closed basin(s), or a group of streams forming a coastal drainage area" (Seaber and others 1987). The third level (6-digit HUC) classification, or accounting unit, constitutes areas within a sub-region, such as watersheds within parts of the MAIA boundary. While most analyses of the MAIA region are based on ecoregion sections or provinces (fig. 5), some tree response indicators (Chapter 13) were also analyzed at 4-digit HUC watershed scales (HUC 4), e.g., those shown with associated river systems in figure 6. Bird diversity analyses (Chapter 24) was done within watersheds at the HUC 8 scale.

The MAIA region's eastern portion drains into the Atlantic Ocean. It is bounded by the Delaware Bay watershed to the north and the Pamlico Sound watershed to the south (fig. 6). In the western portion, the Allegheny and Monongahela watersheds, as well as most of the Kanawha River

watershed, are entirely within the region's boundary. The MAIA region also includes about two-thirds of Big Sandy-Guyandotte, one-fourth of Tennessee River, and one-half of Upper Ohio River watersheds. All western watersheds eventually drain into the Gulf of Mexico via the Mississippi River. In addition, very small sections of Southwestern Lake Ontario (Genesee River), Pee Dee River, Middle Ohio River, and Eastern Lake Erie watersheds are included in the region.

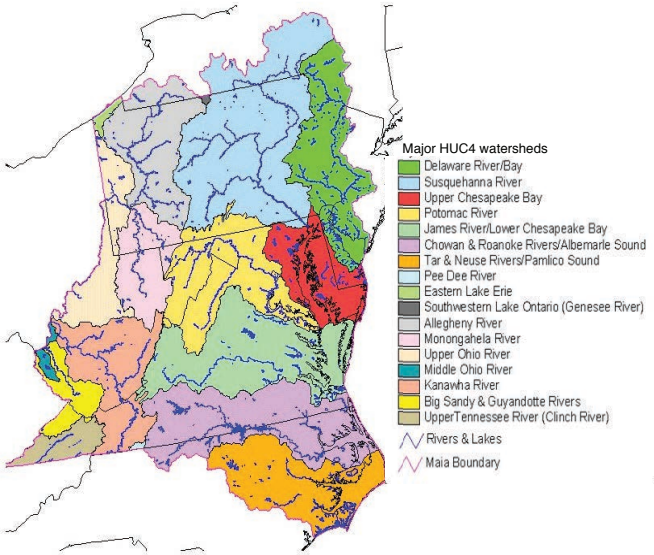


Figure 6— Major 4-Digit Hydrologic Unit Code (HUC4) watersheds in the MAIA region. Some data from the U.S. Forest Service's Forest Health Monitoring program was averaged at HUC4 spatial scales to assess differences with ecoregion section values and provide potential for comparisons with water condition. Source: United States Department of Interior's Geological Survey; (http://water.usgs.gov/GIS/huc.html).

Major Roads

The number and size of roads are important indicators of forest health and sustainability because they reflect human population density and use in an area. Roads serve as corridors for introduction of exotic, invasive species (plants, animals, insects, and pathogens); they segment contiguous areas of forests into fragments and create barriers for many wildlife species; and they increase management challenges where human habitation meets undeveloped forest areas (increased urban-wildland interface). The eastern MAIA region contains a relatively high number of major interstate highways (fig. 7) and urban areas (fig. 8); only central Pennsylvania and West Virginia are relatively free from a divergence of major Interstate highways and urban areas.

These primary Interstate highways converge in major cities such as Washington, DC, Baltimore, Philadelphia, and Pittsburgh (fig. 8).

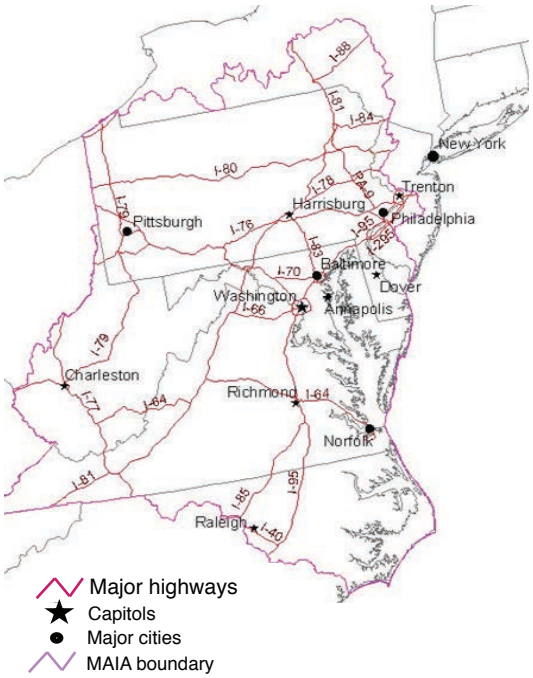


Figure 7—Major roads in the MAIA region, indicating major sources for fragmenting forest ecosystems, introduction of exotic invasive species, and other urban-wildland interface issues. Source: Environmental Systems Research Institute, Inc, Data & Maps; (http://esri.com/).

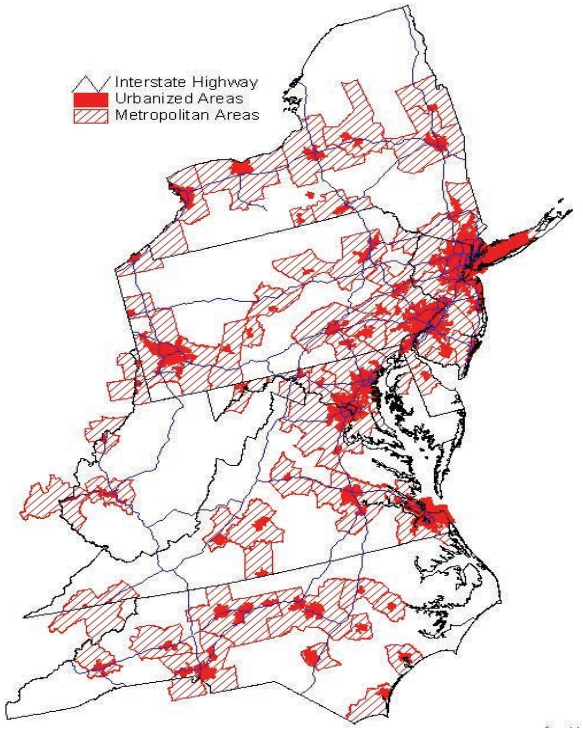


Figure 8—Metropolitan and urban areas in the MAIA region. Source: Environmental Protection Agency Region III; (http://www.epa.gov/region03/index.htm).

Chapter 4.

Forest and other Land Uses

[1]Stephanie Fulton, [2]James Steinman, [3]Evan Mercer, and [3]Kenneth W. Stolte
[1]Region 4, U.S. Environmental Protection Agency
[2]Northern Research Station, USDA Forest Service
[3]Southern Research Station, USDA Forest Service

Thirty-nine percent (43 million acres) of the 110 million acres of land (including land owned by the Federal Government) in the Mid-Atlantic region was classified as nonforest land, and 61 percent (67 million acres) was classified as forestland[1] (http://www.fia.fs.fed.us/). Ninety five percent (63.5 million acres) of the region's forestlands were classified as timberland (i.e., forests where the mean annual increment is greater than 20 cubic feet per acre per year), two percent were classified as reserved forests (not managed for timber production or harvest), and almost one percent was considered woodland (forests where the mean annual increment is < 20 cubic feet per acre per year) (fig. 9).

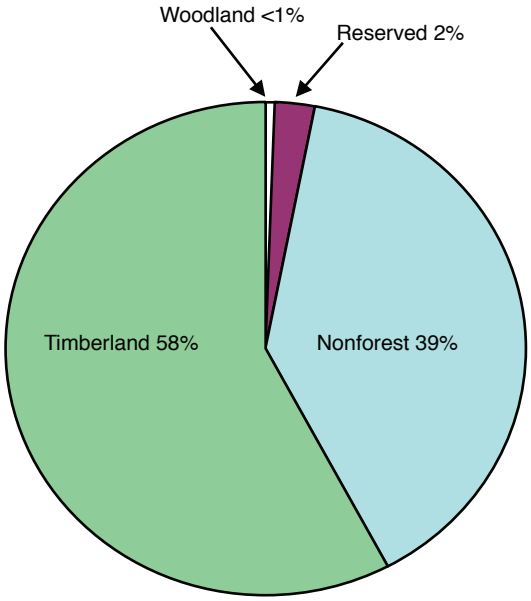


Figure 9—Classification of all lands in the MAIA region. Woodland is defined as forest land with less than 10 percent stocked with timberland species and with at least 10 percent crown cover in timberland and woodland species. Reserved land is defined as land that is withdrawn from all timber utilization by a public agency or by law. Source: US Forest Service Forest Inventory and Analysis Eastwide Database; (http://www.fia.fs.fed.us/).

[1]Forestland in fig. 9 has timberland, woodland, and reserved land components, and includes Federal lands. It is defined in the US Forest Service Forest Inventory and Analysis Program's Eastwide Database as land at least 10 percent stocked by forest trees of any size, or formerly having such tree cover, and not currently developed for nonforest uses. See glossary for additional information.

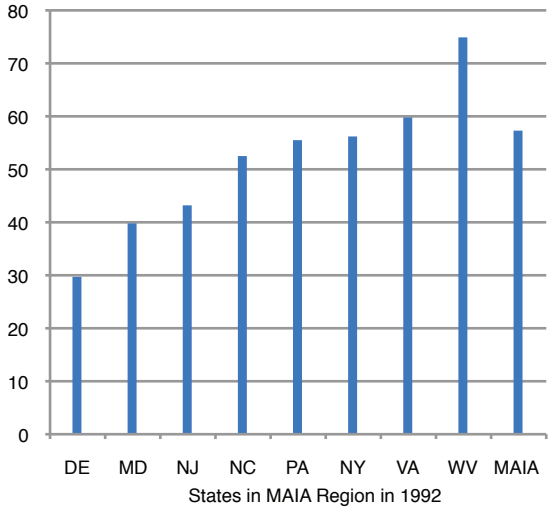


Figure 10—Non-federally owned urban and rural land that is classified as forestland in each State and in the entire MAIA region in 1992. Some counties in New Jersey (~12 counties), North Carolina (~47 counties), and New York (~25 counties) included in MAIA region. Source: USDA 2000; (http://www.nrcs.usda.gov/technical/NRI/).

The percentage of all non-Federal land[2] (urban and rural) under forest cover within MAIA State boundaries in 1992 ranged from 29.7 percent in Delaware to 74.9 percent in West Virginia (fig 10). For the entire MAIA region, forests covered 57.3 percent of all non-Federal land (USDA 2000).

Within the five States contained entirely in the MAIA region (Delaware, Maryland, Virginia, West Virginia, and Pennsylvania), the total surface area is about 80,993,900 acres (USDA 2000) (fig. 11). In 1997 forests covered about 42,100,000 acres of that area, or roughly 52 percent of the total surface area, including Federal lands and water. The

[2]Figures 10 and 11 do not include Federal lands because Federal lands are excluded from the National Resources Inventory data set. Forestland in fig.10 is defined by the 1997 National Resources Inventory (USDA 2000) as a *land cover/use* category that is at least 10 percent stocked by single-stemmed woody species of any size that will be at least 4 meters (13 feet) tall at maturity. Also included is land bearing evidence of natural regeneration of tree cover (cut over forest or abandoned farmland) and not currently developed for nonforest use. Ten percent stocked, when viewed from a vertical direction, equates to an areal canopy cover of leaves and branches of 25 percent or greater of the area. The minimum area for classification as forest area is 1 acre, and the area must be at least 100 feet wide. See glossary for additional information.

amount of forest land in 1997 was highest in Pennsylvania (15,477,900 acres), followed by Virginia (13,315,800 acres), and West Virginia (10,581,500 acres). In four of the five States, forests comprised the primary land cover and use; in Delaware crop lands were the primary land cover (484,500 acres). Other primary land cover and uses in 1997 included crop land, pasture land, total agriculture (crop and pasture lands combined), urban land, Federal lands, and water (fig. 11) (USDA 2000). Agriculture comprised the second largest land cover type in Pennsylvania, covering 7,316,100 acres; crop lands covered 5,471,200 acres; and pasture lands 1,844,900 acres. Water bodies constituted 471,700 acres in Pennsylvania; Federal lands (723,900 acres) and developed (urban) lands (3,983,200 acres) accounted for the other land cover types. Similar land cover patterns were found in the other four States (fig. 11).

Although the area in urban landscapes increased greatly from 1982 to 1997 in most States (see Urbanization, chapter 11), the area of developed or urbanized lands remained relatively small compared to areas maintained for agricultural uses (crops and pasture) and forest. Within the entire five-State area in 1997, urbanized lands comprised about 8,943,900 acres (11.04 percent) of the total MAIA region, compared to 18,222,400 acres (22.5 percent) in agricultural uses, and 42,100,000 acres (52 percent) in forest (fig. 11). The area providing fresh water was about 4,517,800 acres (5.6 percent), and Federal lands covered about 4,781,600 acres (5.9 percent). Other minor land uses in the region (USDA 2000) included acreage managed under the Conservation Reserve Program (promoting conversion of highly erodible crop land into vegetative cover) and Minor/Other Rural land (farm structures, windbreaks, etc.).

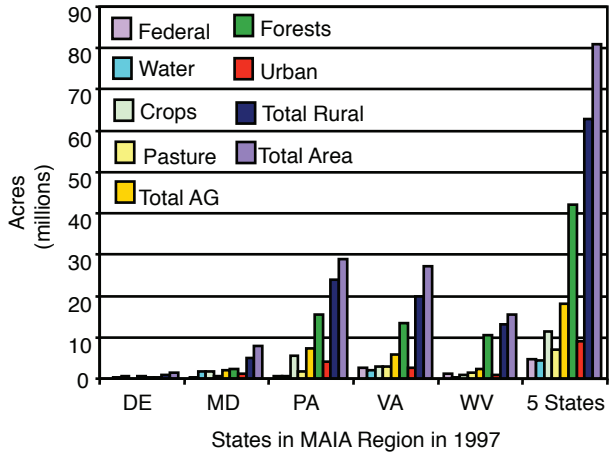


Figure 11—Acres of primary land use types five States contained entirely in the MAIA region in 1997 and the total for all five States. Total agriculture is the combination of cropland and pastureland. Total area is total surface area including land and water cover. Source: USDA 2000; (http://www.nrcs.usda.gov/technical/NRI/).

Current Forest Extent

Contiguous forested land was found concentrated in the northern (Allegheny Mountains) and western (Appalachian Mountains) mountainous areas where—because of the rugged terrain and relatively poor soils—human habitation has been restricted to the fertile valley bottoms. In those areas forest stretches across the landscape in vast contiguous blocks (580 hectares with at least 90 percent forest cover), which are broken only by small communities and transportation networks (fig. 12). Intact forests provide many important ecological functions that have come to be valued by human society, including clean water and air,

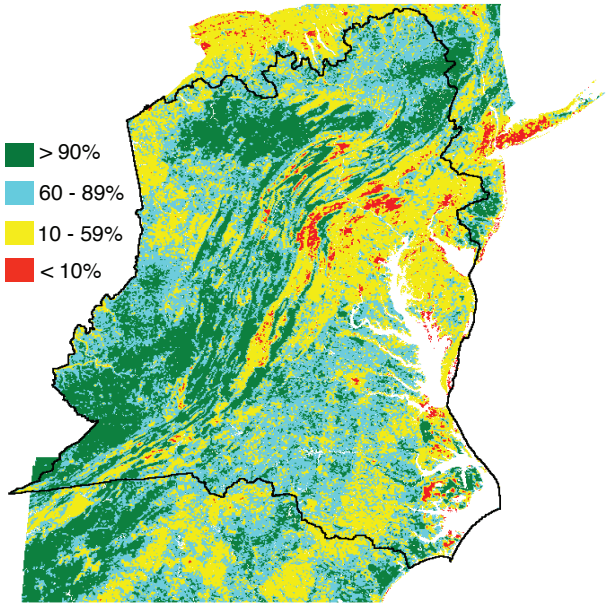


Figure 12—Forest cover within 1457 acre landscape units in the MAIA region ca. 1990. The fine-scale (0.22 acres per pixel) land cover map from EPA's MRLC was generalized to show the proportion of forest cover within the 1457 acre units. Areas with more than 90 percent forest cover are highly interior forested landscapes; in contrast to highly fragmented units with less than 10 percent forest cover. A 60 percent cover threshold separates interior forests from more fragmented forests. Source: Loveland and Shaw 1996; Riitters and others 2000; U.S. Environmental Protection Agency's early 1990s Multi-Resolution Landscape Consortium (MRLC); (http://www.epa.gov/mrlc/).

biological diversity, high natural resource productivity, fish and wildlife habitat, as well as quality-of-life factors such as recreation opportunities and aesthetic enjoyment.

Large tracts of contiguous forest provide relatively rare, high quality, interior forest habitat not found in smaller forest areas. Interior forest habitat is that portion of a forest that lies far enough away from the forest edge to offer shelter from predators, catastrophic weather events (e.g., wind storms), and human disturbance (e.g., introduction of exotic species). Interior forest habitat is negatively affected by stressors associated with *edge effects* that include increased light and temperature; easier access by nuisance wildlife and invasive plant species, and disturbances associated with a variety of human uses (O'Connell and others, 1998 and 2000). The severity of impacts to forest ecosystems by high edge-to-area ratios depends on forest fragment size relative to perimeter length. The smaller the patch, the greater the ratio value of perimeter to area, and a greater impact on the breadth of forest values. Effects occur mainly along the edge, and extend some distance into the forest interior, decreasing towards the center of a forest's interior. Effects on forest wildlife and plant species composition vary by species; some are more sensitive than others to changes in forest microclimate and competition from nuisance wildlife and invasive plant species. Forests along streams and rivers, known as riparian forests, provide critical shade, shelter, and food resources for fish and other aquatic and terrestrial life. The largest, most dense blocks of contiguous forest in the Mid-Atlantic are located in the north-central and central Appalachians.

The distribution of forests in the Mid-Atlantic region, when evaluated at landscape scales, clearly shows how forests are interspersed primarily with agriculture, urban, and other development land uses (fig.13). Although the total amount of forestland in the region changed only slightly between 1982 and 1992 (USDA 2000), the pattern of change was distributed unequally across the region. Generally, watersheds nearest the Chesapeake Bay and their tributary rivers have far less forest than watersheds in the mountainous western portion of the region.

The permanent conversion of forest to non-forest uses, referred to as deforestation, is concentrated in the Piedmont and Coastal Plain regions along the I-95 corridor (fig. 8). This area contains some of the most densely populated urban centers on the East Coast, including Norfolk-Chesapeake, VA; the Washington DC-Baltimore, MD corridor; and the Wilmington, DE-Philadelphia-Allentown, PA corridor. Another region of low forest cover is the area surrounding Pittsburgh, PA in the Erie/Ontario Lake Hills and Plains region.

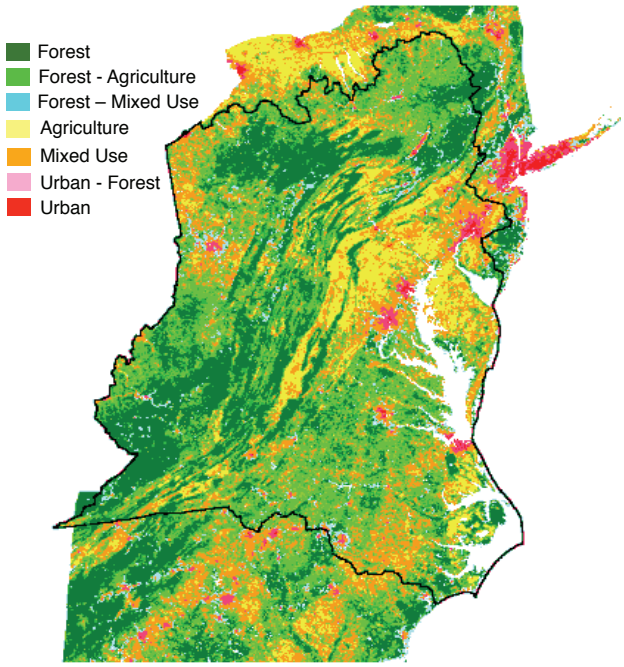


Figure 13—Landscape Pattern Types (LPT) in the MAIA region ca. 1990. The fine-scale (0.22 acres per pixel) land cover map from EPA's MRLC indicated the relative proportions of forest, agriculture, and developed cover types within the 1457 acre units; LPTs were labeled by the relative amount of three land cover types within the surrounding 1457 acre units. The 19 LPT categories of MRLC were condensed into 7 LPT aggregates to simplify regional patterns. Source: Wickham and Norton 1994; Loveland and Shaw 1996; Jones and others 1997; Riitters and others 2000; U.S. Environmental Protection Agency's early 1990s Multi-Resolution Landscape Consortium (MRLC); (http://www.epa.gov/mrlc/).

Chapter 5.

Urban Forests

Danial Twardus
Forest Health Protection, USDA Forest Service

It is hard to come up with a comprehensive definition of urban forest, but most will agree that it may be simply a row of trees along their residential street, or a cluster of trees in the town squares or community parks. Most citizens realize and appreciate the numerous benefits of trees in the urban environment, whether they occur in large or small clusters, along the undeveloped city limits, or as individual decorative trees in one's front yard. Most commonly, though, an urban forest is valued for the aesthetic nuance it provides, as well as the shade and micro-habitats that can be found there. The benefits and values of urban forests also include biodiversity, flood control and water preservation during droughts, refilling aquifers, water quality maintenance, temperature and climate control, energy conservation, and recreational activities such as bird watching and hiking.

In addition, evidence is emerging that the physical environment in the inner city—especially areas with substantial concentrations of trees—can have social benefits, such as enhancing developmentally important recreational activities for urban youth (Taylor and others 1998). Urban, non-timber forest products may also represent important economic, nutritional, and cultural resources to a variety of groups, especially minority and low-income urban residents. Non-timber forest products include plants like ginseng, edible mushrooms, pine needles, etc. An initiative in Baltimore and Philadelphia identified more than 30 different products found in urban forests (Community Resources 1999).

Public interest in reducing the loss of urban forests is apparent in the growth of Tree City USA. This program, administered by the National Arbor Day Foundation, provides recognition to cities, towns, military installations, and other governmental entities, which are showing a commitment to preserving urban forests[3]. Such commitment includes an active community tree-care ordinance, where a designated party is responsible for tree care and the citizenry maintains an active community forestry program. Participation in Tree City USA by communities in the Mid-Atlantic States is relatively high (a total of 173 communities participate), and in 1997 Pennsylvania had the most participating communities (table 5).

Table 5—Tree City USA program in MAIA States circa 2000

State	Number of Tree City Communities
Maryland	39
Delaware	4
Washington, DC	1
Virginia	33
West Virginia	16
Pennsylvania	80
Total	173

Source: (http://www.arborday.org/programs/treeCityUSA.cfm).

Urbanization causes significant changes in land cover: Natural surfaces are covered by concrete, asphalt, and other impervious materials that significantly alter the aerodynamic, reflective, thermal, and moisture properties of a landscape. Such changes bring with them "urban climates" that are characterized by elevated temperatures and reduced air quality. Trees can have a major ameliorative influence on the human environment in urban areas. They interact with wind, air temperature, humidity, and solar and long-wave radiation, thereby helping conserve energy. Urban tree canopies reduce summer cooling costs by shading structures and people. Although the effects of tree shade on heating and cooling energy use vary with building type, building orientation, and tree type and location, energy savings can be substantial. In Chicago, shade trees were reported to have just that effect. A single 25-foot tree was estimated to reduce annual heating and cooling costs by as much as 2 to 4 percent (McPherson 1993).

Sanders (1986) developed models of the relationships between urban forest cover and the amount of storm water runoff in Dayton Ohio. He found that the 22 percent tree canopy cover in Dayton lowered water runoff by about 7 percent. His model also predicted that if tree cover of impervious surfaces was increased to 50 percent an additional reduction of 5 percent in water runoff would be realized. That is, if Dayton had a 50 percent urban tree cover that would produce a 12 percent decrease in water runoff.

[3]Schweitzer, Tina. 1999. Personal Communication. Tree City USA Coordinator, National Arbor Foundation, 211 North 12th Street, Nebraska City, NE, 68508.

Air pollution affects most major cities and the forests that surround them, especially in high population-density urban areas of the Mid-Atlantic region. Air pollutants of greatest concern are carbon dioxide (CO_2) and other gases that contribute to global warming; sulfates (SO_x), nitrates (NO_x), and other anions that contribute to acidic deposition (harming plants, buildings, and people); ozone (O_3); metals; and particulates (Stolte 1997). Climate change gases have the potential to alter temperature and moisture patterns globally, and consequently disrupt many natural and social systems. Sulfates, nitrates, and other anions acidify precipitation and degrade soils, forests, and water systems; and they damage paint, buildings, statues, and other objects. Ozone is a strong oxidant that injures plants by reducing growth and causing mortality, and it degrades rubber and other materials. Metals accumulate in the food chain and cause many developmental problems in animals and humans. Particulates, especially small sizes classified as PM10, reduce visibility, clog machinery, and cause respiratory problems in humans.

Urban forests help remove air pollution by intercepting particulates and absorbing gaseous pollutants, and by returning clean oxygen to the air. However, only a little research has been published on the removal of atmospheric pollution by urban trees. In an experiment in Chicago, Nowak (1994a, 1994b) estimated that under typical in-leaf (growing season) daytime conditions in 1991, one ha of urban tree cover removed 0.0007 lb/acre/hr of CO_2, 0.0037 lb/acre/hr of SO_2, 0.004 lb/acre/hr of NO_2, 0.005 lb/acre/hr of PM10, and 0.011 lb/acre/hr of O_3.

Urban tree values

American Forests (1999) conducted an analysis of tree cover in the Chesapeake Bay region to determine how the landscape had changed over time. That analysis assessed the value of ecological features using data from satellite images spanning the 24-year period from 1973 to 1997. Major findings were:

- The ecology of the southeastern portion of the Chesapeake Bay watershed had changed dramatically since 1973. Forest tree cover declined, and urban development expanded. Areas with high vegetation and tree canopy coverage declined from 55 percent of the area in 1973 to 38 percent of the area in 1997.
- In the Baltimore-Washington corridor, areas with heavy forest cover declined from 55 percent in 1973 to 37 percent in 1997. Tree cover loss was estimated to contribute to a 19 percent increase in water runoff, and a reduction of the corridor's ability to remove 9.3 million pounds of pollutants. The remaining 37 percent cover was estimated to provide the equivalent of $340 million in one-time capital investment that would be necessary to build storm-water retention facilities if the remaining 37 percent cover of trees and other vegetation was removed.

Most people now living in the MAIA region only have contact with trees or fragments of forest, or with isolated trees in the urban and suburban environments near their homes. Perhaps for that reason, there has been considerable interest in maintaining local urban/suburban forests. Such interest is reflected in the amount of smart growth, sustainability awareness, and expanded green-space programs. The media have played a significant role in fostering that interest.

We used two designations to classify urbanized areas—urban areas and metropolitan areas—based on geographic entities defined by the U.S. Census Bureau (Dwyer and others 2000). Urban areas are cities, towns, and villages with at least 2,500 people, or a population density of at least 384 people per square kilometer, within restrictively defined incorporated or unincorporated areas. Metropolitan areas are a county or group of counties that contained, or were associated with, a large population center. Metropolitan areas included not just a large city, itself, but also the surrounding lands that are economically and socially integrated with the city's core.

Figure 8 shows the urban areas (UA) and metropolitan areas (MA) in the Mid-Atlantic region. Within the region's total land area, urban areas covered from 1.7 percent of the total land area in West Virginia to 30.6 percent in New Jersey; metropolitan areas occupied from 16.1 percent in West Virginia to 100 percent in New Jersey. In the multi-State MAIA area tree coverage ranged from an average 10.3 percent in urban areas and 49.7 percent in metropolitan areas, compared to an average 3.5 percent in urban area and 24.5 percent in metropolitan area for the U.S. (excluding Alaska and Hawaii) (table 6). With the exception of West Virginia all MAIA States had more than 34 percent classified as MA.

The tree cover in Mid-Atlantic metropolitan areas constituted over 15.5 billion trees, and within urban areas over 735 million trees provided cover. (Dwyer and others 2000) (table 6). Tree populations in metropolitan areas were highest in New York (> 4.5 billion trees). Virginia was highest in urban areas (> 156 million trees), followed by New Jersey (>143 million trees) and Pennsylvania (>139

Table 6—Tree cover in metropolitan areas (MA) and urban areas (UA) by State area and tree cover in MAIA region and the U.S. circa 2000

State	Total Area[a]	Area MA	Area UA	State[b] MA	State[b] UA	Tree Population MA	Tree Population UA	Tree Cover[c] MA	Tree Cover[c] UA	State Tree Cover[d] MA	State Tree Cover[d] UA	Trees per Capita MA	Trees per Capita UA
	----------thousand acres----------			----percent-----		-------million-------		-----percent----		-----percent----			
DE	1,590,192	828	140	52.0	8.8	213	13	50.9	46.3	58.2	9.0	384	27
MD	7,930,094	4,530	1,118	57.1	14.1	851	89	46.5	40.1	53.2	11.1	192	21
NJ[e]	5,579,730	5,580	1,708	100.0	30.6	1,597	144	56.6	41.4	100.0	22.3	207	20
NC[e]	34,455,837	11,987	1,586	34.8	4.6	4,357	139	52.5	42.9	31.4	3.4	996	36
NY[e]	34,804,462	17,992	2,501	51.6	7.2	4,598	133	44.7	26.3	43.9	3.5	278	8
PA	29,482,538	14,404	2,066	48.9	7.0	3,733	139	48.7	34.4	43.5	4.2	370	16
VA	27,370,151	10,068	2,191	36.8	8.0	3,648	157	53.3	35.3	34.4	4.9	764	27
WV	15,643,123	2,497	268	16.1	1.7	891	23	65.6	42.2	13.4	0.9	1,191	33
U.S.[f]	991,628,365	488,986	69,407	24.5	3.5	74,426	3,821	33.4	27.1	24.5	2.8	377	17

[a]Includes land and water area combined.
[b]Percent of total State area covered by metropolitan areas (MA) and urban areas (UA).
[c]Percentage of metropolitan and urban areas covered by trees.
[d]Percentage of total state tree cover within metropolitan areas and urban areas.
[e]Includes entire State, not just counties in MAIA region.
[f]Includes District of Columbia, but not Alaska and Hawaii.
Source: Dwyer and others 2000.

million trees). These three States also ranked nationally in the top 10 States in terms of tree populations within urban areas (Dwyer and others 2000).

Tree cover within metropolitan areas of the MAIA region averaged 53.3 percent, nearly 20 percent higher than the national average (33.4 percent) (table 6). The metropolitan areas of West Virginia had the most tree cover (65.6 percent) followed by New Jersey (56.6 percent), Virginia (53.3 percent), and North Carolina (52.5 percent). Within urban areas only, average tree cover in the region was 40.4 percent, compared with a national average of 27.1 percent. Delaware, Maryland, New Jersey, North Carolina, and West Virginia all had over 40 percent tree cover in urban areas.

In the MAIA States, high levels of tree cover in both the MAs and UAs resulted in significant proportion of each State's total tree cover being in those classifications (fig. 14). West Virginia, which had a relatively small portion of surface area in MA or UA classifications, had only 13.4 percent of tree cover classified as in MAs and only 0.9 percent in UAs.

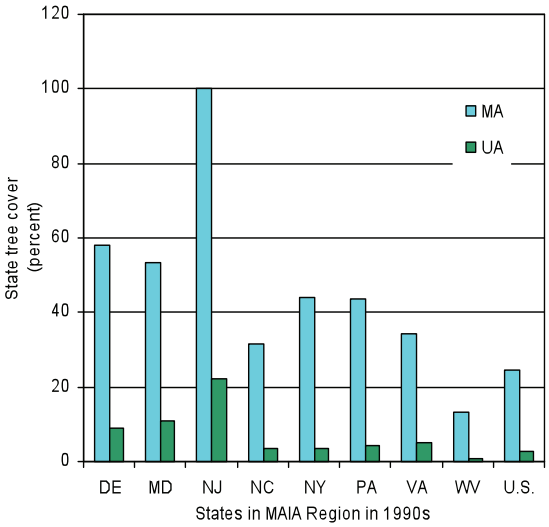


Figure 14—Total tree cover in metropolitan areas (MA) and urban areas (UA) in MAIA region states and the U.S. in 1990's. U.S. area includes District of Columbia but not Alaska or Hawaii. Source: Dwyer and others 2000.

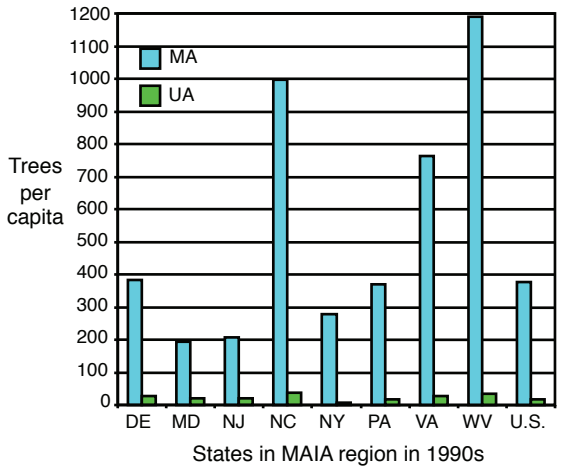


Figure 15—Trees per person (per capita) in metropolitan areas (UA) and urban areas (UA) in MAIA region states and the U.S. in 1990's. U.S. includes District of Columbia but not Alaska or Hawaii. Source: Dwyer and others 2000.

The other MAIA States had more than 31.4 percent of total tree cover in classified as MA. New Jersey, at 100 percent, was followed by Delaware (58.2 percent), and Maryland (53.2 percent). National averages for percent of total tree cover were lower for MA (24.5 percent) and UA (2.8 percent) than all MAIA States except for West Virginia (fig. 14).

The average number of trees per capita within the Mid-Atlantic MAs was 456, compared to a national average of 377. Within MAIA urban areas the number per capita averaged over 23 trees, compared to a national average of 17 (table 6). The highest number of trees per capita was found in the more rural States of West Virginia, North Carolina, and Virginia (figure 15). In fact, North Carolina and West Virginia were among the top 10 States in the Union with 996 and 1,191 trees per capita, respectively (Dwyer and others 2000).

In comparison with national averages, the region's MAs and UAs have substantial tree cover and numbers of trees. The benefits of urban forests in the MAIA, as well as programs of urban forestry and urban-rural-forest interface concerns, are important—not only because of the large population densities, but also because of the relatively large portion of the States' tree cover within metropolitan and urban areas.

Street Trees

Even though they make up only a small percentage of overall urban forest, street trees are one of the most obvious components of urban forest. In one study, street trees accounted for only 10 percent of a city's trees, yet produced 24 percent of the total leaf cover, which is vital in reducing air pollution (Nowak 1994a).

Street trees grow in very stressful environments. The average city tree lives only 32 years, and trees in downtown areas live only 7 to10 years, in stark contrast to the expected life span—up to 200 years—in rural settings. In the District of Columbia, about 5 percent of the urban trees die each year (DC Environmental Agenda 1999). Some common stressors affecting urban trees are pollution, soil compaction, nutrient deficiencies, drainage problems, salt uptake, construction damage, insects, and pathogens (Stoyenoff and others 1997). Research efforts are underway to determine which trees grow best in urban environments with the least amount of maintenance. This study found that the most commonly planted trees in an urban environment are Norway maple, flowering pear, linden, green ash, honey locust, and red maple (Stoyenoff and others 1997).

Street tree assessments were conducted in New Jersey (New Jersey Department of Environmental Protection 1996), Delaware (Valenti 1998), and Maryland (Cumming and others 2001). The New Jersey and Delaware assessments were statewide, while the Maryland assessment was limited to the most densely populated eastern counties. All three assessments showed a lack of species diversity in street tree populations. In New Jersey, approximately one-half of street trees constituted four species: three of which were maples. In Delaware, 51 tree species were identified, but 42 percent were maple species. In Maryland, 43 species were identified but nearly 30 percent were maples.

The three State assessments also measured tree conditions in terms of crown condition and damage. In New Jersey, 30 percent of trees inventoried showed signs of crown dieback or, generally, were in poor condition. In Delaware, 46 percent of sampled trees showed some signs of crown dieback, and in Maryland, some damage was observed on nearly 40 percent of trees. The Maryland survey, restricted to roadside trees along public rights of way, showed that only 15 percent of all survey plots had any trees, indicating a significant depletion of the roadside inventory.

While street trees have aesthetic value, they also can be a source of conflict in terms of sidewalk and road upheavals, power line conflicts, and tree hazard potential. In New Jersey and Delaware, roughly one-quarter and two-thirds of trees were associated with sidewalk damage, respectively.

STRESSORS

Chapter 6.

Air Pollution

Kenneth W. Stolte and John W. Coulston
Southern Research Station, USDA Forest Service

Stressors

Public and private concerns about forest health and sustainability often are expressed in terms of the topic 'forest stress.' discussed because stressors are found in forest ecosystems. Indeed, without forest stressors forest health hardly would be an issue. But what is a stressor, how do we determine whether or not it is significant, and what stressors are important when discussing forest health and sustainability? One dictionary defines stress (Stein 1982) as:

- A physical pressure, pull, or other force exerted on one thing by another
- The action on a body of any system of balanced forces whereby strain or deformation results
- The amount of such action, usually measured
- Any stimulus that disturbs or interferes with the normal physiological equilibrium of an organism
- To make excessive demands upon
- Any force or pressure tending to alter shape, cause a fracture, etc.
- A deformation of a body or structure as a result of an applied force

So, stress is a force, action, or substance that causes a negative or undesirable event because of the type, magnitude, or duration of that event, or its synergy with other forces, causing strain, deformation, change, or damage to an object or unit. Whether a force becomes a stress is determined by its magnitude, and the resistance or resiliency of an object or system encountered by that force. For example, is an object's or system's capacity to resist or tolerate changes caused by external forces sufficient not to cause negative effects?

Therefore, the stability of an ecological community depends on the homeostatic responses of the constituent species to forces (Ricklefs 1979). Healthy forest ecosystems exist within the constraints of long-term physical and biological factors that control the type, number, and condition of species that can grow, reproduce, and complete their life cycles.

Figure 2 conceptually demonstrates how a variety of forces, actions, or objects can interact with forest ecosystems. The observed responses indicate changes and stasis or movement in either a non-sustainable direction (e.g., changes in nutrient cycling, reductions in size); or, conversely, movement or stasis in a sustainable direction (e.g., resistance to change, network integrity).

Forest ecosystems typically are altered by one or more of three major groups of stressors:

1. A broad group of biotic or biological influences that are new to a forest ecosystem (e.g., introduced exotic invasive species)

2. A broad group of abiotic or physical influences new to a forest ecosystem (e.g., air pollution, change in climatic temperatures or precipitation)

3. Biotic or abiotic influences that have co-evolved with forest ecosystems for many centuries or millennia, but have increased significantly in magnitude (severity, extent, and/or duration) and thus the influence or effect is new (e.g., native insects and pathogen epidemics; increased severity of weather events like drought)

New influences, such as air pollution, exotic species, and many others, can combine with epidemic outbreaks of established native insects, catastrophic fires, and other disturbances, to produce a stressor complex(es) to which extent forest ecosystems have not adapted; and these new stressor combinations may, potentially, significantly alter forest ecosystem structure and function. Conversely, normal fire regimes, native insects and pathogens, damage from storms, and other disturbance events, which are within historic levels of severity, usually perform a necessary function in maintaining healthy and sustainable forest ecosystem. Indicators of components and processes that show a measurable response to stressors provide information on the severity of impacts on the general condition and functioning of forest ecosystems. The vitality of a forest ecosystem can be estimated by the intensity of impact to, or alteration of, forest components and processes (e.g., growth, reproduction, mortality) caused by stressors or stressor complexes.

Forest vitality, one of the major criteria presented in the Santiago Declaration (Anon. 1995a, 1995b), was a primary focus area of the FHM program in 1990. Beginning in 1993, the Montreal Process eventually led to the signing of the Santiago Declaration in 1996. At that time, many FHM indicators initially developed in 1990 and later proposed to

MAIA stakeholders, are the same as the Montreal Process Criteria and Indicators (table 1). Other indicators developed for and used by the FHM and FIA programs, such as crown dieback, tree damage, mortality volumes, were not explicitly identified as indicators of forest vitality in the Santiago Declaration but fit into the general category of biological indicators of key processes that, when disrupted, can lead to changes in forest health and sustainability.

Airborne pollutants may affect forest resources by varying degrees in the MAIA region. Forest resources may be exposed to air pollutants deposited in the forest environment but cause no measurable effects, have slight effects but little impact on overall ecosystem functioning, or cause serious impacts that disrupt key ecological processes and thus affect many attributes of an ecosystem (Smith 1974). We have begun a process to determine what effects, if any, are occurring or have occurred in the MAIA region as a direct result of exposure to air pollutants. Our first step was to identify ecoregion sections that have repeatedly been subjected to high ozone exposure or relatively high levels of wet ion deposition, although we did not attempt to establish relationships between air pollutants and forest condition, other than by relating ozone exposures to injury on ozone bioindicator plants (USDA Forest Service 1995). We are primarily interested in long-term average, cumulative exposures of ozone or wet deposition of ionic air pollutants which possibly would help researchers interpret any future changes in forest ecosystems attributed to air pollution.

To evaluate ozone air pollution, we used the W126 index values developed by Lefohn and Runeckles (1987) to average 4 years of data (1993 to 1996) and identify ecoregion sections that consistently had received the highest average annual ozone exposures. TheW126 index statistically weighs acute, episodic ozone exposure peaks, i.e., when hourly ozone concentrations exceed 0.10 ppm (parts per million), with less elevated, chronic ozone exposures to obtain an overall ozone severity exposure index. This index reflects the detrimental effects that high, acute ozone exposures, interspersed with elevated but lower chronic ozone exposures, have on susceptible bioindicator plant species (documented susceptibility to ozone air pollution). We then compared the 4-year average W126 ozone exposures to the amount of plant injury (symptoms on foliage unique to ozone damage) found on bioindicator plants.

We also analyzed average annual wet deposition of known toxic ions reported by the National Atmospheric Deposition Program (NADP) (Lynch and others 1996), between 1979 and 1995. Our purpose was to identify where wet ion deposition was the highest and, therefore, had a high potential to affect forest ecosystems sensitive to such deposition.

W126 Ozone Exposures and Bioindicator Plant Responses

Ozone is a naturally occurring component of the upper and lower atmosphere. Ozone in the lower atmosphere injures susceptible plant species by entering the leaves during photosynthesis. It can be detrimental to forest ecosystems if exposures are high enough to damage leaves of susceptible species. The amount of leaf injury determines whether significant reductions in net photosynthesis and impairments of other physiological processes are occurring (Miller and others 1996). Plant species shown to be susceptible to ozone exposure are known as ozone bioindicator species (Manning and Feder 1980). As with other plant injuries or diseases, the presence of any amount of ozone injury is of concern. Foliar injury from ozone indicates that exposures were high enough to injure bioindicator species, and also may have injured other species found at the same site that had not been monitored for ozone injury (because they are inaccessible trees, or not yet identified as susceptible with definitive injury symptoms).

If any of the susceptible species on a site are keystone species, i.e., species identified as vital to a particular vegetative association, there is potential for significant changes in the affected ecosystem (Smith 1974). For example major impacts from ozone air pollution occur in some mixed-conifer forests in the southern California mountains because of the high ozone susceptibility of ponderosa and Jeffrey pines, which are major components of the lower elevation mountain forests (Miller and others, 1996). The final assessment report of the National Acid Precipitation Assessment Program (NAPAP) characterized ozone as “the pollutant of greatest current concern with respect to regional scale impacts on North American forests” (NAPAP 1991).

FHM and FIA have developed a system of ozone biomonitoring ground plots throughout the U.S., including much of the MAIA region (USDA Forest Service 1995). The sites contain ozone-sensitive trees, shrubs, or herbs that are bioindicators for ozone air pollution, and identified areas where ozone exposures had been sufficiently elevated to injure susceptible species (Manning and Feder 1980). To evaluate ozone as a stressor in MAIA forests, we calculated the average ozone exposures between 1993 and1996. We used the W126 index, a biological index that weights hourly tropospheric ozone concentrations to measure their

effects on ozone sensitive plants (Lefohn and Runeckles 1987). The W126 index was then summed over the growing season to reflect levels of tropospheric ozone harmful to susceptible plant species in the Southern Appalachian region (SAMAB 1996). The W126 ozone exposures and the FHM bioindicator plot responses were mapped to compare ozone exposures with ozone-induced foliar injury on ozone-susceptible plant species.

The Southern Appalachian Man and the Biosphere (SAMAB) program (SAMAB 1996) correlated W126 values, and the number of hours when ozone exposures equaled or exceeded 0.10 parts per million (ppm), with exposure effects for selected Southern Appalachian tree species (table 7). Minimal or no response of forest trees were found when ozone exposures are growing season W126 index values of less than 5.9 or 6 hours of less at 0.10 ppm. Level 1 responses (only highly susceptible species affected, such as black cherry) occurred at growing season W126 index values between 5.9 and 23.7. Level 2 responses occurred on only highly susceptible and moderately susceptible species affected, and Level 3 responses occurred on all ozone-susceptible species, even tree species considered moderately tolerant of ozone, such as red oak (table 7).

Spatial interpolation of ozone exposures from 1993 to 1996 showed that average annual ozone exposures were relatively high (Level 2) throughout most of the MAIA region (fig. 16). While the distribution of average ozone exposure throughout the region was relatively uniform, the amount of ozone-induced foliar injury on sensitive plant species differed throughout, with the highest levels (Bioindicator Response Category Level 3) found mostly in the northern and western parts of the region (fig. 16). Injury to ozone-susceptible species was highest in 1998, with ecoregion sections 221I – Southern Cumberland Mountain, 212G – Northern Unglaciated Allegheny Plateau, M221B – Allegheny Mountains, and 221D – Northern Appalachian Piedmont (Bailey, 1995) (fig. 5) containing many plots with relatively high ozone injury index values (>25), indicating severe foliar injury (Stolte and others, 2005). The difference in response of ozone-sensitive plants to relatively homogeneous levels of ozone exposure is likely due to site factors primarily affecting plant uptake of ozone, such as varying soil moisture and nutrients common

Table 7—Eastern U.S. tree species susceptibility to ozone air pollution and range of exposures inducing ozone-injury foliar symptoms circa 2000

Response Category[a]	Common Name	Genus and Species	Effects Range W126	Effects Range Hours ≥ 0.10 ppm
Level 1	black cherry	*Prunus serotina*	≥ 5.9 to 23.7	≥ 6 to 50
	slash pine	*Pinus elliotti*		
Level 2	green ash	*Fraxinus pennsylvanica*	≥ 23.8 to 66.5	≥ 51 to 134
	sycamore	*Platanus occidentalis*		
	tulip poplar	*Liriodendron tulipifera*		
	white ash	*Fraxinus americana*		
	white pine	*Pinus strobus*		
Level 3	American beech	*Fagus grandifolia*	≥ 66.6	≥ 135
	loblolly pine	*Pinus taeda*		
	pitch pine	*Pinus rigida*		
	red maple	*Acer rubrum*		
	red oak	*Quercus rubra*		
	shagbark hickory	*Carya ovata*		
	Virginia pine	*Pinus virginiana*		
	white oak	*Quercus alba*		

[a]The Southern Appalachian Assessment, Atmospheric Technical Report, July 1996 has references of studies on forest tree sensitivity to ozone that were used to compile this list of tree species.
Source: Lefon and Runeckles 1987; Lefon and Foley 1992; The Southern Appalachian Assessment—Atmospheric Technical Report 1996.

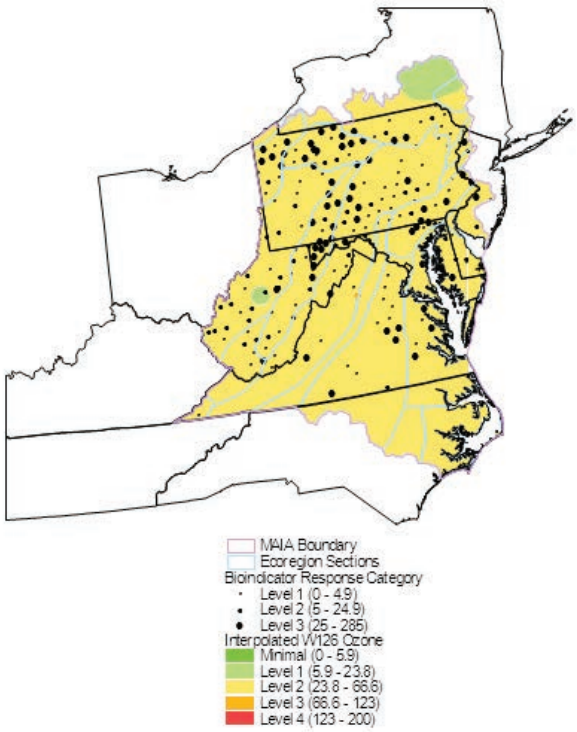


Figure 16—Average annual ozone concentrations (W126 index values) in the MAIA region for 1993 to1996 (colored polygons). Legend also shows ozone concentration values for other parts of U.S. for comparison. Closed circles are average ozone injury index values recorded at FHM ozone biomonitoring sites around the same time. Source: Ozone data from EPA database (http://www.epa.gov/); ozone bioindicator response data from USDA Forest Service National Forest Health Monitoring program; (http://fhm.fs.fed.us/).

in areas with varying topography, and other factors that affect gas exchange within plant leaves.

The potential for elevated ozone levels to alter forest ecosystem structure and function was higher in these ecoregion sections than in other areas of the MAIA region. The potential for reduced photosynthesis and growth of black cherry, green ash, sycamore, tulip poplar, white ash, and white pine tree species in the region is high, particularly in areas where soil moisture is adequate for foliar uptake of ozone during the normal gas-exchange processes of photosynthesis. That is, the average ozone exposures in most of the MAIA region from 1993 to 1996 were high enough to have affected photosynthesis and growth of these ozone-susceptible tree species.

An estimate of the potential loss of black cherry growth—one of the most ozone-susceptible (table 7) and commercially valuable tree species—due to ozone exposure was made by comparing 1990 ozone levels with the distribution and volume of black cherry in the Region. Our analysis indicated that the total biomass loss of black cherry due to ozone exposure could range from 0.1 to 25 million green pounds (mgp) in some eastern counties to about 3,250 to 12,000 mgp in other western and northern counties (fig. 17). This analysis emphasized the potential threat of ground-level ozone to the productivity and diversity of ozone-susceptible species in the MAIA region.

Research efforts included investigating relationships among bioindicator plant responses to ozone and impacts to forest ecosystems in the surrounding areas. Our goal was to relate ozone injury to bioindicator plant species to growth, reproduction, or mortality of trees and other plant species in the area. Additional monitoring and research are needed to understand the full impacts of ozone exposure on forest ecosystems in the MAIA region.

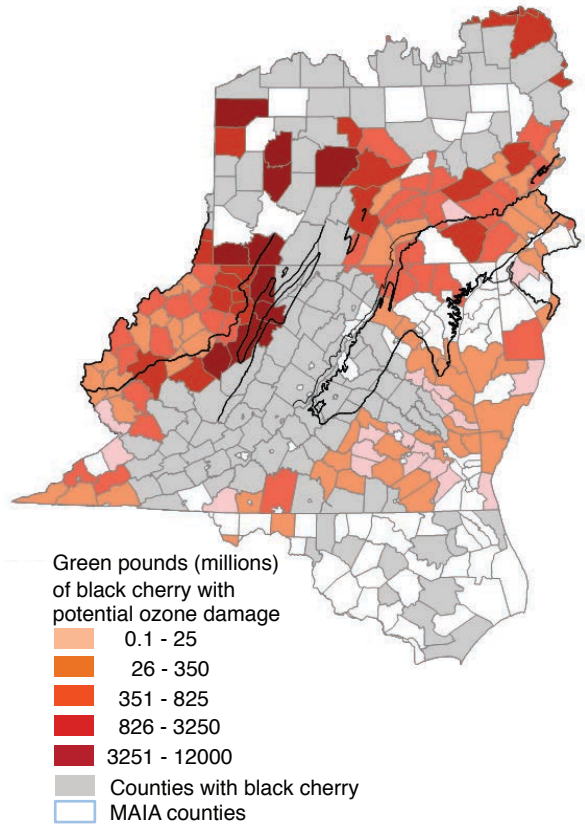


Figure 17— Potential biomass loss in black cherry in 1990 where the distribution of this species overlapped areas with phytotoxic ozone exposures in the MAIA region. Source: North Carolina State University 1996; (http://www.ncsu.edu/).

Wet deposition of ions

Wet deposition of nitrogen and sulfur-based ions and/or acidified precipitation may be damaging forests through direct impacts on tree foliage and indirect impacts on soil acidity, nutrient cycling, soil fertility, fine tree roots, and tree nutrition (NAPAP 1991). The deposition of these ions can acidify forest soils, mobilize toxic aluminum and other trivalent cations, reduce the uptake of nutrient cations, and leach nutrient cations and toxic aluminum from soils into aquatic systems (Tomlinson 1990). In trees, reduced calcium—an element that is poorly translocated in tissues of some tree species—may result in dieback of the growing parts of the crown. The loss of potassium and magnesium—both readily translocated within tree tissues—results in the loss of older foliage as these nutrients are moved from older tissues to the growing points of the tree. Depending on severity, loss of foliage can result in increased transparency of the tree crown. The long-term effects and the spatial distribution of areas affected are still uncertain.

For several decades, there has been significant concern about effects of air pollution on forest health, e.g., decreased growth, increased mortality, and change in species diversity. When serious, such effects may be classified *declines*; however, a major conclusion in the final assessment report of NAPAP was that "the vast majority of forests in the United States are not affected by decline," and that "no consistent relationship between forest health and acid deposition could be found." (NAPAP 1991) In localized high elevation forests with red spruce species, acid deposition has played an exacerbating role in the decline of already stressed forests.

The NAPAP assessment recommended that ground-based monitoring should be conducted in fixed-area plots and include periodic measurement—not only of tree growth but also soil properties, understory vegetation, insects, pathogens, and other ecosystem components (NAPAP 1991). Such efforts would aid in the interpretation of tree and canopy information. NAPAP recommendations have become an integral part of the Detection Monitoring components of the FIA and FHM programs, as well as the Intensive Site Monitoring component of FHM. Nationally standardized annual estimates of tree growth, crown condition, tree mortality, lichen species richness, physical and chemical properties of soils, and insect and disease sketch-mapping have been incorporated into the FHM and FIA programs. Such estimates can be used in future analyses to evaluate the condition of forests in areas exposed to relatively high depositions of toxic ions and tropospheric ozone concentrations.

The spatial interpolation of data about wet deposition of sulfate, nitrate, and precipitation pH from the National Atmospheric Deposition Program, National Trends Network, and Canadian Air and Precipitation Monitoring programs for the period from 1979 to1995 provided us with an estimate of forest area in the MAIA region receiving increased wet atmospheric deposition in kilograms per hectare per year (kg/ha/yr).

Average annual wet deposition levels were relatively high in the central-western and northwestern parts of the region (fig. 18). Wet sulfate deposition was relatively high (28.9 to 36.6 kg/ha/yr) in ecoregion sections M221A (Northern Ridge and Valley), M221B (Allegheny Mountains), 221E (Southern Unglaciated Allegheny Plateau), and 212G (Northern

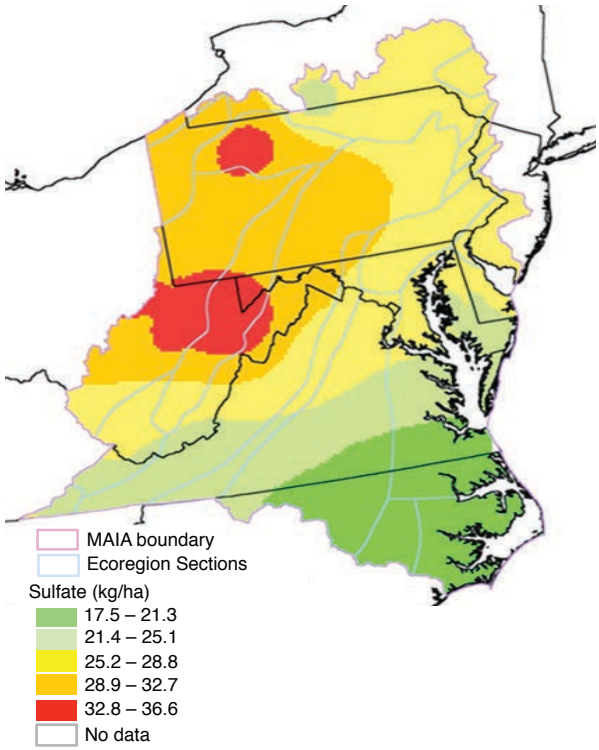


Figure 18—Average annual wet deposition of sulfate in MAIA region from 1979 to 1995. Data from this16-year period were averaged and interpolated at 5-km grid scale to estimate average annual wet deposition at ecoregion section spatial scales. Source: 1996 National Atmospheric Deposition Program's National Trends Network; (nadp.sws.uiuc.edu/).1996 Canadian Air and Precipitation Monitoring; (http://www.msc-smc.ec.gc.ca/capmon/index_e.cfm).

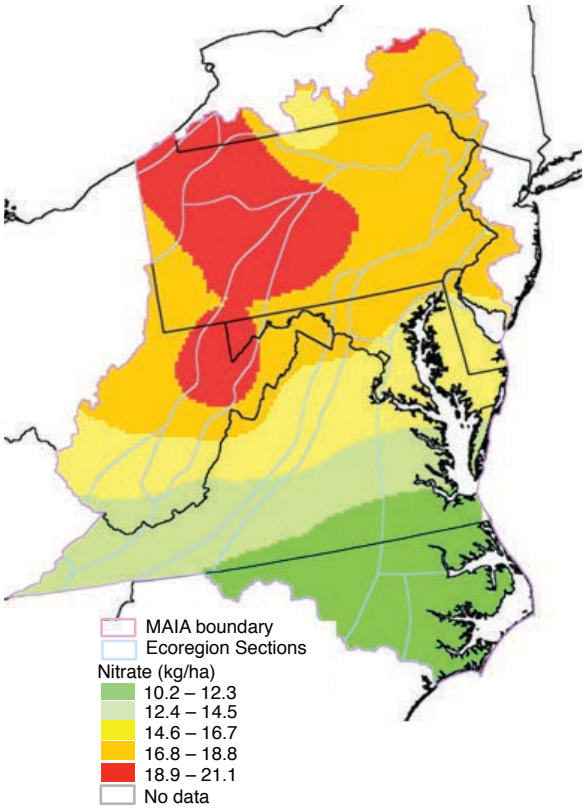


Figure 19—Average annual wet deposition of nitrate in MAIA region from 1979 to 1995. Data from this16-year period were averaged and interpolated at 5-km grid scale to estimate average annual wet deposition at ecoregion section scales. Source: 1996 National Atmospheric Deposition Program's National Trends Network; (nadp.sws.uiuc.edu/); 1996 Canadian Air and Precipitation Monitoring; (http://www.msc-smc.ec.gc.ca/capmon/index_e.cfm).

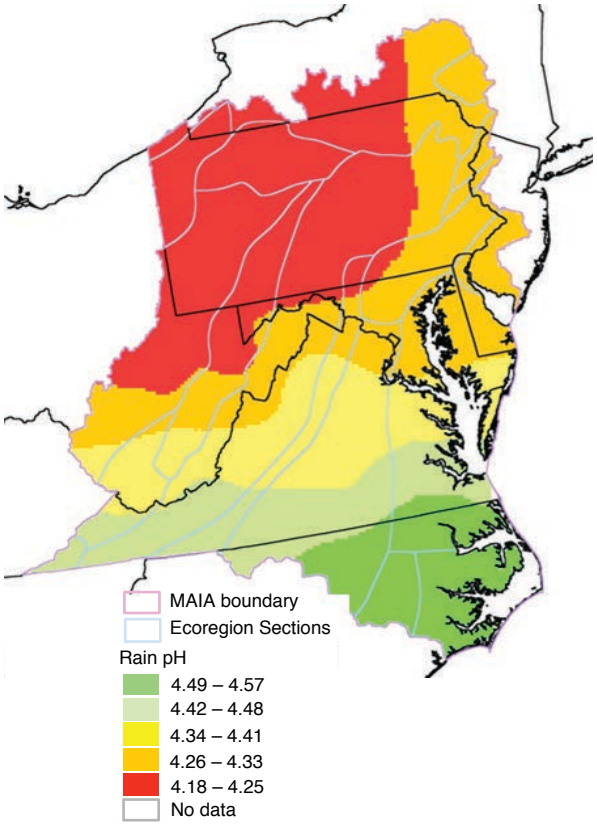


Figure 20—Average annual precipitation pH in the MAIA region from 1979 to 1995. Data from this 16-year period were averaged and interpolated at 5-km grid scale to estimate average annual wet deposition at ecoregion section scales. Source: 1996 National Atmospheric Deposition Program's National Trends Network; nadp.sws.uiuc.edu/ and 1996 Canadian Air and Precipitation Monitoring; (http://www.msc-smc.ec.gc.ca/capmon/index_e.cfm).

Unglaciated Allegheny Plateau), and 221F (Western Glaciated Allegheny Plateau) (Bailey, 1995).

Wet nitrate deposition also was high (16.8 to 21.2 kg/ha/yr) in these same ecoregion sections (Bailey 1995) and, additionally, extended north and east into ecoregion sections 212F (Northern Glaciated Allegheny Plateau), M212E (Catskill Mountains), 221B (Hudson Valley), 221A (Lower New England), 221D (Northern Appalachian Piedmont), and 221C (Upper Atlantic Coastal Plain) (fig. 19).

The average annual pH of precipitation was relatively low (4.18 to 4.34 pH) in all the northern half of the MAIA region (fig. 20), with large areas in the central western and northwestern region receiving the highest annual average acidic precipitation of 4.18 to 4.26. Southeastern areas in North Carolina received the lowest acidic precipitation (4.49 to 4.57). There was a pronounced gradient of acidic precipitation extending north-to-south over the entire region.

The analyses of sulfate, nitrate, and precipitation pH deposition from 1979 to 1995 clearly showed that the north and particularly the northwest areas of the MAIA region received substantially more annual wet deposition then the southern portion of the regions. It is very likely that the 1990 Clean Air Act Amendments (EPA 2008) will significantly reduce the amount of wet pollutant deposition in the MAIA region in the near future.

Sufficient data on forest condition from the FIA and FHM monitoring programs may soon be available to evaluate any relationships that might exist between the wet depositions of ions in the MAIA region (NADP 1998) and the condition of forest ecosystems. As additional data on the condition of the forests become available from these and other forest monitoring programs, perhaps it will be possible to determine whether or not there is any connection between the deposition of these and other ions, the acidity of precipitation, and the condition of forest ecosystems.

Chapter 7.

Fire Regimes

Kenneth W. Stolte and John W. Coulston
Southern Research Station, USDA Forest Service

Historic frequency and severity

Fire has been a powerful regulatory mechanism in forest ecosystems for hundreds of millions of years. It is a natural part of the environment, and fire-dependent ecosystems are adapted to a particular fire frequency and intensity. Forest ecosystems will remain in their natural state only if the fire regime they are adapted to is maintained (Kimmins 1987). Forests have always burned—the frequency and intensity of burning depends on fuel buildup, weather conditions, topography, and the frequency of ignition sources. In earlier centuries, lightning was the primary source of ignition. Influencing either the frequency or intensity of the fire cycle can change the species composition and age structure—as well as soil characteristics—of a fire-adapted community (Kimmins 1987).

Post-colonial settlement people have altered historic fire regimes through activities such as fire suppression, tree harvesting at select stages, and using fire as a silvicultural tool. Historic (pre-Columbian) fire regimes in the MAIA region have been surmised from tree ring analyses and fire scars (fig.21) (Fire Science Laboratory 1999b), and those data have indicated that fires in most of the region burned often (0 to 35 years) and were of low severity, while fires in the northern and southwestern part of the MAIA region burned less frequently (35 to 100+ years) and with mixed severity.

Changed Fire Regime Conditions

Changed condition classes are used to categorize departures from historic fire regimes in terms of 5 ecosystem attributes (Fire Science Laboratory 1999a): disturbance regimes, disturbance agents, smoke production, hydrologic function, and vegetative attributes. Changed condition-class 1 represents a relatively small deviation from ecological conditions compatible with historic fire regimes. Changed condition-class 2 is a deviation from ecological conditions compatible with historic fire regimes that would require some silvicultural management to restore historic conditions. Changed condition-class 3 represents a major deviation from the ecological conditions compatible with historic fire regimes that would require a major effort such as harvesting and replanting to restore historic conditions.

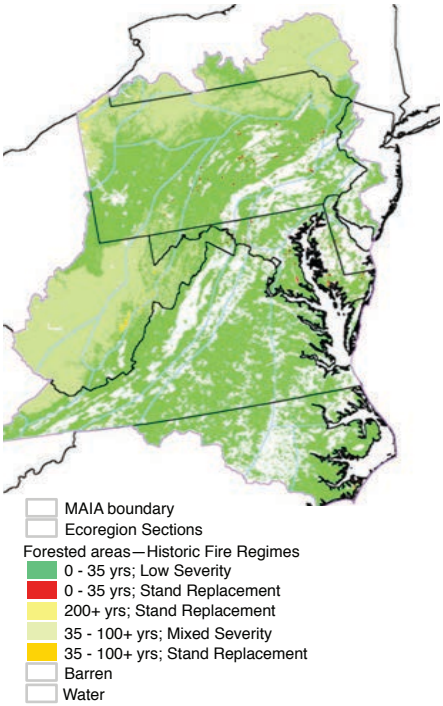


Figure 21—Historic fire regimes constructed from tree ring analyses and fire scars in the MAIA region. Source: USDA Forest Service; (http://www.fs.fed.us/fire/fuelman/index.htm).

Figure 22 shows deviations from ecological conditions compatible with historic fire regimes for forested areas in the MAIA region. Fire regimes in most of the region have been significantly altered, and moderate (changed condition-class 2) to significant (changed condition-class 3) amounts of treatments (burning and/or silvicultural treatments) would be necessary to restore the forests to historic fire regimes (fig. 21). Current fire conditions in the MAIA region indicate the most pronounced deviations from historic regimes are in northern Pennsylvania/southeastern New York, eastern West Virginia, eastern Maryland, and the Piedmont section of Virginia, specifically in ecoregion sections 212F (Northern Glaciated Allegheny Plateau), 212G (Northern Unglaciated Allegheny Plateau), southern half of M221B (Allegheny Mountains), M221C (Northern Cumberland Mountains), southwestern part of M221A (Northern Ridge and Valley), 231A (Southern Appalachian Piedmont), and northern part of 232A (Middle Atlantic Coastal Plain) (Bailey1995). Much of the remaining parts of the western and northern parts of the MAIA region

are in condition-class 2 (moderate change in ecological conditions). Only parts of eastern North Carolina and parts of Virginia are in condition-class 1, which means the ecological conditions in these areas are similar to conditions found in historic fire regimes.

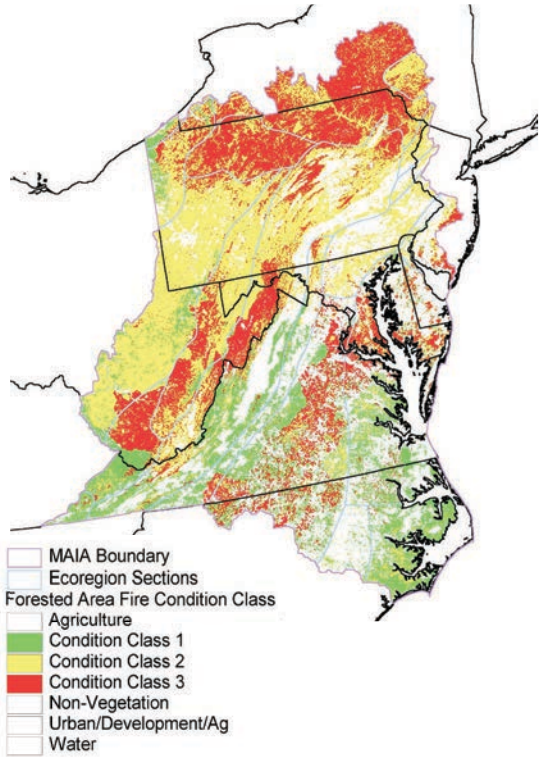


Figure 22— Condition classes of changes in current fire regimes from historic fire regimes in forested areas in the MAIA region. Higher numbered condition classes indicate increasing amounts of silvicultural treatments would be needed to restore historic fire regimes. Source: USDS Forest Service; (http://www.fs.fed.us/fire/fuelman/index.htm).

Chapter 8.

Climate Change

Henry W. Walker
Region 3, Office Research & Development,
US Environmental Protection Agency

Climatic forces constitute the major factor influencing the distribution of forests and other ecosystems, and thus climate is an essential component when delineating ecoregion boundaries (Bailey 1995). Forests in the Mid-Atlantic have been affected by variability in temperature and precipitation on time scales of centuries, millennia, and longer, including the successive glacial advances and retreats in the Northern Hemisphere during the Holocene period (1.8 million years ago to present). Evidence of such longer-term climate variability comes from ice and marine sediment cores, and from pollen counts in freshwater and marine sediment cores (Brush 1984b). Pollen counts also help document related redistribution and alteration of species composition in forests on both sides of the North Atlantic (Brush 1986).

Additional evidence of inter-annual and longer time-scale climate variability can be inferred from the width, density, and chemical composition of tree rings, which provide information on precipitation, temperature, and other variations in weather patterns. Analysis of tree ring cores indicates that the 20th century was a period of rapid warming (Mann and others 1998, 1999), consistent with other finding that the Earth is in a period of relatively rapid climate change (IPCC 2001).

Large changes in forest composition along the U.S. east coast may result from greenhouse gas increases linked to global climate change (Iverson and Prasad 1998). An analysis by the Mid-Atlantic Regional Assessment (MARA) of potential impacts on the Mid-Atlantic region due to climate variability and change is underway. MARA is a component of the larger U.S. National Assessment of the Potential Consequences of Climate Variability and Change, and will include a detailed assessment of potential impacts on the Nation's forests (National Assessment of Climate Change 2000). Primarily, the National Assessment is based on two state-of-the-art global climate models, one from the Hadley Centre for Climate Prediction and Research in Great Britain, and the other from the Canadian Climate Centre (CCC). Both models estimate transient climate changes over the next 100 years, assuming unconstrained, "business-as-usual" emissions of greenhouse gases and sulfate aerosols (National Assessment of Climate Change 2000).

Both models predict future trends in air temperature change, with increases in maximum air temperature of 4 to 8 °F expected for the Mid-Atlantic region by the end of the 21st century. Potential precipitation changes in the region are much more uncertain. Both models predict slight increases in the frequency and intensity of winter storms; however, the spatial resolution of these two models is not fine enough to predict thunderstorms or hurricanes. Some models with higher spatial resolution for the Mid-Atlantic region (Crane and Hewitson 1998, Jenkins and Barron 1997) project increases in precipitation during the 21st century similar to the larger increases in precipitation predicted by the Hadley model.

A recently released MARA overview report includes a two-page summary of potential positive and negative impacts of climate change on forests in the region (MARA 2000). One major uncertainty is whether trees, as relatively long-lived organisms, will be able to migrate in pace with the anticipated rate of climate change. Forest migrations, which are limited by necessarily related adjustments in forest species composition, can take longer than anticipated changes in climate.

To evaluate possible long-term changes in the region's forests, we constructed scenarios using new equilibrium climate conditions anticipated under doubled CO_2 atmospheric concentrations. The results suggest that forest growth could increase in the region, but that the "mix" of species under new equilibrium climate conditions likely would change. In addition, potential increases in extreme climate events could negatively affect forest production. Although there is still a high degree of uncertainty, it appears that a relatively rapid shift in dominant forest types might foster invasive species and decrease biodiversity. Coldwater fish (e.g., trout) and certain types of birds are among the species likely to be adversely affected (MARA 2000).

A more detailed study of potential effects of climate variation and change on both forests and the practice of forestry is based on a regional forest management survey and uses results from five different climate change models. those models use scenarios run to equilibrium under

doubled atmospheric CO_2 concentrations (McKenney-Easterling and others 2000). The survey results indicate that forest operations are more likely to be affected by severe weather—including high winds and extremes in precipitation—than by temperature extremes. Results from all five models suggest there is a potential for a new equilibrium regional climate that is generally hotter and drier. In a region better suited to moist conditions, this could lead to large reductions in tree species in forest types such as the maple-beech-birch forest in the northern and higher-elevation western portion of the MAIA region (fig. 23), and the oak-gum-cypress forest in the southern coastal portion (MARA 2000, McKenney-Easterling and others 2000). An online database is available at http://www.nrs.fs.fed.us/atlas/ with maps and geographic distribution centers for 134 species of trees and 10 major forest types in the East. The site presents current and possible future distributions based on 3 widely-accepted climatic change models (Prasad and Iverson 2000).

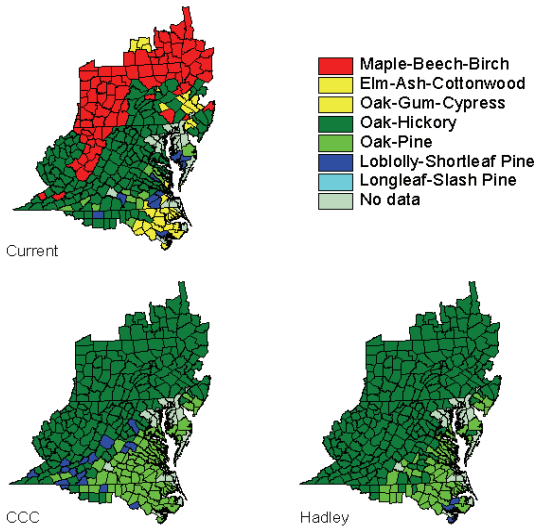


Figure 23— Dominant forest types under circa 2000 climate regimes, and potential forest type distributions under Canadian Climate Center and Hadley doubled CO_2 equilibrium climate scenarios. Source: Mid-Atlantic Regional Assessment (MARA) Team; (http://www.nhbs.com/mid_atlantic_regional_assessment_of_climate_change). Pennsylvania State University; (http://www.psu.edu/).

Chapter 9.

Insects and Pathogens

William G. Burkman and Edward Yockey
Southern Research Station, USDA Forest Service

Native insects and pathogens (primarily fungi) have been a natural selective force in forests for millennia, and are a balanced part of a healthy ecosystem. However, their populations can reach epidemic proportions and reduce growth or kill otherwise healthy trees. Such outbreaks also can affect other resources valued by society, such as aesthetics, recreation, water, and wildlife. But usually these species are maintained in equilibrium within healthy forest systems, and what are primarily killed include weakened and senescent trees, which makes room for new, vigorous forests. Insects and fungi also help recycle nutrients by decomposing plants to replenish the soil with vital nutrients necessary for plant growth, and thus are essential to stable, healthy forest ecosystems.

The natural loss of trees through native insect and pathogen activities often becomes a source of contention among the human population, which needs lumber and other wood products. Human use and management activities also change the way native organisms affect forest ecosystems, sometimes causing unexpected and unwanted consequences. Fusiform rust and several root disease organisms exemplify how organisms of relatively little consequence in the past now cause severe and widespread reductions in tree growth and increased mortality. Forest resource managers face the continuing challenge of meeting society's needs within the broader and inevitable context of natural insect and pathogen cycles.

The threat of invasive species (insects, pathogens, plants, and animals) being introduced into the United States will increase as global interactions increase, solidifying a continuing threat to U.S. forests. Introduced species are not integral to native forests, have not evolved as part of indigenous forest ecosystems, and often cause new and often devastating effects that can change forest ecosystems forever. For example, chestnut blight, white pine blister rust, gypsy moth, Dutch elm disease, and beech bark disease have disrupted major forest ecosystems by greatly reducing or eliminating keystone tree species from their native habitats.

Insects

Gypsy moth is an introduced species that has become one of the most troublesome defoliating insect in the eastern United States. In addition to its impacts on forest ecosystems, the insect is a serious "people pest," requiring the expenditure of millions of dollars for control throughout the MAIA region and elsewhere east of the Mississippi. Much of this control is focused on urban and other forested environments used by people for recreation, and not on more-typical forest ecosystems. Since it was introduced into the U.S. in 1869 by a French naturalist living in Massachusetts, the gypsy moth has firmly established itself in the Northeast, and has spread south and west to become a significant pest in all MAIA States (fig. 24).

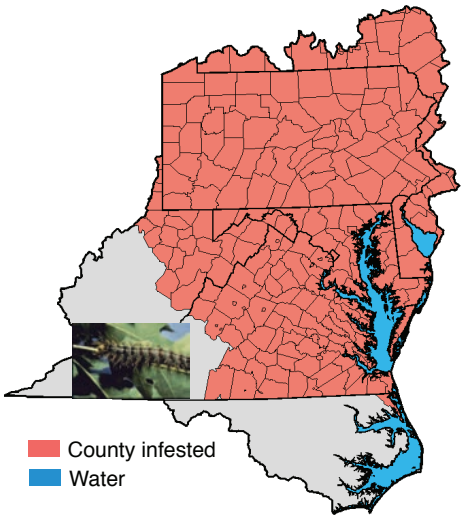


Figure 24—Gypsy moth infested areas in MAIA counties through 1998 with gypsy moth caterpillar photograph (inset). Source: USDA Forest Service, Forest Health Protection; (http://www.fs.fed.us/foresthealth/).

In 5 years this common pest defoliated over two million acres in States with areas in the MAIA region. Land in the Commonwealth of Virginia alone accounted for about 63 percent of the total cumulative defoliation after 1994 (table 8), and extensive defoliation occurred there in 2 years (1994 and 1995). Although gypsy moth defoliation was down for three subsequent years, the infested area continued to expand slowly in a southwesterly direction, eventually spreading to roughly 75 percent of counties in the MAIA area, i.e., approximately 3.3 percent of all forested land. As insect damage spreads, changes in the composition of forest types followes. Favored species (e.g., the white oak species group) have declined as a percentage of the forest cover, to be replaced by non-favored species such as yellow poplar and red maple.

Table 8—Gypsy moth defoliation in MAIA region states circa 2000

State	1994	1995	1996	1997	1998	Cumulative
	acres					
Delaware	60,728	65,462	534	n/a	n/a	126,724
Maryland	93,147	93,864	11,148	596	448	199,203
North Carolina	*n/a*	*n/a*	*n/a*	*n/a*	*n/a*	*n/a*
New York	480	200	16,285	2,200	9,455	28,620
Pennsylvania	17,957	132,487	9,027	2,292	31,611	193,374
Virginia	452,475	850,000	*n/a*	*n/a*	*n/a*	1,302,475
West Virginia	50,257	102,971	70,726	476	650	225,080
Totals	674,594	1,244,984	107,720	5,564	42,164	2,075.026

n/a = data not available.
Source: The USDA Forest Service, Forest Health Protection program; http://www.fs.fed.us/foresthealth/.

Gypsy moth infestations alternated between 2- to 4-year periods when moth populations were scattered, and there was little noticeable defoliation, to periods when moth populations were high and defoliation was very apparent. The insect has four life stages: egg, larva, pupa, and adult. Only the larvae stage (caterpillars) damages trees. Short-range dispersal takes place as larvae are wind-blown on silken threads. Long-range dispersal occurs through inadvertent human transport via vehicles, relocated lawn furniture, transport of firewood, etc. The female of the European variety does not fly.

Gypsy moths generally favor hardwood host species, such as oaks, apple, sweet gum, alder, basswood, gray and white birch, poplar, willow, and hawthorn. Less favored species are Eastern cottonwood, hemlock, Atlantic whitecedar, pines, and Eastern spruces. The gypsy moth clearly dislikes ash, yellow poplar, redcedar, American holly, maples, and certain shrubs such as mountain laurel, arborvitae, and rhododendron.

Impacts of gypsy moth depend on the severity and timing of defoliation, as well as the hosts' ability to tolerate the damage. Trees growing on poor, dry sites, for example, are less likely to survive than those on fertile, moist sites. If < 50 percent of the crown is defoliated, most trees are able to produce a second flush of foliage by midsummer. Healthy trees are typically able to tolerate one or even two consecutive defoliations of over 50 percent. Nevertheless, defoliation predisposes trees to attack by secondary insects and pathogens—such as the two-lined chestnut borer and the *Armillaria mellea* fungus—that can cause or contribute to tree death.

Natural and artificial controls play an important role in holding gypsy moth populations in check. Among the natural enemies are mice, birds, raccoons, shrews, parasitic and predaceous insects, and other arthropods such as spiders. One of the most significant and exciting natural checks is the soil-borne fungus *Entomophaga.* This insect-attacking fungus is credited with causing a recent collapse of gypsy moth populations throughout the insect's Eastern range. Nonetheless, there is some evidence that populations are again rebuilding in a few localized areas.

Although insidious and sometimes relatively slow, gypsy moth infestations can result in pronounced changes in native forest ecosystems. As favored host trees are depleted through defoliation, the subsequent effects on fauna and other flora can be dramatic. Overstory defoliation can result in sudden and intense sunlight reaching the forest floor, causing a dramatic increase in shade-intolerant vegetation. This, in turn, can lead to population shifts that favor some species and harm others. For example, white-tailed deer populations might increase due to increased forage, while black bear might suffer due to loss of acorn production and hibernation sites. Although less obvious, effects on invertebrates and plants can be just as significant.

Scientists are hopeful that the gypsy moths' natural enemies such as the *Entomophaga* fungus will help hold populations in check. Nevertheless, entomologists and biometricians project that by the year 2025, all counties in the MAIA States, except extreme southwestern Virginia and western North Carolina, will be completely infested with gypsy moth.

Hemlock woolly adelgid is an introduced insect first discovered in Virginia in 1956. Since then, it has spread both northeast and southwest along the spine of the Appalachian Mountains (fig. 25). Although a relatively minor pest in its native Asia, the hemlock woolly adelgid presents a significant threat to Eastern hemlock and Carolina hemlock in every MAIA State, because of the potential to

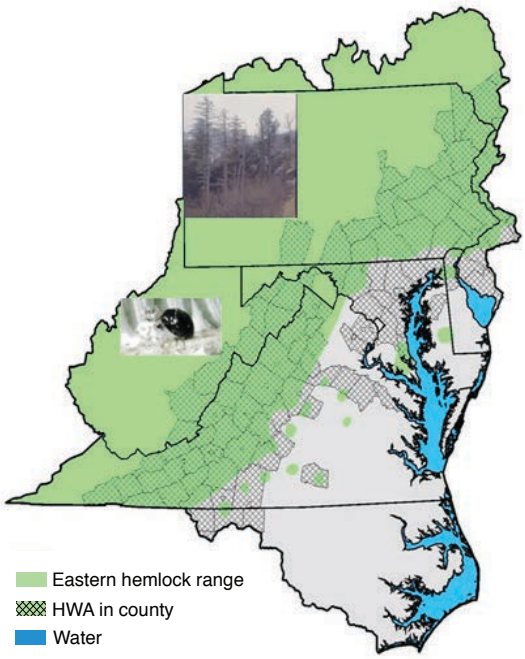


Figure 25—Hemlock woolly adelgid (HWA) infested areas in MAIA counties through 1997. Photographic insets show hemlock trees killed by HWA, and biological control ladybird beetle feeding on HWA. Source: USDA Forest Service, Forest Health Protection; (http://www.fs.fed.us/foresthealth/).

wipe out its host in a manner reminiscent of the chestnut blight. The probable loss of hemlock trees will have serious consequences for the composition and functioning of Appalachian forests, especially in riparian areas bordering streams, rivers, and lakes.

The insect itself is extremely small and aphid-like, covered with a white, woolly, waxen *floc* that it secretes. The hemlock woolly adelgid size and woolly egg masses facilitate long-distance dispersal by wind and migrating birds. In the Eastern U.S., the hemlock woolly adelgid attacks Eastern hemlock and Carolina hemlock, feeding on the young branches and twigs of its host, normally near the needles. All stages of the insect suck sap from the host, apparently injecting a saliva-borne toxin into the hemlock as it feeds. Late winter and early spring feeding by the adelgid can retard or prevent the spring flush of new needles. Symptoms of infestation include yellowing, desiccation, and death of older needles. Dieback of major limbs can occur within 2 years, progressing from the bottom of the tree upward.

The ecological consequences of hemlock loss may be profound. A dense hemlock canopy creates obvious and distinctive microclimates and an acidic duff layer, conditions conducive to the development of unique plant and animal communities. The cooling effect on the forest floor from hemlock canopies can be dramatic, with temperatures significantly lower than in adjacent open areas or even in nearby hardwood stands. Hemlocks also provide a cooling effect for mountain streams critical to the survival of organisms ranging from arthropods to brook trout. Several endangered plant and animal species are found in forest types where hemlock is an important or primary component of the ecosystem. Impacts from hemlock loss can be almost immediate—or long-term through the alteration of natural successional processes. With no other tree species suited to closely fill its ecological niche, the threat of disappearing hemlock stands is particularly alarming—both from an aesthetic and an ecological perspective.

The hemlock woolly adelgid continues to spread relatively rapidly, and infestations now cover about one-fourth of the hemlock's range in the eastern U.S. Much of the successful establishment and rapid spread of the hemlock woolly adelgid in North America has been attributed to a lack of natural predators and parasites. In Asia, hemlock woolly adelgid populations are largely held in check by natural enemies, and the insect is rarely troublesome unless trees are growing on poor sites or are otherwise exposed to some stressful condition.

Despite the formidable threat posed to the eastern hemlock forests by the hemlock woolly adelgid, scientists are working hard to ameliorate its impact. Improved detection and monitoring methods will allow us to quickly identify areas of new infestation and respond with the best technology available. Intense efforts are underway to identify and possibly introduce natural enemy species from Asia. In high-value stands such as are found in recreation areas, efforts are continuing to screen environmentally acceptable insecticides. Such insecticides might enable us to keep the hemlock gene pool intact until controls are available for the general forest environment.

The southern pine beetle is a native insect that is one of the most destructive insect pests of pine in the Southeastern United States. Although best known for its outbreaks in the Deep South, southern pine beetle is found throughout much of the MAIA region, and is troublesome as far north as Pennsylvania, New Jersey, and the Delmarva Peninsula (fig. 26).

Southern pine beetle infestations occur sporadically throughout the insect's range in groups of infested trees

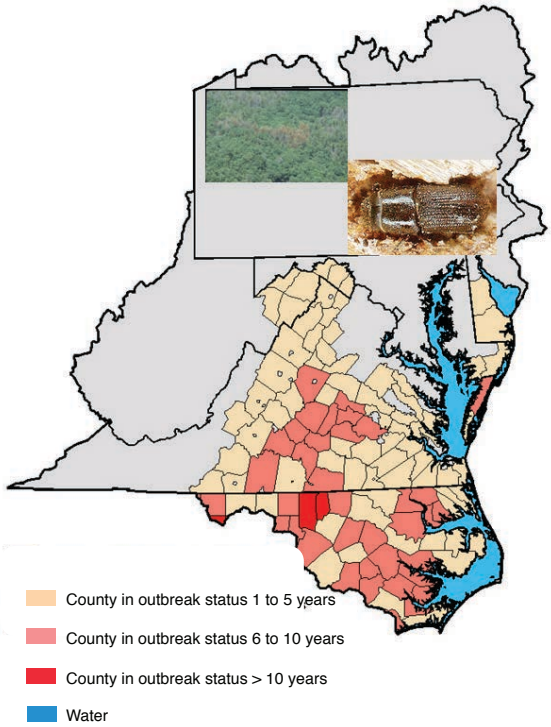


Figure 26—Southern pine beetle outbreak infestations in MAIA counties over 30 year period (ca. 1968 to1998). Photographic insets show an example of resulting tree damage and mortality, and beetle in a carved gallery. Source: USDA Forest Service, Forest Health Protection; (http://www.fs.fed.us/foresthealth/).

called *spots* that are composed of just a few trees or hundreds of acres of forest. Forest entomologists define an outbreak as one or more multiple-tree spots per thousand acres of host type in a county. The southern pine beetle attacks all species of southern pine, but prefers loblolly, shortleaf, pitch, pond, and Virginia pines. In some years infestations are so rare as to be difficult to find; in others, millions of trees are lost throughout much of the insect's range. In most years, southern pine beetle is in outbreak status somewhere in the South. In the 1973 to 1977 outbreak, the beetle killed trees containing an estimated 4.5 billion board feet of sawtimber throughout the southern United States.

Over 5 years, southern pine beetle outbreaks were highly variable in intensity and distribution throughout the South (table 9). Natural enemies such as predaceous and parasitic insects, woodpeckers, and diseases can hold southern pine beetle populations in check during non-outbreak years, but under epidemic conditions these natural controls seem to have little effect. Extremely prolonged cold temperatures,

Table 9—Southern pine beetle infestations in MAIA states circa 2000

State	1994	1995	1996	1997	1998
	acres				
North Carolina	532	3,719	2,825	1,117	769
Virginia	1,094	140	50	91	54
Totals	1,626	3,859	2,875	1,208	823

Note: Southern pine beetle rarely reaches outbreak status in Delaware, Maryland, Pennsylvania, and West Virginia but is present in these States and causes isolated mortality. Source: The USDA Forest Service, Forest Health Protection program; (http://www.fs.fed.us/foresthealth/).

however, can have a dramatic effect. When temperatures drop to 0 °F for several days, southern pine beetle brood mortality can be high. Sustained temperatures at or above 95 °F in the Gulf States have also been shown to kill broods.

The most practical insect controls are preventative treatments. Any effort land managers can make to propagate vigorous, healthy trees will discourage southern pine beetle infestations. In managed areas, stands should be thinned before they become dense and stressed, and harvested before reaching an overly mature stage. Trees struck by lightning or otherwise heavily stressed should be removed wherever possible. In areas subject to flooding, improved drainage can reduce the potential for southern pine beetle population buildup. Roads should be carefully constructed to reduce the possibility of erosion or water table changes.

Since the mid-1980s, private and community interests have been at odds with foresters over management of the southern pine beetle. Large areas of the South have been converted from old fields to vast, unnatural, monotypic, even-aged pine stands. While such areas offer relatively cost-efficient management, they require intensive silvicultural treatments to ward off potentially catastrophic pine beetle buildups. As political pressures mount to prevent clearcutting and even-aged monotypic management, large areas of pine reach physiological maturity simultaneously, creating conditions ideal for southern pine beetle outbreak. Overly mature, dense natural pine stands also favor the buildup of southern pine beetle populations. Some groups support uneven-aged, mixed forest types that will preclude large-scale beetle outbreaks, an approach that is complicated by the need to preserve the red-cockaded woodpecker, an endangered bird that nests in old-growth pines, which are highly susceptible to southern pine beetle attack. Because the bird will not nest in dead trees, land managers are repeatedly confronted with

the dilemma of protecting the endangered species by not cutting old-growth trees, or preventing outbreaks of southern pine beetle by removing them.

Pathogens

Butternut canker attacks butternut, a small- to medium-sized, shade intolerant, relatively short-lived (< 75 years) tree associated with several mixed mesophtytic hardwood forest types in the eastern U.S. Butternut seeds are valued as a food source by both wildlife and humans. The wood is used for furniture, paneling, and carving. Due to the introduction of butternut canker, butternut is being eliminated throughout its range (fig. 27). Butternut canker is thought to be an exotic pathogen because of its rapid expansion and the resulting elimination of butternut from most locales. Butternut canker was first reported in southwestern Wisconsin in 1967, but an examination of killed trees in North Carolina and South Carolina determined that the fungus was present in the southern MAIA region—probably in the 1950s (Schlarbaum and others 1997).

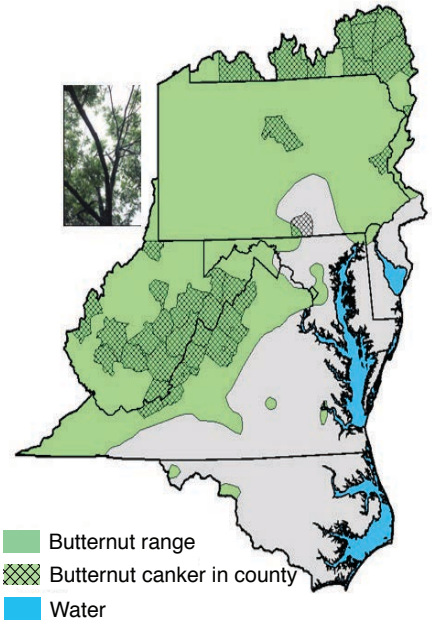


Figure 27—Butternut species range and documented butternut canker distribution. Butternut canker is thought to be distributed throughout the range of butternut but is often not observed because of the scarce and sporadic distribution of Butternut trees. Photographic inset shows an example of a sparse tree crown, the obvious typical first sign of infection. Source: USDA Forest Service, Forest Health Protection; (http://www.fs.fed.us/foresthealth/).

This fungus infects the tree initially through bud, leaf scars, and possibly bark openings. The fungal spores are spread by rainfall and likely also by insects and birds. The fungus also is seed-borne and has been isolated in black walnut husks. Cankers are most commonly found on the main stem, at the base of the tree, and on exposed roots. The fungus kills the trees as the expanding perennial cankers girdle the trunk. The fungus also has been found in black walnut seedlings and is also causing mortality in that species.

A cure for butternut canker has not yet been found, but some of the butternut population remains disease-free or are able to inhibit canker expansion, raising the hope that a viable population of resistant individuals will remain to preserve the biodiversity of eastern forests. The USDA Forest Service (www.fs.fed.us) has placed a harvest restriction on healthy butternuts in national forests. The potential elimination of butternut has serious implications for the biodiversity of eastern forests, and loss of highly valued wood products, especially in the northern range of butternut where black walnut is not found.

Beech bark disease is a disease-complex that results when American beech trees are attacked by the beech scale insect, which provides entry points for invasion by two primary fungi: *Nectria coccinea var. faginata* (Lohman, Watson, and Ayers) and *N. galligena* (Bres.). These two fungi eventually kill the host tree. Beech bark disease causes significant mortality and defect in American beech trees—particularly large, older trees—reducing the prominence of American beech in forest stands.

American beech is a slow-growing, common tree found east of the Mississippi River and generally throughout the MAIA region. It is a major component of three forest cover types and a minor component of 17 others. American beech is a major mast (nut) producer throughout its range; the nuts are eaten by a large variety of birds and mammals. Beech wood is used in flooring and furniture as well as other lumber products. The wood is favored for fuelwood because of its high density and burning qualities.

Beech scale was accidentally introduced into Nova Scotia around 1890. By the 1930s, beech bark disease was causing pockets of mortality in south central Maine and scale insect was found in eastern Massachusetts. Beech bark disease subsequently spread northward into Quebec and west and south throughout New England, New York, New Jersey, and eastern Pennsylvania (fig. 28). An isolated 70,000-acre area was discovered in northeastern West Virginia in 1981, and expanded to include six counties in West Virginia and one county in Virginia.

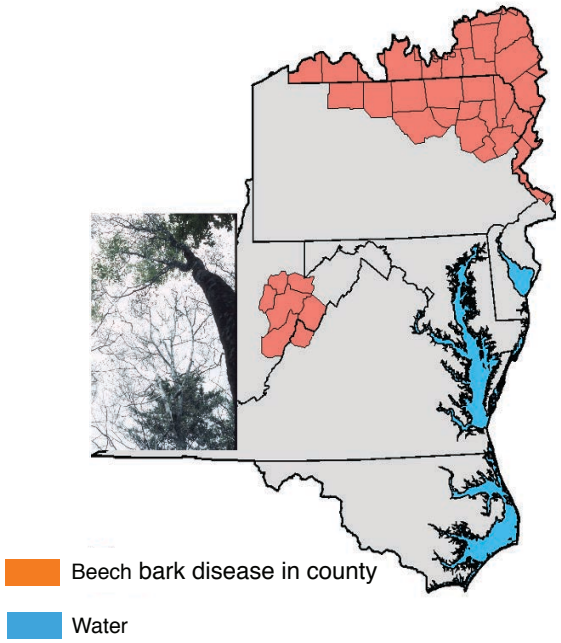


Figure 28—Beech bark disease is a disease-complex of American beech trees attacked by the beech scale insect that creates entry points for invasion by two primary killing fungi: Nectria coccinea var. faginata and N. galligena. The map shows incidence by MAIA counties through 1998. Inset shows an example of thinning tree crowns symptomatic of established beech bark disease infection centers. Note: American beech is generally found throughout the MAIA region. Source: USDA Forest Service, Forest Health Protection; (http://www.fs.fed.us/foresthealth/).

Within the MAIA region, the primary zone of infestation is the advancing front (large populations of beech scale and initial infection due to *Nectria*), but the acreage contained within the killing front (presence of both the scale insect and *Nectria* fungus, as well as mortality rates as high as 50 percent) is increasing. In the aftermath zone (high mortality in large trees, as well as heavy infestation with *Nectria* in remaining, small trees), some individuals can be found that are disease-free. These individuals seem to be resistant to the beech scale and subsequently will avoid infection by *Nectria.*

The impact of beech bark disease is most apparent in the size of remaining American beech trees in infested areas, where a large proportion of study plots have smaller d.b.h. (diameter at breast height) trees than in areas that have not been infested. American beech trees do not produce large quantities of seed until they have reached 40 to 60 years (Tubbs and Houston 1990). The seed is eaten by a wide variety of birds and mammals—in fact American beech is the only widespread nut producer in northern hardwood forests. Smaller, generally younger trees produce less seed, and account for a reduced availability of mast for wildlife species. Although American beech is not a highly favored lumber species, it is widely used for flooring, furniture, pulp, charcoal, and fuelwood. Reduced growth and increased mortality will affect the availability of American beech for logging operations.

Beech bark disease cannot be controlled in forested areas at the landscape scale. Extremely cold air temperatures have been shown to be lethal to beech scales not protected by snow. The beech scale can be controlled with insecticides in urban environments and on high-value ornamentals to reduce potential for *Nectria* infection. Vigorous trees free of beech bark disease have been found within heavily infected areas, and subsequent research has shown that those trees are resistant to beech scale. Such resistant trees offer the best hope for continued presence of American beech in eastern forests.

Since the initial report of dogwood anthracnose 20 years ago in New York and Connecticut, the number of flowering dogwoods has been declining throughout the species' range. Generally associated with the Appalachian Mountains, dogwood anthracnose has been found as far south as central Alabama. The disease's origin is unknown but, due to its rapid spread across the landscape, it is believed to have been introduced from outside the U.S. Another theory asserts that changes in the environment have modified the host-pathogen relationship in ways that have allowed the fungus to become a significant pathogen. The disease is prevalent throughout the northern, north-central, and western portions of the MAIA region (fig. 29). In areas outside the range of flowering dogwood in forestlands, counties with areas infested are those where primarily ornamental dogwood trees are found.

Flowering dogwood is a small understory tree found throughout the eastern U.S., except in the Lake States. Dogwood's primary value is as an ornamental tree, due to the spring emergence of white, showy, petal-like bracts which are not actual flowers. The fruit is a valuable wildlife food in the fall and winter. Because its leaves decompose rapidly, flowering dogwood is a known soil-improver, and its leaf litter is an important source of calcium in forest soils.

Initial infection by dogwood anthracnose is signaled by spots on the lower leaves, which then spread to small twigs and branches. The disease generally progresses from the lower crown to the upper portions of the tree, as evidenced by the dieback of twigs and branches. The fungus can cause a canker that may kill larger trees in 2 to 3 years.

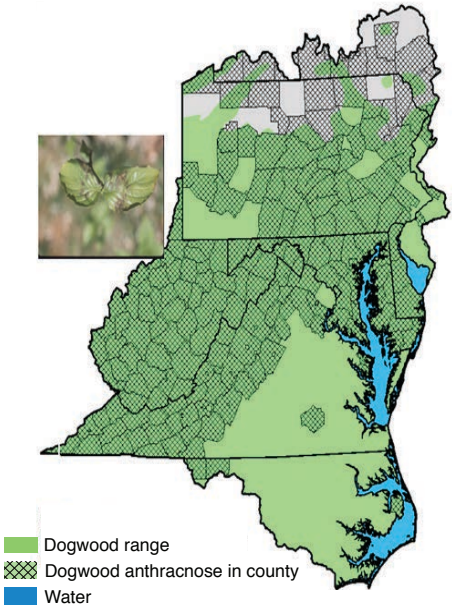


Figure 29—Range of dogwood species and distribution of dogwood anthracnose in MAIA counties through 1998. Inset shows an example of leaf blotching indicative of dogwood anthracnose infection. Source: USDA Forest Service, Forest Health Protection; (http://www.fs.fed.us/foresthealth/).

The disease affects dogwoods of all sizes, but is most severe on seedlings and understory trees. Vigorous, stress-free trees generally are more disease resistant. Britton and others (1996) have demonstrated that simulated acidic deposition can increase the disease's severity, primarily through changes in nutrient availability, not actual foliar damage.

Control procedures are not available for dogwood trees in the forest environment, but techniques are available to address the disease in high value settings such as recreation sites or urban areas. Management practices that favor vigorous trees will help keep remaining trees healthy. Avoidance of mechanical injury reduces the ability of the fungus to infect trees. Watering by sprinklers should be avoided, because that only spreads the fungus from infected leaves to uninfected leaves.

The reduction in numbers of flowering dogwoods has implications for the biodiversity of eastern forests, especially along the Appalachian Mountain range. Due to its value as an ornamental tree, a wildlife food source, and its ability to improve soil nutrient status for forested environments, any reduction in flowering dogwood is significant. The area infected with dogwood anthracnose is continuing to increase, but at a slower rate. Although dogwood anthracnose probably will not eliminate flowering dogwood from forest and urban settings, research on control methods are still needed.

Chapter 10.

Exotic Invasive Plant Species

Kenneth W. Stolte and John W. Coulston
Southern Research Station, USDA Forest Service

Forest ecosystems can also be threatened by exotic invasive plant species. Invasive plants are often successful competitors with native vegetation because they are new to the environment, and have few if any natural agents to keep them from becoming dominant. Invasive plants are often successful where disturbance has created an opportunity for establishment, but they can also be very successful in areas with low disturbance because they thrive in soils rich in carbon and nitrogen—the same type of sites that often have a high diversity of native plants (Stohlgren and others 1999). Analyses of data from 279 plots in 7 different areas of the U.S., collected as part of methods research to monitor the understory component of forests by the FHM program, indicated that exotic plant species accounted for a significant part of the total flora, and also accounted for a disproportionate amount of the flora's cover, or abundance (Stolte 1997).The FIA and FHM programs have fully developed protocols for monitoring understory native and exotic plant species diversity (www.fs.fia.fed.us) that will facilitate analysis of the spread of common exotic invasive species when this indicator is implemented in the southern United States.

An agreement between the USDA Forest Service, Forest Health Monitoring program and the Biota of North America Program (BONAP) in 1998 and 1999 focused on characterization of native and exotic plant species in the MAIA region. BONAP (http://www.bonap.org/) analyzed data from herbarium, literature, and other sources to document the county-level occurrence of native and exotic plant species in New Jersey, Delaware, West Virginia, Virginia, and North Carolina; a few counties in eastern Pennsylvania and New York; and no counties in Maryland.

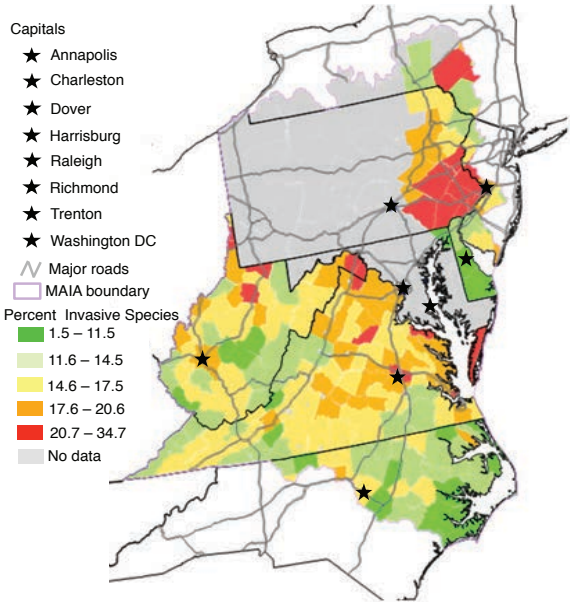


Figure 30—Exotic invasive plant species by county in the MAIA region compiled from herbaria and other data sources. Source: Biota of North America Program; (http://www.bonap.org/).

The analysis showed that invasive species were relatively scarce in some counties, accounting for only 1.5 to 11.5 percent of the total flora (fig. 30). In other counties, sometimes but not always near major cities, exotic plant species accounted for 20.6 to 34.7 percent of the total flora. The highest concentration of invasive species in the region, based on this county-level analysis, was found between the cities of Harrisburg, PA, and Trenton, NJ, in western New Jersey. Trenton is very near to the large city of Philadelphia, in eastern Pennsylvania. This analysis indicated that exotic plant species are well-established in some portions of the region, and the probability for continued expansion is very high.

Queen Anne's lace or wild carrot was the most common exotic plant species found in the counties evaluated to date, occurring in 93 percent of the counties evaluated (table 10). Other common exotic plant species were red clover in 91 percent of counties, and narrowleaf plaintain found in 90 percent of counties. Seven other exotic species were found in 80 percent or more of the counties. Additional evaluation of the rest of the counties in the MAIA region would provide a good baseline for many of these exotic plant species, the number of counties where they occurred, and what species have continued to spread throughout the region over time.

Table 10—Exotic plant species in MAIA region states circa 2000

Rank	Common name	Occurrence in MAIA counties
		-----percent-----
1	Queen Anne's lace	93
2	*Red clover*	91
3	*Narrowleaf plaintain*	90
4	Ox-eye daisy	88
5	*Sheep sorrel*	88
6	Barnyard grass	88
7	*White clover*	84
8	*Yellow sweet clover*	84
9	*Woolly mullein*	81
10	*Asiatic day flower*	80

Note: County-level records for Maryland, and the western counties of Pennsylvania and New York, were not obtained. Source: The Biota of North America Program; (http://www.bonap.org).

Chapter 11.

Urbanization, Fragmentation, and Land Use Change

Stephanie Fulton[1], Evan Mercer[2], and Kenneth W. Stolte[2]
[1]Region 4, U.S. Environmental Protection Agency;
[2]Southern Research Station, USDA Forest Service

Rapid expansion of urban development in and adjacent to forests results in increasingly fragmented forest ecosystems and presents serious problems for natural resource managers and urban planners, as well as others who are charged with implementing government policies and services. Urban expansion into forests increases the risk of wildfires, disruption of established animal populations, introduction of exotic invasive species, and degradation of water quality—resulting in injury or degradation of people, homes, and businesses. Such expansion results in a general increase in the need to protect water quality, wildlife, and forest health, while attempting to meet the social and recreational needs of people and provide for public health and safety.

Population Growth

In 1990, with 171,129 square miles of land area and a population of about 35 million people, the average population density for the entire MAIA region was 204 people per square mile (psm) (table 11). Between 1970 and 1990, the region's population had grown by 4.3 million people. Most States in the region exhibited population growth rates from 21 percent (Delaware and Maryland) to 30 percent (Virginia) in those two decades, except in Pennsylvania (0.7 percent), West Virginia (2.8 percent), and New York (6.9 percent). New Jersey experienced the highest average population density in all three censuses (608 psm in 1990), followed by Maryland (489 psm), Delaware (341 psm) and Pennsylvania (265 psm). The lowest population density in 1990 was in West Virginia (75 psm).

Table 11—Population changes in the MAIA region 1970 to 1990

	Population			Population density			
	1970	1980	1990	1970	1980	1990	Change 1970-1990
	---------millions of persons--------			-- persons per square mile--			---percent---
Delaware	548.1	594.3	666.2	277	308	341	23.1
Maryland	3,923.8	4,217.0	4,781.5	397	429	489	23.2
New Jersey[a]	2,947.5	3,360.7	3,714.6	476	547	608	27.7
New York[a]	2,611.5	2,703.5	2,792.8	120	125	129	7.5
North Carolina[a]	2,409.5	2,751.3	3,133.5	104	118	135	29.8
Pennsylvania	11,800.8	11,863.9	11,881.6	262	264	265	1.1
Virginia	4,651.5	5,346.8	6,187.4	117	135	156	33.3
West Virginia	1,744.2	1,949.6	1,793.5	73	81	75	2.7
Total/Average	30,636.9	32,787.2	34,951.0	179	192	204	14.1

[a] Population figures for NJ, NY and NC are totals for the subset of counties in the MAIA region. Source: 1970, 1980, and 1990 Census of Population and Housing, Bureau of Census, U.S. Department of Commerce; (http://www.census.gov/population/www/).

Aggregating population statistics at the State level, however, masks important geographic variability. Evaluation of population density at the county level provides a more in-depth view of population density changes. Figure 31 shows the 1990 population densities for all counties in the region, and illustrates the variability across the MAIA region, sometimes within relatively short distances. Most of the region's counties fell into the three lowest density categories, between 6 and 150 psm. The highest population densities occurred in urban and metropolitan areas and along the Interstate highway corridors that connect those populated areas (figs. 7 and 8).

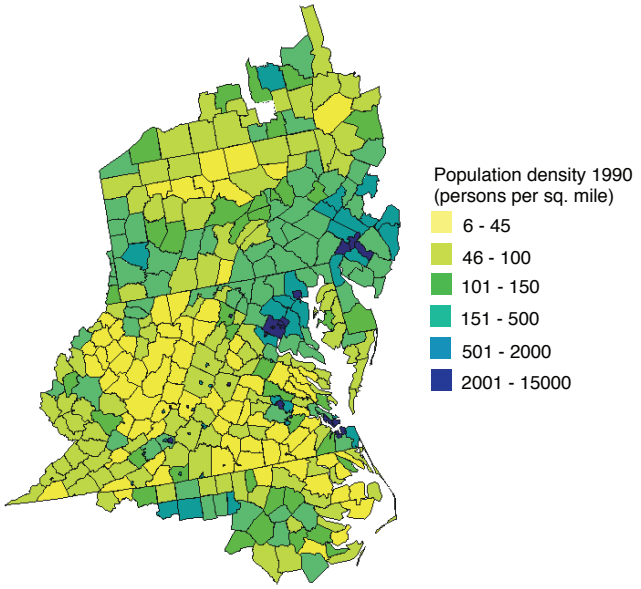


Figure 31—Population density in MAIA region in 1990. Source: U.S. Census Bureau; (http://www.census.gov/population/www/).

The average population density in the Mid-Atlantic States had increased dramatically from1950 to 1990, with much of the eastern part of the MAIA region experiencing population increases greater than 15 percent (fig. 32).

Much of the western portion of the region (including almost all of West Virginia and large portions of southern Virginia, eastern North Carolina, and central and western Pennsylvania), however, actually had experienced significant decreases (minus 2 percent to greater than minus 15 percent) in population density since 1950. In contrast, between 1970 and 1990 the number of counties that exhibited decreasing population densities, particularly those greater than minus 15 percent, dropped dramatically (fig. 33).

It appears that a large rural-to-urban migration during the 50s and 60s reduced the population in large areas of West Virginia, Virginia, North Carolina, and Pennsylvania, while populations in urban areas increased dramatically throughout the region. During the 70s and 80s, populations increased in almost all counties in the region. In many

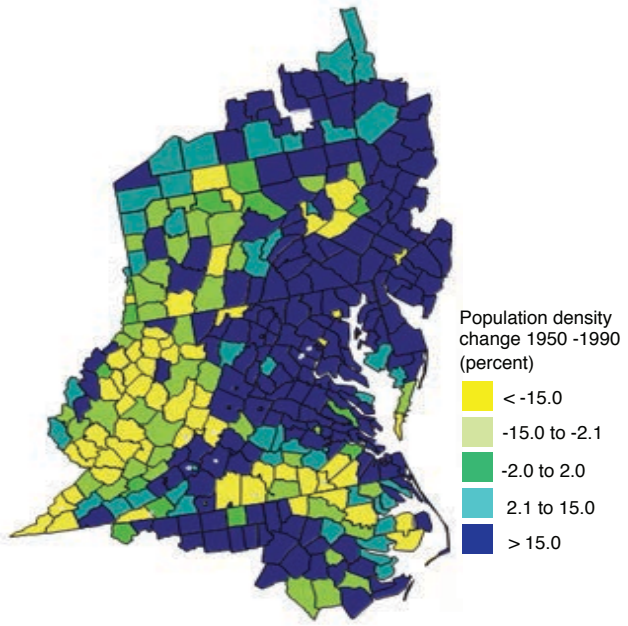


Figure 32—Change in population density 1950 to 1990. Source: U.S. Census Bureau; (http://www.census.gov/population/www/).

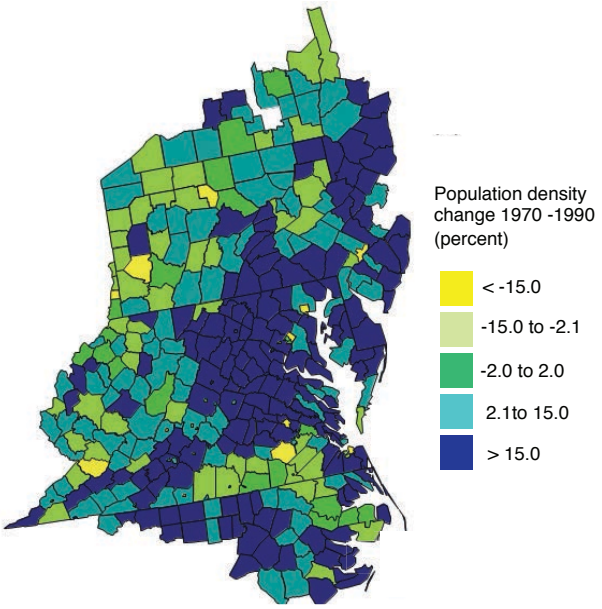


Figure 33—Change in population density in MAIA region 1970 to 1990. Source: U.S. Census Bureau; (http://www.census.gov/population/www/).

counties, however, the increased population density between 1970 and 1990 had not yet balanced the decreases that occurred between 1950 and 1970. This suggests that issues involving the interactions of forests and humans may be quite different between the rapidly urbanizing Eastern seaboard and the inland rural areas of West Virginia, Pennsylvania, southeastern Virginia, and North Carolina.

Increasing human population densities had two important effects on the forest management of private lands and on timber supply. The most obvious impact occurred in areas of rapid population growth where areas in forest land were reduced as they were converted to urban or residential uses. Impacts to forest lands also were felt in areas of moderate population density, where landowners' expectations of large future returns from converting forest lands to urban and residential uses may reduce long-term investments in forest management.

Table 12—Population density and management of forestland in western Virginia circa 2000

Population Density	Probability of Commercial Forest Management
--persons per square mile--	--------percent--------
0	82
20	75
45	50
70	25
100	10
>150	0

Source: Wear and others 1999.

An empirical analysis in western Virginia found a significant relationship between population density and forest management (Wear and others 1999). Table 12 presents the relationship between population density and the probability that forests will be managed for commercial timber production.

Wear and others (1999) found that estimates of timberland based on purely physical measures (e.g., those that ignore population impacts) may overstate timber supply by roughly 40 percent, and that the transition from rural to urban uses of forest lands occurs at 20 to 70 psm, with a 75 percent probability of forest management occurring at 20 psm and 25 percent at 70 psm. The probability of actual forest management approaches zero at 150 psm. MAIA counties with greater than 50 percent probability of forest management are those shaded yellow in figure 31 (6 to 45 psm). The light green counties (46 to 100 psm) have between 10 and 50 percent probability that forest management will occur in them. In all the remaining counties, forest management is highly unlikely either now or in the future.

Fragmentation

Increases in population growth usually result in the conversion of forest land to non-forest uses, either urban development (generally permanent conversion) or agriculture (which sometimes reverts back to forest or converts to urban development). Forestry practices can also fragment the forest—sometimes only temporarily—but generally leaving the land forested, albeit with changes in forest size, shape, and species composition (e.g., introduction of monoculture pine plantations). Figure 34 shows the location and degree of forest fragmentation in the Mid-Atlantic States. Generally, forest practices had much less impact on the degree of fragmentation than agricultural or urban uses (compare figs. 13 and 34). Not only was the quantity of forest land reduced, but also the quality of forest habitat was reduced as patterns of loss transformed the forest landscape into small, isolated patches of trees.

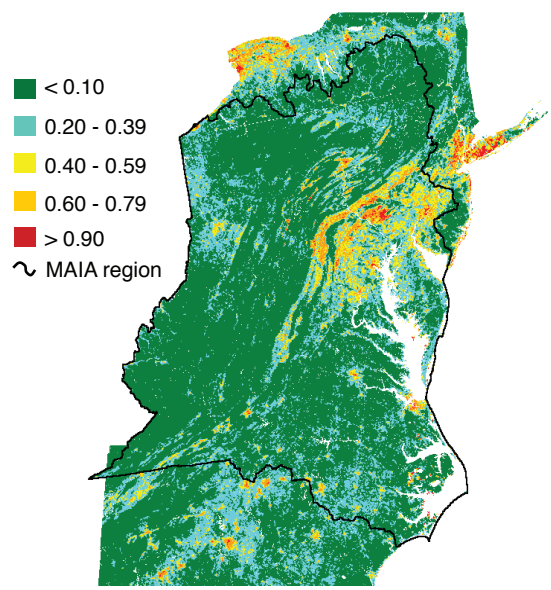


Figure 34—Forest fragmentation in the MAIA region1990 to 1993. Note: Changes in forest fragmentation were not analyzed due to lack of regional remote sensing data. Source: Riitters 1999.

Fragmentation is a process by which larger, contiguous forest lands are broken into smaller, physically isolated fragments or forest islands, surrounded by human-modified environments converted to agricultural, urban, or residential land uses (Society of American Foresters 1998). The ecological consequences of forest fragmentation include: (1) an irreversible loss of habitat, (2) increased sedimentation in aquatic systems, (3) changed stream conditions, (4) changed forest microclimate, (5) increased edge-dwelling wildlife, (6) decreased forest connectivity, (7) increased opportunities for human-wildlife conflicts, and (8) species extinction or loss of species diversity. As forest loss and fragmentation increase, remaining forest patches become smaller and more isolated, the amount of high quality interior forest habitat is reduced, overall forest connectivity is negatively affected, and the safe movement of wildlife between remaining forest patches is impeded.

Generally, areas with high levels of forest fragmentation (fig. 34) are associated with areas of high population density (figs. 8 and 13), net growth in population (figs. 32 and 33), and net loss in forest land areas (figs. 12 and 13). Conversely, low levels of forest fragmentation (fig. 34) are associated with retention of forest cover (fig. 12), low human population densities (fig. 31), and low-to-negative change in human populations (figs. 32 and 33).

The movement of urban populations to the suburbs is another component of increasing urbanization. Generally, while increased population density can be associated with forest fragmentation as city centers have expanded; another result of urban population growth has been ever-expanding rings of low-density housing at the cities edge. This phenomenon often is referred to as the *urban-rural interface.* Developers purchase large, relatively cheap (compared to "in-city"), unbroken tracts of forest land, divide them into smaller parcels, and create large suburban housing developments, which come complete with strip malls, additional roads, and other indicators of unbridled growth. Results of increased urbanization include loss of forest cover and increased impervious surface areas which, in turn, lead to a heightened likeliness of flash flooding and increased volume of downstream flows, easier access for exotic invasive species, and more human disturbance of what little forest remains.

Generally, conversion of forests into crop and pasture lands to support growing populations in the 18th and 19th centuries had the greatest impact on forest fragmentation, and many of these lands were later converted to urban development, as shown in figs. 12 and 13. Urban growth in the Mid-Atlantic States was concentrated in three regions: (1) the Allegheny Plateau (2) along Interstate 95 corridor from Philadelphia, PA, to Raleigh NC, and 3) the Piedmont, Blue Ridge Mountains, and Northern Ridge and Valley areas (figs. 5, 8, 13, and 33). Watersheds in the first two regions contained some of the most highly populated urban centers in the East, including Erie and Philadelphia, PA; Newark, NJ; Baltimore, MD; Washington, DC; and Norfolk-Chesapeake, VA (fig. 8). Forest cover in these regions was low (fig. 12) and highly fragmented (fig. 34)—the remaining forest patches were small and supported little high quality interior forest habitat.

Agricultural and forest-related uses increased (figs. 12 and 13), and forest fragmentation decreased appreciably (fig. 34), the farther one moved from urban centers (fig. 8), particularly away from coastal areas. The Piedmont and coastal regions of Virginia, west and northwestern West Virginia, and western and northeastern Pennsylvania, as well as the entire Northern Ridge and Valley region, were primarily a mix of forest and agricultural land uses (fig. 13). Forest cover was relatively low and highly variable in those areas, when compared to the densely forested north-central and central Appalachians and Blue Ridge Mountains; still, forest fragmentation was lower than near the more urban areas (fig. 34). The Pennsylvania Ridge and Valley region and DelMarva Peninsula were predominantly rural agricultural land. Over time, however, as agriculture began to dominate the landscape and forest cover decreased, forest fragmentation increased—although it never approached the degree found in urban areas. Generally then, this landscape was characterized by low-to-medium forest cover (fig. 12) and low-to-medium levels of forest fragmentation (fig. 34).

Land Use Change

The overall forestland base (table 13) of non-Federal rural lands within the MAIA region remained fairly steady from 1982 to 1992 (fig. 35), and decreased only slightly (by 247,100 acres, or 0.42 percent) between 1982 and 1992 (USDA 2000).

States only partially within the MAIA region showed increased forest cover (New York 4.64 percent), and decreased forest cover (North Carolina 6.75 percent and New Jersey 3.25 percent). The area of agricultural lands (pastureland and cropland; table 13) decreased by 2,686,000 acres between1982 and 1992; and as a direct result the total area of rural lands decreased by 2,495,300 (fig. 35).

The relatively small decrease in forestlands, compared to the much larger decrease in agricultural lands, and hence total rural lands, resulted in an overall 1.4 percent increase

Table 13—Land use and cover categories in National Resources Inventory reports

Land cover and use	Management purpose	Management type	Type of cover
Minor land and other rural land	Support or ancillary structures or land-types to other land cover/use	Ranges from intensive (e.g., farmsteads) to minimal (e.g., barren land)	Farmsteads, farm structures, windbreaks, barren land, marshland
Pastureland	Introduced forage plants for livestock grazing	Cultural treatments, including fertilization, weed control, reseeding or renovation, and control of grazing	Cover of grasses, legumes, and/or forbs. Single species, grass mixture, or grass-legume mixture
Cropland	Production of adapted crops	Cultivated and non-cultivated	Row or close-grown crops, or hayland or pastureland
Forestland	Growth and maintaining tree-based ecosystem for multiple uses (timber, recreation, aesthetics, etc.)	Ranges from intensive plantation management to non-intrusive protection of wilderness areas	10 percent or greater stocking (23 percent cover from above) with trees (> 4 meters tall maturity). Natural regeneration of tree cover and not developed for non-forest use. Minimum 1 acre in size and at least 100 feet wide

Source: USDA 2000; (http://www.nrcs.usda.gov/technical/NRI/).

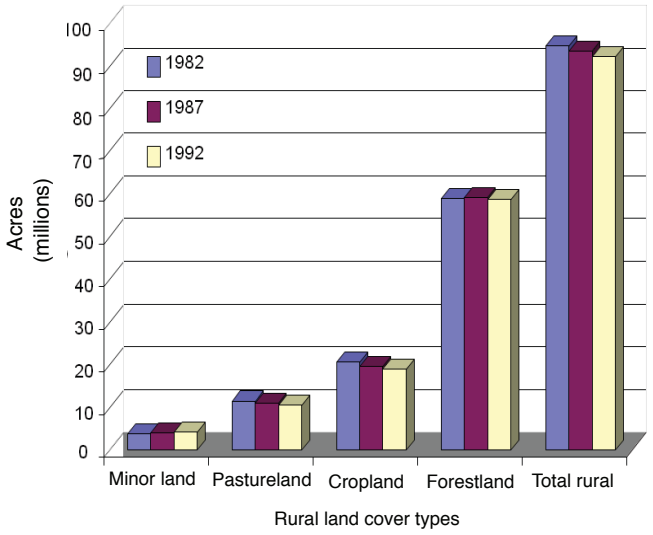


Figure 35—Non-federal rural land cover area in MAIA region in 1982, 1987, and 1992. Source: USDA 2000; (http://www.wa.nrcs.usda.gov/technical/NRI/).

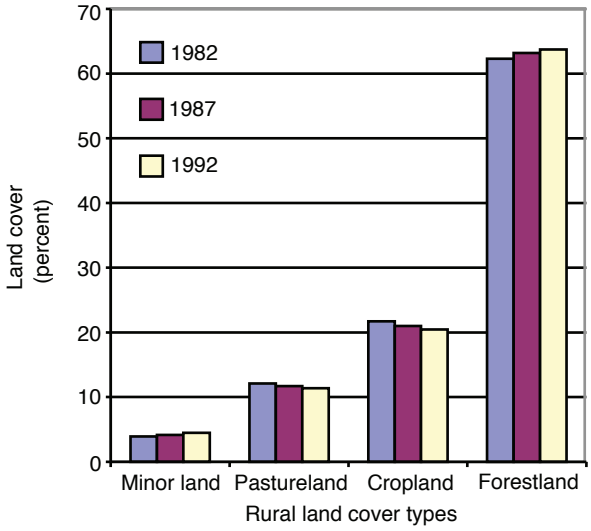


Figure 36—Land cover share of total non-federal rural land in MAIA region in 1982, 1987, and 1992. Source: USDA 2000; (http://www.wa.nrcs.usda.gov/technical/NRI/).

in the percentage of forest cover in rural lands from 1982 (62.3 percent) to 1992 (63.7 percent) (fig. 36) (USDA 2000). During this same period the percentage of total rural acreage used as cropland (table 13) decreased by 8.32 percent, as did pastureland use (8.55 percent); and in 1992 these uses accounted for 20.4 percent (down from 21.7 percent in 1982) and 11.4 percent (down from 12.1 percent in 1982) of the total rural lands, respectively (fig. 36).

We also evaluated land use in 1997, and the changes from 1982 through 1997, for the five States located entirely within the MAIA region (Delaware, Maryland, Pennsylvania, Virginia, and West Virginia). These evaluations were based on NRI data for four reporting periods (1982, 1987, 1992, and 1997) for the most common land cover types that include forest, cropland, pasture, urban (table 13), Federal ownership, and water bodies (USDA 2000). In 1997 this five State area covered about 80,993,900 acres and was composed primarily of 62,909,900 acres of rural lands (77.7 percent), which constituted 11,354,000 acres of cropland (14.0 percent), 6,868,400 acres of pasture (8.5 percent), 42,100,000 acres of forests (52.0 percent), and 2,587,500 acres (3.2 percent) of other land uses including minor land (farm structures, wind breaks, other man-made edifices) (table 13) and Conservation Reserve Program lands (fig. 37). Other land-use types covered 8,943, 800 acres of developed (urban) lands (11.04 percent); 4,781,600 acres of Federal ownership (5.90 percent); and 4,517,700 acres of water bodies (5.58 percent).

The amount of forest land declined slightly from 1982 to 1997 in all five States except West Virginia, but remained the most common land type of all rural lands, except in Delaware, where crop lands were more abundant (fig. 38) (USDA 2000). The water-body acreage increased slightly in Maryland, Virginia, and West Virginia; and the amount of Federal lands increased slightly in Maryland and Virginia.

The amount of urban lands increased during this same period, and crops and/or pasture lands decreased by about

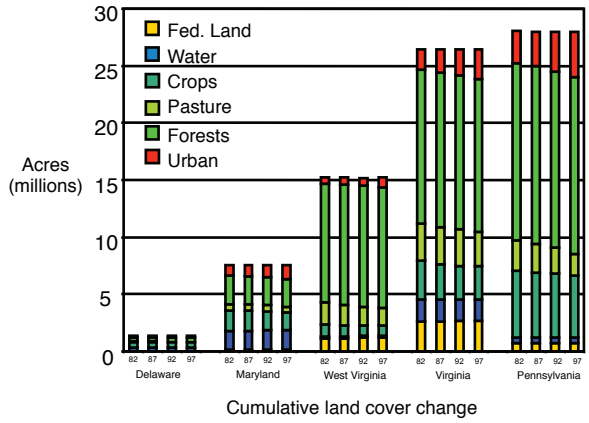


Figure 38—Land cover in five MAIA region states in 1982, 1987, 1992, and 1997. Source: USDA 2000; (http://www.wa.nrcs.usda.gov/technical/NRI/).

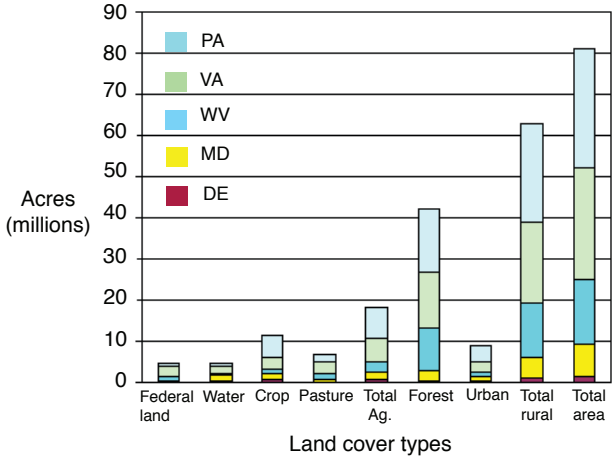


Figure 37—Land cover in five MAIA region states in 1997. Total agriculture is the combination of crop lands and pasture lands. Source: USDA 2000; (http://www.wa.nrcs.usda.gov/technical/NRI/).

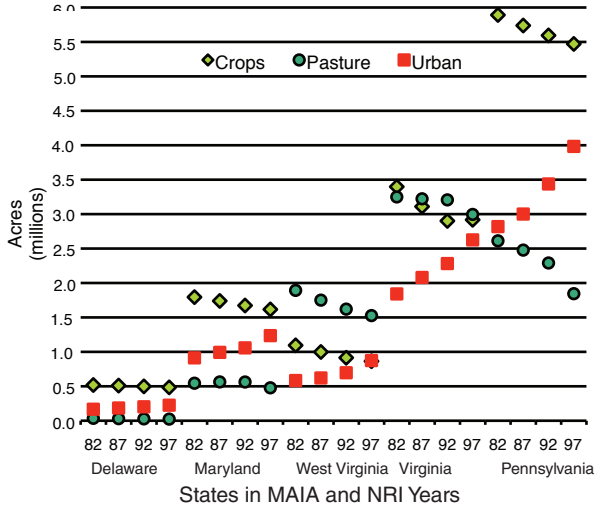


Figure 39—Urbanization changes in primary land cover types in five MAIA region states 1982 to 1997. Source: USDA 2000; (http://www.wa.nrcs.usda.gov/technical/NRI/).

the same extent. Figure 39 shows changes only in the number of acres of urban, crop, and pasture lands in the five State region; but it clearly shows that the amount of urban lands in each of the five States increased between 1982 and 1997, occurring at the expense of crop and/or pasture lands. Urban lands primarily replaced crop and pasture lands in West Virginia and Pennsylvania, and primarily replaced crop lands in Delaware, Maryland, and Virginia.

The percentage increase in urban lands from 1982 to 1997 was substantial in all five States entirely within the MAIA region, including Delaware (35.0 percent), Maryland (35.4 percent), Pennsylvania (41.3 percent), Virginia (42.6 percent), and West Virginia (49.6 percent) (fig. 40) (USDA 2000). Within that time frame the average percent increase of urban lands in the five States was 41.4 percent. The percent change in all agriculture lands (crops and pasture) was highest in West Virginia (about 20 percent). Pasturelands decreased by over 25 percent in Delaware and Pennsylvania, and croplands decreased by 14 percent or more in Virginia and West Virginia.

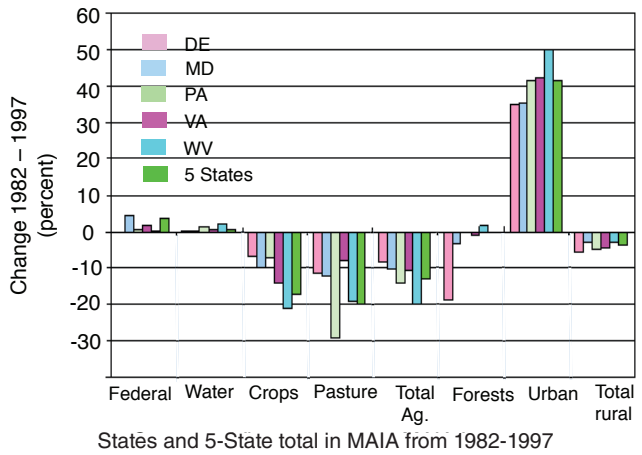


Figure 40—Differences in land cover types for five MAIA region states 1982 to 1997. Total agriculture is the combination of crop and pasture lands.Source: USDA 2000; (http://www.wa.nrcs.usda.gov/technical/NRI/).

Figure 41 shows the percent contribution of decreases in land cover types that offset the increases in other land cover types, particularly urbanization, for each State entirely in the MAIA region. Increases in urban lands were offset by relatively large decreases in percent of crop and pasture lands. Crop land reductions ranged from 0.13 percent in Delaware to 21.1 percent in West Virginia, with an average of 17.4 percent for all States. Pasture land reductions ranged from 7.8 percent in Virginia to 32.7 percent in Delaware, with an average of 20.0 percent for all States. Decreases in crop and pasture lands by State can be

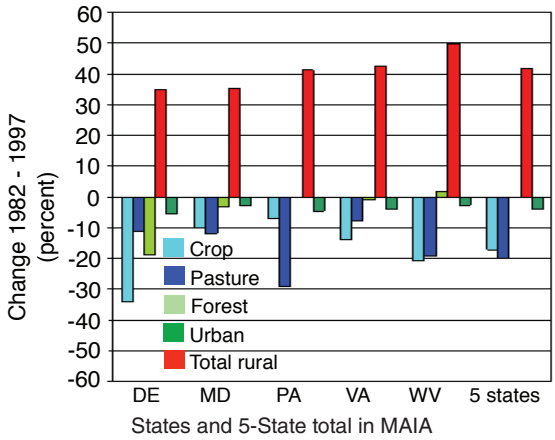


Figure 41—Differences in major land cover types in five MAIA region states 1982 to 1997. Source: USDA 2000; (http://www.wa.nrcs.usda.gov/technical/NRI/).

expressed as a percent decrease in total agricultural lands, i.e., Delaware—minus 8.3 percent, Maryland—minus 10.5 percent, Pennsylvania–minus 14.0 percent, Virginia—minus 11.0 percent, and West Virginia—minus 20.0 percent. The average decrease in agricultural uses in all five States was 13.4 percent.

As a result of the increases in urban use, total rural land area decreased in the affected States (fig. 41): Delaware by 5.6 percent, Maryland by 3.1 percent, Pennsylvania by 4.7 percent, Virginia by 4.1 percent, and West Virginia by 2.9 percent. The average decrease in total rural lands for all States was 4.0 percent. Delaware also had a decrease in forest lands (5.1 percent), as did Maryland (3.5 percent), Pennsylvania (0.2 percent), and Virginia (1.0 percent). In West Virginia forest cover increased by 1.6 percent.

Relative contributions toward the percent change in land cover for all five States showed an average 45.4 percent increase (fig. 42); where urban growth constituted 41.4 percent, Federal lands 3.3 percent, and changes in the area of water-bodies 0.7 percent (USDA 2000). These increases, coupled with decreases in or conversion from crop, pasture, and forest land (17.4, 20.0, and 0.25 percent, respectively), resulted in significant changes in land use. West Virginia had the largest percentage land-use change—an increase of 53.3 percent due to increased urban growth (49.6 percent), forest cover (1.62 percent), water-body acreage (1.96 percent), and Federal lands (0.08 percent) and decreases of 21.1 percent in croplands and a 19.53 percent in pasturelands.

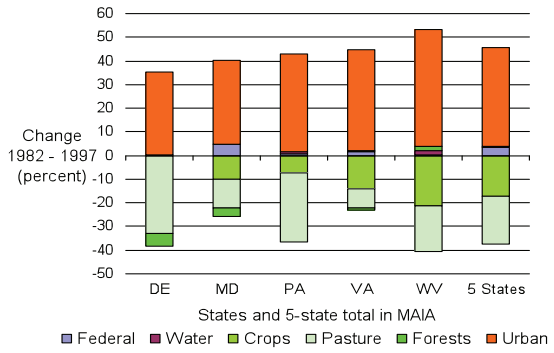


Figure 42—Contribution of land cover types to rural lands in five MAIA region states 1982 to 1997. Source: USDA 2000; (http://www.wa.nrcs.usda.gov/technical/NRI/).

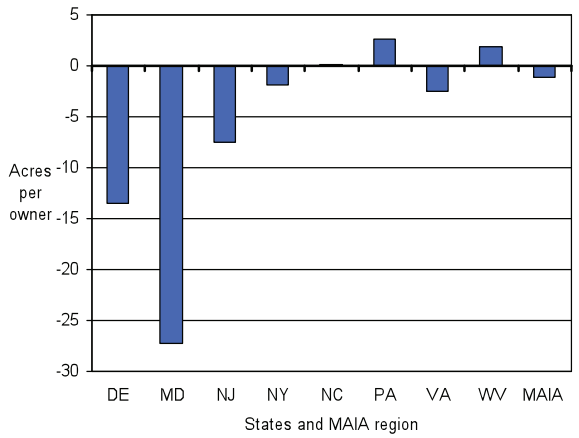


Figure 43—Change in number of acres per owner of forest land by State and MAIA region 1978 to 1994. Source: Birch 1996a, Birch 1996b, Birch 1996c.

The myriad land use changes from 1982 to 1997 resulted in an annual loss rate of 176,347 acres of rural lands in all five States combined. At this rate all rural lands in these five States will be converted to non-rural uses in 357 years.

Change in Forest Land Ownership Patterns

Parcelization is the breaking-up of single contiguous land ownerships into smaller tracts or parcels with increased numbers of owners. We found the size of private forest land ownership decreased in some States between 1978 and 1994 (Birch 1996a, 1996b, 1996c). The largest decrease in acres per owner occurred in Maryland (27.3 acres) and Delaware (13.5 acres) (fig. 43). There were slight increases in the number of acres per owner in Pennsylvania and West Virginia. The average acres per owner decreased by 1.18 for the whole MAIA region.

The number of owners for the entire region increased by 11 percent, while the number of acres of forest land owned increased by only 2 percent between 1978 and 1994 (Birch 1996a, 1996b, 1996c). This resulted in a decrease in the percent acres of forest land ownership by 61.1 percent in Maryland, 40.3 percent in Delaware, 33 percent in New Jersey, and 8 percent or less in Virginia, New York, and Pennsylvania (fig. 44). The percent acres of forest land ownership between 1978 and 1994 increased only in West Virginia (4.6 percent) and North Carolina (0.4 percent).

Another way of evaluating parcelization is the number of owners and acres by parcel. Table 14 shows that in 1994 there was a significant number of large forest tracts in the MAIA region. Large tracts of forests (>1,000 acres) were owned by only 0.13 percent of all owners and represented 21.1 percent of all forests (Birch 1996a, 1996b, 1996c). The smallest forest parcels (l to < 20 acres) belonged to 74.2 percent of all owners and accounted for only 12.7 percent of all forest land. Owners with parcels 20 to 999 acres represented 25.8 percent of all owners and represented 66.1 percent of all forest land.

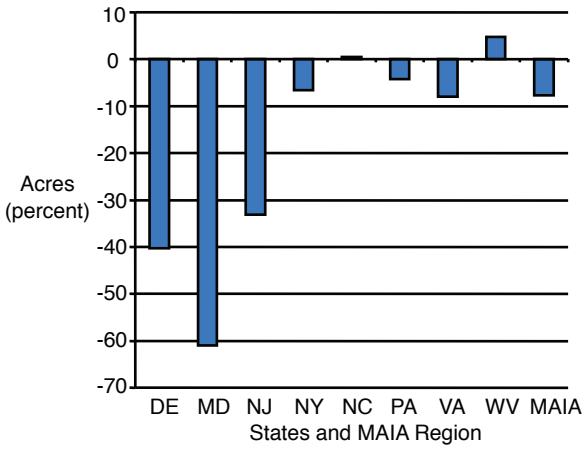


Figure 44—Change in acres per owner share of forest land by State and MAIA region 1978 to 1994. Source: Birch 1996a, Birch 1996b, Birch 1996c.

Table 14—Owners and size of forest parcels in MAIA region in 1994

Parcel Size Class	Owners		Acres	
-----acres----	-thousands-	----percent---	---thousands--	----percent---
1-9	895.2	60.5	2,577	6.3
10-19	202.1	13.7	2,611	6.4
20-49	194.5	13.2	5,757	14.1
50-99	106.6	7.2	7,003	17.2
100-199	46.7	3.2	5,627	13.8
200-499	27.6	1.9	5,797	14.2
500-999	4.7	0.3	2,777	6.8
1000-4999	1.8	0.1	2,912	7.2
5000+	0.4	0.03	5,655	13.9

Source: Birch 1996a, 1996b, 1996c.

These data suggest that parcelization is continuing in most of the region that will ultimately lead to increased fragmentation of forest land. Coupled with forest fragmentation, parcelization will eventually reduce the size of forest land management areas, making economically viable forest management unlikely, if not impossible (Wear 1996). If these trends continue, the quantity and quality of forest resources from the MAIA region will decline.

Chapter 12.

Bird Communities and Ecological Condition

Laura E. Jackson
Research Triangle Park Laboratory, Office of Research and Development, U.S. Environmental Protection Agency

Two recent studies in the Mid-Atlantic region suggested significant impacts of forest fragmentation on forest bird species richness (Cam and others 2000a, 2000b) and on overall ecological condition, as defined by a Bird Community Index (O'Connell and others 2000).

An examination of North American Breeding Bird Survey (BBS) data (Robbins and others 1989) across EPA Region III (Pennsylvania, Maryland, Delaware, Virginia, and West Virginia) showed that the number of forest-breeding songbird species observed along survey routes increased significantly with the number of 8-digit watersheds (<http://water.usgs.gov/GIS/huc.html>) that had high proportions of interior forest habitat (Cam and others 2000a, 2000b). Similarly, species richness of these forest songbirds decreased significantly with degree of forest fragmentation and proportion of edge habitat in the watershed (see *Richness Analysis of Breeding Bird Survey Data* in Technical Appendix D for details of methods).

Comparable associations were observed when relative species richness was substituted as the dependent variable. Relative species richness was defined as the proportion of the pool of forest bird species that had been recorded since 1966 at five or more BBS routes within an 80-km radius, and within the same State and physiographic stratum as the analysis route. This metric accounted for possible variations in species richness attributable to physiographic heterogeneity across the large study region, rather than to the degree of forest fragmentation. This definition necessitated omission of certain routes from analysis where route density was lower than five within the 80-km radius.

Positive correlations between the two richness measures and percent forest/interior forest were also found at the scale of the 39.4-km survey route with a 0.8-km buffer. These results provided scale-specific data that corroborated previous research showing that large blocks of unfragmented forest, containing habitat remote from edge influences, supported more forest songbird species than forest habitat contained within a heterogeneous mix of land-cover patches (Boulinier and others 1998b, Kareira and Wennergren 1995, Herkert 1994, Tilman and others 1994, Robbins and others 1989).

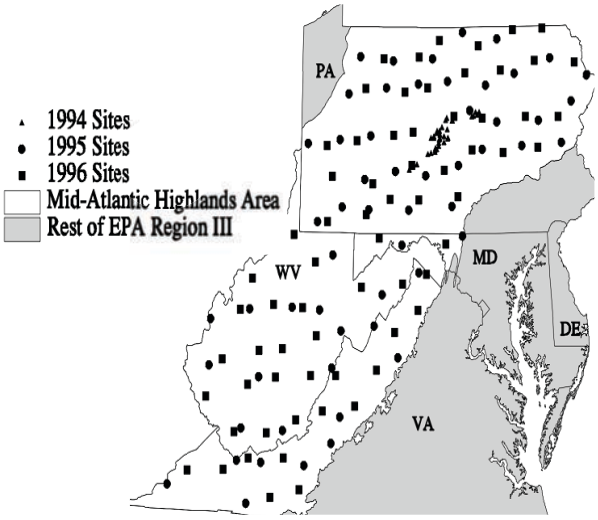


Figure 45—Breeding-bird sample site locations within the Mid-Atlantic highlands area (MAHA). Source: O'Connell and others 1998, 2000.

In another study, O'Connell and others (2000) developed a bird community index (BCI) as an indicator of the overall ecological condition in the Mid-Atlantic highlands area (MAHA) (<http://www.epa.gov/ecoplaces/ecosystems.pdf>). Basically, the MAHA is the higher-elevation western portion of the MAIA region. A probability based sample of 126 sites throughout the MAHA section of EPA Region III was completed in 1995 and 1996 (fig. 45).

Songbird counts and vegetation measures were taken along a 1-km transect at each site. Vegetation pattern at the landscape scale was derived from aerial photography of the circular area (79 ha) surrounding each transect. Bird species were classified according to behavioral and physiological response guilds (groups with similar nesting, foraging, and other life-history traits), which were selected to reflect aspects of ecosystem composition, structure, and function. Distinct species assemblages correlated highly with known levels of ecological condition measured at intensive research sites in the study region. This correlation enabled the development of the BCI as a condition indicator. Because songbirds occur throughout forested and other landscapes, the BCI is intended to integrate ecological conditions across a large assessment region that exhibits diverse land cover and intensities of human use.

The BCI was restricted to the mountainous portion of EPA Region III in order to reduce environmental variability. The index was calibrated to ecological properties specific to MAHA at the regional scale, and is not intended for use at local scales or in other physiographic areas without recalibration.

The BCI identified five categories of ecological condition (fig. 46).

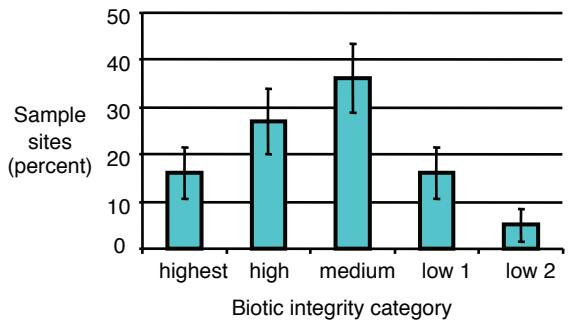


Figure 46—Distribution of 126 MAHA sample sites in five condition categories determined by the Bird Community Index (BCI). Error bars represent 95 percent confidence interval estimate for the percentage of land area in MAHA supporting these five categories. Source: O'Connell and others 2000.

Based on properties of the bird community at each of the 126 sample sites, 16 percent of the MAHA was assessed to be in the highest (excellent) condition; 27 percent was in high (good) condition; 36 percent was in medium (fair) condition, and 21 percent was in low-1 and low-2 (poor) condition. Two distinct bird communities ranked equally low on the condition gradient, and were found to be associated with different landscape types.

Figure 47 shows the spatial distribution of BCI scores across the MAHA.

These were statistically associated with percent forest in the 79-ha landscapes (fig. 48). Forest fragmentation was positively associated with a high percentage of exotic species, nest predators and parasites, omnivores, and multi-brooded species with a life-history strategy of rapid proliferation. This is a classic profile of opportunistic behaviors indicative of simplified or otherwise disturbed systems. Fragmentation was negatively correlated with a high percentage of insectivores, single-brooded species, foraging specialists, and other specialist guilds. In the

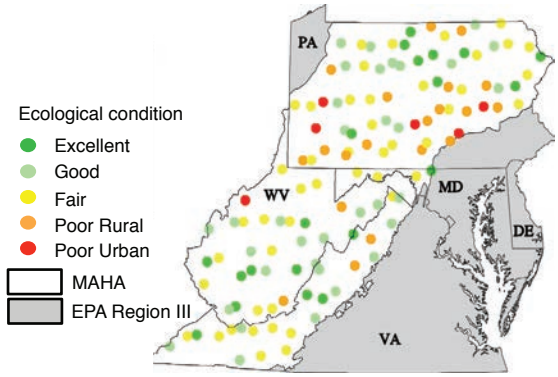


Figure 47—Spatial patterns of BCI scores across MAHA. Note: data from individual sites are not statistically representative of subregional conditions. Source: O'Connell and others 2000.

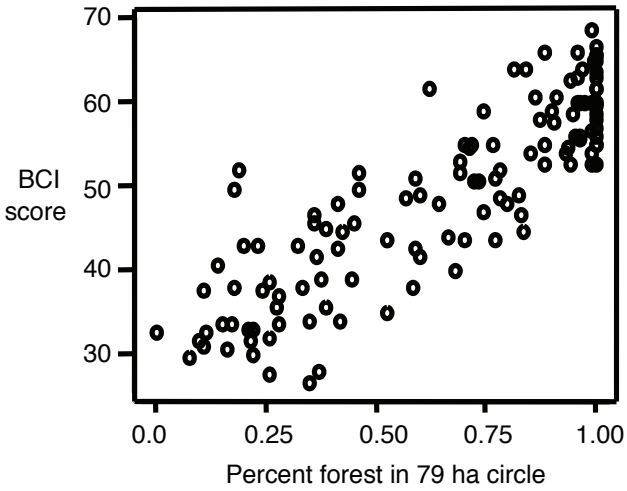


Figure 48—Bird community index (BCI) scores along a gradient of forest cover. Source: O'Connell and others 2000.

MAHA, these guilds indicate a system relatively unaffected by human activity, supporting multiple trophic levels and microhabitats, and populated with disturbance-sensitive species with low intrinsic rates of population increase.

Figure 49 depicts the associations between land-cover composition and ecological condition as defined by the BCI. On average, sites with less than 28 percent forest cover, more than 60 percent agricultural/herbaceous cover, or more than 30 percent residential/commercial cover were associated with songbird communities indicative of poor ecological condition (low-1 and low-2) in the study area. Sites with more than 87 percent forest cover were associated with songbird communities indicative of good or excellent ecological condition. Landscape pattern was not

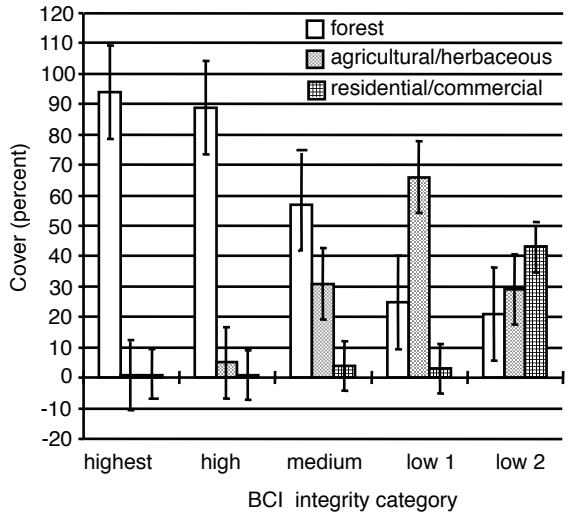


Figure 49—Mean and standard deviation of percent forested, agricultural/herbaceous, and residential/commercial land cover of sites in five condition categories determined by BCI scores. Source: O'Connell and others 2000.

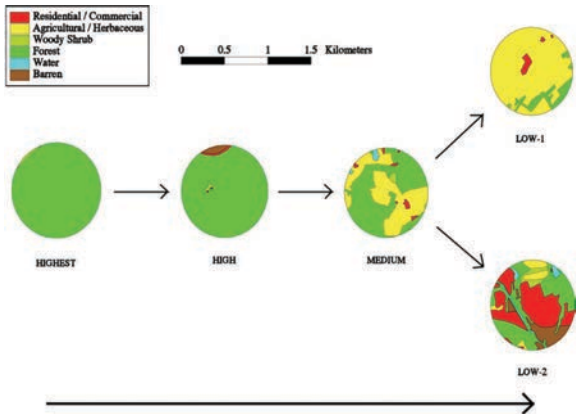


Figure 50—Land-cover configuration of representative sites in each condition category determined by BCI scores. Source: O'Connell and others 2000.

sufficient to distinguish the two upper categories (highest and high); yet significantly different bird communities distinguished good from excellent sites. Vegetation measurements on the ground revealed that sites in excellent condition supported a taller (~24 m) and more closed (~60 percent) tree canopy than sites in good condition (~20 m and ~47 percent, respectively).

Figure 50 provides examples of landscapes that were associated with the five categories of ecological condition defined by the BCI. Two categories of poor condition (low-1 and low-2) emerged from the bird community data. Although the bird communities were statistically distinct in each category, they received equivalent index scores and therefore were interpreted as reflecting equivalent ecological condition. Association of these bird communities with their respective landscapes revealed that one community predominated in a rural landscape (low-1), while the other predominated in an urban landscape (low-2). The conclusion is that poor ecological condition may be observed in both landscape types. However, additional work later ranked urban (low-2) below rural bird communities (low-1) on a gradient of ecological condition, due to the impoverishment of species observed in the former (O'Connell and others 2000).

This study quantified critical thresholds of land-cover change where significant shifts occurred in the composition of forest bird communities and ecological condition. It also demonstrated that bird community composition and overall system condition are related to land-cover pattern at spatial scales of at least 79 hectares. This represents a minimum landscape scale, where even small land-cover changes could affect ecological condition.

FOREST RESPONSE

Chapter 13.

Tree Crowns, Damage, and Mortality

Mark J. Ambrose
Department of Forestry, North Carolina State University

The previous chapter focused on stressors that may be adversely affecting the health and sustainability of forests in the MAIA region. However, forests differ in their ability to tolerate or resist various stressors (fig. 2). The presence of stressors sometimes can be directly measured (e.g., air pollution, insects and pathogens, invasion of exotic species). However it is often very difficult to determine relationships between the type and severity of stressors and forest ecosystem responses, because often there are multiple stressors affecting multiple ecosystem processes and components simultaneously, as well as other confounding factors. In order to determine if stressors, measured or not, are negatively affecting forest ecosystems, long-term forest health monitoring systems with indicators that are linked to ecological processes have been developed and established by the USDA Forest Service (e.g., FIA, FHM, NRI, and others), collaborating State agencies, academia, and private groups.

This chapter discusses early signs or symptoms (indicators) that appear when ecosystem resiliency has been exceeded—specifically, indicators that are symptomatic of changes in one or more fundamental ecosystem processes. For example, the condition of tree crowns can be directly related to the essential process of photosynthesis.

Indicator data from approximately 350 forested plots from 1991 through 1998 in Virginia, Maryland, Delaware, and New Jersey; from 1995 through 1998 for West Virginia; from 1995 and 1998 for Pennsylvania; and in 1998 for North Carolina were collected and analyzed by FHM program staff. No plot data had been collected in New York as of 1998. Forest Health Monitoring data in the MAIA region were analyzed both by major watershed subregion (HUC-4 scale) (<http://water.usgs.gov/GIS/huc.html>) (fig. 6) and by ecoregion section (Bailey 1995) (fig. 5). Major watersheds are convenient land units for analyzing how forest health indicators vary across the MAIA region, and allow forest condition to be related to the condition of associated aquatic systems.

Analysis by Ecoregion Section

Because the MAIA region was delimited in part by political boundaries and in part by ecological boundaries, some ecoregion sections and some watersheds are entirely within the region, while others are only partially within it. If an ecoregion section (or the portion of a section for which FHM data were available) was entirely within the MAIA boundary (fig. 5), we estimated average indicator values for the section. Such estimates were clearly representative of forest condition within the region. However, several ecoregion sections extended beyond the region's borders. If at least one-third of the area of an ecoregion section was within the MAIA region, we estimated a value for the ecoregion section using data from both inside and outside of the region. If less than one-third of the ecoregion section was within the MAIA region, we made no estimates, because they probably would not be representative of conditions within the region's boundary. No estimates were made for the parts of ecoregion sections in New York, because there were no FHM plots in that State as of 1998. Data from ecoregion sections M221B-Allegheny Mountains and M221C-Northern Cumberland Mountains (Bailey 1995) were combined for the analysis because neither section had enough forested plots to be analyzed independently (see Technical Appendix B). The geographic extent of ecoregion sections that were analyzed for indicators of forest response are shown in figure 51.

Maps associated with the ecoregion analyses display indicator values for the MAIA region, while the map legends show the entire range of values found for ecoregions throughout the U.S. in the analyses for the *FHM 2001 National Technical Report* (Conkling and others 2005). This allowed us to compare indicator values from the MAIA region with those from the rest of the Nation's forests.

Analysis by Watershed

We have included within the MAIA region some watersheds in their entirety and, just as we included some entire ecosystems, we included portions of other watersheds that extend beyond the region. Analyses of watershed condition were performed at the hydrologic unit code 4 level (HUC4) (fig. 6) (USGS 1974-1987; (<http://water.usgs.gov/GIS/huc.html>). All data collected from plots actually within the MAIA region were used in those analyses. If the majority of a watershed was within the MAIA region, all data from the FHM program for that watershed were used to calculate a watershed-level estimate for each indicator, including data from plots outside the MAIA region. If the majority of the watershed was outside of the MAIA region, only data from within the region were used to estimate indicator values for the portion of the watershed in the region.

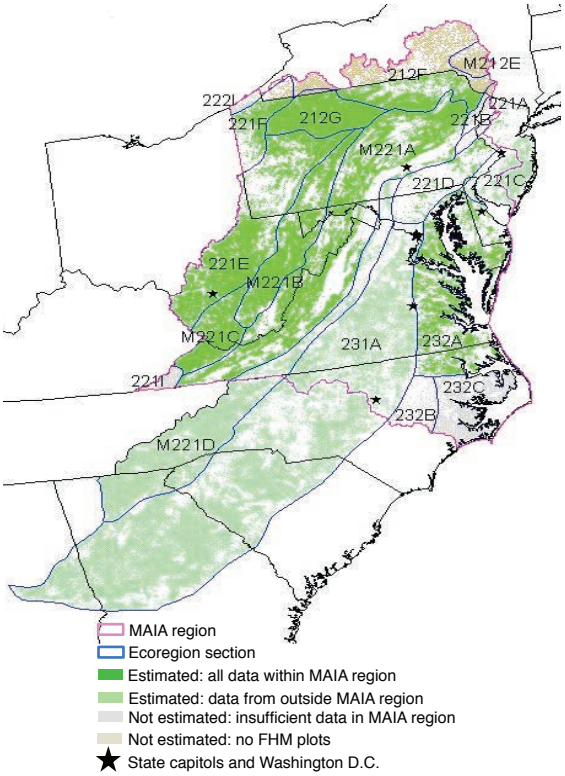


Figure 51—Ecoregion sections and analytical approaches used were based on amount of data available only within the MAIA region. Some ecoregion section averages (light green color) were based on plot data in the same ecoregion section that was both within and external to the MAIA region. The plot value was the actual value if the plot was measured in 1998, and an estimated value based on previous plot measurements otherwise. Source: National Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

In some watersheds we could not conduct analyses because of an insufficient number of forested plots. In those cases, data from adjacent watersheds were combined for the analysis. For example, data from the Pee Dee River and the Kanawha River watersheds were combined, and data from the Middle Ohio River and the Big Sandy-Guyandotte River watersheds were combined (fig. 6). No indicator values were presented for the eastern Lake Erie or the southwestern Lake Ontario watersheds because no forested FHM plots fell within the portions of those watersheds that are in the MAIA region.

Tree Crown Condition

Tree crown condition is an important indicator of individual tree and forest stand health. A large number of studies have related crown condition to tree growth and productivity for a variety of trees species (Lewis and Conkling 1994). A tree with a dense, full crown generally has a relatively high *leaf area index* that is based on the surface area of all leaves compared to the surface area of the ground below the tree crown. Thus trees with high leaf area indices have crowns with dense foliage that produce dark shade below the tree. Productivity rates or the rates of tree growth tend to increase as leaf area indices approach 8, due to increases in effective light interception (Salisbury and Ross 1978). Thus substantial loss of leaf area reduces the efficiency of light interception, interferes with the processes of photosynthesis, reduces the amount of carbon fixed as sugars, and reduces tree growth (Weinstein and Beloin 1990). In fact, there are suggestions that radiation interception, as a function of leaf area index, is the primary determinant of tree growth in single-species stands (Vose and Swank 1990). In loblolly pine plantations, loblolly productivity is directly related to leaf area index or leaf biomass (Voss and Allen 1988). Thus trees with relatively full, highly foliated large crowns have a high potential to maximize gross photosynthesis because they are able to use more solar radiation and carbon dioxide available during the growing season to produce sugars. Dense, full crowns are also indicative of a greater resilience to some stressors, such as Western pine beetles, because the tree is physiologically more active and more successful at pushing out boring insects with tree sap (Miller and others 1996).

FHM measured and FIA continues to measure several variables related to the amount and fullness of foliage and the vigor of the apical growing points of the crown. Two of these variables are the mortality of terminal twigs in the sun-exposed portions of tree crowns *(dieback)* and the *transparency* of the foliage of the tree crown relative to background conditions (i.e., the sparseness of crown foliage).

Crown dieback in the FHM and FIA programs is an ocular estimate of the percent mortality (0 to 100 percent) of the terminal portion of branches that were less than 1 inch in diameter and in the upper, sun-exposed portion of the crown (Burkman and others 1995). Foliar transparency is an ocular estimate of the sparseness of foliage recorded as the percent of sky or other background visible through the live, normally foliated portion of the crown (0 to 100 percent). Foliar transparency increases as the number and size of leaves or needles decreases.

We calculated plot-level averages for dieback and transparency in all plots containing at least five trees. For each ecoregion section and watershed in the MAIA region,

an average crown indicator value was estimated using a generalized least-squares (GLS) procedure (see *Analysis using Generalized Least Squares Models* in Technical Appendix B). The analyses used current as well as all prior plot measurements simultaneously to estimate the status and periodic annual change in crown indicators for each analysis unit (ecoregion section or watershed). Hardwoods and softwoods were analyzed separately for each crown indicator.

The FIA monitoring design uses a rotating panel sample, in which all plots are not measured every year (Stolte 1997). For each plot not measured in 1998, average dieback and transparency values were estimated from past measurements of that plot and from past and present measurements of other plots. The average plot values shown on the crown indicator maps were actual values if the plot was measured in 1998 and estimated values if otherwise.

The relationships between crown condition and tree health often are complex. Because crown condition is one critical factor of a tree's current photosynthetic potential, it provides an indication of present and potential growth. Large, healthy crowns indicate a potential for high productivity. Conversely, a poor crown condition indicates a reduction in productivity. Extremely poor crown condition indicates that trees have insufficient leaf surface area to maintain basic photosynthetic functions, and growth will cease and mortality may soon follow. Poor crown condition (high transparency and/or dieback) may be a direct response to one specific stressor or may be a response to multiple stressors (stress-complex) (Houston 1981).

A conceptual framework was developed for interpreting the complex relationships between crown condition and tree health tree crown condition data. This framework was based on the rather different physiological responses of hardwood and softwood trees to stressors. Various stressors affect the condition of tree crowns of different species in different ways. In response to certain stressors, many hardwoods species prematurely lose foliage, or produce fewer leaves or leaves of smaller size. Thus, foliar transparency can be seen as an indicator of present or fairly recent stress in hardwoods. In hardwoods, crown dieback often occurs when stressors over preceding growing seasons have reduced a tree's ability to photosynthesize or have otherwise interfered with the growth process. Reserves of carbohydrates become depleted in a stressed tree, and it is unable to produce sufficient viable buds on many twigs for the following growing season. Twigs not producing new buds die, and a portion of the crown dies back. Therefore, crown dieback of hardwoods can be an indicator of cumulative stress over multiple growing seasons. Under prolonged stress, one would expect to see an increased transparency response followed in time by increased dieback, and eventually, by early mortality. Under these conditions, crown dieback serves as a symptom (indicator) of a decline disease complex; examples have been documented for a number of important timber species (Houston 1981).

Hardwood dieback also may occur as the result of severe, acute stress affecting the terminal portions of tree crowns (e.g., insect attack, severe winter weather damage to twigs). In such cases, high dieback can occur with little or no increase in transparency. Where transparency does increase it will seldom lead to increased dieback in response to a transient stressor (e.g., short-term drought) or a low-level chronic stressor (e.g., low populations of defoliating insects). If the stress is alleviated within a short time, the tree recovers.

Similarly, many softwood species will lose foliage prematurely in response to stress, increasing their transparency. In most softwood species, however, dieback usually does not occur except as the result of severe, acute stress affecting a portion of the tree (e.g., bole damage, foliar disease, and root damage). More typically, softwood crowns affected by prolonged stress first lose lower branches, reducing the percent live crown on the tree bole, and the foliage becomes sparser, resulting in continually increasing transparency. Thus, in softwoods under prolonged stress, dieback response generally does not follow an increase in transparency. However, mortality can be expected to follow either significantly increased transparency or crown dieback.

Poor crown condition may indicate a transient stressor that may result in reduced productivity or may indicate prolonged or multiple stressors that may lead directly or indirectly to early mortality. Often the actual cause of tree mortality will not be the primary stressor but a secondary pathogen or insect that can take advantage of the trees' weakened state (Houston 1981).

The challenge of interpreting crown indicator data is discerning whether changing crown conditions are due to transient stressors or more prolonged or multiple stressors, and then determining whether the tree response to stressors is significant enough to be of concern. The interpretation is further complicated by the fact that crowns of different tree species vary in their responses to environmental stressors. Interpretation of transparency

is also difficult because tree species vary significantly in transparency levels associated with healthy or normal tree crowns. Therefore, where areas with relatively high crown dieback or transparency are identified, a more detailed analysis may be required to understand whether these values are normal for that area, are associated with problems affecting a particular species, or indicate a broader forest health issue.

Hardwood and Softwood Crown Condition by Ecoregion Section

Most of the ecoregion sections in the MAIA region had relatively low (12 to 19 percent) hardwood transparency levels (fig. 52) compared with those seen in other ecoregion sections of the U.S. (transparency levels as high as 42 percent [Conkling and others, 2005]). Only a few scattered plots had relatively high transparency levels (38.1 to 58 percent).

Hardwood transparency was increasing throughout much of the eastern MAIA region (0 to 2 percent per year), and decreasing in the western MAIA region (0 to minus 2 percent per year) (fig. 53).

Hardwood dieback was very low (2.8 to 4.7 percent) throughout most of the MAIA region (fig. 54), and even the MAIA ecoregion section with the highest average dieback (4.7 to 7.0 percent) was low compared to the rest of the U.S., where the highest average hardwood dieback levels were 13.6 to 21.7 percent (Conkling and others 2005).

Hardwood dieback levels had decreased or were staying about the same in most of the MAIA region (fig. 55), with some increase in dieback (less than 2 percent per year) in northeastern Pennsylvania.

Average softwood transparency values in the Appalachian Mountain area (Southern Unglaciated Allegheny Plateau, Northern Ridge and Valley, Allegheny Mountains, and Northern Cumberland Mountains (Bailey 1995) were 21.8 to 35.7 percent (fig. 56), the highest observed in the eastern U.S. (Conkling and others 2005).

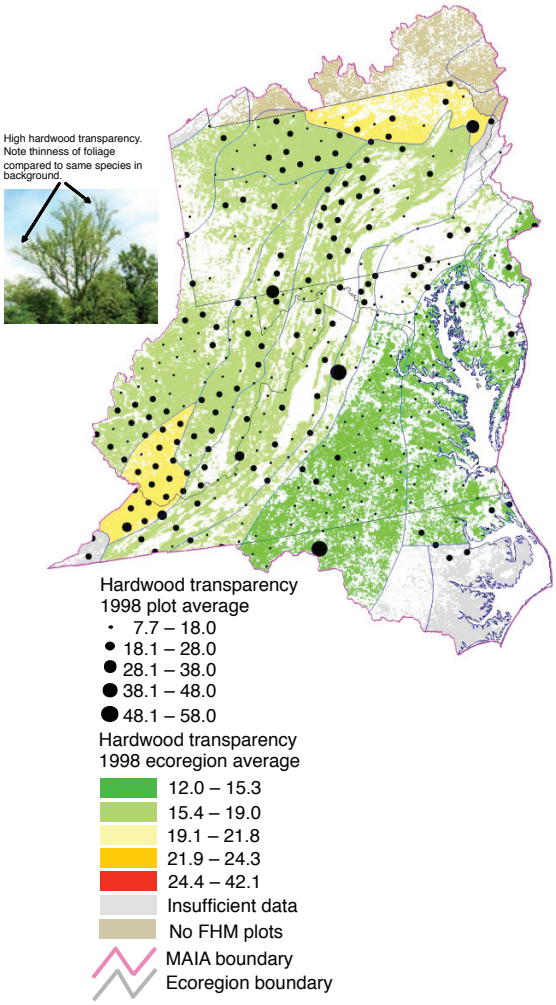


Figure 52—Average percent foliar transparency of hardwood trees in 1998 by ecoregion section in the MAIA region (colored polygons), derived from the average foliar transparency of hardwood crowns at each FHM plot (solid black dots) in each ecoregion section in 1998. The plot value was the actual value if the plot was measured in 1998, and an estimated value based on previous plot measurements otherwise. Note legend also gives average percent foliar transparency for hardwood trees in other ecoregion sections in the U.S. in 1998 for comparison to MAIA region. Data collected 1991-1998 in Delaware, Maryland, New Jersey, and Virginia; in Pennsylvania in 1995 and 1998; in North Carolina in 1998. Source: National Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

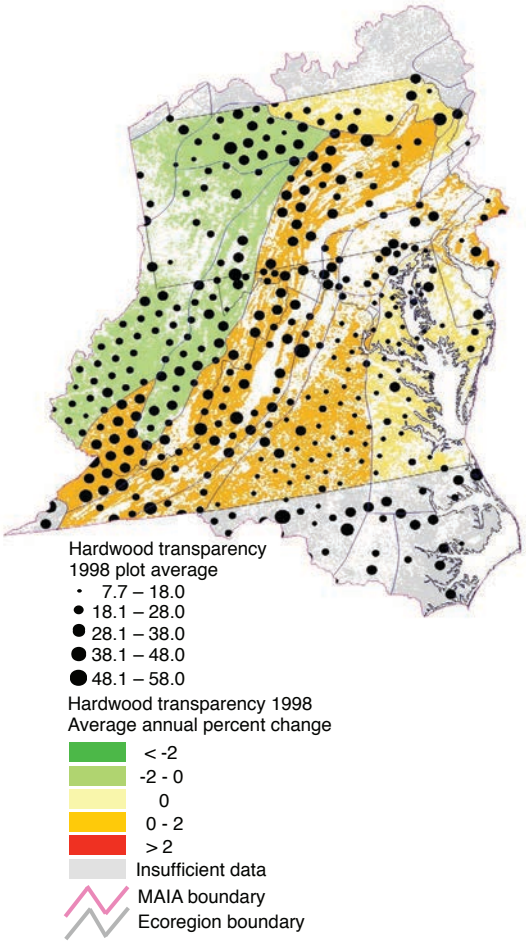


Figure 53—Average annual change in percent foliar transparency of hardwood tree for the period of record for each State by ecoregion section (colored polygons), derived from the average foliar transparency of hardwood crowns at each FHM plot (solid black dots) in each ecoregion section in 1998. The plot value was the actual value if the plot was measured in 1998, and an estimated value based on previous plot measurements otherwise. Note legend also shows annual percent change in hardwood foliar transparency for ecoregion sections outside of the MAIA region for perspective. Data collected 1991-1998 in Delaware, Maryland, New Jersey, and Virginia; in Pennsylvania in 1995 and 1998; in North Carolina in 1998.
Source: National Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

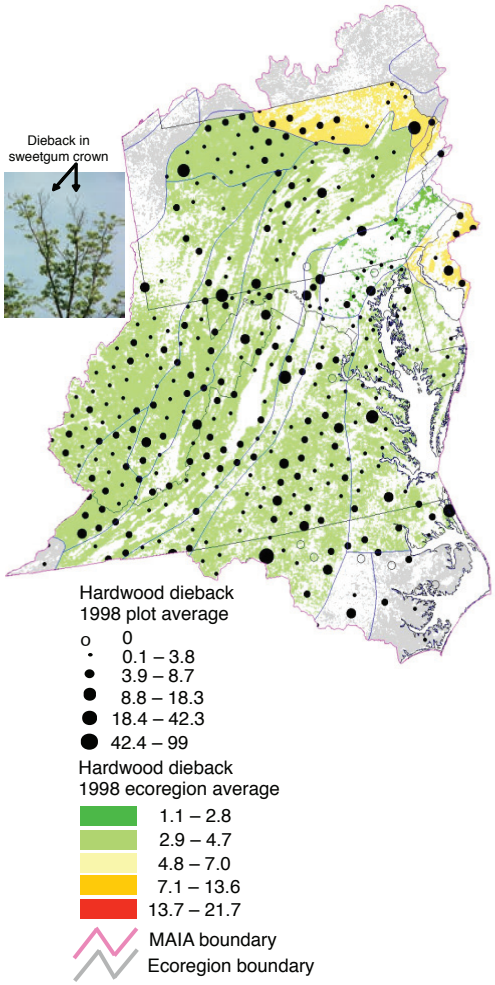


Figure 54—Average percent crown dieback of hardwood trees in 1998 by ecoregion section (colored polygons) in the MAIA region derived from the average crown dieback of hardwood crowns at each FHM plot (solid black dots) in each ecoregion section in 1998. The plot value was the actual value if the plot was measured in 1998, and an estimated value based on previous plot measurements otherwise. Note legend also gives average percent crown dieback for hardwood trees in other ecoregion sections in the U.S. in 1998 for comparison to MAIA region. Data collected 1991-1998 in Delaware, Maryland, New Jersey, and Virginia; in Pennsylvania in 1995 and 1998; in North Carolina in 1998. Source: National Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

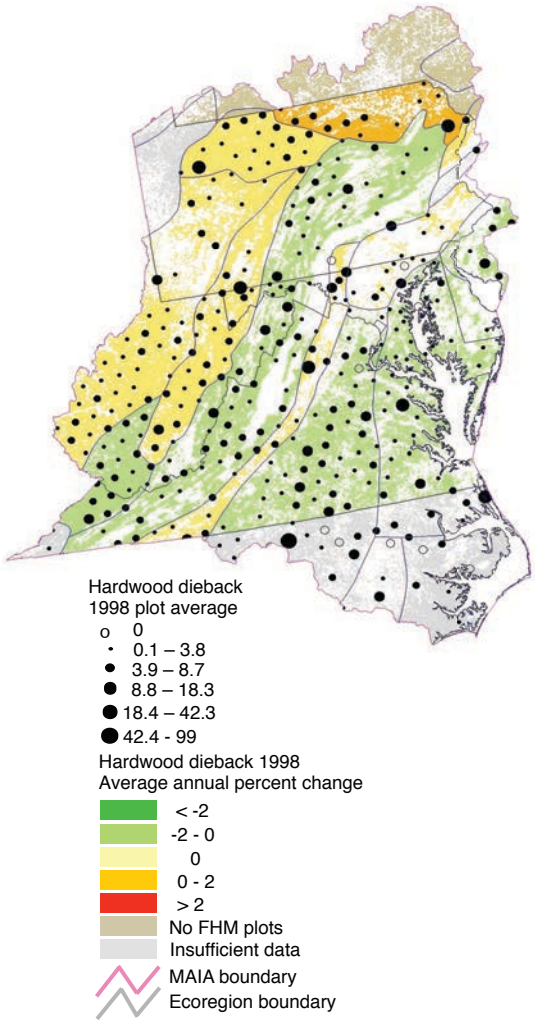


Figure 55—Average annual change in percent dieback of hardwood trees for the period of record for each State by ecoregion section (colored polygons), derived from the average foliar dieback of hardwood crowns in each FHM plot (solid black dots) in each ecoregion section in 1998. The plot value was the actual value if the plot was measured in 1998, and an estimated value based on previous plot measurements otherwise. Note legend also shows annual percent change in hardwood dieback for ecoregion sections outside of the MAIA region for perspective. Data collected 1991-1998 in Delaware, Maryland, New Jersey, and Virginia; in Pennsylvania in 1995 and 1998; in North Carolina in 1998. Source: National Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

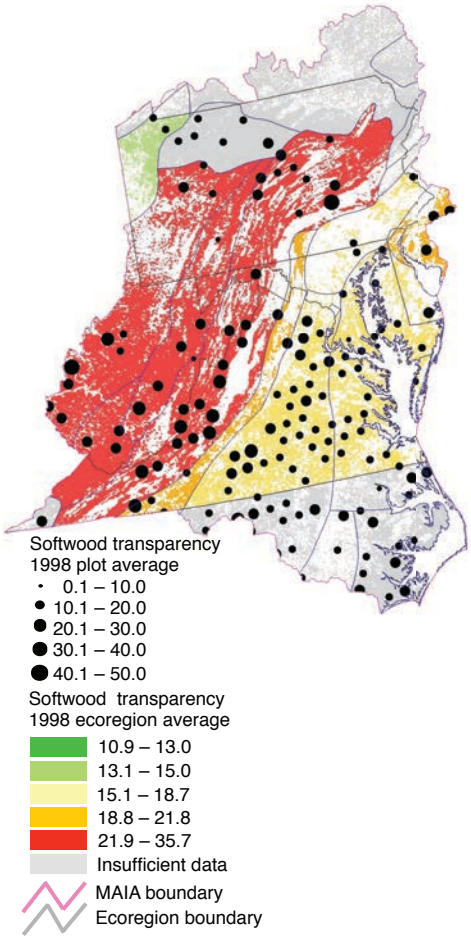


Figure 56—Average percent foliar transparency of softwood trees in 1998 by ecoregion section (colored polygons). The black circles show the average softwood foliar transparency at each FHM plot in 1998; the plot value is the actual value if the plot was measured in 1998 and an estimated value based on previous measurements otherwise. Data collected 1991-1998 in Delaware, Maryland, New Jersey, and Virginia; in Pennsylvania in 1995 and 1998; in North Carolina in 1998. Source: National Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

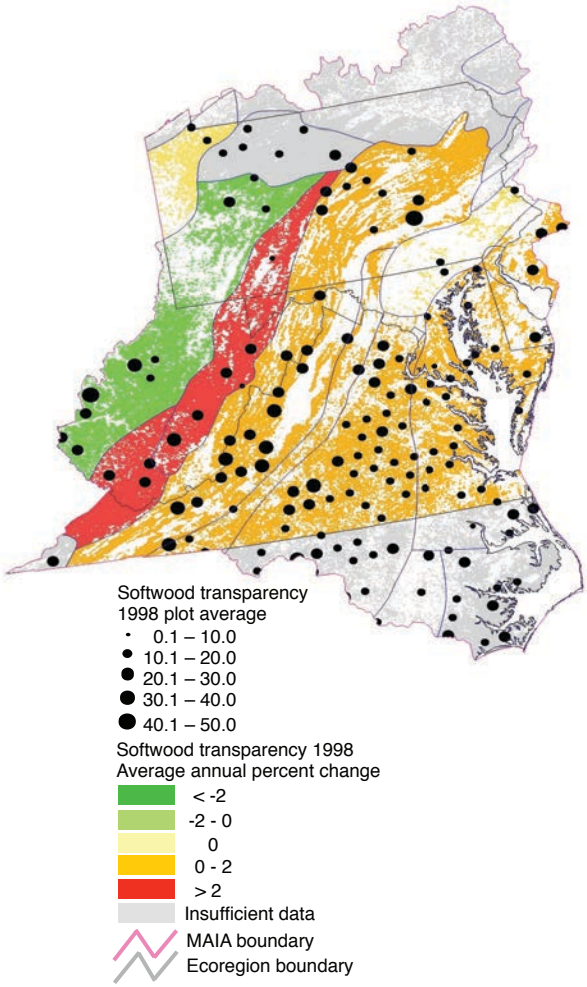


Figure 57—Average annual change in percent foliar transparency of softwood trees for the period of record in each State by ecoregion sections (colored polygons). The black circles show the average foliar transparency of softwood crowns at each FHM plot in 1998. Data collected 1991-1998 in Delaware, Maryland, New Jersey, and Virginia; in Pennsylvania in 1995 and 1998; in North Carolina in 1998. Source: National Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

Softwood foliar transparency was also observed to have increased in all of these ecoregion sections except for the Southern Unglaciated Allegheny Plateau (Bailey 1995) in the western part of the MAIA region (fig. 57).

Most of the high softwood crown transparency values were found in Virginia pine. Other species with high transparency included shortleaf pine, table mountain pine, and pitch pine. Analysis of these and other data by Burkman and Bechtold (2000) revealed a real decline in the crown condition of Virginia pine in this region. Natural pests and anthropogenic stressors may be contributing to declining health of the pines in this area, but the main reason appears to be the age structure of pine stands. Most of the pine stands in the region originated between 1880 and 1920 during a period of farm abandonment and reforestation. For a variety of reasons, there were only a few young pine stands replacing these old, senescing stands (Conkling and others 2005).

Softwood transparency in the western part of the MAIA region appeared to indicate a change in forest composition rather than a broad forest health issue. Older pine stands were succeeding to hardwoods. If these trends continued, the affected pine species would probably become a significantly smaller component of forests in the west (Conkling and others 2005).

Dieback of softwood tree crowns was highest (4.6 to 6.7 percent) in the Northern Ridge and Valley, and the Southern Unglaciated Allegheny Plateau ecoregions, as well as the Upper Atlantic Coastal Plain of New Jersey and Delaware (Bailey 1995) (fig. 58).

Most of the Coastal Plain and Piedmont had low dieback levels (0 to 3 percent) relative to other parts of the country, where softwood dieback average levels as high as 6.7 to 19 percent were observed (Conkling and others 2005). Dieback levels also were relatively low (1.3 to 3 percent) in the Allegheny and Northern Cumberland Mountains (Bailey 1995). Softwood dieback appeared to have increased only in the Southern Unglaciated Allegheny Plateau (0 to 2 percent per year), and was stable throughout the rest of the MAIA region (fig. 59).

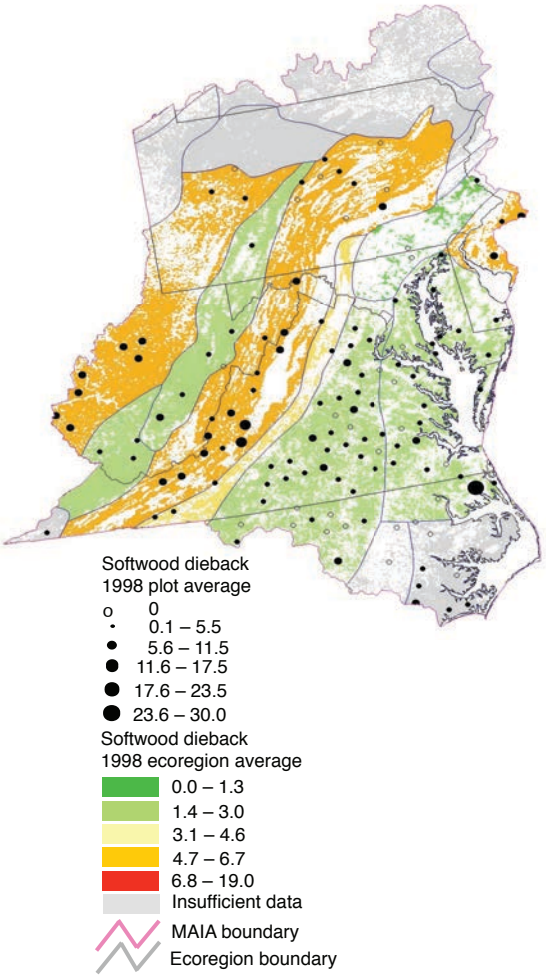


Figure 58—Average percent crown dieback of softwood trees in 1998 by ecoregion section (colored polygons). The black circles show the average softwood crown dieback at each FHM plot in 1998; the plot value is the actual value if the plot was measured in 1998 and an estimated value based on previous measurements otherwise. Data collected 1991-1998 in Delaware, Maryland, New Jersey, and Virginia; in Pennsylvania in 1995 and 1998; in North Carolina in 1998. Source: National Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

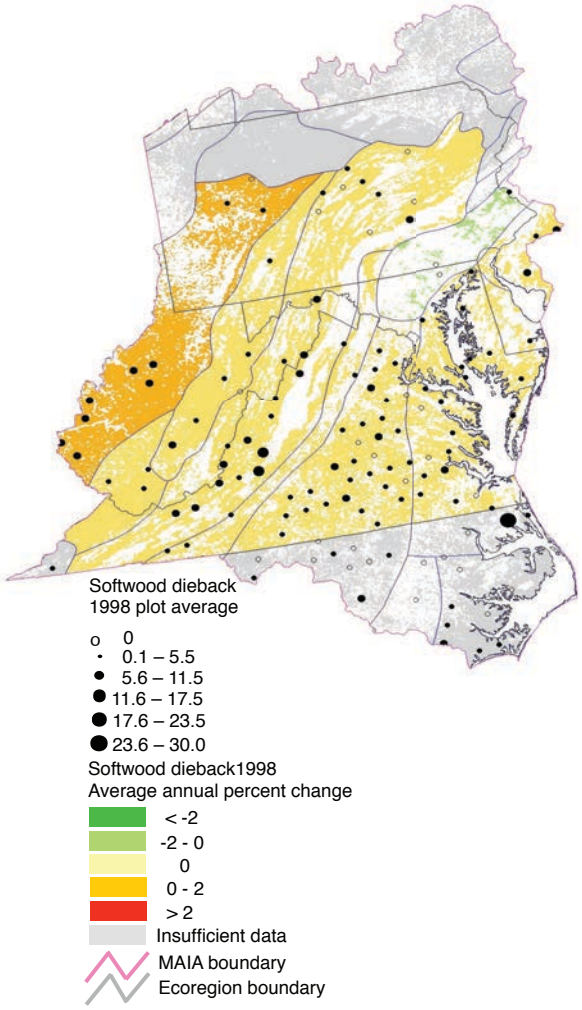


Figure 59—Average annual change in percent dieback of softwood trees for the period of record in each State by ecoregion sections (colored polygons). The black circles show the average dieback of softwood crowns at each FHM plot in 1998. Data collected 1991-1998 in Delaware, Maryland, New Jersey, and Virginia; in Pennsylvania in 1995 and 1998; in North Carolina in 1998. Source: National Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

Hardwood and Softwood Crown Condition by HUC-4 Watershed

In this section the condition of hardwood and softwood tree crowns was evaluated by watershed (fig. 6), using hydrologic unit codes size 4 (HUC-4) of the U.S. Geological Survey (http://water.usgs.gov/GIS/huc.html). Our purpose was to compare crown condition within watersheds and begin development of a framework for evaluating relationships between forest condition and aquatic conditions in the region. Because this type of analysis was done only for the MAIA region, and not the rest of the U.S., it was not possible to directly compare watershed conditions with other watersheds outside the MAIA region. The maps displayed in the figures in this section differ from those presented in the ecoregion analyses. Here, the range of watershed values observed in the MAIA region are divided into three categories for mapping purposes, even when the range of observed values is very narrow.

Hardwood foliar transparency was lowest (12.8 to 16.0 percent) in watersheds draining into Albemarle Sound and most of those draining into the Chesapeake Bay (fig. 60). Transparency was higher in watersheds draining into the Ohio River system and into Delaware Bay, as well as in the Susquehanna River watershed.

Hardwood crown dieback levels were low throughout the MAIA region (fig. 61). The highest dieback level observed in a MAIA region watershed was only 5.5 percent, and only a few scattered plots had dieback levels above 10 percent. The difference in average dieback between the best and worst watersheds was very small, about 3.4 percent.

Figure 62 shows foliar transparency values of softwoods by watershed in 1998. Softwood transparency was highest in the western part of the MAIA region, with average values of 25.1 to 31.7 percent in the Monongahela, Kanawha, Big Sandy/Guyandotte, and Middle Ohio River watersheds. Lower transparency values (12.3 to 25.0 percent) were observed throughout the rest of the MAIA region.

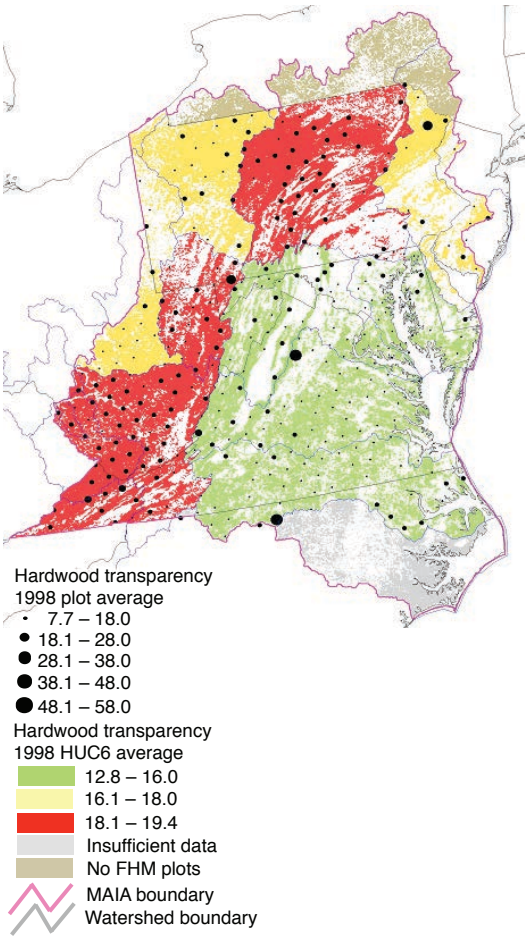


Figure 60—Average percent foliar transparency of hardwood trees in 1998 by major HUC4 watershed (colored polygons). The black circles show the average foliar transparency of hardwood crowns at each FHM plot in 1998; the plot value is the actual value if the plot was measured in 1998 and an estimated value based on previous measurements otherwise. Data collected 1991-1998 in Delaware, Maryland, New Jersey, and Virginia; in Pennsylvania in 1995 and 1998; in North Carolina in 1998. Source: Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

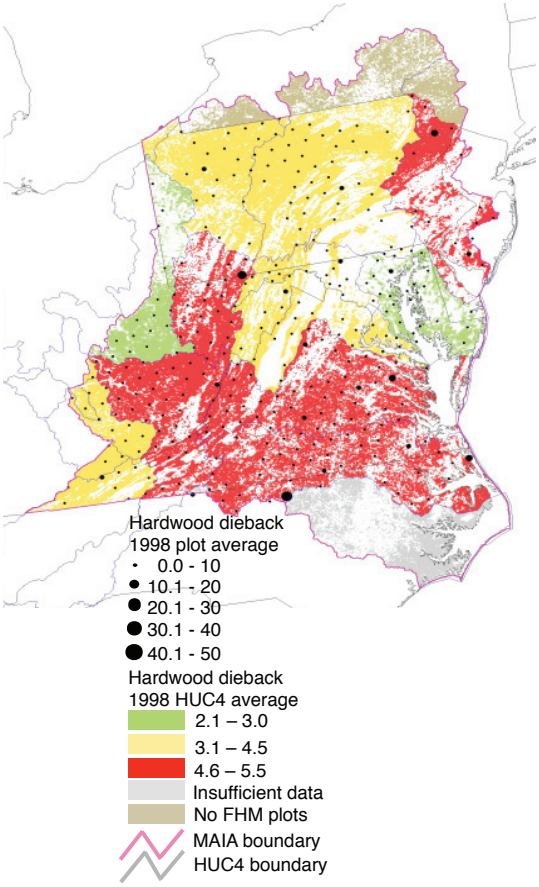


Figure 61—Average percent crown dieback of hardwood trees in 1998 by major HUC4 watersheds (colored polygons). The black circles show the average hardwood crown dieback at each FHM plot in 1998; the plot value is the actual value if the plot was measured in 1998 and an estimated value based on previous measurements otherwise. Data collected 1991-1998 in Delaware, Maryland, New Jersey, and Virginia; in Pennsylvania in 1995 and 1998; in North Carolina in 1998. Source: National Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

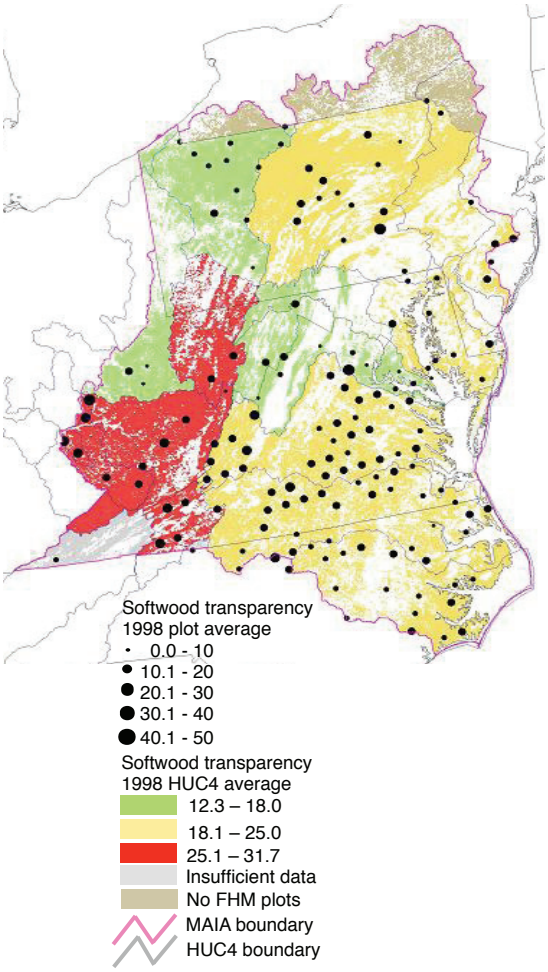


Figure 62—Average percent foliar transparency of softwood trees in 1998 by majorHUC4 watershed (colored polygons). The black circles show the average softwood foliar transparency at each FHM plot in 1998; the plot value is the actual value if the plot was measured in 1998 and an estimated value based on previous measurements otherwise. Data collected 1991-1998 in Delaware, Maryland, New Jersey, and Virginia; in Pennsylvania in 1995 and 1998; in North Carolina in 1998. Source: National Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

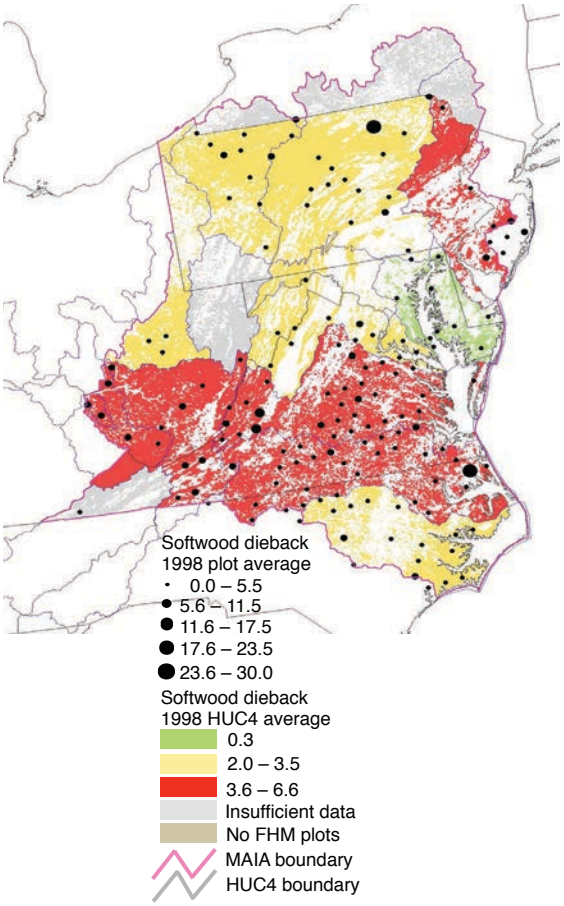


Figure 63—Average percent crown dieback of softwood trees in 1998 by major HUC4 watershed (colored polygons). The black circles show the average softwood crown dieback at each FHM plot in 1998; the plot value is the actual value if the plot was measured in 1998 and an estimated value based on previous measurements otherwise. Data collected 1991-1998 in Delaware, Maryland, New Jersey, and Virginia; in Pennsylvania in 1995 and 1998; in North Carolina in 1998. Source: National Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

The highest average softwood crown dieback by HUC-4 watershed in the MAIA region was 6.6 percent (fig. 63). There was a band of high dieback (3.6 to 6.6 percent) in the watersheds of southern West Virginia and most of Virginia, as well as high dieback in the Delaware Bay watershed. The lowest softwood dieback value (0.3 percent) was found in the Upper Chesapeake Bay watershed.

Tree Damage

Damage caused by pathogens, insects, storms, and human activities can significantly affect the growth, reproduction, and mortality of trees. In the field, we recorded tree damage if it was considered serious enough to increase the probability that a tree would be infected by lethal pathogens, that the tree would die prematurely, or that the growth and/or reproduction of the tree would be seriously depressed. To be recorded, damages must have met or exceeded set thresholds,e.g., > 20 percent bole circumference with an open wound, > 30 percent of the foliage damaged by > 50 percent (Mielke and others 1995). A score of zero does not necessarily mean that a tree is free of disease, storm, or defoliator damage: insect pests or pathogens may be present and possibly affect long-term tree productivity but will not be recorded unless levels exceed the predetermined thresholds. Also, because damages are not attributed to particular causal agents, FHM damage indicators are not appropriate for estimating how widespread insects or pathogens may be.

We estimated a damage severity index (DSI) score for each damaged tree. The DSI score was based on three variables: type of damage symptom, location of damage on the tree, and severity of the damage (Mielke 1999). The location of injury often affects the impact on the tree, e.g., injury near the base of the tree is more serious than injury near the apex of the tree because parts of the crown can be lost without killing the tree. Similarly, some damage symptoms are more serious than others. Open wounds, for example, can heal if they do not become infected, and therefore are not as serious as cankers, which are caused by fungal

Table 15—Damage severity index example based on type, location, and severity rating circa 2000

Type	Severity	Location						
Damages 1 and 3	Percent circumference affected	Roots	Roots, stump, lower bole	Lower bole	Lower and upper bole	Upper bole	Crown-stem	Branches
Cankers, galls, wounds	20-29	20	20	20	20	20	10	5
	30-39	30	30	30	30	30	15	10
	40-49	40	40	40	40	40	20	15
	50-59	50	50	50	50	50	25	25
	60-69	60	60	60	60	60	30	40
	70-79	70	70	70	70	70	35	55
	80-89	80	80	80	80	80	40	70
	90-99	90	90	90	90	90	45	85

Source: The National Forest Health Monitoring program; (www.fhm.fs.fed.us).

species that often kill the bark and cambium. A symptom's severity is simply an estimate of the area affected; which is to say that a canker affecting 80 percent of the tree-bole circumference is more serious than a similar canker affecting 30 percent of the tree-bole circumference. A DSI score for each damage occurrence was based on these three variables (table 15). The index value associated with each particular combination of damage type, location, and severity was instituted following several workshops with Federal, State, and university experts in forest pathology and entomology[4].

Up to three damages per tree could be scored: the scale ran from 0 to a theoretical maximum of 300, with zero indicating no damage above the minimum threshold being recorded and 300 indicating three damages of maximum severity. In reality, individual tree damage index scores rarely exceeded 90; trees usually died before damage levels get much higher. Generally, a high damage index indicates multiple damages, severe types of damage, and/or extensive damage, with the damages often occurring near the base of trees.

Our analysis of tree damage used only the most recent measurement of each forested plot through 1999. The mathematical formula for our plot-level damage index is presented in Technical Appendix B. Damage severity indices were determined for individual trees, and then averaged at the plot level separately for softwoods and hardwoods. The average percent of trees damaged per plot for softwoods and hardwoods was calculated for each major watershed.

Interpreting tree damage and its relationship to forest health is complex—tree damage is the result of a variety of processes, both deterministic and stochastic. Some processes are anthropogenic, some part of natural disturbance regimes, and others are secondary impacts that result from changes in forest ecosystems that sometimes follow forest management practices.

The number of trees damaged, and average damage values, were quite low throughout the MAIA region, especially for softwoods. No watershed in the region had more than 20 percent average of softwood trees damaged (fig.64), while nationwide some ecoregion sections had 30 to 40 percent of softwoods damaged (Conkling and others 2005).

[4]Personal communication. 2001. Manfred Mielke, USDA Forest Service, Northeastern Area State and Private Forestry, 1992 Folwell Avenue, St. Paul, MN, 55108

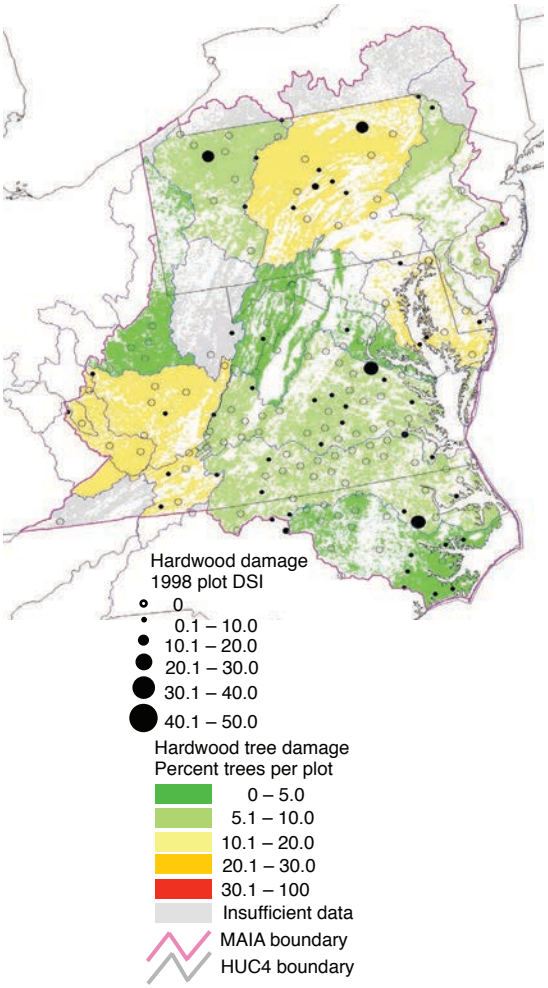


Figure 64—Average percentage of hardwood trees per plot that had any significant damage to the roots, trunk, bole, or crown by major HUC4watershed (colored polygons). The closed circles indicate the Damage Severity Index (based on the type of damage, severity, and location on the tree) value of hardwood trees on each FHM plot. Data collected 1991-1998 in Delaware, Maryland, New Jersey, and Virginia; in Pennsylvania in 1995 and 1998; in North Carolina in 1998. Source: National Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

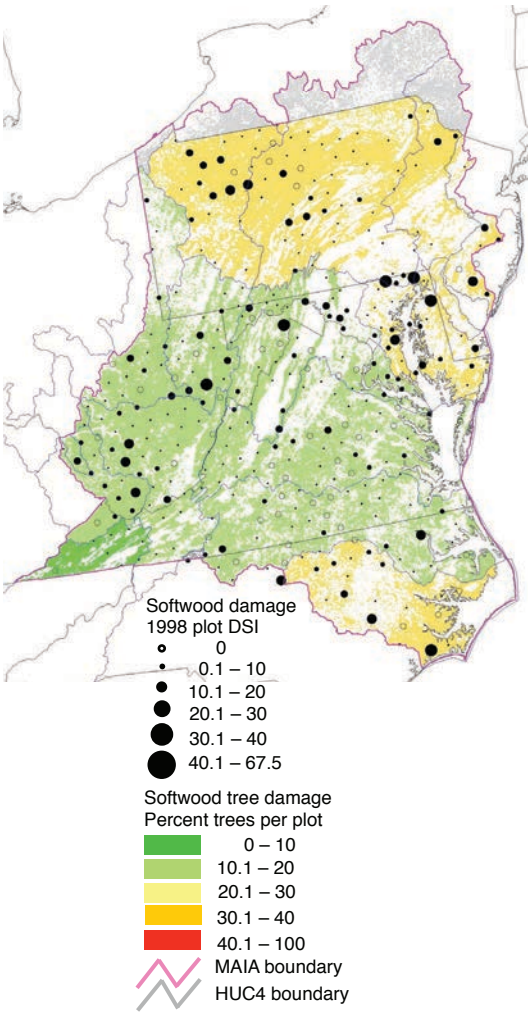


Figure 65—Average percentage of softwood trees per plot that had any significant damage to the roots, trunk, bole, or crown by major HUC4 watershed (colored polygons). The black circles indicate the Damage Severity Index (based on the type of damage, severity, and location on the tree) value of softwood trees on each FHM plot. Data collected 1991-1998 in Delaware, Maryland, New Jersey, and Virginia; in Pennsylvania in 1995 and 1998; in North Carolina in 1998. Source: National Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

Hardwood damage levels were somewhat higher but still relatively low compared with values observed elsewhere in the U.S. (Conkling and others 2005). Hardwood damage was highest (20 to 30 percent) in northern watersheds and the Pamlico watershed in the southeast MAIA region (fig. 65). There were scattered individual plots with very high average DSI values (e.g., 40.1 to 67.5) that might be related to localized outbreaks of insects or pathogens, storm events, or some other causal agents.

Tree Mortality

Loss of tree volume due to mortality is a natural part of any forest ecosystem. Annual mortality, in terms of wood volume per acre, was based on trees that had died since the monitoring plot was first established. Because different forest types grow under different conditions at different rates, a simple measure of mortality volume is not a good measure of forest health on a national basis. For example, greater tree volume may be lost in a healthy forest in the Southeast than the total standing volume in some dry western forests. A better mortality indicator is the ratio of annual mortality volume to gross volume growth (MRATIO). A MRATIO value >1 indicates that mortality is exceeding growth, and that live standing volume actually is decreasing. The MRATIO can be large if an over-mature forest is senescing and losing a cohort of older trees. If forests are not naturally senescing, a high MRATIO (>0.6) may indicate high mortality due to some acute cause (insects or pathogens), or generally deteriorating forest health conditions.

Another aspect of tree mortality is the size of the trees that have died relative to the surviving trees. We also calculated the ratio of the average dead tree d.b.h. (diameter at breast height) to the average live tree diameter (DDLD ratio) for each plot where mortality occurred. Low (much <1) DDLD ratios usually indicate competition-induced mortality typical of young, vigorous stands, while relatively higher ratios (>1) likely indicate mortality of larger trees associated with senescence or some external stress factors such as insects or pathogens.

Analysis of tree mortality by ecoregion section gave moderate MRATIO values throughout most of the MAIA region (MRATIO of 0.3 to 0.6) (fig. 66). This means that for every cubic foot of wood gained in annual growth, between 0.3 to 0.6 cubic feet of wood was lost to mortality. The MRATIO was lowest (0.10 to 0.30) in the Allegheny Mountains and the Northern Cumberland Mountains (Bailey 1995).

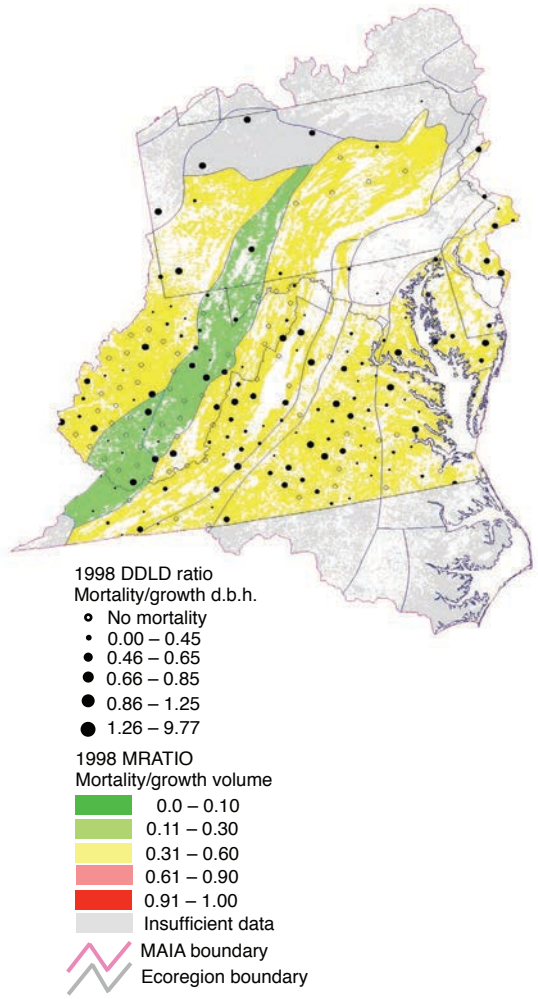


Figure 66—Tree mortality volume by ecoregion section expressed as the ratio of annual mortality volume to annual growth volume (colored polygons). Mortality ratio volumes of 1.0 indicate that there was no net gain in tree volume on the plot. The closed black circles represent plot-level values of the ratio of the average diameter of trees that died to the average diameter of the surviving trees on each plot (DDLD ratios). DDLD ratios of 1.0 indicate that on average the trees that died were as large as the surviving trees. Data collected 1991-1998 in Delaware, Maryland, New Jersey, and Virginia; in Pennsylvania in 1995 and 1998. Source: National Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

Mortality volume relative to growth volume (MRATIO) for hardwood and softwood trees combined was highest in the Allegheny watershed (MRATIO = 0.79) (fig. 67). An MRATIO of 0.79 means that for each cubic foot of wood produced, 0.79 cubic feet of wood was lost to mortality, and only 0.21 cubic feet of wood was gained.

The DDLD ratio values for several plots within the Allegheny watershed also were relatively high (0.81 to 2.8). This suggests that stands were either over mature and senescing, or other stressors, such as insects or pathogens, were affecting forests in the area. MRATIOs were relatively low in the Susquehanna, Upper Ohio, and Upper Tennessee watersheds (0.23 to 0.30). Mortality estimates for watersheds located mostly in Pennsylvania were derived from a very small sample. Multiple plot measurements had been taken only from one fourth of plots in Pennsylvania as of 1998, so it was difficult to reach any strong conclusions about the relative health of forests in the Allegheny and Susquehanna watersheds from mortality data alone.

In general the process-linked indicators of tree crown condition, damage, and mortality suggested much of the mid-Atlantic forests were in relatively good condition compared to other forests in the U.S. These indicators show that major regional-scale stressors were not substantially affecting long-term survival or productivity of MAIA forests. Some plots and localized areas might have warranted more intensive study. For example, mortality ratios in the Allegheny watershed (fig. 67) could be re-evaluated, as more data from Pennsylvania become available (<www.fia.fs.fed.us>), and especially if any of these areas also have higher crown dieback or transparency. Other issues were the annual increase in foliar transparency of hardwoods (fig. 53), and the condition (fig. 56) and annual change (fig. 57) in softwood transparency in the western region. Individual plots with relatively high crown dieback and transparency, damage, and mortality ratios may be harbingers of future more-widespread problems (figs. 52 to 67).

Tree condition problems affecting individual species, or groups of species, may not have been detected because our analyses were grouped by hardwood and softwood species. Health problems of species with few trees present would likely be masked in watershed-based analyses (that often combine multiple forest types and more tree species), in contrast to ecoregion section analyses that typically contain fewer major forest types and tree species. Also, additional research is needed to definitively relate crown indicators to specific causal agents.

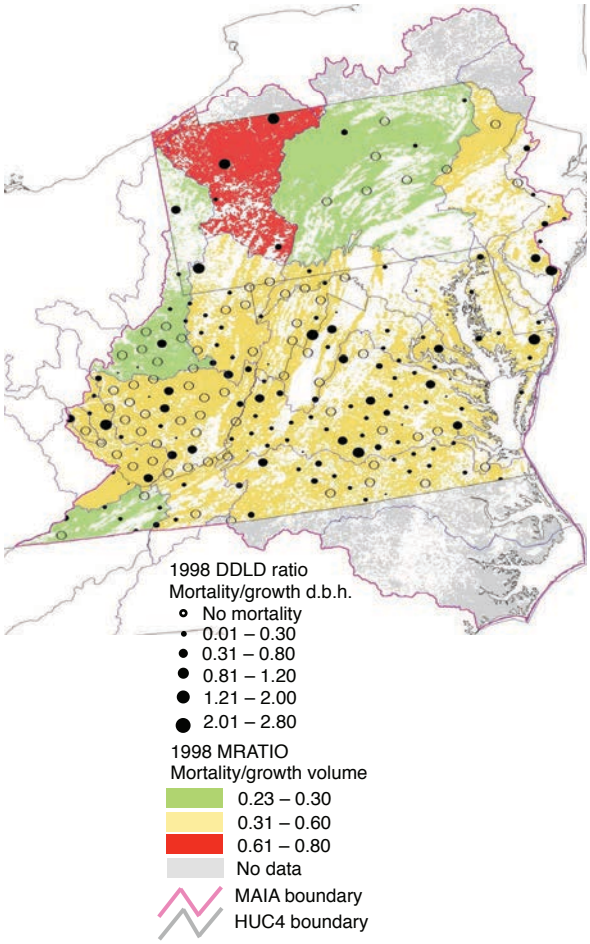


Figure 67—Tree mortality by major HUC4 watershed expressed as the ratio of annual mortality volume to annual growth volume (colored polygons). The black circles represent the ratio of the average diameter of trees that died to the average diameter of the surviving trees on each plot. Data collected 1991-1998 in Delaware, Maryland, New Jersey, and Virginia; in Pennsylvania in 1995 and 1998. Source: Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

PRODUCTIVITY

Chapter 14.

Tree Volume, Age, Growth, and Density

James Steinman
Northern Research Station, USDA Forest Service

Measures of tree productivity provide important general indicators of the health and sustainability of forest ecosystems. Standing volume, forest types and species, and rate and amount of tree volume change are included in our measure of tree productivity. Tree productivity directly relates to environmental factors such as soil fertility and toxicity, root system vitality, climate, weather, insects and pathogens, air quality, and other factors. Measurement of tree volume provides a baseline to compare future tree growth and thereby establish changes in volume, and to identify areas where tree growth is low or high. Areas with good tree growth are indicative of supportive soil, climate, weather, and other conditions; which contribute to the value of those areas for recreation, wildlife habitat, and timber and non-timber forest products.

The collective growth rate of forest stands over time is a common measure of tree productivity. The measurement of growth is strongly correlated with increase in total tree biomass, i.e., the accretion of wood volume in boles (stem or main trunk) of trees. We typically express net rates of tree-stand growth by deducting the volume lost to mortality and harvesting from increments of stem wood gained on living trees. Negative changes in tree volume indicate that more tree volume has been lost to mortality and/or harvesting than gained in growth (see MRATIO in Chapter 13). Increasing declines in net tree volumes should be evaluated to determine causal agents, suggest remedial activities, and possibly mitigate additional loss of timber.

Tree Growth Rates

Forest stands in counties of the MAIA region had an average growth rate of 50 cubic feet of wood volume per acre each year (fig. 68), but counties in the southeast portion often averaged more than 60 cubic feet. The lowest growth rates, 40 cubic feet, occurred in West Virginia, central Pennsylvania, and parts of the northeast coastal areas. Annual growth rates in individual stands ranged from 0 to 200 cubic feet per acre. The loblolly pine–shortleaf pine forest type, within which a high proportion of plantation stands are managed for timber production, had 90 cubic feet per acre, the highest average annual rate of new tree volume in the region. Plantation stands are typically harvested more often, a strategy to keep trees in a high growth-rate stage. Maple–beech–birch stands had the lowest average annual rate among all forest types at 30 cubic feet per acre. Cool northern climates and the inherently slow growth rates of sugar maple and American beech explain the relatively lower productivity of these forest types.

Standing Volume of Forest Stands

The volume of wood in forests represents the cumulative production of trees since their establishment as seedlings. The distribution of standing tree volume (cubic feet per acre) in forests of the MAIA region is shown in figure 69.

The average volume was 1,800 cubic feet per acre. The oak–gum–cypress forest type on the coastal plain had an average volume of 2,400 cubic feet per acre, the highest of any forest type. Those stands tended to be older and denser than those composing the other forest types, and therefore had greater accumulations of wood volume. Hardwood stands in western Virginia, central West Virginia, and north-central Pennsylvania, and conifer stands in northern North Carolina,

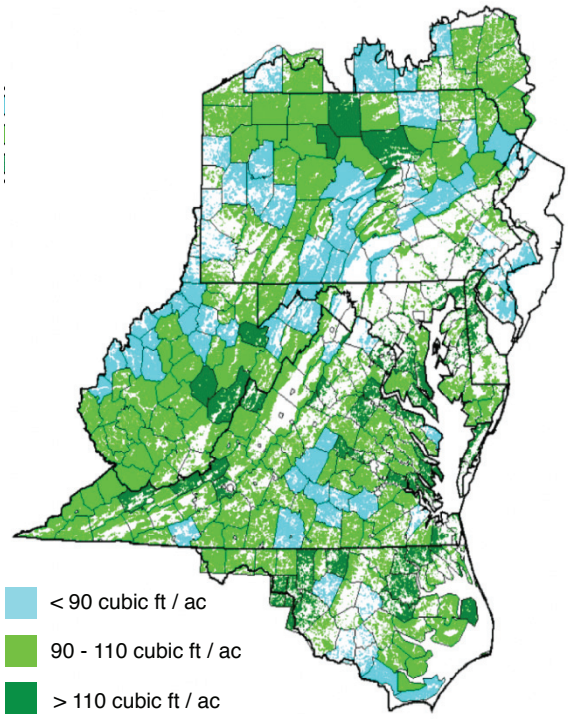


Figure 68—Forest stand productivity in MAIA counties as average net annual growth of trees (gross volume growth minus mortality volume) in cubic feet per acre per year circa 2000. Source: Forest Inventory and Analysis program data; (http://fia.fs.fed.us/).

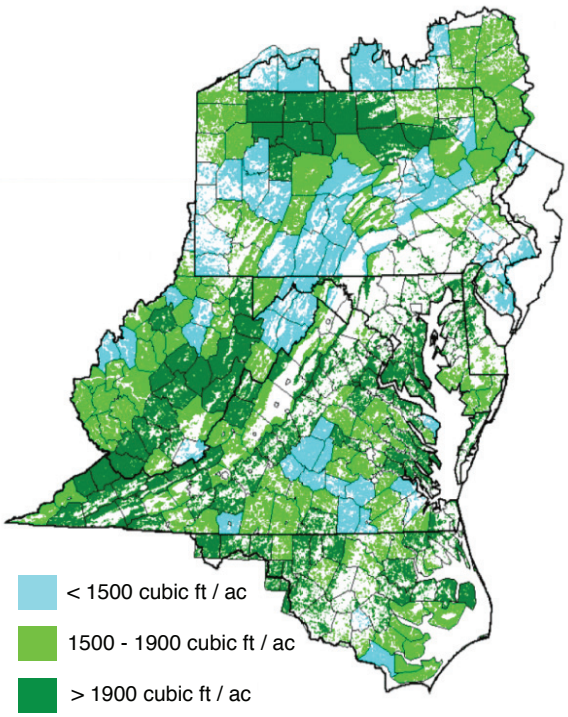


Figure 69—Forest stand volume in MAIA counties as average tree volume in cubic feet per acre prior to 2000. Source: Forest Inventory and Analysis program data; (http://fia.fs.fed.us/).

also had counties where tree volumes exceeded 1,900 cubic feet per acre. Although loblolly pine stands in southeast Virginia and northeast North Carolina had greater growth rates (fig. 68), they had an average volume of only 1,800 cubic feet per acre, about on-par with the regional average. It is typical for younger stands to have lower volume but relatively high annual growth rates.

Forest Stand Age

Tree productivity is partly reflected in a stand's age, and stand age usually will determine the average tree size for a given forest type. The growth rate of regenerating forest stands generally increases as trees age and establish large crowns and root systems. Conversely, growth rates of stands begin to decline as tree canopies close, and nutrients and other resources become limiting. We aged tree stands by counting the number of annual growth rings on site trees, which are trees in a stand meeting specified criteria for determining stand age (USDA Forest Service 1995; <www.fia.fs.fed.us>).

The forest stands in the MAIA region averaged 60 years, and ranged from 0 to 120 years. About half of the region's forest area had stands with more than half the trees 10-inches diameter at breast height (d.b.h.) or greater. Trees ≥10-inches d.b.h. were harvestable by industry standards (<www.fia.fs.fed.us>), but are relatively immature ecologically. Only 20 percent of forest stands were in an immature stage of development (< 10 inches d.b.h.), or considered immature from a timber production point of view (fig. 70).

Most of these immature stands were loblolly pine stands in the Piedmont region of Virginia and North Carolina that were younger than the regional average, with many stands composed of relatively smaller, fast-growing trees (fig. 68). Many oak–hickory forests in West Virginia, parts of western Pennsylvania, and southern New York were the oldest and considered harvestable by industry standards (fig. 70). Many oak–gum–cypress forest stands found on the coastal plain of Virginia and North Carolina also were older and had larger average-sized trees than the rest of the forested region (fig. 69).

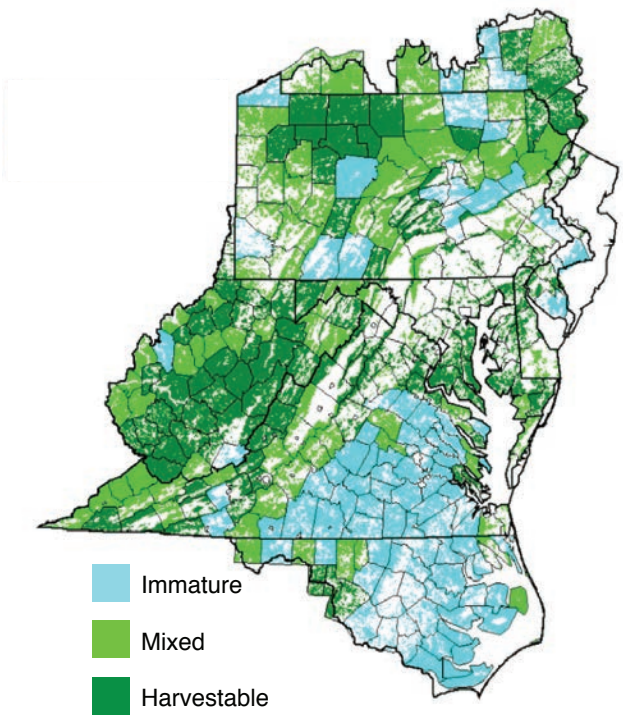


Figure 70—Age of forest stands in MAIA counties based on average size of trees. Ages relate to industry standards for potential tree harvest, and do not reflect ecological maturation. Immature = stands with the majority of trees 5 through 10 inches diameter at breast height (d.b.h.); mixed = stands with about half the trees 5 to 10 inches d.b.h. and half the trees greater than 10 inches d.b.h.; and harvestable = stands with the majority of trees greater than 10 inches d.b.h. Source: Forest Inventory and Analysis program data; (http://fia.fs.fed.us/).

Forest Stand Density

The productivity of forest stands also is related to how trees are spaced. Stand productivity is higher in stands where trees fully occupy all available growing space. A commonly used measure of occupancy is stand basal area (ba), which is the total cross-sectional area of all tree stems at 4.5 feet above the ground per unit area. In the MAIA region, forest stands averaged 100 ft^2 ba per acre. Most counties averaged 90 to 110 ft^2 ba per acre, some averaged < 90 ft^2 per acre, and only a few counties had > 110 ft^2 ba per acre (fig. 71). Individual stands of oak–gum–cypress forest type in coastal areas were the most dense in the region, with up to 250 ft^2 ba per acre. In a few counties this forest type also had the highest average stand basal area (125 ft^2 per acre) of all forest types. Equal proportions of stands with high, medium, and low densities occurred in forests in counties of the MAIA region, and stands of various densities were distributed evenly across the landscape (fig. 71).

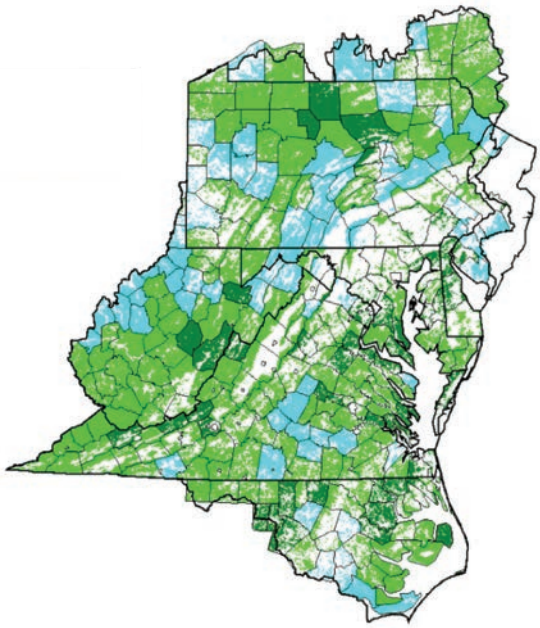


Figure 71—Forest stand density in MAIA region counties, in basal area feet squared per acre (ft^2 / ac) per unit area circa 2000. Source: Forest Inventory and Analysis program data; (http://fia.fs.fed.us/).

Chapter 15.

Market Benefits

Evan Mercer and Aruna Murthy
Southern Research Station, USDA Forest Service; American Express, US-Global Financial Services

Timber Production

To analyze timber production in the MAIA region, we examined the variety of timber products made from 1970 to 1990, highlighted changes that occurred, and provided an insight into spatial patterns of timber production. Because there is not a single source of data that covers the entire region, we used data from various surveys of wood-product manufacturers to compile a data set of timber product output for various States in the region. Unfortunately, the surveys were conducted in different years for different products in different States. Therefore, we used the decades of 1970, 1980, and 1990 as common periods of reference to analyze timber production. Data from the most recent survey in each decade provided estimates of annual timber production for a representative year in each decade.

Quantities of Sawlogs and Pulpwood

Annual timber production in the MAIA region was 1,021 million cubic feet (mmcf) during the 1970s, increasing to 1,137 mmcf in the 1980s (table 16; figs. 72 and 73). Sawlog production figures for 1990 were available for only three States (New York, North Carolina, and Virginia), so discussion of general timber production trends for the entire MAIA region was limited to 1970 and 1980.

Figure 72—Volume of annual hardwood production in the MAIA region in decades 1970s and 1980s. Source: Forest Inventory and Analysis program data; (http://fia.fs.fed.us/).

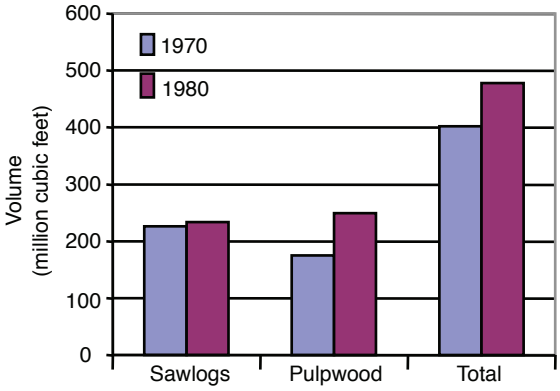


Figure 73—Volume of annual softwood production in the MAIA region in decades 1970s and 1980s. Source: Forest Inventory and Analysis program data; (http://fia.fs.fed.us/).

During the 1970s, sawlogs constituted 60 percent of total timber production. However, during the 1980s, the sawlog share of total volume declined by 6 percent to 54 percent, while the pulpwood share increased to 46 percent. Total pulpwood production expanded in the region by about 25 percent in the 1970s and 1980s. Overall, the hardwood share of both sawlogs and pulpwood total volume produced was higher than softwoods in the region. Hardwoods accounted for 62 to 63 percent of all sawlogs produced.

Virginia accounted for about one third of total timber production in the MAIA region in the 1970s and 1980s (fig. 74), with North Carolina second, even though only 47 counties in North Carolina were evaluated as part of the MAIA region. Sawlogs accounted for 59 percent of Virginia's timber production in the 1970s, 56 percent in the 1980s, and 52 percent in the 1990s (table 16). North Carolina's portion of the region yielded 26 percent of total production in the 1970s, and 32 percent in the 1980s (fig. 74). By the 1990s, total production in North Carolina exceeded Virginia (table 16).

Timber production in Pennsylvania was lower than in Virginia and North Carolina (fig. 74), and was dominated by hardwoods, comprising 92 percent (1970s) and 94 percent (1980s) of total timber production (table 16). In contrast, hardwoods comprised about half of Virginia's production, and about a third of North Carolina's total timber production in the 1970s, 80s, and 90s. Other States'

Table 16—Annual hardwood and softwood sawlog and pulpwood production in the MAIA region in 1970s, 80s, and 90s

Region	Year[a]	Softwood Sawlogs	Softwood Pulpwood	Softwood Total	Hardwood Sawlogs	Hardwood Pulpwood	Hardwood Total	Grand Total	Share of Total MAIA production
		-------million cubic feet--------			----- million cubic feet------			-mmcf[d]-	---percent--
Delaware									
	1970	1.0	3.8	4.8	1.0	0.9	1.9	6.6	0.7
	1980	0.4	1.5	1.9	2. 5	0.2	2.7	4.6	0.4
	1990	*NA*[b]	2.1	*NA*	*NA*	0.6	0.6	*NA*	*NA*
Maryland									
	1970	5.3	14.8	20.1	18.3	9.1	27.4	47.5	4.7
	1980	6	17.9	23.9	16.2	10.3	26.5	50.4	4.4
	1990	*NA*	8.6	*NA*	*NA*	7.4	7.4	*NA*	*NA*
New Jersey									
	1970	0. 5	1.8	2.3	2.5	0.09	2.6	4.8	0.5
	1980	0.3	0.8	1.1	2	0.2	2.2	3.3	0.3
	1990	*NA*	0.1	*NA*	*NA*	0.01	0.01	*NA*	*NA*
New York									
	1970	6.1	3.2	9.3	56.8	3.7	60.5	69.8	6.8
	1980	*NA*	5.8	5.8	*NA*	4.5	4.5	*NA*	*NA*
	1990	9.5	5.2	14.7	40	4.7	44.7	59.4	NA
North Carolina									
	1970	111.1	73.8	184.9	36.2	48.4	84.6	269.5	26.4
	1980	127.2	118.3	245.5	43.2	76.6	119.8	365.3	32.1
	1990	165.9	104.7	270.6	49.3	92.2	141.5	412.1	*NA*
Pennsylvania									
	1970	8.7	4.8	13.5	95	72.5	167.5	181	17.7
	1980	7.2	4.7	11.9	109.4	91.8	201.2	213.1	18.7
	1990	*NA*	4.9	*NA*	*NA*	55	55	*NA*	*NA*
Virginia									
	1970	91.4	67.9	159.3	108.3	70.8	179.1	338.4	33.2
	1980	92	94.2	186.2	130.6	83.5	214.1	400.3	35.2
	1990	92.6	105.1	197.7	113	84.4	197.4	395.1	*NA*
West Virginia									
	1970	2.9	5	7.9	69.3	25.8	95.1	103	10.1
	1980	0.99	5.9	6.89	76	17.2	93.2	100.1	8.8
	1990	*NA*	7.9	*NA*	*NA*	24.7	24.7	*NA*	*NA*
Mid-Atlantic Region									
	1970	226.9	175.1	402.0	387.4	231.3	618.7	1020.7	-----
	1980	234.1	249.1	477.4	379.9	284.3	659.7	1137.1	-----
	1990[c]	268.0	238.6	506.6	202.3	269.0	471.3	977.9	-----

[a]Surveys were conducted in different years for different products. The decades of 1970, 1980 and 1990 were considered a common point of reference. The data from the most recent survey in that decade is used for reporting sawlog production. Pulpwood production corresponding to that timeframe, or closest to that timeframe, is reported.
[b]NA = Not available.
[c]Includes only New York, Virginia, and North Carolina for 1990.
[d]mmcf = million cubic feet.
Source: Timber Product Output and pulpwood surveys conducted by the USDA Forest Service; (http://fa.fed.us).

contribution to the manufacture of timber products was relatively small. West Virginia produced smaller quantities of sawlogs and pulpwood from softwoods (7 to 8 mmcf) than all the other larger MAIA States in 1970 and 1980, but production of hardwood sawlogs and pulpwood was about 12 times greater (95 mmcf from hardwoods in 1970 and 93 mmcf in the 1980s) (table 16). Other MAIA States had a more balanced production of sawlogs and pulpwood from softwoods and hardwoods.

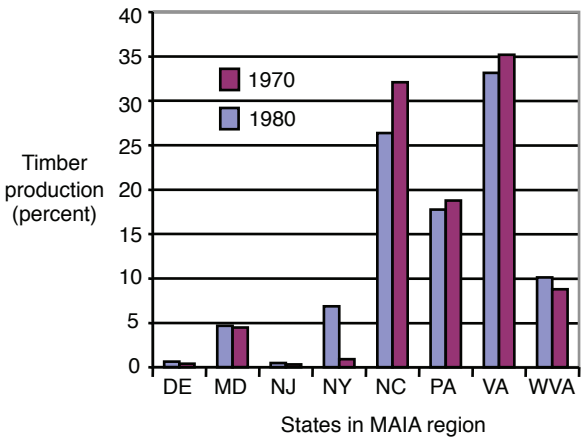


Figure 74—Share by State of total timber production in the MAIA region in decades 1970s and 1980s. Source: Forest Inventory and Analysis program data; (http://fia.fs.fed.us/).

Sawlog Removals of Species and Species-Groups

The market value of sawlogs depends on a number of factors such as species, physical and chemical characteristics of the wood, availability, and demand. Table 17 summarizes the average annual removal of sawlogs, by species and species groups, by State and the whole MAIA region during the 1980s. Timber removal records from the USDA Forest Service Eastwide Database (Hansen and others 1992) provided the volume (in thousand board feet) of sawtimber harvested by species-groups and species. We used the average annual sawlog removals from the most recent forest survey in each State prior to 2000 for this analysis.

For the entire MAIA region the softwood share of total sawlog removals was 38 percent while the hardwood share was 62 percent (table 17). Hardwoods were predominant in New Jersey (100 percent), Pennsylvania (95 percent), West Virginia (93 percent), New York (84 percent) and Maryland (82 percent). The softwood share of total removals was greatest only in Delaware (54 percent) and North Carolina (60 percent). The proportion of softwood removal in Virginia was 42 percent, and proportion of softwood removals in the remaining States was small.

Shortleaf and loblolly pines accounted for 48.1 percent of all softwood removals in the region (table 17). About 83 percent of total softwood sawlog removals in North Carolina and Maryland, and 91 percent in Delaware, were loblolly and shortleaf pines. Hemlock accounted for 57 percent of total softwood removals in Pennsylvania and 33 percent in New York. Red oaks, white oaks, and yellow poplar accounted for nearly 61 percent of total hardwood removals for the region. Yellow poplar sawlog removals were 41 percent of hardwood in Maryland, 24 percent in Virginia, 21 percent in North Carolina, 14 percent in West Virginia, and 11 percent in New Jersey.

In North Carolina and New Jersey sweet gum sawlog removals were 18 percent and 14 percent, respectively, of the total hardwood removals in those States (table 17). In other States sweet gum either was not removed or removed only in small amounts. Maple sawlogs were removed predominantly in New York and Pennsylvania at 32 percent and 18 percent, respectively, of the total hardwoods removed in those States. North Carolina and Virginia yielded 11 percent and 2 percent, respectively. About 17 percent of total hardwood removals in Pennsylvania, and 10 percent in New York, belonged to the ash-walnut-cherry species group. In the remaining States it was less than 2 percent.

Timber Inventories

Timber volume numbers are affected by growth, mortality, and removals. Timber inventories in different years for different States in the MAIA region were averaged by State for the period 1970 to 1990 and included sawlog information in the 1990s for only New York, North Carolina, and Virginia. Timber was estimated at 102.9 billion ft^3 of total growing stock, of which 57.9 billion ft^3 (56 percent) was in sawlog form (table 18)[5]. Over 63.5 million acres of timberland were in growing stock (table 2), which averaged about 1,620 ft^3 per acre. Hardwoods comprised about 80 percent of the growing stock in the

[5]Growing-stock volume is defined as the volume expressed in cubic feet (ft^3) of solid wood in trees with diameters at breast height (d.b.h.) of ≥ 5 inches from a 1-foot stump to a minimum 4.0-inch top diameter of tree bole. Sawtimber volume is the volume in the sawlog portion of sawtimber trees with dimensions of at least 9.0-inches d.b.h. for softwoods (between 1-foot stump and 7-inch top) or 11.0-inches d.b.h. for hardwoods (between 1-foot stump and 9-inch top). See text in Introduction and Glossary for more detail.

Table 17—Average annual sawlog removals in MAIA region States in 1980s

SPECIES	WV	WV	MD	MD	DE	DE	NJ	NJ	VA	VA	PA	PA	NY	NY	NC	NC	MAIA Region	MAIA
	-mmcf[a]*-*	*-%*[b]*-*	*-mmcf-*	*-%-*	*-mmcf-*	*-%-*	*-mmcf-*	*-%-*	*-mmcf-*	*-%-*	*-mmcf-*	*-%-*	*-mmcf-*	*-%-*	*-mmcf-*	*-%-*	*-mmcf-*	*-%-*
Softwoods																		
Hemlock	6824	20.9	0	0	0	0	0	0	0	0	21495	56.8	14864.5	32.8	0	0	43294	1.9
Yellow pines	11275	34.6	0	0	0	0	0	0	0	0	0	0	852.5	1.9	160500	13.4	172664	7.8
White and red pine	159	0.5	0	0	0	0	0	0	91650	10.9	13147	34.8	20009.6	44.1	3900	0.3	128956	5.8
Shortleaf/loblolly	0	0	46945	82.8	13843	90.9	0	0	16514	2.0	3172	8.4	0	0	991900	82.9	1072558	48.1
Spruce-balsam fir-cypress	14355	44.0	0	0.0	0	0	0	0	0	0.0	0	0	6991	15.4	33800	2.8	55205	2.5
Other softwoods	0	0.0	9734	17.2	1393	9.1	0	0	734348	87.2	0	0	2663.9	5.9	6600	0.6	754858	33.9
Total softwoods	32,613	100	56,679	100	15,236	100	0	0	842,512	100	37,814	100	45,382	100	1,196,700	100	2,227,536	100
Hardwoods																		
Yellow/sweet birch	6254	2	0	0	0	0	0	0	0	0	8693	1	2109	1	0	0	17060	0
Hickory	18124	4	5913	2	0	0		0	48264	4	16549	2	5746	2	22700	3	117311	3
Beech	25175	6	1407	1	0	0	1063	8	0	0	23802	3	21531	9	8000	1	81005	2
Yellow poplar	59219	14	110277	41	0	0	1436	11	282073	24	46246	7	3464	1	173000	21	675814	19
Other hardwoods	19956	5	50094	19	4392	34	0	0	126282	11	28623	4	36513	15	23300	3	289248	8
Sweet gum	0	0	16175	6	1101	8	1777	14	0	0	0	0	0	0	145500	18	164582	5
Ash-walnut-cherry	38533	9	0	0	0	0	290	2	22074	2	117923	17	24936	10	17300	2	221097	6
Select white/red oaks	107363	26	49404	19	3293	25	1729	13	299124	26	59737	9	60115	25	126600	16	707508	20
Other white/red oaks	92475	22	0	0	3463	27	6738	52	281596	24	267006	38	8160	3	130100	16	789705	22
Hard maple	40980	10	33392	13	717	6	0	0	85312	7	126068	18	47342	20	0	0	333884	9
Soft maple	0	0	0	0	0	0	0	0	0	0	0	0	28971	12	73400	9	102383	3
Tupelo and blackgum	3374	1	0	0	0	0	0	0	20667	2	930	0	0	0	85700	11	110671	3
Total hardwoods	411,453	100	266,662	100	12,966	100	13,033	100	1,165,392	100	695,577	100	238,887	100	805,600	100	3,610,267	100
Softwood percent	7		18		5		0		42		5		16		60		38	
Hardwood percent	93		82		46		100		58		95		84		40		62	
Total	444,066		323,341		28,202		13,033		2,007,904		733,391		284,268		2,002,300		5,837,802	

[a] mmcf = million cubic feet.

[b] percentage of total state production.

Source: USDA Forest Service Eastwide Database. Forest inventories were conducted in different States in different years, hence the table does not have a consistent time period.

Table 18—Hardwood and softwood growing stock and sawtimber volume in MAIA region States circa 2000

Growing stock[a]						
State	Softwood	Hardwood	Total	Softwood Share	Hardwood Share	State share of MAIA
	million cubic feet			*percent*		
Delaware	176	468	644	27	73	0.6
Maryland	813	3,662	4,475	18	82	4.4
New Jersey	521	1,522	2,042	25	75	2.0
New York	2,365	8,322	10,686	22	78	10.4
North Carolina	6,368	8,360	14,728	43	57	14.3
Pennsylvania	2,332	22,453	24,785	9	91	24.1
Virginia	6,648	19,839	26,487	25	75	25.7
West Virginia	1,219	17,823	19,041	6	94	18.5
Total MAIA[c]	20,440	82,448	102,888	20	80	100

Sawtimber[b]									
State	Softwood		Hardwood		Total		Softwood share	Hardwood share	State share of MAIA
	-mmcf-	*-pgsv[c]-*	*-mmcf-*	*-pgsv-*	*- mmcf-*	*-pgsv-*	*percent*		
Delaware	115	65.3	239	51.1	354	55.0	32	68	0.6
Maryland	506	62.2	2,075	56.7	2,582	57.7	20	80	4.5
New Jersey	483	92.7	1,361	89.4	1,844	90.3	26	74	3.2
New York	1,674	70.8	3,943	47.4	5,616	52.6	30	70	9.7
North Carolina	4,204	66.0	4,810	57.8	9,014	61.2	47	53	15.6
Pennsylvania	1,557	66.8	11,144	49.6	12,701	51.2	12	88	21.9
Virginia	3,801	57.2	11,506	58.0	15,307	57.8	25	75	26.4
West Virginia	805	66.0	9,665	54.2	10,469	55.0	8	92	18.1
Total MAIA	13,145	64.3	44,742	54.3	57,887	56.3	23	77	100

[a]Growing stock volume is the cubic-foot volume of sound wood in trees at least 5.0-inches dbh from a 1-foot stump to a 4-inch top..
[b]Sawtimber volume is the growing-stock volume in the sawlog portion of sawtimber-size trees:
Softwoods: volume between 1-foot stump and 7-inch top for sawtimber trees 9.0-inches dbh and larger
Hardwoods: volume between 1-foot stump and 9-inch top sawtimber trees11.0-inches dbh and larger.
[c]Percent of growing stock volume.
Source: USDA Forest Service Eastwide Database; Hansen and others 1992.

MAIA region and softwoods the remaining 20 percent. North Carolina was the only State with a relatively higher share of growing stock in softwood (43 percent), while West Virginia (94 percent hardwood) and Pennsylvania (91 percent) were the two States that had the greatest volume in hardwood growing stock. The share of sawtimber varied little (51 to 61 percent) among the States in the region, except for New Jersey's sawtimber share of total growing stock at 90 percent.

Changes in Timber Inventory

Changes in timber volume in forest stands result from the interaction of growth, removals, and mortality. FIA data provided estimates of average net annual growth of growing stock, average annual removals, and average annual mortality of growing stock and sawtimber trees[6]. Net changes in growing stock and sawtimber inventories obtained from the FIA data were calculated by subtracting

[6]Annual net growth is the change in growing stock volume between surveys (divided by the number of growing seasons to produce average annual net growth). Mortality is the estimated net volume of growing stock trees at the previous inventory, which died from natural causes before the present inventory (divided by the number of growing seasons between surveys to produce average annual mortality). Removals are losses that occur for reasons other than natural causes, and include harvesting for products, cultural operations such as timber stand improvement and land clearing, and logging residues.

Table 19—Inventory of growing stock and sawtimber volume change in MAIA region States circa 2000

States	Growing stock			Sawtimber		
	All species	Softwoods	Hardwoods	All species	Softwoods	Hardwoods
	percent					
Delaware	1.60	0.99	1.83	4.91	-2.59	8.52
Maryland	2.77	2.15	2.91	8.79	5.05	9.70
New Jersey	2.34	2.10	2.42	7.66	6.16	8.20
New York	1.99	2.11	1.96	13.69	12.34	14.27
North Carolina	0.40	0.40	2.96	4.10	2.30	5.67
Pennsylvania	1.40	2.52	1.29	11.80	13.29	11.59
Virginia	0.92	-0.08	1.25	8.17	6.20	8.83
West Virginia	2.28	1.93	2.30	14.65	11.02	14.96
Total area	1.44	0.89	1.83	10.03	6.75	11.00

Source: USDA Forest Service Eastwide Database, Hansen and others 1992.

average annual removals from average net annual growth. To avoid double-counting, we did not subtract average annual mortality, because FIA reports the average net annual growth volume after subtracting mortality volume. The percent net changes in growing stock and sawtimber inventories were then calculated for the individual States and the MAIA region to determine the percentage of growing stock and sawtimber volume that were carried over from one year to the next.

An average of about 1.44 percent of the growing stock volume was carried over from one year to the next in the MAIA region from 1970 through 1990 (table 19). The percent of growing stock volume added to the inventory every year was higher for hardwoods (1.83 percent per year) than for softwoods (0.89 percent per year). The average percent of sawtimber volume carryover was higher than growing stock volume at 10.0 percent per year, and like the growing stock volume, the increase in hardwood sawtimber (11.0 percent per year) was higher than softwood sawtimber (6.8 percent per year).

All States in the MAIA region recorded increased average growing stock volume for all species (table 19). The largest increase across all species was in Maryland, where the annual increase was 2.77 percent. Virginia (0.92 percent per year) and North Carolina (0.40 percent per year) recorded the lowest increases in total growing stock volume. Hardwood growing stock volume increased in all States, but the largest increases occurred in North Carolina (2.96 percent per year) and Maryland (2.91 percent per year). The softwood growing stock inventory declined in Virginia by 0.08 percent per year, but there was an increase in softwood sawtimber volume of 6.2 percent per year, possibly because more pulpwood-sized material was being removed from Virginia, which was consistent with the increasing pulpwood production observed in Virginia over the previous two decades. Among the other States, where softwood growing stock volume increased, Pennsylvania had the largest annual net volume increase of 2.5 percent per year), and North Carolina the smallest annual net volume increase of 0.40 percent per year.

Sawtimber volume grew in all States across all species, but the largest increases occurred in West Virginia (14.7 percent per year) and New York (13.7 percent per year) (table 19). However, Delaware's softwood sawtimber volume declined by 2.6 percent per year. Softwood sawtimber volume increased the most in Pennsylvania (13.3 percent per year), followed by New York (12.3 percent per year) and West Virginia (11.0 percent per year). Hardwood sawtimber volume grew in all the States, but the largest increases occurred in West Virginia (15.0 percent per year) and New York (14.3 percent per year).

Table 20 summarizes the average net annual growth, average annual removals, and average annual mortality of growing stock volume and sawtimber inventories for individual States and the whole MAIA region. Average net annual growth of growing stock for the MAIA region was 3.06 billion ft^3, a net increase of growing stock volume of 2.97 percent per year—all States had an increase of > 2.1 percent per year. Net growth (4.0 percent per year) and removals (3.6 percent per year) of all growing stock volume were highest in North Carolina. Loss of growing stock volume due to mortality averaged 0.55 percent per year throughout

Table 20—Growing stock and sawtimber volume removals, growth, and mortality in MAIA region States circa 2000

Growing stock							
State	Total volume	Average net annual growth	Average annual removal	Average annual mortality	Total volume net growth	Total volume removal	Total volume mortality
	million cubic feet				*percent*		
Delaware	643.90	13.50	3.20	4.11	2.10	0.50	0.64
Maryland	4474.90	163.29	39.27	27.34	3.65	0.88	0.61
New Jersey	2042.20	53.79	6.09	10.14	2.63	0.30	0.50
New York	10686.10	294.33	81.21	44.90	2.75	0.76	0.42
N. Carolina	14727.80	594.80	536.30	96.10	4.04	3.64	0.65
Pennsylvania	24784.50	631.74	284.05	176.93	2.55	1.15	0.71
Virginia	26487.00	801.61	558.72	161.33	3.03	2.11	0.61
W. Virginia	19041.30	505.17	71.06	46.66	2.65	0.37	0.25
Total MAIA	102887.70	3058.23	1579.90	567.51	2.97	1.54	0.55

Sawtimber							
State	Total volume	Average net annual growth	Average annual removal	Average annual mortality	Total volume net growth	Total volume removal	Total volume mortality
	million cubic feet				*percent*		
Delaware	354.10	45.59	28.20	11.47	12.88	7.96	3.24
Maryland	2581.60	565.20	338.36	68.08	21.89	13.11	2.64
New Jersey	1843.90	154.36	13.03	22.52	8.37	0.71	1.22
New York	5616.40	1053.38	284.27	97.38	18.76	5.06	1.73
N. Carolina	9014.00	2371.70	2002.30	235.80	26.31	22.21	2.62
Pennsylvania	12701.00	2441.20	942.51	339.67	19.22	7.42	2.67
Virginia	15307.00	3270.42	2019.34	456.64	21.37	13.19	2.98
W. Virginia	10469.10	1978.06	444.03	101.75	18.89	4.24	0.97
Total MAIA	57887.10	11879.90	6072.04	1333.32	20.52	10.49	2.30

Note: The latest FIA surveys by State in the MAIA region before 2000 were used to compute the proportion of growing stock volume removed per year; the average net annual growth; average annual removals; and average annual mortality as a percent of the total growing stock volume and sawtimber volume for the MAIA region. The ratio of average annual removals to growing stock volume is often called the production intensity.

Source: USDA Forest Service, Eastwide Database; Hansen and others 1992.

the MAIA region (table 20). In most States, the average annual mortality rate of growing stock volume was 0.50 to 0.71 percent—only West Virginia and New York had average annual mortality rates < 0.50 percent.

New Jersey had the lowest percent annual net growth of total sawtimber volume (8.4 percent per year) and North Carolina the highest (26.3 percent per year), followed by Maryland (21.9 percent per year) (table 20). Percent average annual net growth, removals, and mortality were higher (20.5, 10.5, and 2.3 percent, respectively) for sawtimber volume than for growing stock volume. Percent average annual mortality of the total sawtimber volume was highest in Delaware (3.24 percent) and lowest in West Virginia (0.97 percent).

The ratio of average annual removals to growing stock volume is an indicator of the proportion of growing stock volume removed per year, sometimes called *production intensity.* For the MAIA region as a whole, 1.54 percent of

the total growing stock volume was removed per year, about half of the 2.97 percent net growth of growing stock volume (gross volume growth minus average mortality of 0.55 percent of growing stock volume) (table 20). Softwoods accounted for 2.8 percent per year, and hardwoods accounted for 0.96 percent per year of the growing stock removals. Production intensity was highest in the southern part of the region, with 3.6 percent of growing stock volume removed per year in North Carolina and 2.1 percent in Virginia. Pennsylvania was the only State in the northern part of the region with production intensity greater than one percent (1.2 percent per year).

The production intensity for the MAIA region's sawtimber averaged 10.5 percent, again about half the average 20.5 percent net volume growth (gross volume growth minus average mortality loss of 2.3 percent). The highest proportion of sawtimber removals were in North Carolina (22.2 percent), Virginia (13.2 percent), and Maryland (13.1 percent) (table 20). These States also had the highest percent net growth of sawtimber wood, led by North Carolina (26.3 percent), Maryland (21.9 percent), and Virginia (21.4 percent). Average sawtimber volume lost to mortality was relatively low in the region (2.3 percent), ranging from 0.97 percent in West Virginia to 3.2 percent in Delaware.

Timber Markets

Hardwood and softwood timber products from the MAIA region found only a partial market in the area. Timber was exchanged in several different market areas, and each State created markets with adjoining States. For example, Pennsylvania exported roundwood to Ohio and Canada, and imported timber from New York and Maryland. Virginia, on the other hand, exported and imported roundwood from North Carolina, West Virginia, Delaware, Maryland, Kentucky, Tennessee, and Georgia.

Sawlog production was highest in the southern MAIA region (Virginia and North Carolina), where major timber products were softwood sawtimber for structural lumber and softwood pulpwood for paper goods. Timber production in the central and northern part of the region emphasized hardwood sawtimber for the furniture, housing, and pallet industries. Although the primary market for low-grade hardwood lumber was the pallet industry, low-quality hardwoods were increasingly harvested for pulpwood production. Pennsylvania led the Northeast in supplying high quality hardwood for furniture stock in secondary processing mills in-State and along the Eastern seaboard, particularly along the southeast (Wharton and Bearer 1994).

Additional information on the movement of some hardwood and softwood products within and among some MAIA states is given in the following paragraphs. Although this information can become dated in a relatively short time because of changing economics, it is useful to begin to understand some of the complexities in movement of wood products produced in the MAIA region.

Virginia was the biggest producer of hardwood and the second biggest producer of softwood sawtimber in the MAIA region (see table 16). Johnson (1994) reported that Virginia retained about 84 percent of roundwood produced and 92 percent of its sawlog production for domestic processing and manufacture in 1989 and 1992. Yet the State was still a net importer (57 percent of total volume) of softwood sawlogs. In 1992, Virginia mills received softwood sawlogs from North Carolina (73 percent), West Virginia (16 percent), Tennessee (6 percent), Kentucky (3 percent), Maryland (2 percent), and Delaware (negligible).Virginia was also a net importer of pulpwood; imports exceeded exports by 64 percent. Imports of pulpwood, about 49.2 million ft^3, came primarily from North Carolina (67 percent), West Virginia (22 percent), Maryland (8 percent), and Delaware (3 percent), with negligible amounts from Kentucky and Tennessee. Exports of pulpwood, totaling 30 million ft^3, went to North Carolina (47 percent), Tennessee (20 percent), Maryland (18 percent), Pennsylvania (13 percent), and Georgia (2 percent).

About 5 percent of roundwood intended to become sawn products in mills through 1988 was retained for use by the primary wood-using mills in Pennsylvania. The remainder was exported, mostly to mills in Ohio and Canada. Ohio mills were the largest importers of Pennsylvania sawlogs, receiving nearly 3.5 mmcf or 93 percent of the total exported to the State. Nearly all sawlogs received from by Ohio mills were red oak and white oak species. Ohio mills imported Pennsylvania oak, partially to augment supplies of oak produced by the State, and because Ohio mills were willing to pay competitive prices for roundwood (Wharton and Bearer 1994).

Overall, Pennsylvania was a net importer of roundwood. Slightly more than 11 mmcf—all in sawlogs—was imported in 1988. Most imported wood was red oak from New York and Maryland, States that provided 27 percent of all sawlogs imported. Other important species included

white oak (12 percent), black oak (9 percent), and ash (8 percent). Ash was a minor species in Pennsylvania, and was imported from New York to meet demands for bats and tool handles. Oak was also imported to meet demands for furniture and other dimension stock such as cabinets and toys. Most of the oak sawlogs were imported from Maryland, but a significant amount of white oak was imported from Ohio to manufacture cooperage (e.g., barrels, casks, butter churns) in Pennsylvania (Wharton and Bearer 1994).

North Carolina was a net exporter of industrial roundwood and pulpwood between 1988 and 1990, and sawlog exports equaled imports in 1988. Exported sawlogs went to mills in South Carolina and Virginia, and imports came from Georgia, South Carolina, Tennessee, and Virginia. In 1990 North Carolina pulpwood was exported primarily to South Carolina (31 mmcf), as well as Tennessee and Virginia. About 23 percent of pulpwood imports in North Carolina in 1990 were from Virginia, and the remainder came from Tennessee, Georgia, Kentucky, and West Virginia (Davenport 1990).

Although nearly three-fourths of the lumber processed in Delaware in 1985 came from sawlogs harvested within the State, Delaware was a net exporter of sawlogs. Nearly twice as many sawlogs were exported than were imported from other States. About 35 percent of Delaware's hardwood sawlogs went to Maryland sawmills in 1985. Delaware imported maple, beech, hickory, and locust logs (Wharton and Nevel 1990). During the same period, West Virginia was a net exporter of sawlogs, with most exports going to Virginia and Ohio. The State's imports came from Maryland, Kentucky, and Pennsylvania (Widmann and Murriner 1990).

Timber Prices

In the timber market, like many other markets, the interplay of supply and demand determines prices. Price fluctuations are the response generated by changes in supply and demand, but only sometimes indicating the scarcity of product. In economics, scarcity is the problem of ever-increasing human needs and wants in a world of finite resources. That is, society does not have sufficient productive resources to fulfill all wants and needs (<http://en.wikipedia.org/wiki/Scarcity>). Prices may be influenced by pressure from either seller or buyer: by sellers who reduce harvest when prices fall, and by buyers who purchase fewer products when prices rise. Changes in the behavior of either party will affect quantities of different species being exchanged in the market, as well as their prices. The general rules for market adjustment are: (1) an increase in the supply of timber will lower the price and, conversely, a decrease in the supply of timber will increase the price, given the demand holds constant; (2) an increase in demand for timber with no change in supply not only will raise the demand, but also raise prices in the short term, i.e., not necessarily for very long. Conversely, a decrease in demand with no change in supply lowers the quantity demanded, and lowers prices in the short run but, again, not necessarily for a long time.

We examined trends in the prices of different timber products as indicators of changes in the scarcity of timber in the region and the performance of the region's timber market. Prices were adjusted for inflation using the all-commodity producer price index (1982=100). We used regression techniques to estimate trends in real prices, choosing two different data sources to analyze prices: the Timber Market Report, published quarterly by Pennsylvania State University, School of Forest Resources, Cooperative Extension Service (<http://www.naturalresources.umd.edu/ResourcesStumpage.html>); and Timber Mart South (http://www.tmart-south.com/tmart/index.html>), published by the Daniel B. Warnell School of Forest Resources, University of Georgia.

The two different data sources were modified by Prestemon and Pye's (2000) weighting procedure to compare prices among Pennsylvania, North Carolina, and Virginia from 1977 to 1997. Hardwood prices in Pennsylvania were representative of prices in the northern MAIA region with its higher proportion of hardwood production. Prices in Virginia and North Carolina were representative of prices in the southern MAIA region, where softwood production is relatively higher.

We considered two different prices for the analysis: stumpage price (the price of standing trees) and delivered log price (the price paid to the mill).To analyze trends we used delivered log prices, when available, because they capture market scarcity better than stumpage prices; they are reported by species and grade; and are measured at a common point. However, we used stumpage prices for hardwood pulpwood, because delivered prices were not available.

In Pennsylvania, we analyzed delivered log prices to identify sawlog price trends for white oak, red oak, and black cherry (by grade). We chose these species because they rank among the top five in terms of removals, and because prices by grade were available for Pennsylvania. Prices in Pennsylvania were reported for four regions,

Table 21—Average annual rate of price change for hardwood (1987-1997) and (1977-1997) softwood sawlogs and pulpwood by grade circa 2000

Product	Location	Year	Grade			
			Low[a]	Medium[a]	High[a]	Average
			----------percent----------			
Sawlogs						
Red Oak	Pennsylvania	1987-1997	4.91	4.19	4.98	--[b]
White oak	Pennsylvania	1987-1997	*NS*[c]	4.38	7.09	--
Black cherry	Pennsylvania	1987-1997	13.66	13.76	17.68	--
Pine	Western North Carolina	1977-1997	--	--	--	*NS*
Pine	Central/far eastern NC	1977-1997	--	--	--	2.14
Pine	Western Virginia	1977-1997	--	--	--	3.86
Pine	Eastern Virginia	1977-1997	--	--	--	3.98
Pulpwood			--	--	--	
Pine	Western North Carolina	1977-1997	--	--	--	-0.74
Pine	Central/far eastern NC	1977-1997	--	--	--	-1.08
Pine	Western Virginia	1977-1997	--	--	--	1.45
Pine	Eastern Virginia	1977-1997	--	--	--	1.73
Hardwood	Pennsylvania	1987-1997	-19.12	-19.3	-22.29	--

[a]Corresponds to Forest Service's F3, F2, and F1, respectively: F3 (=low), F2 (=medium), and F1(=high) grades.
[b] -- Model did not apply to the referenced combination.
[c]No significant change in prices.

and we analyzed price trends for the State using average prices across those regions. For delivered pine sawlogs and pulpwood to North Carolina and Virginia mills, we analyzed data from two regions in each State—western North Carolina, and central and far eastern North Carolina; and western and eastern Virginia.

Business cycles also affect timber markets, so analysis of price trends should start and end at comparable points in the market cycle, i.e., the peak should be compared to a peak and a trough to a trough. We used performance of the hardwood lumber market as an indicator to determine the timing of a peak or trough in the market. For our analyses we examined trends discernable from 1987 to 1997 using Pennsylvania prices; and from 1977 to 1997 using Virginia and North Carolina prices. These time-frames were comparable because they represented points in an upward portion of the hardwood market cycle.

Sawlog Prices

We analyzed price trends for three species of hardwoods (red oak, white oak, and black cherry) in Pennsylvania between 1987 and 1997, and for pine sawlogs in North Carolina and Virginia between 1977 and 1997. The three hardwood species represented about 38 percent of the total sawlog production in the MAIA region (table 21). Prices were reported for low, medium and high-grade saw logs—these grades correspond to the Forest Service's log grades F3, F2, and F1, respectively.

Our analyses showed that black cherry had the highest price increase percentage per year, followed by white oak and red

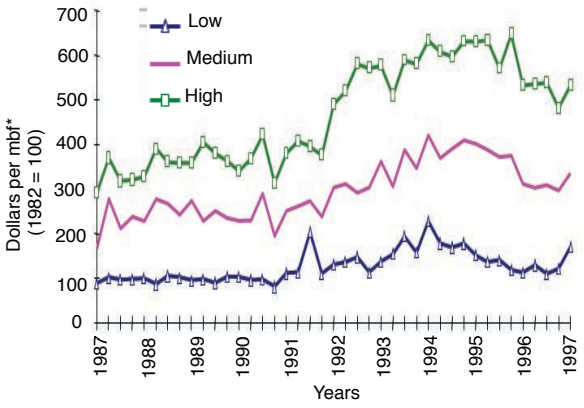


Figure 75—Real prices of delivered black cherry sawlogs in Pennsylvania 1987 to 1997. Low, medium, and high grades correspond to USFS grades F3, F2, and F1, respectively. Source: Pennsylvania Woodlands, Timber Market Report, quarterly reports; (http://www.sfr.psu.edu/TMR/TMR.htm).

oak (table 21). Real prices for the highest grade black cherry in Pennsylvania increased an average 17.7 percent per year between 1987 and 1997, while the low and medium grades grew over 13.7 percent per year (table 21) despite annual fluctuations (fig. 75). Black cherry prices were relatively stable until the fourth quarter of 1991, then prices increased by about 50 percent for the highest grade logs in the fourth quarter of 1992, compared to prices in fourth quarter of 1991. Overall, we observed a general, increasing trend in prices since 1992. The observed price increases were the result of a combination of various factors. Black cherry logs were the second most exported species from the U.S. to Europe in 1992, accounting for 21 percent of the total log exports to Europe (Luppold 1994). Also, the furniture and cabinet industry in Pennsylvania fostered a large demand for black cherry.

The demand for veneer to make cabinet, furniture, or architectural millwork also increased. Of the total amount of black cherry available in the region, higher quality timber was relatively scarce. Increasing concern for sustainable management of this environmentally and economically important species contributed to declining timber harvests in the Kene National Forest in Pennsylvania, resulting in less cherry timber available in the region and more pressure on the supply. Black cherry has a thin margin: a small increase in demand can affect pricing significantly, even though the total demand volume does not change much. All of these factors, i.e., greater demand in both domestic and international markets, along with reduced supply, combine to make black cherry a very valuable commodity in Pennsylvania.[7]

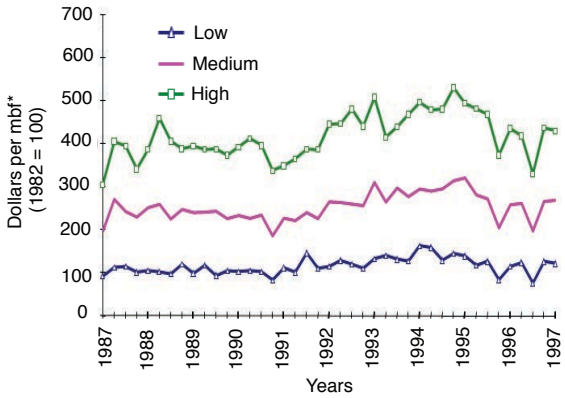


Figure 76—Real prices of delivered red oak sawlogs in Pennsylvania by low, medium, and high grades from 1987 to1997, corresponding to USFS grades F3, F2, and F1, respectively. Source: Pennsylvania Woodlands, Timber Market Report, quarterly reports; (http://www.sfr.psu.edu/TMR/TMR.htm).

[7]Johnson, J. 1998. Personal Communication. Harwood Market Report, P.O. Box 241325, Memphis, TN, 38124-1325

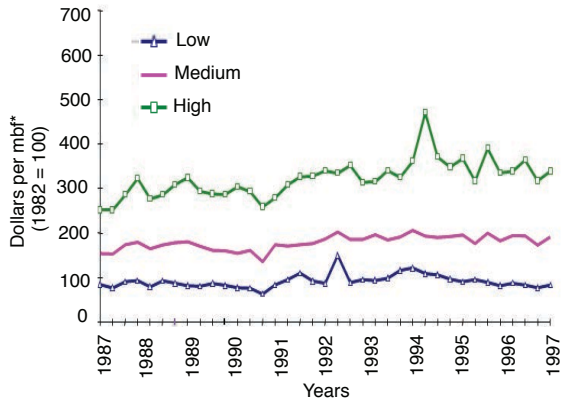


Figure 77—Real prices of delivered white oak sawlogs in Pennsylvania 1987 to 1997, by low, medium, and high grades, corresponding to USFS grades F3, F2, and F1, respectively. Source: Pennsylvania Woodlands, Timber Market Report, quarterly reports; (http://www.sfr.psu.edu/TMR/TMR.htm).

Prices for species of red oak and white oak sawlogs also increased significantly in Pennsylvania from 1987 to 1997 (table 21; figs. 76 and 77)—prices for higher-grade sawlogs increased faster than for lower grades. High-grade red oak and high-grade white oak grew 5.0 percent and 7.1 percent per year, respectively (figs. 76 and 77), compared to medium-grade red oak (4.2 percent), medium-grade white oak (4.4 percent), and low-grade red oak (4.9 percent); all indicated an overall increase in demand for most grades of both red and white oaks.

The source of demand for sawlogs is the demand for products made from them. An understanding of the demand for lumber improves our understanding of the fluctuations of sawlog prices. Since the early 1970s, red oak has become an important furniture lumber and the dominant species for kitchen cabinets and millwork (Luppold 1997). During the 1980s, red oak lumber demand was strong in both domestic and international markets. Red oak lumber was exported to Canada, Asia, and Europe (Luppold and Baumgras 1996), and was the most important species exported to Asia and Canada in the mid-1980s. Demand by the major secondary hardwood processors in the mid-1980s tended to favor red oak of all grades, while the flooring industry has traditionally used lower-grade red oak. Although red oak is plentiful in Eastern forests, high-grade timber is not. Consequently, the strong demand, coupled with a finite supply, has led to higher real prices (Luppold 1993).

Considerable demand for white oak lumber in the international market may have accounted for the increasing

prices of white oak sawlogs in Pennsylvania. In 1981, almost equal volumes of red and white oak were exported to Europe from the U.S.; but by 1990, white oak exports were more than 3.5 times greater than red oak exports. White oak was the most widely exported species to the European market in 1990, and individual market shares for white oak ranged from under 20 percent in Denmark and Italy to more than 80 percent in Spain. Red oak was in high demand in some countries, and accounted for roughly 40 percent of the market in Luxembourg and Belgium, 30 percent in France, and a very small amount in most Scandinavian countries (Hansen and others 1991).

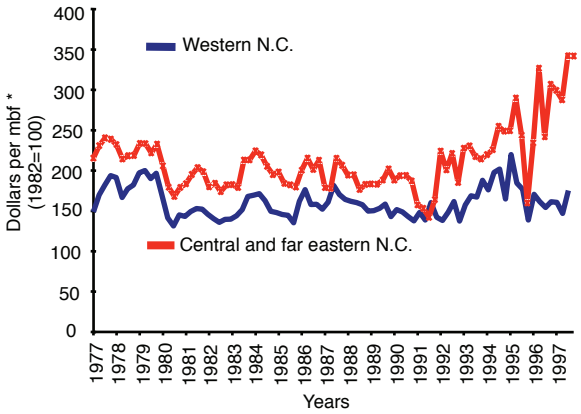


Figure 78—Real prices of delivered pine sawlogs in western and central-far eastern North Carolina 1977 to1997. Source: Timber Mart South, monthly and quarterly reports; (http://www.tmart-south.com/).

In central and eastern-coastal North Carolina from 1977 to 1997, the prices for pine sawlogs increased at an annual rate of 2.14 percent (table 21 and fig. 78).

Prices of pine sawlogs in central and far eastern North Carolina increased dramatically after 1991, while prices for pine sawlogs declined by 0.1 percent in western North Carolina. Pine sawlog prices in western and eastern Virginia grew almost twice as fast as in North Carolina, increasing 3.86 and 3.98 percent per year in western Virginia and eastern Virginia, respectively (table 21 and fig. 79). Southern softwood markets were very strong in the 1990s, when strong housing markets coincided with declines in wood supply from public lands in the western U.S. This trend was expected to continue (Wear 1996).

Pulpwood Prices

We examined stumpage price trends for the low-, medium-, and high-grade hardwood pulpwood in Pennsylvania between 1987 and 1997. Prices for all three grades of hardwood pulpwood in Pennsylvania declined significantly by 19.1, 19.3, and 22.3 percent, respectively (table 21 and fig. 80). Prices for all three grades were higher in the third quarter of 1989 and 1990 (around $13 per ton), began again to rise sharply in the second quarter of 1991, and peaked much higher in the second and third quarter of 1992. At its height, the price of the highest grade hardwood pulpwood

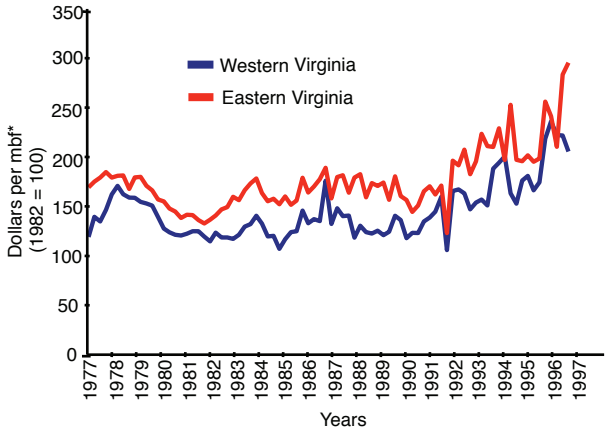


Figure 79—Real prices of delivered pine sawlogs in western and eastern Virginia 1977 to 1997. Source: Timber Mart South, monthly and quarterly reports; (http://www.tmart-south.com/).

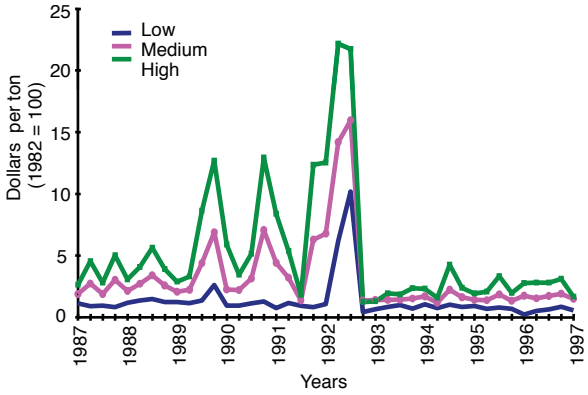


Figure 80—Real prices of stumpage hardwood pulpwood in Pennsylvania by low, medium, and high grades, corresponding to USFS grades F3, F2, and F1, respectively, 1987 to 1997. Source: Pennsylvania Woodlands, Timber Market Report, quarterly reports; (http://www.sfr.psu.edu/TMR/TMR.htm).

had risen to around $22 per ton. In the fourth quarter of 1992 the prices crashed back to 1987 levels, and remained down through 1997. Higher prices in 1989, 1990, and 1992 were a reflection of scarcity during those times.

Total pulpwood production in Pennsylvania increased by about 40 percent between 1968 and 1988. Pulp processors relied heavily on chipped-residues to handle production increases, aided by increased sawlog portions that made manufacturing residue more available. The chipped-residue portion of total pulpwood production increased from 23 to 37 percent from 1968 to 1988 (Wharton and Bearer 1994). This indicated a decreased demand for higher and medium-grade pulpwood that resulted in lower prices. However, in 1990, hardwood pulpwood production declined by almost 34 percent compared to 1988 production. And, even though softwood pulpwood increased by about 11 percent in 1990 compared to 1988, the production of hardwood and softwood pulpwood together declined by 32 percent in 1990.The production of hardwood pulpwood dropped by about 8.7 percent in 1993 from pulpwood production in 1990 (Widmann 1987, 1988, 1989, 1990, 1996). An artificial demand for hardwood pulpwood created in anticipation of this drop in production in 1993 might have been the cause of increased stumpage prices of hardwood pulpwood in 1989, 1990, and 1992 (fig. 80).

We also examined the delivered-price trends for pine pulpwood in North Carolina and Virginia between 1977 and 1997. All grades of pine pulpwood prices declined in North Carolina—at an average annual rate of 0.74 percent in the western part of the State, and 1.08 percent in the central and far eastern part of the State between 1977 and 1997 (table 21 and fig. 81).

Pine pulpwood production and harvesting increased in North Carolina from 1986 to 1992. Almost all of this growth was in southwestern North Carolina, where softwood pulpwood production expanded by 53 percent. Pulping capacity in that region was relatively small, however, suggesting that mills in Georgia and Tennessee were drawing increasing amounts of material from this region. This indicated that hauling distances and zones of procurement for pine pulpwood were expanding, foreshadowing increasing demand for pulpwood timber (Wear 1996). From 1978 to the end of 1981, prices declined in both regions of North Carolina, then increased until the middle of 1985. After 1985, we observed no real trend in prices except for normal fluctuations in the market cycle.

Pine pulpwood prices increased in Virginia between 1977 and 1997 (table 21 and fig. 82). They increased at an annual rate of 1.45 percent in western Virginia and 1.73 percent in eastern Virginia. Eighty-four percent of roundwood products cut for pulpwood was retained for processing at Virginia pulp mills. Imports of nearly 49 million cubic feet exceeded exports by 64 percent, making the State a net importer of pulpwood (Johnson 1994), and suggested that

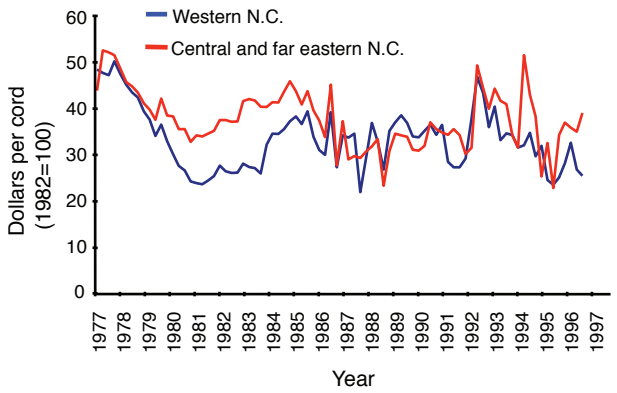


Figure 81—Real prices of delivered pine pulpwood in western and central-eastern North Carolina 1977 to 1997. Source: Timber Mart South, monthly and quarterly reports; (http://www.tmart-south.com/).

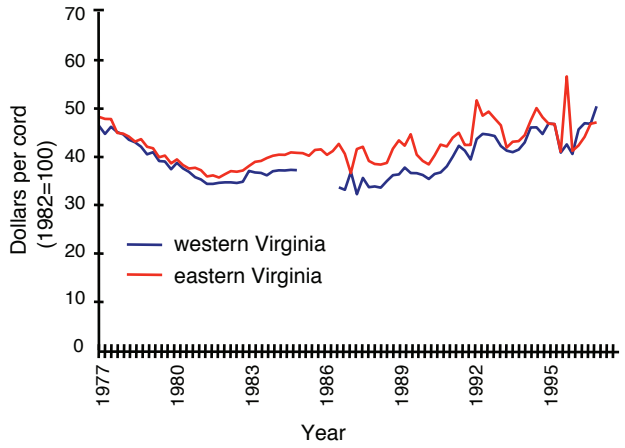


Figure 82—Real prices of delivered pine pulpwood in western and eastern Virginia 1977 to 1997. Note discontinuity in years 1985 and 1986 due to unavailability of price data. Source: Timber Mart South, monthly and quarterly reports; (http://www.tmart-south.com/).

pine pulpwood was relatively scarce in Virginia during those 2 decades.

The Demand for Timber

The total demand for timber in the U.S. depends on the demand for materials in the housing, furniture, paper, packaging and other timber-dependent industries. A change in aggregate demand can result from changes in population or consumer income, changes in prices, changes in tastes, and other common market fluctuations. We categorized timber demand at two levels: (1) at primary timber processing industry facilities, where timber is first converted into wood products to meet consumer demand; and (2) at secondary timber processing industry facilities, where primary timber products are further processed into final products.

Primary timber processing industries include logging and harvesting operations; producers of solid-wood commodities such as softwood and hardwood lumber, structural and nonstructural panels, and a wide variety of other wooden products, e.g., pallets, treated fence posts, ladders, and picture frames; and producers of fiber-based commodities such as pulp, paper, and paperboard (Haynes 1990). Secondary timber processing industries include house construction, kitchen cabinets, wood containers, and other wood products manufacturers.

To analyze the demand for timber, we first examined the national timber assessment by Haynes (1990) for changes in population, per capita disposable personal income, and performance trends of the wood product industries in the U.S. Then, assuming that the annual rate of change in population and per capita disposable personal income followed national trends, we considered implications of these changes on demand for timber within the MAIA region. We also observed the productive capacity of wood products industries in the region to further understand what changes had occurred.

National Trends

The following projections of U.S. demand for timber are derived from Haynes (1990) national assessment of past and future timber situations in the U.S. for the years 1989 to 2040. Upward trends in population and per-capita disposable personal income imply a greater demand for and consumption of wood and paper products. As a result, we expect consumption of wood and wood-fiber products to continue to grow over the next 5 decades, especially the consumption of paper and fiberboard. We also expect more growth in hardwood roundwood consumption than in softwood roundwood consumption.

We projected that consumption of hardwood and softwood for pulp will increase through 2040. If present trends continue, hardwood pulpwood use will be higher than softwood pulpwood use, due to a gradual industry shift toward use of high-yield mechanical pulps that require more hardwood—and away from chemical pulping, which uses softwood. Softwood sawtimber stumpage prices and delivered prices for hardwood and softwood sawtimber and pulpwood are projected to increase over the next 4 decades. Due to continuous growth in major shipping uses and a steady decline in the availability of larger timber for higher-quality lumber grades, hardwood lumber prices likely will increase as well.

The demand for softwood lumber, panels, and sawlogs is most strongly influenced by cyclic housing markets. Residential construction has been strong, but forecasts are for a slight decline followed by strong housing demand over the next 3 decades (Haynes and others 1995). Strong national demands for softwood lumber necessarily will keep softwood sawlog prices high in the South (Wear 1996), and we expect that softwood sawlog demand will be strong in North Carolina and Virginia.

National trends indicated a growth in hardwood lumber use in the past 3 decades (Haynes and others 1995). Of the total hardwood lumber consumed in 1977, furnishing applications (appearance applications) accounted for 51 percent, industrial applications for 43 percent, and exports for 3 percent (table 22).[8]

About 22 percent of all hardwood lumber in the U.S. in 1977 was used for furniture products (household, upholstered, and commercial). From 1977 to 1991, total U.S. hardwood lumber consumption increased from 8,384 million board feet (mmbf) to 12,321 mmbf, an increase of about 47 percent. Hardwood lumber consumption for industrial applications increased by 7 percent of total consumption, but declined by about 14 percent of the total for all furnishing applications. This included a 10-percent

[8]Furnishing applications (appearance applications) are uses that give importance to the natural beauty of wood. They include furniture, millwork, cabinets, paneling and flooring, and other products. Industrial applications, on the other hand, are uses that give importance to strength and durability. These include pallets, railroad ties, and other uses (Luppold 1993).

Table 22—U.S. hardwood lumber consumption by major industries for 1977, 1982, and 1991

	1977		1982		1987		1991		1977-1991
	Volume	Total share	Volume	Total share	Volume	Total share	Volume	Total share	Average
	--mmbf[a]--	*-percent-*	*--mmbf--*	*-percent-*	*--mmbf--*	*-percent-*	*--mmbf--*	*------percent-----*	
Furnishing applications[b]									
Wood household furniture	1,250	15	932	11	1058	8	898	7	10.25
Upholstered furniture	354	4	284	3	317	2	283	2	2.75
Commercial furniture	221	3	275	3	425	3	370	3	3
Millwork	498	6	506	6	705	6	613	5	5.75
Kitchen cabinets	244	3	366	4	671	5	602	5	4.25
Flooring	304	4	265	3	476	4	529	4	3.75
Dimension	1,326	16	982	11	1,379	11	1,176	10	12
TV Cabinets	51	1	31	0	20	0	15	0	0.25
Plywood	61	1	93	1	112	1	103	1	1
Furnishing total	4,309	51	3,725	43	5,163	40	4,589	37	42.75
Industrial applications									
Pallets	2,313	28	2,900	33	4,513	35	4,704	38	33.5
Treated products	735	9	819	9	781	6	777	6	7.5
Structural members	247	3	389	4	534	4	437	4	3.75
Other uses	276	3	101	1	308	2	245	2	2
Industrial total	3,571	43	4,209	49	6,136	48	6,163	50	47.5
Miscellaneous products	264	3	403	5	794	6	719	6	5
Exports	240	3	325	4	688	5	850	7	4.75
Total all uses	8,384	100	8,671	100	12,781	100	12,321	100	100

[a] mmbf = million board feet.
[b] also referred to as appearance applications.
Source: Southern Appalachian Assessment (Wear 1996).

decline in total lumber consumption for furniture uses in 1991. Export consumption increased from 3 percent in 1977 to 7 percent of total consumption in 1991. Export demand, the fastest and most consistent growth market for hardwood lumber in the U.S., increased from 240 mmbf in 1977 to 850 mmbf in 1991, an increase of 354 percent resulting from increased demand by Asia and Europe. Increased exports to Japan, Taiwan, and Korea caused Asia to be the fastest growing region for U.S. hardwood lumber exports during the last decade (Luppold 1994).

Local Factors

Most of the national trends we have discussed were observed in this assessment of the MAIA region. For example, the total volume of timber produced increased by about 11 percent between the 1970s and the 1980s (table 16), and the hardwood share of that total used to manufacture sawlogs and pulpwood was higher than the softwood volume used to the same purpose. Total pulpwood production expanded in the region by about 25 percent over the past 2 decades (1970s through 1990s). Although softwood and hardwood pulpwood production increased, the softwood share of total production increased more than the hardwood share. Prices of black cherry, red oak, and white oak increased, and the prices of higher-grade delivered sawlogs increased faster than prices of the lower grades (figs. 75, 76, and 77). Likewise, there has been a trend of price increases for delivered pine sawlogs in central and far eastern North Carolina (fig. 78) and Virginia (fig. 79).

Sawlogs have been the major timber product in the MAIA region. The sources of this demand are the various sawmills located in and around the region. Figures 83, 84, and 85

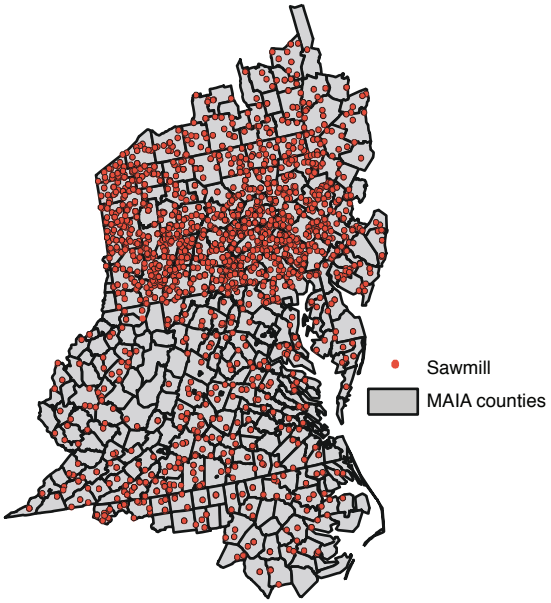


Figure 83—Sawmill locations in the MAIA region in 1990s. Source: Pye 1999.

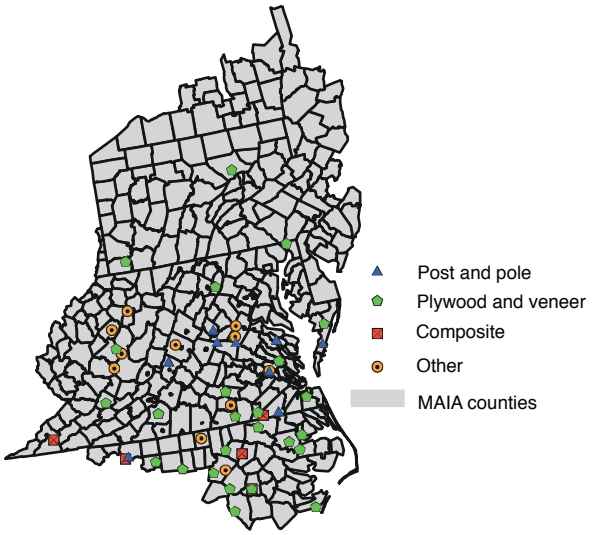


Figure 85—Other mills in the MAIA region in 1990s.The Other category represents miscellaneous mills that produce specialty products like flooring, cabinets, and plywood. Source: Pye 1999.

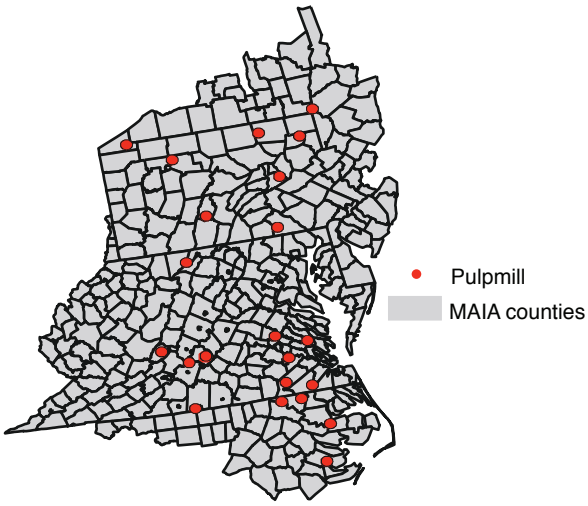


Figure 84—Pulp mill locations in the MAIA region in 1990s. Source: Pye 1999.

show the locations of sawmills, pulp mills, and other mills, respectively (Pye 1999).

Table 23 reports changes in hardwood sawmill production capacity, number of mills, average capacity, and proportion of capacity in large mills for selected States in the Mid-Atlantic region over the past 2 decades (mid 1970s to early 1990s). The average production capacity of hardwood mills has increased from 175 percent to 265 percent over time to meet changing demands for hardwoods, and to accommodate increased hardwood sawlog production.

The decline in the number of hardwood mills since 1991 was tempered by increased average capacity of the remaining sawmills. The growing dominance of the remaining large mills suggests economies of scale or size, increased concentration of productive capacity due to reduced production and distribution costs, and increased access to lumber (table 23). These changes have increased returns on the hardwood resource due to increased productivity and marketing efficiency (Luppold 1995), i.e.,

Table 23—Hardwood sawmill mills and production capacity in MAIA States in mid-1970s to early 1990s

State	Year	Production capacity	Number of mills	Average capacity	Large mills share of total capacity
		---mmbf[a] --		---mmbf---	---percent---
New York	1976	279	116	2.40	27.6
	1980	423	167	2.54	30.1
	1985	447	164	2.73	28.5
	1991	491	155	3.17	50.4
North Carolina	1976	456	290	1.57	35.8
	1983	601	302	1.99	43.9
	1987	759	269	2.82	56.5
	1992	616	208	2.96	62.6
Virginia	1978	569	310	1.84	25.4
	1984	705	348	2.03	40.5
	1987	680	304	2.24	40.3
	1992	717	241	2.98	55.7
West Virginia	1976	458	284	1.61	34.4
	1980	471	281	1.67	39.0
	1986	501	227	2.21	43.3
	1992	580	227	2.55	63.3
Pennsylvania	1975	697	582	1.20	16.1
	1982	943	673	1.40	40.0
	1986	978	574	1.70	37.9
	1991	1029	578	1.78	42.7

[a]mmbf = million board feet
Source: adapted from Luppold 1995

fewer, larger, more efficient mills are economically more cost effective and, therefore, desirable.

Market Benefits/Forest Products Benefits

To quantify the economic importance of forest-based industries in the MAIA region, we examined employment and income generated in the following sectors based on Standard Industry Classifications (SIC): lumber and wood products (SIC 24), furniture and fixtures (SIC 25), and paper and allied products (SIC 26).

Wage Employment

Table 24 summarizes employment related to wood products and the number of wage and salary employees with all forest-based industries, relative to all sectors of the economy in the MAIA region. Total employment in the region averaged 11.9 million employees per year between 1975 and 1995. During this period the SIC 24 produced, on average, about 83,600 jobs per year, SIC 25 produced about 73,100, and SIC 26 provided roughly 87,400. For the entire MAIA region then, forest industries produced an average 244,100 jobs annually (2.04 percent of all wage employment) between 1975 and 1995. At the State level, forest industries contributed more than 2 percent of all employment in North Carolina (2.78 percent), Pennsylvania (2.13 percent) and Virginia (3.32 percent). In the remaining States, forest industry employment ranged from a low of 0.77 percent of all employment in New Jersey to 1.77 percent in West Virginia.

Wage employment in SIC 24 expanded at an average annual rate of 1.32 percent per year between 1975 and 1995, and was the only wood-products sector in the entire MAIA region with any significant change in employment (table 24; fig. 86). Employment in SIC 25 was volatile during those 2 decades, but the overall trend between 1975

Table 24—Average employment (wage and salary) and rate of change in all Standard Industry Classification (SIC) sectors in MAIA region 1975 to 1995

State/sector	Average employment	Average share of total economy employment	Average annual rate of change
	--*thousands*--	----------*percent*----------	
Total MAIA region			
All sectors	11969.8	100	1.93
SIC 24[a]	83.6	0.70	1.32
SIC 25[b]	73.1	0.61	0
SIC 26[c]	87.4	0.73	0
Total SIC 24+25+26	244.1	2.04	
Delaware			
All sectors	248.3	100	2.75
SIC 24	0.82	0.33	3.36
SIC 25	0.5	0.20	5.8
SIC 26	2.5	1.01	0
Total SIC 24+25+26	3.82	1.54	
Maryland			
All sectors	1455	100	2.62
SIC 24	3.8	0.26	0
SIC 25	3.3	0.23	-1.23
SIC 26	9.4	0.65	-0.84
Total SIC 24+25+26	16.50	1.14	
New Jersey[d]			
All sectors	1441.2	100	2.24
SIC 24	2.6	0.18	-1.44
SIC 25	1.8	0.12	0
SIC 26	6.8	0.47	-1.62
Total SIC 24+25+26	11.20	0.77	
New York[d]			
All sectors	972.7	100	1.93
SIC 24	4.7	0.48	0
SIC 25	6.6	0.68	1.15
SIC 26	4.2	0.43	0.84
Total SIC 24+25+26	15.50	1.59	
North Carolina[d]			
All sectors	1439	100	2.46
SIC 24	16.8	1.17	0
SIC 25	16.7	1.16	-1.06
SIC 26	6.5	0.45	1.34
Total SIC 24+25+26	40.00	2.78	

[a] SIC 24 = lumber and wood products
[b] SIC 25 = furniture and fixtures
[c] SIC 26 = paper and allied products
[d] Figures for NJ, NY and NC are totals for the subset of counties in the MAIA region.
Source: Department of Labor, unemployment insurance database ES-202; (http://workforcesecurity.doleta.gov/unemploy/finance.asp).

Table 24 (Continued)—Average employment (wage and salary) and rate of change in all Standard Industry Classification (SIC) sectors in MAIA region 1975 to 1995

Pennsylvania			
All sectors	4042.6	100	0.97
SIC 24	25.4	0.63	2.98
SIC 25	19.3	0.48	0
SIC 26	41.3	1.02	-0.43
Total SIC 24+25+26	86.0	2.13	
Virginia			
All sectors	1893.3	100	3.22
SIC 24	23.3	1.23	0
SIC 25	24	1.27	-0.95
SIC 26	15.5	0.82	1.21
Total SIC 24+25+26	62.80	3.32	
West Virginia			
All sectors	477.7	100	0
SIC 24	6.2	1.30	2.3
SIC 25	0.97	0.20	-2
SIC 26	1.3	0.27	-1.92
Total SIC 24+25+26	8.47	1.77	

[a] SIC 24 = lumber and wood products
[b] SIC 25 = furniture and fixtures
[c] SIC 26 = paper and allied products
[d] Figures for NJ, NY and NC are totals for the subset of counties in the MAIA region.
Source: Department of Labor, unemployment insurance database ES-202; http://workforcesecurity.doleta.gov/unemploy/finance.asp.

and1995 was flat. Following a general upswing between 1975 and 1987, employment in SIC25 consistently declined after1987, falling in 1990 to lower levels than in 1975 (fig. 86).

In contrast, employment in SIC26 was relatively stable between 1975 and 1995, with employment slightly higher in 1995 than in 1975. Between 1975 and 1995, wage employment in all sectors of the MAIA region's economy expanded at a higher average annual rate (1.93 percent) than it did in the forest industry sectors (0 to 1.32 percent) (table 24). As a result, the forest industry share of employment in the MAIA region fell from 2.31 percent in 1975 to 1.74 percent in 1995, averaging 2.04 percent for 1975 to 1995.

Forest industry job growth varied widely from State to State from 1975 to 1995. Overall, Delaware experienced the highest annual rate of growth in employment in the forest industry sector within the whole region, where both SIC24 and SIC25 employment recorded substantial growth, at 3.36 percent and 5.80 percent, respectively (table 24). Other States that increased their average annual

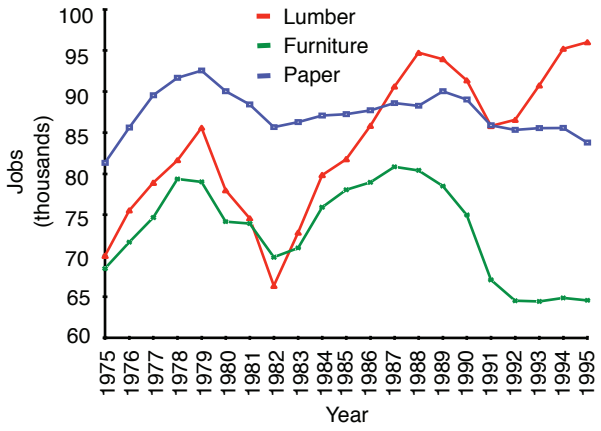


Figure 86—Wage and salary employment (number of employees) in lumber and wood products (SIC 24), furniture and fixtures (SIC 25), and paper and allied products (SIC 26) in the MAIA region 1975 to 1995. Source: Department of Labor, unemployment insurance database, ES-202; (http://www.bls.gov/bls/blswage.htm).

rate of change in employment in SIC24 industries was Pennsylvania (2.98 percent) and West Virginia (2.3 percent) during the same period. In nearby New Jersey, though, there were decreases of 1.44 percent in SIC 24 and 1.62 percent SIC 26 employment. The share of wood-products employment in other States also declined between 1975 and 1995. In Delaware and New York SIC 25 average annual employment increased by 5.8 and 1.15 percent, respectively, while Maryland, North Carolina, Virginia, and West Virginia had average annual employment declines of 1.23, 1.06, 0.95, and 2.00 percent, respectively, in that sector. Employment in SIC 26 grew in New York, North Carolina, and Virginia by 0.84, 1.34, and 1.21 percent, respectively. Negative trends in SIC26 occurred in Maryland, New Jersey, Pennsylvania, and West Virginia by 0.84, 1.62, 0.43, and 1.92 percent, respectively.

Wages and Salaries[9]

Table 25 and fig. 87 summarize the contributions to wage and salary income for SIC 24, SIC 25, and SIC 26, (total forest industries sectors), and all other sectors of the MAIA region's economy from 1975 to 1995.

Real wage and salary income for the entire region's economy averaged $222.3 billion per year between 1975 and 1995, and about 2.02 percent ($4.50 billion) of that total came from forest industries (table 25). Wages in all sectors increased over this period, despite slight decreases in wages in the early 1980's (fig.87).

Wages in SIC 24 increased at the highest annual rate (3.34 percent per year), SIC 25 followed with 1.35 percent, and SIC 26 with 2.16 percent for the whole MAIA region. The lumber and wood products sector (SIC 24) produced an average of $1.25 billion per year in wages and salaries, SIC 25 generated $1.13 billion and SIC 26 an average $2.12 billion. Salary and wage income for the MAIA region economy as a whole (3.81 percent per year) grew faster than wage income for any of the forest industry sectors (SIC 24, 25, or 26) at 3.34, 1.35, and 2.16 percent, respectively (table 25). Among the States in the MAIA region, Delaware experienced the highest average annual growth in wages in all the forest industry sectors (average 5.0 percent per year).

The average annual rate of change for forest industry sectors SIC 24 and SIC 25 exceeded the average annual rate of change for all other sectors (4.19 percent per year) (table 25). The relatively high average annual rate of change in forest

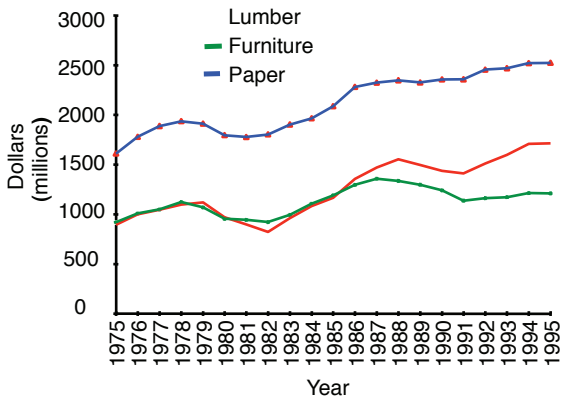


Figure 87—Real wages and salaries (1982=100) for lumber and wood products (SIC 24), furniture and fixtures (SIC 25), and paper and allied products (SIC 26) in the MAIA region 1975 to 1995. Source: Department of Labor, unemployment insurance database, ES-202; (http://www.bls.gov/bls/blswage.htm).

industry wages and salaries in Delaware was led by SIC 25, increasing an average of 7.86 percent per year from 1975 to 1995. Records show a lower average annual growth in wages for all sectors in other MAIA States, even though annual wage and salary growth in some forest industry sectors was higher. For example, wages and salaries in SIC 24 increased 4.18 percent per year in West Virginia, compared to an increase of only 0.67 percent for all other sectors. In North Carolina SIC 26 grew an average of 3.49 percent per year, compared to 3.43 percent for all other sectors, and in Pennsylvania wages in SIC 24 industries grew an average of 4.5 percent per year compared to 2.55 percent for all other sectors.

The average wage per job increased in all forest industries and all other sectors between 1975 and 1995 (fig. 88). The average real wage per job for the entire economy of the MAIA region between 1975 and 1995 was about $18,000—growing from about $16,000 in 1975 to $21,000 in 1995 (an increase of 31 percent). The average wage per job in SIC 24 and SIC 25 was $14,816 and $15,497, respectively; earnings below the MAIA regional average by 18 and 14 percent, respectively. The average wage per job in SIC 26 was $24,000, higher than the regional average for the entire MAIA economy by almost 33 percent. The wage per job in SIC 26 increased by 52 percent between 1975 and 1995, compared to a 40 percent increase in SIC 24 and a 39 percent increase in SIC 25.

[9]All wage and salary figures are in 1982 dollars.

Table 25—Wage and salary average and rate of change in all Standard Industry Classification (SIC) sectors in MAIA region 1975 to 1995

State/sector	Average wage and salary	Average share of total economy wages and salaries	Average annual rate of change
	millions of dollars[a]	----------*percent*----------	
Total MAIA Region			
All sectors	222295.7	100	3.81
SIC 24[b]	1253	0.56	3.34
SIC 25[c]	1128.7	0.51	1.35
SIC 26[d]	2115.8	0.95	2.16
Total SIC 24+25+26	4497.5	2.02	
Delaware			
All sectors	4985.4	100	4.19
SIC 24	12.4	0.25	4.7
SIC 25	7.9	0.16	7.86
SIC 26	54	1.08	2.44
Total SIC 24+25+26	74.3	1.49	
Maryland			
All sectors	27517.2	100	4.69
SIC 24	62	0.23	1.57
SIC 25	55.7	0.20	0
SIC 26	202.9	0.74	1.2
Total SIC 24+25+26	320.6	1.17	
New Jersey[e]			
All sectors	29204.3	100	5.02
SIC 24	45.3	0.16	0
SIC 25	33.5	0.11	2.29
SIC 26	159	0.54	0.88
Total SIC 24+25+26	237.8	0.81	
New York[e]			
All sectors	17014.4	100	3.57
SIC 24	74.3	0.44	2.19
SIC 25	109.5	0.64	2.95
SIC 26	87.4	0.51	3.05
Total SIC 24+25+26	271.2	1.59	
North Carolina[e]			
All sectors	25583	100	3.43
SIC 24	216	0.84	2.55
SIC 25	243.9	0.95	1.45
SIC 26	156.1	0.61	3.49
Total SIC 24+25+26	616	2.40	

[a] 1982=100, prices are adjusted for inflation and expressed in terms of value in 1982.
[b] SIC 24 = lumber and wood products
[c] SIC 25 = furniture and fixtures
[d] SIC 26 = paper and allied products
[e] Figures for NJ, NY and NC are totals of the subset of counties in the MAIA region.
Source: Department of Labor, unemployment insurance database ES-202; (http://workforcesecurity.doleta.gov/unemploy/finance.asp).

Table 25 (Continued)—Wage and salary average and rate of change in all Standard Industry Classification (SIC) sectors in MAIA region 1975 to 1995

Pennsylvania			
All sectors	75782.4	100	2.55
SIC 24	423.1	0.56	4.52
SIC 25	345.4	0.46	1.4
SIC 26	1027.4	1.36	1.78
Total SIC 24+25+26	1795.9	2.38	
Virginia			
All sectors	33680.7	100	5.39
SIC 24	338.7	1.01	3.02
SIC 25	320.5	0.95	0
SIC 26	406.7	1.21	3.65
Total SIC 24+25+26	1065.9	3.17	
West Virginia			
All sectors	8528.3	100	0.67
SIC 24	81.1	0.95	4.18
SIC 25	12.4	0.15	0
SIC 26	22.2	0.26	0
Total SIC 24+25+26	115.7	1.36	

[a] 1982=100, prices are adjusted for inflation and expressed in terms of value in 1982.
[b] SIC 24 = lumber and wood products
[c] SIC 25 = furniture and fixtures
[d] SIC 26 = paper and allied products
[e] Figures for NJ, NY and NC are totals of the subset of counties in the MAIA region.
Source: Department of Labor, unemployment insurance database ES-202; (http://workforcesecurity.doleta.gov/unemploy/finance.asp).

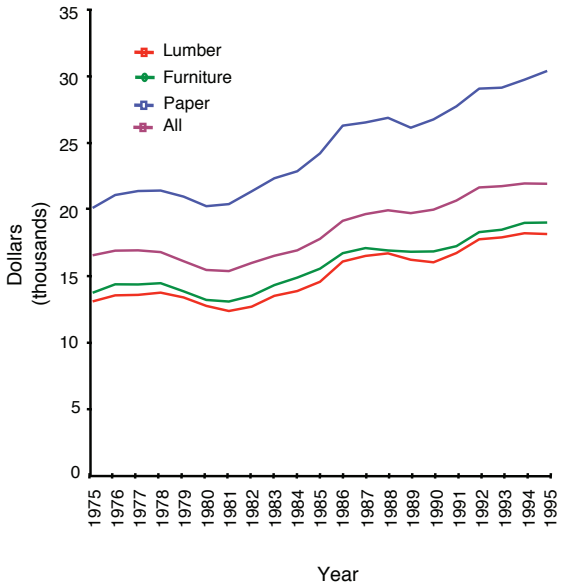


Figure 88—Real wages per job in lumber and wood products (SIC 24), furniture and fixtures (SIC 25), paper and allied products (SIC 26), and all sectors of the MAIA economy 1975 to 1995. Source: Department of Labor, unemployment insurance database, ES-202; (http://www.bls.gov/bls/blswage.htm).

Chapter 16.

Game Species

Robert Eaton
Southern Research Station, USDA Forest Service

Trends in game species populations often are indicators of some aspect of forest health, because changes in animal populations often reflect changes in habitat. Knowledge of the habitat requirements of specific animals, combined with examination of population trends, may provide information about the type, quality, and abundance of vegetative habitats. For example, if historic (or known) game species populations increase with higher harvest rates, probably there is an adequate amount of habitat to support the population of game species; however, it also indicates that the game species population could deplete the resources on which it or other animal species populations depend. Conversely, if historic or known game populations are declining or increasing at a rate lower than expected with increasing harvest rates, it may signal the decline of suitable habitat, over-hunting, and/or the presence of disease or some other limiting factor.

Game species harvest is also an economic concern because income is generated in both a direct and indirect manner. The sale of hunting licenses, firearms, ammunition, and other accessories provides income for the States and commercial interests. Interest in hunting often generates support for more or better habitat for popular game species, as well as general support for improving overall forest ecosystem condition.

In this analysis we focused only on populations of forest game species, but game species may also be used as surrogate habitat indicators for other wildlife species. This is because an adequacy of habitat types for game species indirectly benefits non-game species that require similar habitat. Data used in these analyses were obtained from participating States in the MAIA region, including State jurisdictions external to the MAIA region; therefore, estimates of populations and harvests extend beyond the MAIA boundaries.[10] A limited amount of information on game animals was reported in the MAIA States and region, and there was some scattered information on bobwhite quail and eastern cottontail rabbit from some States.

[10]Curtis H. Flather, personal communication (Research Wildlife Biologist, 4853-Human Dimensions, USDA Forest Service, Rocky Mountain Research Station, 2150 Centre Avenue, Building A, Fort Collins, CO 80526; 970-295-5910).

Black bear populations in the MAIA region increased from an estimated 8,800 in 1975 to 23,800 in 1993, an increase of nearly 171 percent (fig. 89). Harvest of bears also increased from 1,400 in 1975 to almost 4,900 in 1993, a 250 percent increase. However, the number of bears harvested in 1993 represented 20.6 percent of the total population in 1993, a slight increase from the 15.9 percent harvest in 1975. The cause of the increase in bear harvest between 1990 and 1993 may be due to improved tracking of the number of bears killed; but on the other hand, it may simply be due to increased interest in bear hunting. It is also unknown whether the overall population increases of bears since 1975 is due to improved methods of locating and tracking bears, or improvements in the amount and quality of bear habitat.

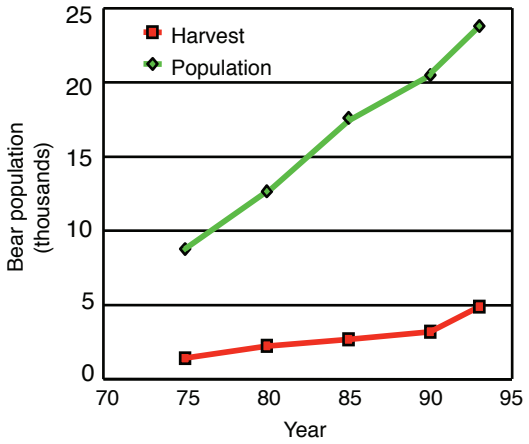


Figure 89—Black bear population and harvest in MAIA region states 1975 to 1993. Estimates include counties outside of the MAIA region for New York, New Jersey, and North Carolina. Source: Curtis H. Flather, personal communication; (http://www.fs.fed.us/rm/analytics/staff/flather.html).

Gray squirrel populations remained relatively steady from 1975 to 1993, falling only about 0.2 percent from about 11,020,000 to roughly 11,000,000, respectively (fig. 90). Squirrel harvest generally declined from 1980 to 1993, from about 4,001,500 to 2,266,500, a reduction of approximately 43 percent. Harvest of gray squirrels in 1993 represented about 20.6 percent of the population, compared to the 1975 harvest of 36.3 percent of the population. The stability of

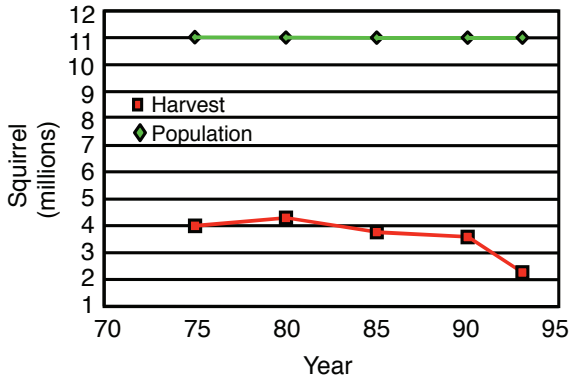


Figure 90—Gray squirrel population and harvest in MAIA region states 1975 to 1993. Estimates include counties outside of the MAIA region for New York, New Jersey, and North Carolina. Source: Curtis H. Flather, personal communication; (http://www.fs.fed.us/rm/analytics/staff/flather.html).

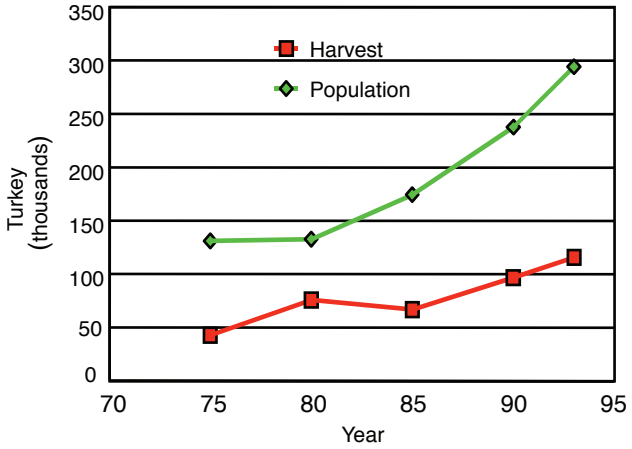


Figure 92—Wild turkey population and harvest in MAIA region states 1975 to 1993. Estimates include counties outside of the MAIA region for New York, New Jersey, and North Carolina. Source: Curtis H. Flather, personal communication; (http://www.fs.fed.us/rm/analytics/staff/flather.html).

squirrel populations, even with declining harvest pressures since 1980, suggests that this species has probably reached the carrying-capacity of environments to support their populations, and populations are more controlled by habitat and other factors then by hunting pressures.

White-tailed deer populations increased between 1975 and 1993 (fig. 91), particularly during the1980 to 1993 period. Populations increased from approximately 2,134,000 in 1980 to about 3,668,000 in 1990, then leveled-off to roughly 3,654,000 in 1993. The estimated overall increase from 1975 to 1993, therefore, was 71.2 percent. Harvest over the same

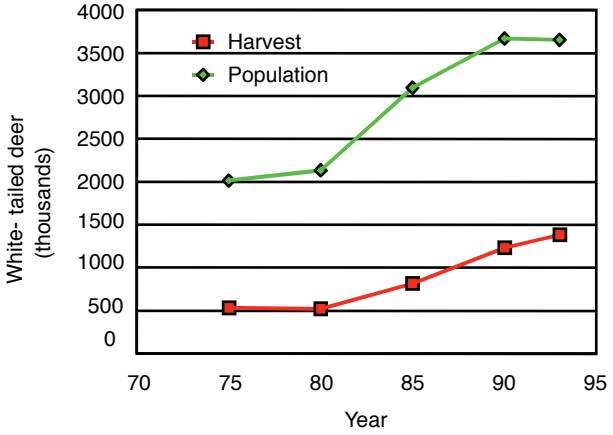


Figure 91—White-tailed deer population and harvest in MAIA region states 1975 to 1993. Estimates include counties outside of the MAIA region for New York, New Jersey, and North Carolina. Source: Curtis H. Flather, personal communication; (http://www.fs.fed.us/rm/analytics/staff/flather.html).

period also increased, but at a faster rate. In 1975 the harvest was approximately 524,700 deer, and by 1993 it had grown to 1,382,600, a 163.5 percent increase. Harvest of white-tailed deer in 1993 represented about 37.8 percent of the population, and in 1975 the harvest was approximately 24.6 percent of the population.

Wild turkey estimated populations and harvests also increased from 1975 to 1993 (fig. 92). Populations rose from approximately 131,000 in 1975 to around 294,000 in 1993, an increase of about 124 percent. Harvest of wild turkey rose from 43,000 to 115,800 during the same period, a 169 percent increase. The harvest rate in 1993 was 39.4 percent of the estimated population, a slight increase from the 32.8 percent harvest rate in 1975.

Eastern cottontail rabbit populations were estimated at 10 million in 1975, and declined to 8 million in 1993, a reduction of 20 percent (no figure available). Estimated harvests of eastern cottontails declined from an estimated 4.5 million in 1975 to 2.4 million in 1993, a decrease of approximately 46 percent. Harvest of eastern cottontails in 1993 was about 30 percent of the total population, compared to 45 percent in 1975. The decline in eastern cottontails from 1975 to 1993, despite reduced harvesting of this species, could be attributed to several factors. The continued maturation of forests, particularly in the central and northern parts of the MAIA region (figs. 68 to 71), would further close tree canopies, reducing the amount of light reaching the forest floor, and therefore reducing understory vegetation important for rabbit cover and food. Another likely reason is the significant reductions in cropland and pasturelands areas

resulting from increases in urban lands (figs. 35, 36, and 39) and reforestation of abandoned farmlands. The former land uses provided a highly desirable source of food for this species. Increases in urban lands also bring increases in dog and cat populations, which could significantly reduce the number of surviving cottontail offspring in each generation. Scattered information on bobwhite quail suggest populations dropped during this period, probably related to many of the same factors relevant to the eastern cottontail rabbits.

The increasing populations of wild turkey, black bear, and white-tailed deer in the MAIA region are other indicators that forests in the region are recovering from the devastating lack of land management concerns in the 17th to 19th centuries. All of these species were nearly extinct in the central and southern forests by the early 1900s (MacCleery 1992). Nonetheless, there is growing evidence that white-tailed deer populations are exceeding the carrying capacity of forests in the eastern States, and are becoming serious stressors of forest ecosystems through overgrazing of understory plants and selective consumption of regenerating tree seedlings (Trumbull and others 1989; Marquis 1981).

SOIL and AQUATIC SYSTEMS

Chapter 17.

Soil Systems

Barbara L. Conkling[1] and Kenneth W. Stolte[2]
[1]Department of Forestry, North Carolina State University
[2]Southern Research Station, USDA Forest Service

Soils are an essential component of healthy forest ecosystems (fig. 2). Plants obtain water and nutrients from the soil, and most of the crucial mineral exchange between the biosphere (world of living creatures) and the inorganic world occurs in the soil (Ricklefs1979). When plants die, they decompose and return mineral nutrients and organic material to the soil. The numerous bacteria, fungi, minute arthropods and worms, termites, millipedes, and other small organisms responsible for decomposition are abundant in the surface layers of the soil where dead organic matter is most plentiful. The activities of these organisms contribute to the development of soil properties in a surface-to-bottom direction, while physical and chemical decomposition of bedrock contribute to soil development in a bottom-to-surface direction. Both surface and bedrock processes are important for a healthy soil system.

Soil serves a variety of purposes, both physical and chemical. It provides five of six environmental components upon which plants, including trees, are dependent: mechanical support, heat, air, water, and nutrients (Brady 1984). Analyses of the estimated extent of human-induced soil degradation on a global basis has indicated that the primary threats to soil stability in forest ecosystems are erosion by water, acidification, loss of nutrients, and compaction (Hudson 1992).

Most of the soil volume is relatively inert, and most of its chemical and biological functions are determined by the clay and organic matter components (Hudson 1992). Soil sampling in the FHM and FIA programs focuses on the biological, chemical, and physical processes of mineral soils and organic matter in the upper 20 inches of soil, with most emphasis on the upper 8 inches. A layer of leaf and wood litter covers the top 8 inches of forest mineral soils composed of O and A, and sometimes B, soil horizons. Specifics about soil collection and analytical methods used in the FHM and FIA programs are in Technical Appendix B *(FHM and FIA Data Analyses).* The thresholds used in our analyses of soil data in this section were derived from standard soil taxonomy and chemistry information and expert opinion.[11]

Fertility

Forest ecosystems recycle most of their nitrogen, phosphorus, and other nutrients through the soil; and soil organic matter is a major component in global carbon cycling. Soil condition can be a good indicator of ecosystem disturbance because many disturbances or types of disturbance affect the surface of the soil, where much of the important biological and chemical activity occurs (Doran and others 1994). Over the last 50 years, soil organic matter levels have increased in many forested areas of the Eastern U.S., as forests recover from past management practices, e.g., clearing for homes and crops, cutting for charcoal, intensive grazing (Trimble 1974).

Fertility is an important attribute of soils, because plants modify the soil and have adapted to survive in soils that cover a wide range of chemical and physical characteristics. For this reason soils are an important component in determining site quality or productivity (Pritchett and Fisher 1987). Bailey (1995) also used soil types as a major factor in differentiating ecoregion units throughout the United States.

The FHM program analyzed mineral soil for percent exchangeable calcium, magnesium, potassium, and sodium. The concentration or percent content of a nutrient is recognized as only one factor used to determine the availability of nutrients to plants. Other factors affecting availability include soil moisture, microbial populations, tree rooting characteristics, and mycorrhizal growth (Brady 1984).

Calcium is an important component in the formation of cell walls, and has been identified as a factor in regulating plant growth rates (Lee and others 1983). Calcium deficiency symptoms appear first in new tissue. Calcium is released from its sources (e.g., bedrock, deposits) by mineral weathering, and is usually the dominant exchangeable cation in the soil. Although calcium is less available at low pH, soils are rarely deficient in the element except in areas where acidic deposition caused accelerated leaching (Tomlinson

[11]Personal communication. 1999. Kimberly Ludovici (retired). Southern Research Station, USDA Forest Service, 3041 Cornwallis Road, Research Triangle Park, NC 27709.

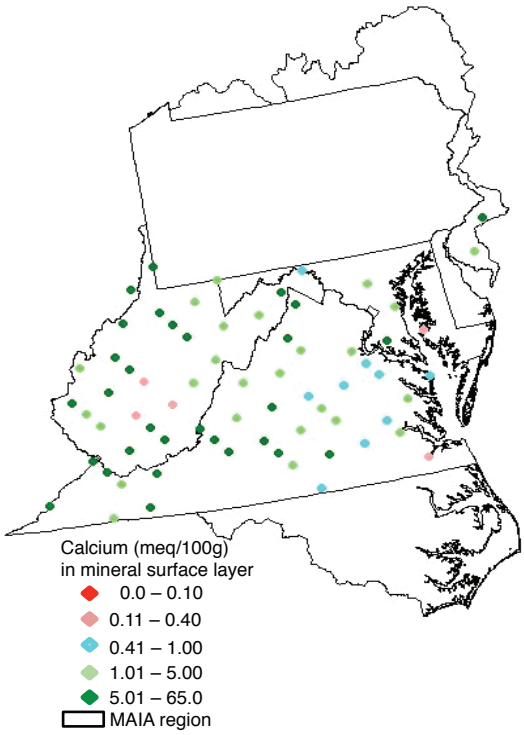


Figure 93—Concentration of exchangeable calcium in soil surface mineral layers (0 to 4 inches) on FHM plots in the MAIA region in 1998. Source: National Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

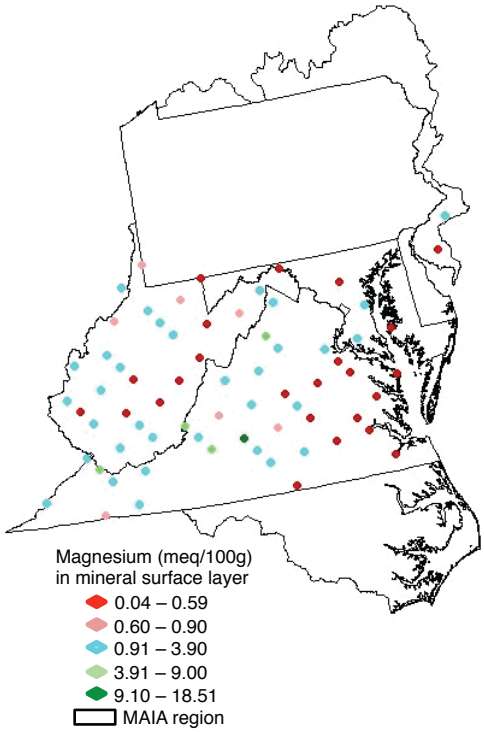


Figure 94—Concentration of exchangeable magnesium in soil surface mineral layers (0 to 4 inches) on FHM plots in the MAIA region in 1998. Source: National Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

1990). Figure 93 shows that the majority of sampled surface soils in the MAIA region were relatively high in exchangeable calcium (1.0 to 65.0 meq per 100g). Some plots in western West Virginia and eastern Virginia had relatively lower values (0 to 1.0 meq per 100g) in localized areas.

Magnesium is a constituent of chlorophyll and a vital part of photosynthesis (Brady 1984). Like potassium, magnesium is mobile in plants, and deficiency symptoms appear first in older tissue. Magnesium, like calcium, is also released from mineral sources into the soil solution. Exchangeable magnesium concentrations (meq per 100g) in a few States in the MAIA region are shown in figure 94. Eastern Virginia, parts of West Virginia, and scattered locations in Maryland had relatively low levels of exchangeable magnesium (0.04 to 0.9 meq per 100g) in surface soils. Relatively moderate levels (0.91 to 3.90 meq per 100g) were found scattered in the central and western parts of the region, and relatively high levels (3.91 to 18.61 meq per 100g) were found at a few sites. As with calcium, the two lowest ranges (0.04 to 0.9 meq per 100g) reflect a higher susceptibility of the soils at those sites to stresses such as acidic input or other acidifying processes.

Acidity

The amount of soil acidity, commonly expressed as pH, is a strong indicator of nutrient availability, and of biological functions such as microbial growth and its effects on rooting (Brady 1984). The pH is the inverse log (base 10) of the hydrogen ion activity—thus a change in pH from 5 to 4 is a ten-fold increase in the number of free hydrogen ions and therefore the acidity of the solution. Soil acidity contributes to the availability of all ions in the soil solution. For example, at higher pH values (> 6.5), ions like calcium,

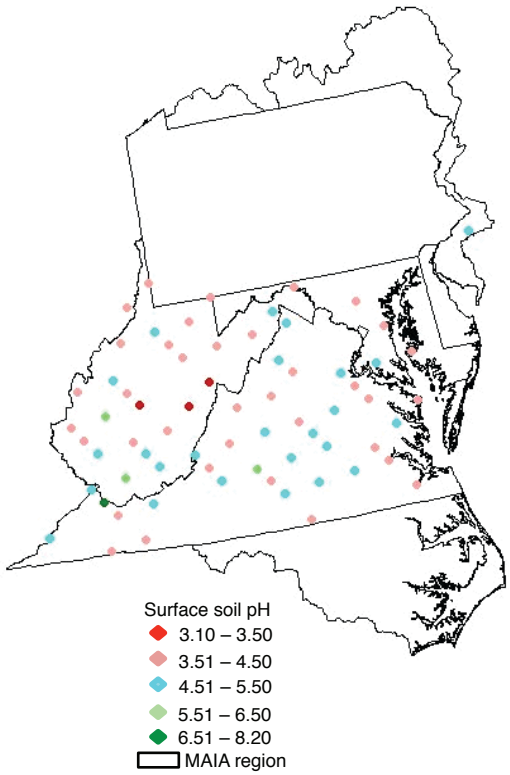


Figure 95—Soil pH values in surface mineral layers (0 to 4 inches) on FHM plots in the MAIA region in1998. Source: National Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

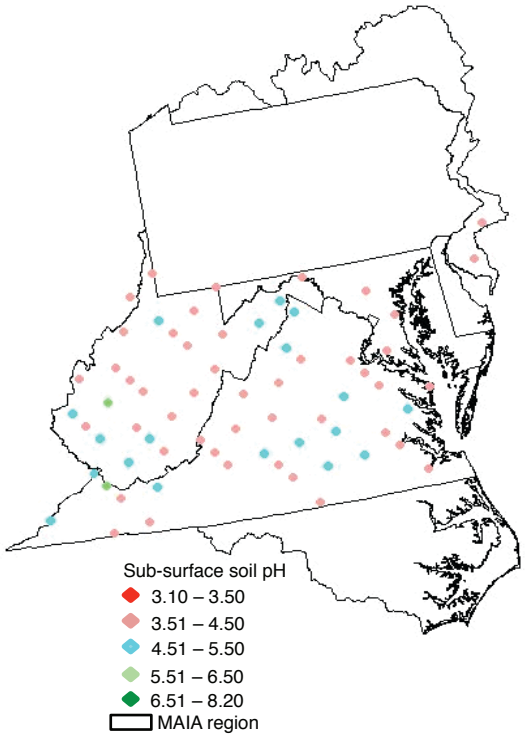


Figure 96—Soil pH values in the subsurface mineral layer (4.1 to 8 inches) on FHM plots in the MAIA region in1998. Source: National Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

potassium, and phosphorus generally are available, and aluminum is chemically bound and not available. When the pH is less than 5, aluminum becomes available and the availability of calcium, potassium, and phosphorus is greatly reduced. At high pH values (>7.0) bicarbonate ions can be found in sufficient quantities to interfere with desired uptake of other nutrient ions.

A major concern is whether some forest soils are receiving acidic precipitation, and if so, is there sufficient buffering capacity in these forest soils to neutralize the added acidity? If not, forest soils can become more acidic and lose essential nutrient cations such as calcium, magnesium, and potassium, and release toxic cations such as aluminum, iron, and manganese in concentrations that can be harmful to plants. Representative pH ranges for some general soil types are: for acid peat soils – about 2.8 to 3.2; for humid region mineral soils – about 5.0 to 7.2; for arid region mineral soils – about 6.8 to 9.0; for alkali mineral soils – greater than 10.

The pH values of some surface-layer soils in the MAIA region are shown in fig. 95. Many of the pH values were low (pH 3.51 to 4.5) and some were very low (pH 3.1 to 3.5), as indicated by the red and dark red symbols. In contrast, a fair number of plots showed soil pH values of 4.51 to 5.5.

The subsurface soil pH values are shown in fig. 96. The pH values lower in the soil profile are similar to the surface layer, with a high number of plots with low pH (3.1 to 4.5) in the subsurface soil layers of West Virginia. These pH values were somewhat lower than reported in other studies, and may be partially attributed to the oven-dried preparation of samples for analysis. Further evaluation is needed on the pH values of MAIA region soils, and the implications of low soil pH values on forest health.

Chapter 18.

Aquatic Systems

Kent Thornton
FTN Associates

Forest Condition and Quality of Streams and Rivers

About 60 percent of the MAIA region is covered by forested lands that are the foundation of an established timber products industry (see Table 2). In addition to providing timber and other products, forests in the region reduce soil erosion and sedimentation, remove atmospheric contaminants, produce oxygen, provide habitat for wildlife (e.g., native and neotropical birds), sustain streams and stream communities, and provide recreational and aesthetic experiences for a large Mid-Atlantic population.

In the early and mid 1990s, scientists, managers, diplomats, and concerned citizens met to formulate an approach for sustaining the world's forests. The Santiago Declaration for the Conservation and Sustainable Management of Temperate and Boreal Forests (Anon. 1995b) identified seven themes necessary to sustain global forests: one of those themes is conservation of forested aquatic ecosystems. While proper forest management leads to sustainable ecosystems, poor forest management practices—particularly during timber harvesting and road construction—can significantly degrade not only the terrestrial ecosystem, but also the associated aquatic ecosystems.

The type of soil in a forested watershed is determined by parent material, climate, vegetation, local topography and, to some extent, the age of the soil. Once formed, soils remain in a state of flux, although they frequently obtain regionally characteristic steady-state properties. The process of soil formation and the stability of mature soil systems can be drastically altered by soil erosion. Steep rock faces, with no soils present, are examples where erosional forces have been greater than the processes that lead to soil formation and buildup. The process of soil formation and buildup usually takes many years in temperate climates, and steady-state soil conditions in healthy watersheds must be protected from soil erosion disturbances to protect associated aquatic systems.

Standard technical definitions characterize soil erosion and extent classes. Surface erosion of soil is the removal of the soil surface by water, wind, ice, or other processes (Warrington and others 1980). Surface soil loss by water specifically involves the detachment of mineral soil particles and organic material from the soil surface. Soil particle detachment from rainfall impacts, or shear from flowing water, can occur when soils are not protected by tree, shrub, herb, and grass plant canopies, plant litter, and mineral or organic surface mulch, which all help mitigate these effects.

Local-scale effects of forest management and timber harvesting practices on stream ecosystems have been studied and well documented in the literature (Bormann and others 1974, Likens and others 1970, 1977). However, forest management and timber harvesting practices have changed substantially in recent decades as best management practices (BMPs) have been identified and implemented. It became apparent that forest management practices on relatively small areas can have large-scale (landscape) impacts (Forman and Godron 1986, Forman and Mellinger 1999, Turner 1989). Because streams are a product of the condition of the watersheds where they are located (Hynes 1975), numerous local-scale impacts might also create large-scale effects on stream quality.

This section reviews local-scale effects of forest management practices on streams in the MAIA region over 20 years, develops a conceptual model of these effects, and analyzes information from a regional stream monitoring program to determine if the effects of forest management practices on streams at the local scale can be detected at a regional scale.

Forest Management Practices and Stream Quality: Regional Effects

Forest management and harvesting practices can have multiple effects on stream quality (Binkley and Brown 1993, Dahlgren and Driscoll 1994, Dietterick and Lynch 1989, Likens and others 1977, Wigley and Roberts 1994), including:

- modified watershed hydrology and water balance,
- erosion and sedimentation,
- habitat alteration,
- chemical contamination, and
- stream biology alteration

These effects are not independent, but rather linked and interactive. For example, the top row of boxes in the conceptual model identifies disturbances that are common in forested watersheds (fig. 97). These disturbances

initially affect the flow and timing of water and often introduce chemical contamination. These primary effects lead to impacts such as erosion of the stream bed or bank (secondary effects) that often lead to loss of nutrients and sedimentation of the stream. As with most natural resource systems, significant changes in habitat often lead to major changes in biodiversity.

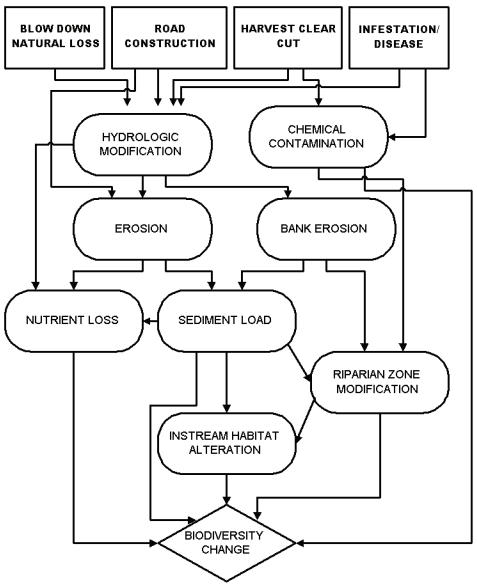


Figure 97—Interactive effects of forest management practices on processes affecting stream quality.

Assessing hydrologic modifications, therefore, can also provide information on soil erosion and stream sedimentation, habitat alteration, nutrient transport, and effects on aquatic species diversity. Figure 98 shows the magnitude of the effects of management and harvesting practices on components and processes in watersheds and associated aquatic systems. It compares two sides of a hypothetical watershed, where one side has experienced significant removal of trees, and the other side had not been disturbed.

On the undisturbed side, water runoff-loss of nutrients-sediment transport and solar inputs to streams are low, and evapotranspiration (ET) and production of coarse (large) woody debris are high. In contrast, the harvested side has relatively low ET, but relatively high solar input to streams, water runoff-loss of nutrients-sediment transport, and input of fine woody debris into streams. Consequently the type, amount, and location of management and harvest activities can have very large effects on water quantity

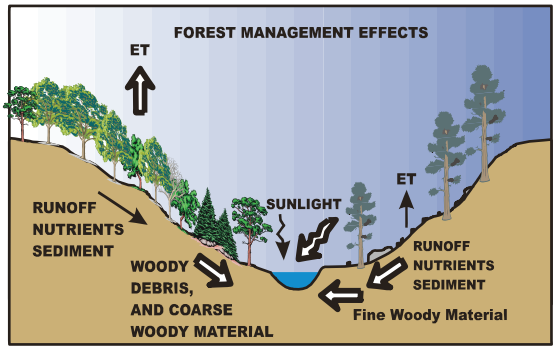


Figure 98—Cumulative effects of forest management on a stream and associated watershed. Changes in arrow sizes reflect changes in the magnitude of the process before and after timber harvest.

and quality, as well as associated biota. See Technical Appendix E (Forest Management Practices and Stream Quality) for a more in-depth discussion of the relationships between the conditions of forested watersheds and associated aquatic conditions.

Direct local-scale effects of forest management practices on stream quality can be readily detected (fig. 98), but it is uncertain whether there is a signature set of cumulative, multiple, local-scale effects that could be detected at a regional scale. To address this question, we obtained information on stream quality from EPA's Environmental Monitoring and Assessment Program's (EMAP) Mid-Atlantic Highlands (MAHA) stream monitoring program (<http://www.epa.gov/reva/vulnerability>). Approximately 360 stream reaches, selected as probability samples using the EMAP sample survey design (Herlihy and others 2000, Stevens 1997), were sampled once during the 1993 to1994 spring season (fig. 99).

Those sites represent over 110,000 stream km in the MAHA region on a 1:100,000 scale map, with most of the streams in Pennsylvania and West Virginia. All of the streams in the Mid-Atlantic Highlands were first-through-third Strahler-order streams (Strahler 1964); very few streams were gauged, however, so we could not determine changes in high and low water flow regimes as a function of land use and forest management practices.

Attributes of riparian and in-stream habitat were measured, including chemistry, fish, benthos, periphyton, and stream metabolism. In addition to riparian and stream measurements, general watershed characteristics (e.g., area, slope, land use/land cover, disturbance) for the watershed upstream from the corresponding stream reach

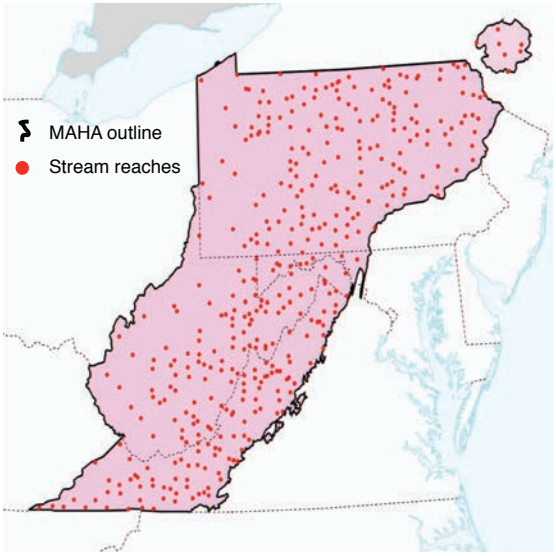


Figure 99—The Mid-Atlantic Highlands Area is circumscribed by the dark border. About 360 stream reaches, representing over 110,000 stream kilometers in the area, were sampled (dots). Source: U.S. EPA; (http://www.epa.gov/reva/vulnerability/).

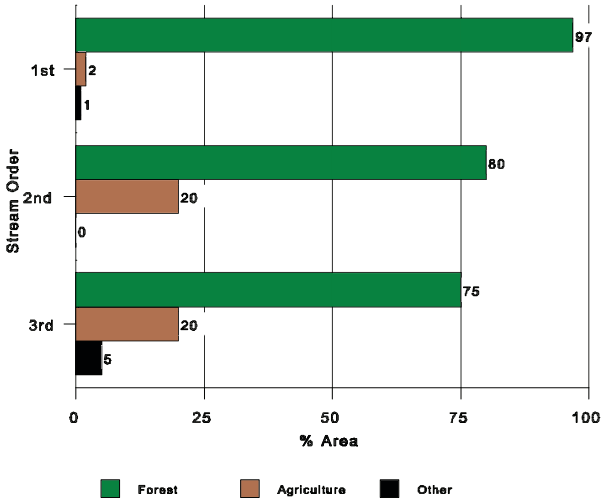


Figure 100—Land use as a function of stream size or order in Mid-Atlantic Highlands Area region. Source: U.S. EPA; (http://www.epa.gov/reva/vulnerability/).

were determined for each stream segment. Our emphasis was on regional-scale management practices that might be observed by analyzing habitat, chemistry, stream insects, and fish information. More information on the MAHA and MAIA programs are on the MAIA (<http://www.epa.gov/owowwtr1/ecoplaces/part1/site15.html>) and EMAP (<http://www.epa.gov/emap>) web sites.

Because almost two-thirds of the MAIA region was covered by forests, we investigated the proportion of forested watersheds associated with different stream orders (fig. 100). We found that 97 percent of the watershed areas associated with first-order streams were forested, only about 2 percent were in agricultural uses, and another 1 percent was in other land use.

About 80 percent of the watershed area associated with second-order streams was forested; the other 20 percent was in agricultural uses. The proportion of watershed areas in agricultural uses for third-order streams was similar to the proportion in second-order streams, but the proportion of forested watershed areas was about 5 percent less (only 75 percent), and about 5 percent was in other land uses.

Another evaluation of the effects of forest management activities and the amount of forest land cover was conducted by partitioning MAHA streams by stream-order and by land use or amount of forest cover. Generally, negative effects of agricultural lands on streams become dominant when the proportion of agricultural use exceeds about 10 percent of the watershed area (Omernik 1977). Therefore, we partitioned each stream-order into two categories–streams in watersheds that had at least 95 percent of the area in forest land cover, and streams within watersheds that had less than 95 percent of the area in forest land cover.

The watersheds classed as at least 95 percent forested had a median forest cover of 99 percent, regardless of stream order (table 26). The median percent forest cover in the < 95 percent forested watershed category was 62 percent for both first- and third-order streams, and 70 percent for the second-order streams. Agricultural uses comprised the other dominant land cover in watersheds having < 95 percent forest land cover. Total area of watersheds was similar within a stream order, regardless of land use (e.g., 11.1 km2 and 14.6 km2), although the mixed land use watershed category (< 95 percent forest cover) had slightly larger median areas.

Soil erosion and sediment delivery to streams, and increased sedimentation, were local-scale effects associated with logging and other forest management practices in a watershed. Much of the erosion and runoff is often due to poor road construction and the increase in road density

during timber harvesting. To assess the potential regional effects of erosion, sediment delivery, and sedimentation, we considered a series of factors—road density, total suspended solids (TSS) concentration, and percent fines (clays and silts) in the stream bed—between the predominantly forested and mixed land use watersheds by stream order (table 27).

For all stream orders, the median road density was 2 to 2.5 times greater in mixed land use watersheds (< 95 percent forested) compared with forested watersheds. Median TSS concentrations were also 2 to 4 times greater in streams that ran through mixed land use watersheds compared with forested watersheds. The percent fines in mixed land use watersheds stream beds were 6 to 14 times greater than in stream beds associated with forested watersheds, regardless of stream order.

A similar approach was used to compare pollutants found in streams in predominantly forested and mixed land-use watersheds. Median nutrient concentrations of nitrate (NO_3) and total phosphorus (TP) were 2 to 4 times greater in streams with mixed land use compared with forested watersheds, regardless of stream order (table 28). Generally, physical and chemical aquatic habitat stressors in MAHA streams were of significant lower magnitudes in forested watersheds than in mixed land-use watersheds.

Benthic insects, particularly the number of species or genera in the three insect orders *Ephemeroptera, Plecoptera,* and *Trichoptera* (EPT), are commonly used as indicators of stream condition affected by sedimentation and chemical contamination (Lenat and Penrose 1996). Greater numbers of EPT genera usually are associated with

Table 26—Watersheds in two forest cover classes and distribution of tree cover in three stream orders in the western MAIA region circa 2000

	Forest cover		Watershed area	
Stream order	Watershed > 95 percent forested	Watershed < 95 percent forested	Watershed > 95 percent forested	Watershed < 95 percent forested
	-----median percent forested-----		*-------median area (km^2)-------*	
1	100	62	1.5	1.9
2	99	70	11.1	14.6
3	99	62	51.4	63.9

Table 27— Watersheds in two forest cover classes and logging road density and water clarity in three stream orders in the western MAIA region circa 2000

	Road density		Total suspended solids		Fine sediments	
Stream order	Watershed > 95 percent forested	Watershed < 95 percent forested	Watershed > 95 percent forested	Watershed < 95 percent forested	Watershed > 95 percent forested	Watershed < 95 percent forested
	---------feet per acre---------		*-------parts per million--------*		*----------percent----------*	
1	9.2	23.9	3.8	7.7	2	17
2	8.1	20.2	2.0	4.4	2	12
3	11.8	21.6	1.2	4.7	1	14

Table 28—Watersheds in two forest cover classes and chemical contamination in three stream orders in the western MAIA region circa 2000

Stream order	Nitrate (NO_3)		Total phosphorus	
	Watershed > 95 percent forested	Watershed < 95 percent forested	Watershed > 95 percent forested	Watershed < 95 percent forested
	----------*parts per million*----------		----------*parts per billion*---------	
1	0.7	1.4	8	22
2	0.4	1.9	6	18
3	0.5	1.9	6	16

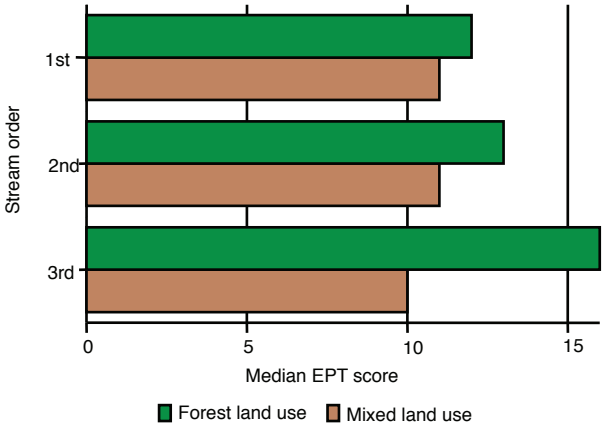


Figure 101—Median Ephemeroptera, Plecoptera, and Trichoptera (EPT) scores by stream order. EPT scores are based on type and number of these three groups of stream insects. Source: U.S. EPA; (http://www.epa.gov/reva/vulnerability/).

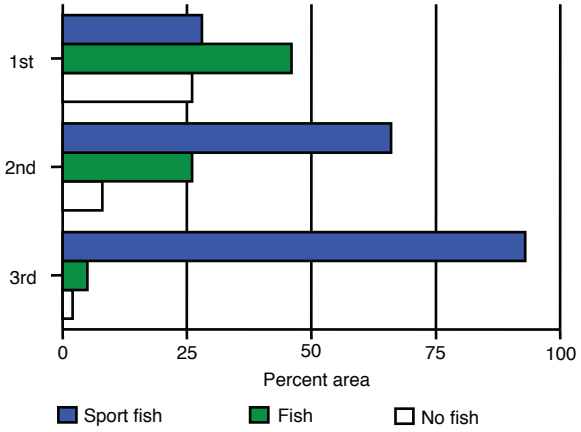


Figure 102—Mid-Atlantic Highlands Area stream miles with sport fish, fish, or no fish. Note that low numbers of fish does not necessarily mean a stream is degraded. Source: U.S. EPA; (http://www.epa.gov/reva/vulnerability/).

better stream condition. In the Mid-Atlantic Highlands Assessment, we compared the number of EPT genera present streamside in forested versus mixed land use watersheds (fig. 101). We found that the number of EPT genera in forested watershed streams was only slightly higher than in mixed land-use watershed streams, and only in third-order streams were the number of EPT genera significantly different in forested watersheds compared to mixed land-use watersheds.

While many fisheries management agencies emphasize the importance of third-order and higher streams for sport fishing, first- and second-order streams are also important in maintaining and sustaining these fisheries (fig. 102). In the MAHA, 74 percent of first-order stream miles contained fish—28 percent were sport fish, 46 percent other fish, and 26 percent had no fish. Finding no fish in 26 percent of first-order streams does not necessarily imply stream degradation, but indicates habitat was unsuitable for fish to exist (e.g., shallow, steep, lack of food).

About 92 percent of the second-order stream miles had fish—66 percent sport fish and 26 percent other fish; and only 8 percent of all second-order stream miles had no fish. About 98 percent of third-order stream miles had fish—93 percent sport fish, 5 percent other fish; and only about 2 percent of third-order streams had no fish. Because first- and second-order streams in the MAHA region were highly forested (fig. 100), and those streams contained significant amounts of sport and other fish (fig. 102), the lower-order streams are necessarily important for sustaining fisheries within the Highlands region. We therefore suspect that even

small watersheds with lower-order streams that are disturbed by forest management practices can have significant effects on stream fisheries.

In a broad sense, a regional indication of forest management practices on impaired stream quality was not apparent, particularly for 1st and 2nd order streams. In fact, stream quality associated with forest land use throughout the MAHA region generally was good, and might provide a reference for what is attainable in other areas or at other spatial scales. Because the effects on first- to third-order streams associated with logging typically are expected to return to pre-harvest variance levels in 2 to 5 years and the rate of harvest in the MAHA region was 1 to 2 percent per year (<http://www.epa.gov/reva/vulnerability>), the likelihood that forest harvesting caused negative regional scale effects on streams is low. Any negative effects on streams from forest management practices are local in scale, and any regional scale effects must be minor and were beyond our ability to detect with the indicators and sampling intensities used in the MAHA studies.

CARBON CYCLES and SEQUESTRATION

Chapter 19.

Tree and Soil Carbon

William D. Smith and Barbara L. Conkling
Southern Research Station, USDA Forest Service and Department of Forestry,
North Carolina State University

Carbon is the fundamental building block of organic life on earth, and therefore cycling of this element is an essential process in all ecosystems. Changes in carbon cycling patterns in forests outside of expected variances can reflect major alterations in forest ecosystems. Plants incorporate carbon into biological systems through photosynthesis—carbon dioxide from the atmosphere is combined with water using energy from sunlight to produce simple sugars and give off oxygen as a waste product. Some carbon is sequestered in woody biomass (above and belowground), some is used in secondary productivity that fosters the growth of insects, birds, animals, and other life forms, and some ends up in the upper soil horizons as dead organic matter which is incorporated into soils. Part of the carbon stored in forest systems later is released into the atmosphere as organic matter decomposes over time. Both forest biomass and forest soils serve as large carbon sinks (carbon deposits) and are, therefore, an essential component of a stable ecosystem and global carbon cycles.

Tree Carbon

Carbon storage in forest biomass is an important factor affecting carbon dioxide concentrations in the atmosphere. Carbon is removed from the atmosphere through photosynthesis in the process of tree growth and is returned to the atmosphere through the decay of dead tree biomass. Approximately one-half of the carbon harvested as biomass is stored for long periods as wood products (Birdsey 1996). A net gain in carbon is the result of high stand-growth rates, relatively low mortality volumes, efficient utilization of harvest trees and salvage of mortality trees, or some combination thereof.

In the 2000 FHM National Technical Report, Stolte and others (2005) reported on their analysis of carbon sequestration in the U.S. by ecoregion province. The amount of carbon stored or lost annually from each FHM plot in the MAIA region was estimated for variable periods from 1991 to 1998. Because the MAIA region contains relatively small portions of three different ecoregion provinces, province-level estimates of carbon sequestration rates do not provide accurate estimates for only those portions of each province in the MAIA region. However, results from the report suggested that carbon sequestration rates in woody biomass for the MAIA region was 1,600 lbs per acre per year. Carbon sequestration rates in trees in the MAIA region were highest (> 60 ft^3 per acre per year) in the middle and lower Atlantic Coastal Plain of the Southeast, moderately high (40 to 60 $feet^3$ per acre per year) in the western and northwestern parts of the region, low (<40 $feet^3$ per acre per year) in the central and north Piedmont areas and western mountain areas, and lowest in the northwestern part of the region (fig. 68).

These results are not surprising, because the rate of carbon sequestration in a given area is a function of inherent site quality (abundant moisture, soil fertility, and moderate rainfall), seral stage, and intensity of forest management (Burns and Honkala 1990). The southeastern Coastal Plain has some of the best conditions for tree growth in the U.S., and includes a high proportion of managed forest plantations with harvest rotation cycles set to get maximum growth rates.

Soil Carbon

Criterion 5 of the Montreal Process Criteria and Indicators refers to the maintenance of forest contributions to global carbon cycles (Anon. 1995a). This criterion includes attention to total forest ecosystem biomass and carbon pools (e.g., standing biomass, coarse woody debris, peat, and soil carbon), and contributions of forest products to the global carbon budget.

In 1995, the plants and soils in forest ecosystems were estimated to account for 60 percent of the terrestrial carbon pool (Lal and others 1995). The status and change in soil carbon over time are important information when analyzing the potential of forested ecosystems to be carbon sinks. Organic matter, usually the largest source of soil carbon, is constantly added to the soil as plant debris. Soil organic matter is then broken down by decomposers that subsequently respire carbon dioxide back into the atmosphere as one part of the decomposition process (Schlesinger 1995). What is important in healthy and sustainable forest ecosystems is that new carbon frequently

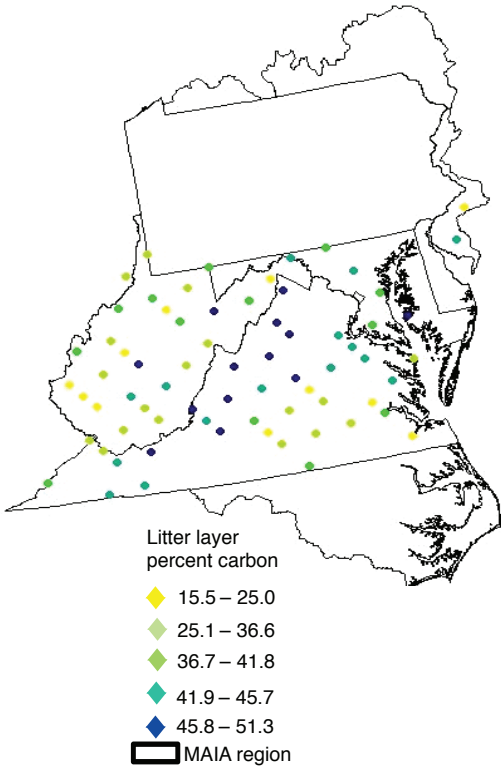


Figure 103—Total carbon by weight in the litter samples collected from forested Forest Health Monitoring plots in the MAIA region in 1998. Source: National Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

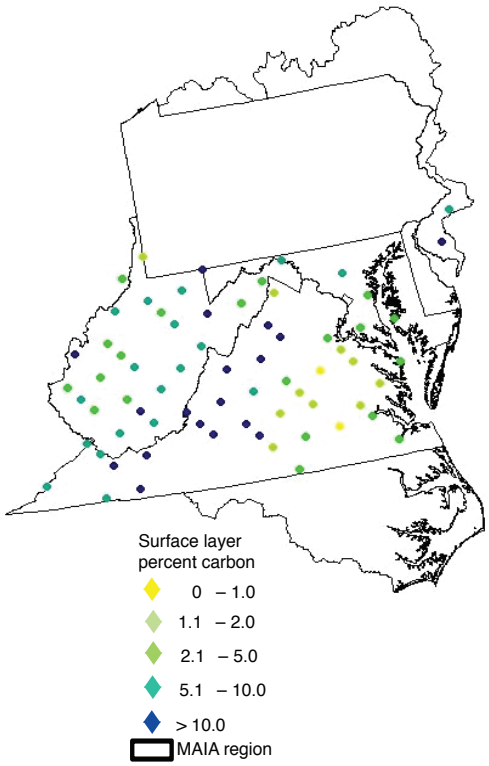


Figure 104—Total carbon by weight in the mineral surface horizon (0 to 4 inches) collected below the litter floor on forested Forest Health Monitoring plots in the MAIA region in 1998. Source: National Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

becomes available to the soil system in litter layers, and that some of the carbon in the litter layers becomes incorporated over time in the surface and subsurface soils.

The methods used by the FHM program prior to 2000 for collecting and analyzing soil samples from FHM plots was given in Chapter 17 (Soil Systems) and in Technical Appendix B. Generally, samples were taken from 3 holes at 3 depths (surface layer, 0 to 4 inches, and 4 to 8 inches). In 1998, bulk density was not recorded on each soil sample, so carbon values in this report were given as percents, and not absolute values.

The percent total carbon from the 1998 FHM soils data are presented in fig. 103 (litter layer), fig. 104 (0 to 4 inch surface layer below the litter), and fig. 105 (4.1 to 8 inches below the bottom of the surface layer). In figs. 104 and 105, which display data on the mineral samples, any soils with an organic horizon were included. The Soil Conservation Service (1975) considered a soil layer to be an organic horizon if it lost 20 percent or more in weight when burned in a laboratory analysis. Because only 1 year of data were available for this report, increases or decreases in percent total carbon could not be determined.

There were areas of relatively low (15.5 to 36.5 percent) total carbon in the litter layer of soils in eastern Virginia and scattered through West Virginia (fig.103). In contrast, the mountainous Allegheny, Northern Ridge and Valley, and Blue Ridge Mountain areas had relatively high (36.6 to 51.3 percent) total carbon in the litter layer. There also were areas of relatively low (0.0 to 2.0 percent) total carbon in the surface layers in eastern Virginia (fig. 104) and in the subsurface layers in eastern Virginia and western West Virginia (fig. 105). Relatively high (2.1 to >10.0 percent) total carbon was found in both the surface and subsurface

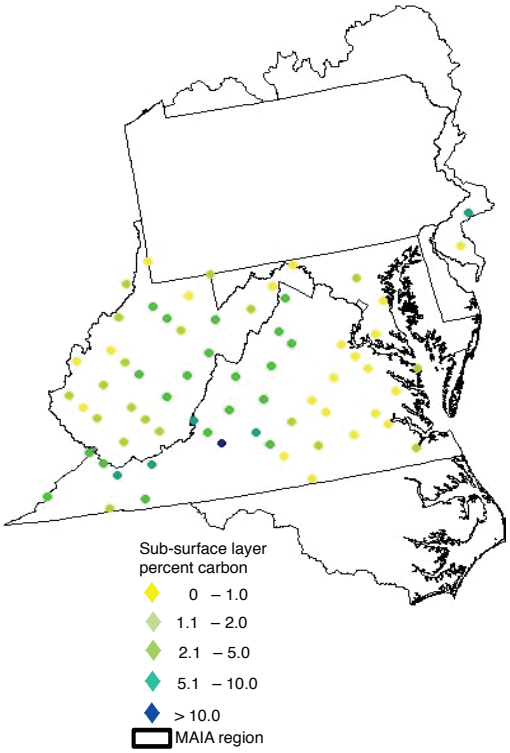


Figure 105—Total carbon by weight in the mineral subsurface horizon (4.1 to 8 inches) on forested Forest Health Monitoring plots in the MAIA region in 1998. Source: National Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

layers of the mountainous Allegheny, Northern Ridge and Valley, and Blue Ridge Mountain areas.

The higher carbon found in the litter and surface soils of the mountainous compared to other areas in the MAIA region areas reflects the differences between the relatively unmanaged forests of the mountains, with high inputs of carbon to the soil, and the intensively managed plantations (e.g., eastern Virginia) where a lot of carbon is removed as wood products (see figs. 68 to 70). The most important issues are the relative amounts of total organic carbon in soils, determined from existing and newly-established long-term monitoring plots, and whether this carbon is decreasing or increasing over time based on the reevaluation of monitoring plots within different ecological strata that are sometimes further stratified for different management objectives.

BIOLOGICAL DIVERSITY

Chapter 20.

Ecosystem Diversity

Stephanie Fulton
Region 4, U.S. US Environmental Protection Agency

Biological diversity, or biodiversity, is a measure of the variety of life as entities we call species. Biodiversity occurs at multiple spatial scales (Ricklefs 1979), and high biodiversity often is equated with ecosystem stability, lack of extreme, non-historic disturbances, and resilience to stressors (Rapport and others 1985). However, biological diversity may temporarily increase within an ecological unit due to the introduction of exotic, invasive species, although such increases are not indicative of long-term health and stability. The initial increase in diversity through addition of one or more exotic species will be temporary, because these non-native, highly invasive species tend to survive and thrive at the expense of native species, and often eventually decreases overall biological diversity (Stohlgren and others 1999).

Biological diversity can be evaluated at one or more primary spatial scales: ecosystem, community, species, or genotypes within a species. In this section we address ecosystem, community, and species diversity, but not genotypic diversity. All four levels are interconnected and hierarchical: diversity of genotypes within a species; species diversity within a community; community diversity within an ecosystem; and ecosystem diversity within a biome. If biodiversity is affected at one organizational level, changes will occur primarily at lower levels of organization; but changes also may affect higher organizational levels. For example, the virtual loss of the dominant American chestnut tree species in Eastern forests (Schlarbaum and others 1997) resulted in severe reductions in the numbers of genotypes of this species, and also affected community level diversity, because considerable numbers of other plant and animal species were greatly impacted by the loss of this once dominant overstory tree. The physical, structural complexity within any level is another aspect of diversity—structural diversity.

Biological diversity is important at all levels because of the connectivity among genotypes, species, communities, and ecosystems. Many types of stressors, both human-induced and natural, can lead to changes in diversity at one or more spatial scales. As conditions change and habitats are altered by forces, diversity often is affected to some degree; and changed conditions often are an indication of new or exacerbated endemic stressors on ecosystems. Based on extensive fossil records, many more genotypes, species, communities, and ecosystems have become extinct than exist in total today. For example, ice ages in past millennia greatly altered diversity at all levels. Therefore, loss of biological diversity at any spatial scale that might occur as a result of human activity, e.g., tree harvest, fire suppression, climate change, or other activities, must be evaluated relative to normal variations in biological diversity due to natural forces. That is, to gain a true appreciation of how human activities affect biological diversity, it is necessary to compare the nature and magnitude of changes in biological diversity found in appropriate reference conditions or from historical records.

Biological diversity within any ecological unit is limited based on the nature of the ecological unit, which primarily is determined by environmental factors of temperature, precipitation, geology, topography, and the type and magnitude of endemic disturbance forces. Biological diversity is highest when temperatures are moderate, precipitation plentiful, topography creates different physical niches, and disturbances generally are not catastrophic. Thus biological diversity is often highest in tropical ecosystems, and much lower in boreal ecosystems. Therefore, comparing biological diversity between forest types, watersheds, or other ecological units in the MAIA region is not valid, but comparing biodiversity within an ecological unit over time is the most informative use of this indicator.

Inventorying, monitoring, and interpreting biological diversity (e.g. forest types, plant species, bird species) within ecological units identifies the relative diversity among units, and establishes baseline conditions against which to compare future conditions. Long-term monitoring of biological diversity within ecological units, and the factors that influence diversity, provide a mechanism to differentiate changes in naturally caused diversity from human-induced factors. In this report we discuss ecosystem diversity (or landscape diversity), community diversity (forest types), and diversity of species (trees, lichens, and birds). In future analyses we can evaluate changes in biological diversity over time and estimate the role that human activities may play in causing any observed changes.

Patterns of forest loss are as important as the amount of forest lost, because they determine the size and shape of remaining forest patches, how remaining forest patches are connected, and other alterations that may affect ecosystem diversity. Removal of small forest patches can have disproportionate negative effects on wildlife habitat if the patches removed are fragments that provide corridors for wildlife movement between larger blocks of forest. Patches of suitable habitat must be close enough together for particular species to move from patch-to-patch, and avoid poor habitats, as well as predators. Both rate and pattern of forest conversion and forest fragmentation are strongly influenced by regional socioeconomic patterns of land ownership, land use, and resource consumption.

Patterns of land use greatly affect the natural environment and produce landscape-scale patterns defined by the size, shape, and distribution of land-use types present in a region. Forests are one type of landuse; and the size, shape, species-composition, and connectedness of forest patches across the landscape affect ecosystem processes. Some ecological processes affected by size and distribution of forest fragments include the rate of spread of wildfires, native insects and pathogens, and invasive species; the movement and survival of forest-dependent wildlife species and birds; water quantity and quality; soil erosion, loss of soil, and sedimentation or loss of aquatic systems; and myriad other forest components and processes. For example, forests stabilize stream banks and provide shade and important food sources for many aquatic species. Streamside—or riparian—forests also filter nutrients, toxins, and sediments that flow into streams from adjacent urban and agricultural areas. The type, length, and width of riparian forests determine the filtering capacity, as do the type, amount, and proximity of agricultural and urban lands adjacent to them.

Relative impacts of conversion of forestland to non-forest uses, and subsequent increases in forest fragmentation, vary by land-use type. The spatial pattern of landscape units representing the relative proportion of forest, agriculture, and urban landuse in the MAIA region showed these land use types distributed unequally across the landscape. The eastern section of the MAIA region contained agriculture and urban land use types interspersed with small fragments of forest, and only a few large, unbroken tracts of forests remained in the western Region (fig. 12). That is, the number of patches of multiple landuse types was highest in the eastern MAIA region, and much lower in the western sections (fig. 13).There were relatively large areas in central Pennsylvania, western Maryland, Virginia, and North Carolina, and almost all of West Virginia where forest land-use types dominated the landscape. These same areas contained numerous large forest patches with more than 90 percent forest cover (fig. 12). Generally, watersheds closest to the Chesapeake Bay and its tributaries have suffered the greatest loss of forests and the most increases in number and diversity of land-use types (figs. 12 and 13), while some watersheds in the western, mountainous portion of the region still contain relatively intact forest ecosystems.

Chapter 21.

Community Diversity

James Steinman and Stephanie Fulton
Northern Research Station, USDA Forest Service and
Region 4, U.S. Environmental Protection Agency

Forest Types

Forest types constitute the grouping of tree species that are often found together—a type of community diversity that is normally determined by the relative abundance of the most common species in a forest (Eyre 1980). Associations of the same tree species occurring in different species' abundance often constitute a different but related forest type. It follows then, that a commonly occurring association of species may be classified into one or more forest types based on the relative abundance of trees in each species.

The amount of each forest type varied by State and primarily was determined by the intersection of State boundaries ((fig. 1) and Bailey's (1995) ecoregion sections) (fig. 5; table 4)—which is to say, how well individual tree species were adapted to local ecological conditions (climate, soil, and topography) within a State. Additionally, the natural distribution of these forest types has been modified by human activities.

The Mid-Atlantic deciduous forest is among the world's most floristically diverse regions. Forests of the MAIA region contain species common to other Eastern forests, numerous *endemic* species (common within the MAIA region but not in other places), and many rare, endangered, or threatened plant species. The occurrence of seven different forest types—oak–hickory, maple–beech–birch (or northern hardwoods), loblolly–shortleaf pine, other conifers, oak–gum–cypress (or oak-gum), elm-ash, and aspen-birch—illustrate the high community diversity (forest type communities) in the MAIA region. The most common forest type was oak-hickory, which accounted for 32 percent of the total land area, followed by maple-beech-birch at 12 percent of the area; other conifer followed at 7 percent, loblolly-shortleaf pine at 6 percent, oak-gum-cypress at 2 percent, elm-ash at 1 percent, and aspen-birch at less than 1 percent (fig. 106). Most of these forest types were found in relatively distinct areas within the region, although there was some overlap among a few of the forest types.

We found that Pennsylvania, Virginia, and West Virginia, in that order, had the most cover of all forest types in the MAIA region (fig. 107). Virginia, Maryland, New York,

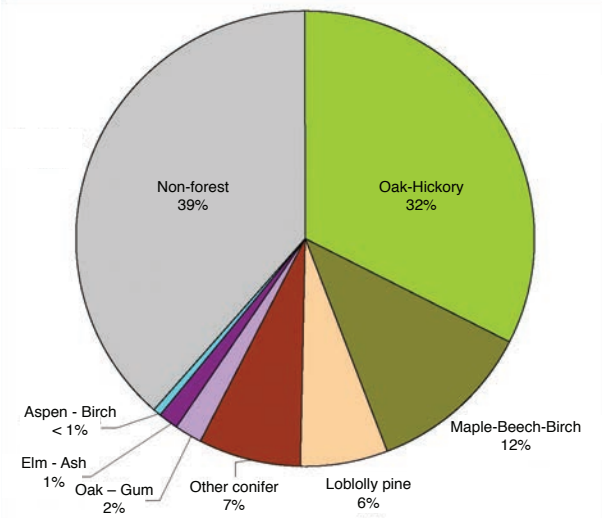


Figure 106—Forest stand distribution in the MAIA region. Maple-beech-birch are also known as Northern Hardwoods.
Source: USDA Forest Service, Forest Inventory and Analysis program's Eastwide Database; (http://www.srs.fs.fed.us/sustain/data/fia/eastwideguide.htm).

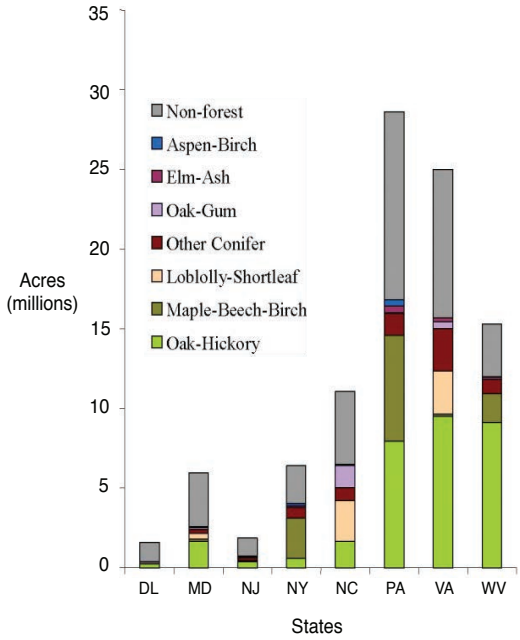


Figure 107—Forest cover type by MAIA region state. Source: USDA Forest Service, Forest Inventory and Analysis program's Eastwide Database; (http://www.srs.fs.fed.us/sustain/data/fia/eastwideguide.htm).

North Carolina, and Pennsylvania each had five or more forest types. West Virginia and New Jersey each had four forest types. Pennsylvania had the largest area covered by all forest types (about 17 million acres), and the most acres of maple-beech-birch and aspen-birch forests. Virginia had the largest acreage of oak-hickory, and North Carolina had the largest area of oak-gum-cypress.

More than 53 percent of the MAIA region's forests were in the oak–hickory forest type, which covered almost one-third (32 percent) of the total area (fig. 106). West Virginia, central Virginia, most of Pennsylvania, and most central and northern coastal areas had large areas in the oak-hickory type (fig. 108). Upland oaks and hickory species dominated the oak–hickory forest type. Typically about 40 percent of the trees were various oak species, and 5 percent were different hickory species. Red maple and yellow-poplar each contributed an additional 11 percent to the composition of the oak-hickory type, and were commonly predominant within some forest stands. Over 20 other tree species comprised the remaining 44 percent of oak-hickory forests, including elm, maple, and black walnut.

The elimination of fire, a common disturbance regime before European settlement, heavy deer browse, and other factors have contributed to lack of oak regeneration on many oak-hickory sites. In many regions, we found a notable shift in relative stocking from oak species to red maple and tulip poplar. In the absence of prescribed burning or other silvicultural activities, and if deer populations were to remain high, oak regeneration would remain low, increasing concerns that oak species will continue to dominate Eastern forests for the foreseeable future. The future domination of oak species may be confined only to dry or frequently disturbed sites.

Maple-beech-birch, sometimes referred to as Northern Hardwoods type, occupied 12 percent of the region and, as indicated by that name, sugar maple, red maple, American beech, yellow birch, and sweet birch accounted for over half of the composition of this forest type. Another 25 percent of the trees comprised black cherry, white ash, and eastern hemlock species; elm, basswood, and white pine were minor components of that type group. The maple–beech–birch type primarily was found in southern New York, northern Pennsylvania, and at higher elevations in the western MAIA region (fig. 109).

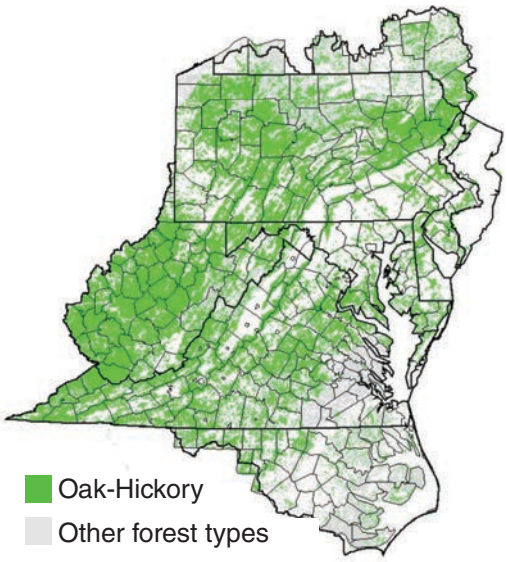


Figure 108—Oak-hickory forest-type distribution in the MAIA region. Source: USDA Forest Service, Forest Inventory and Analysis program's Eastwide Database; (http://www.srs.fs.fed.us/sustain/data/fia/eastwideguide.htm).

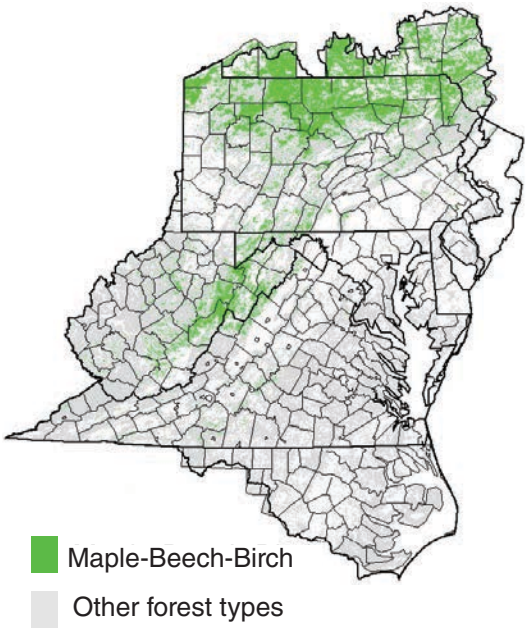


Figure 109—Maple-beech-birch forest-type distribution in the MAIA region. Source: USDA Forest Service, Forest Inventory and Analysis program's Eastwide Database; (http://www.srs.fs.fed.us/sustain/data/fia/eastwideguide.htm).

The other-conifer forest type (forest areas dominated by pines other than loblolly-shortleaf) covered 7 percent of the region and was dominated by hemlock, shortleaf pine, Virginia pine, white pine, red pine, red spruce, or balsam fir, either singly or in combination. This forest type included scattered areas of eastern hemlock and white pine in Maryland, New York, Pennsylvania, Virginia, and West Virginia; pitch pine in New Jersey; and some pond pine in the Piedmont area of Virginia and North Carolina (fig 110).

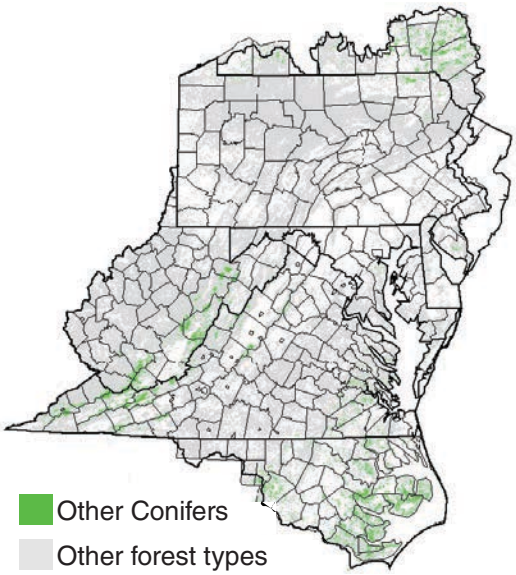


Figure 110—Other Conifer forest-type distribution in the MAIA region. Source: USDA Forest Service, Forest Inventory and Analysis program's Eastwide Database; (http://www.srs.fs.fed.us/sustain/data/fia/eastwideguide.htm).

Loblolly-shortleaf pine forest type covered 6 percent of the MAIA region's land base (fig. 106). This forest type occurred primarily in the Piedmont areas of Virginia and North Carolina (fig. 111). Almost half of the trees in this forest type were loblolly pine, and 16 percent were Virginia pine. Although historically included in the name of this forest type, shortleaf pine only represented 2 percent of this type group's composition. More abundant tree species included sweet gum, red maple, and red oak; hickory and black gum were minor components.

The other less common forest types had more localized distributions within the region. The oak–gum–cypress forest type accounted for only 2 percent of land area in the MAIA region (fig. 112), but provided critical bottomland hardwood habitat for many other tree species. This forest type occurred

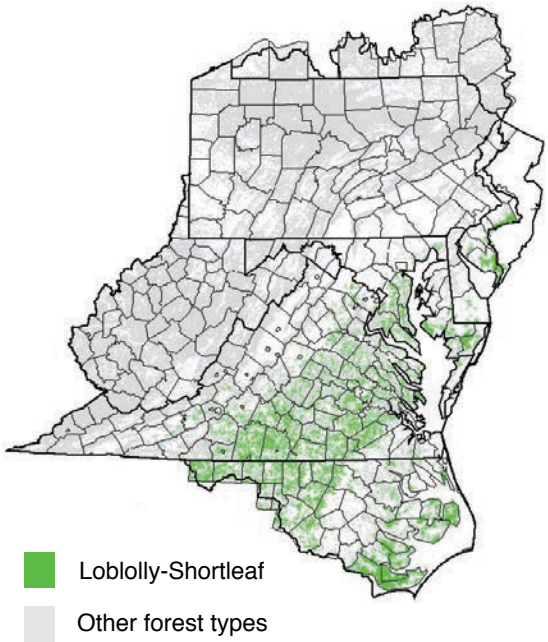


Figure 111—Loblolly-shortleaf pine forest-type distribution in the MAIA region. Source: USDA Forest Service, Forest Inventory and Analysis program's Eastwide Database; (http://www.srs.fs.fed.us/sustain/data/fia/eastwideguide.htm).

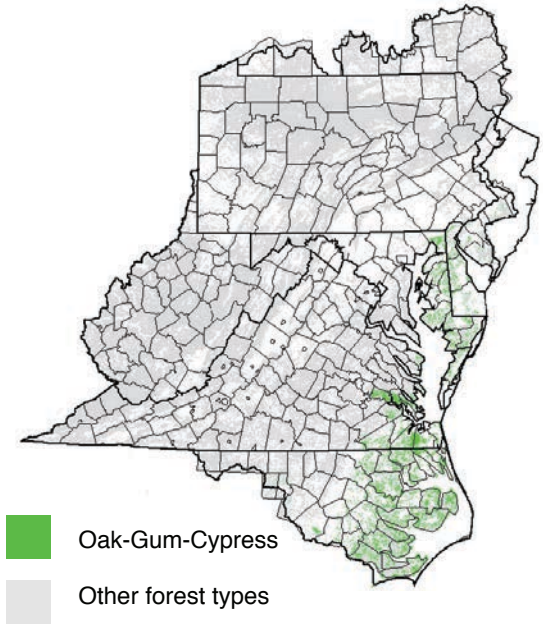


Figure 112—Oak-gum-cypress forest-type distribution in the MAIA region. Source: USDA Forest Service, Forest Inventory and Analysis program's Eastwide Database; (http://www.srs.fs.fed.us/sustain/data/fia/eastwideguide.htm).

only on poorly drained areas near the Atlantic coast, because wet-site species of white oaks, swamp tupelo, and bald cypress are best adapted to those conditions and therefore dominated this forest type in those areas.

The elm–ash forest type was relatively rare, occupied only 1 percent of the land area, but it also provides critical bottomland hardwood habitat for many other species. We found aspen–birch, the rarest type, in the high-elevation mountainous regions, where it occupied only about 0.5 percent of the region's land area.

Tree Genera

Similar to the number of forest types, the number (richness) of tree genera in a region also reflects plant diversity at a community level. Although Pennsylvania and Virginia had the most acreage of all forest types combined, and Virginia and Maryland contained six of the forest types found in the region (fig. 107), West Virginia had the largest area with the highest number of tree genera; a majority of western counties there had forests containing more than seven genera (fig. 113).

In south-central and western Pennsylvania a few counties also had more than seven tree genera. The reason for high richness of tree genera may be that both West Virginia and western Pennsylvania have mild, temperate growing climates with abundant rainfall; varied topography of plateaus and mountain ranges, and ecotonal areas where the natural ranges of many northern and southern species coincide. West Virginia and western Pennsylvania (with the exception of the Pittsburg area-fig. 8) had relatively little urban development, and were covered with highly contiguous forests (fig. 12) where forest land-use was the primary use (fig. 13). Thus tree genera richness was highest in areas of the MAIA region where physical factors supported a high number of genera, and human disturbance to forest ecosystems was lowest.

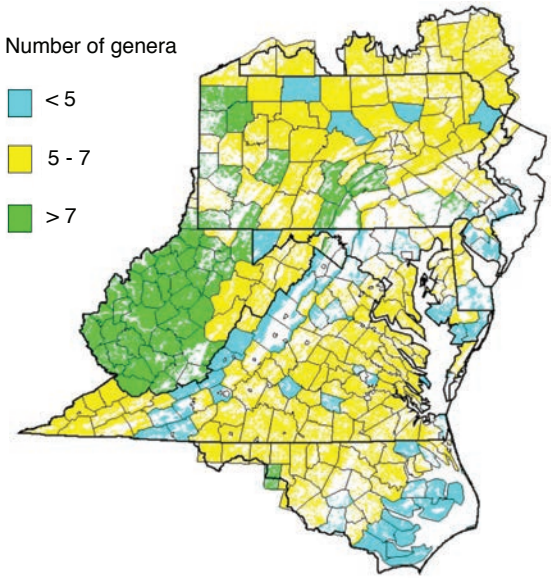


Figure 113—Distribution of tree genera in the MAIA region. Source: USDA Forest Service, Forest Inventory and Analysis program's Eastwide Database; (http://www.srs.fs.fed.us/sustain/data/fia/eastwideguide.htm).

Tree genera richness was lowest (fewer than five genera) (fig. 113) in coastal areas associated with oak–gum–cypress swamps (fig. 112), where fewer kinds of tree species are well adapted to wet sites. Drought-prone areas along the Ridge and Valley sections (ecoregion section M221A-Bailey 1995) in Pennsylvania that continue along the border of Virginia and West Virginia (fig. 5) also are areas where growing conditions limit the number of species and, therefore, tree genera adapted to poor site conditions. Such sites were dominated by drought-tolerant species such as chestnut oak and hickory. Although many forest stands in the Piedmont area of Virginia and North Carolina are loblolly pine plantations, five to seven genera of tree species typically were found within these forests and in much of the MAIA region. Most counties in the MAIA region often had five to seven genera, and only a few counties had fewer than five.

Chapter 22.

Structural Diversity

James Steinman
Northern Research Station, USDA Forest Service

Stand Size Class and Stand Age

The distribution of different sized trees in a forest is an indicator of structural diversity and maturity of that forest's ecosystem. Landscapes with forest stands of different ages are more likely to accommodate a wide complement of uses, and to be more resilient to a variety of stressors. Individual forest stands with different tree sizes also are more likely to provide multiple benefits such as wildlife habitat, human recreation, and other uses.

Following the widespread abandonment of agricultural lands in the early 1900s (when improved farming practices produced more food on less land), and reductions in the use of wood for fuels, fences, railroad ties, and other purposes, much of the MAIA region's forests reestablished on old fields.

About 55 percent of the region's forests now contain trees that are considered harvestable (where the average tree is > 5-inches d.b.h.) by regional timber-industry standards. Although these trees may be considered merchantable from a timber harvest perspective, from an ecological perspective they are still only maturing. Most counties with stands of merchantable, harvestable trees were in northern Pennsylvania, West Virginia, western Virginia, and parts of North Carolina (fig. 69). Twenty-nine percent of the region's forests were either immature (where the average trees were saplings > 1-inch but < 5-inches d.b.h.), and 16 percent were in regeneration (where average trees were seedlings < 1-inch d.b.h.) (fig. 114).

Distribution of stand sizes by State showed that Pennsylvania, Virginia, and West Virginia had the most forested acres in harvestable and immature classes (fig. 115). Virginia and Pennsylvania had the most acres of forest in the regeneration phase, and the forests of Maryland, Delaware, and New Jersey had the lowest acreage of forests in all three stand size classes. States in the MAIA region had more than 33,000,000 acres of forests in a harvestable state of development, but ecologically still maturing.

For the 20 most common forest tree species in the Mid-Atlantic region (table 29) we compared the distribution of number of trees among five diameter classes (1- to 5-inches;

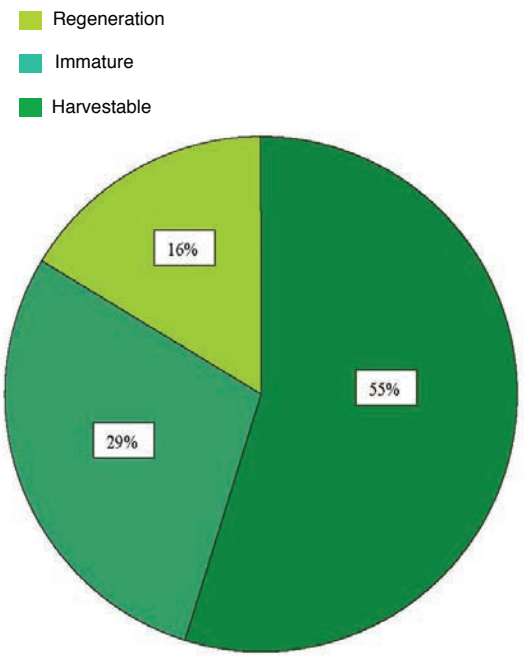


Figure 114—Forest stand-size class distribution in the MAIA region based on three commercial classes: harvestable (sawtimber-sized stands where average tree is greater than 5 inches d.b.h.); immature (pole-sized stands where average tree is a sapling less than 5 inches d.b.h.); and regeneration (stands where average tree sapling is 1- to 4.9 inches d.b.h.). Size classes reflect commercial standards for harvesting and not ecological maturity. Source: USDA Forest Service, Forest Inventory and Analysis program's Eastwide Database; (http://www.srs.fs.fed.us/sustain/data/fia/eastwideguide.htm).

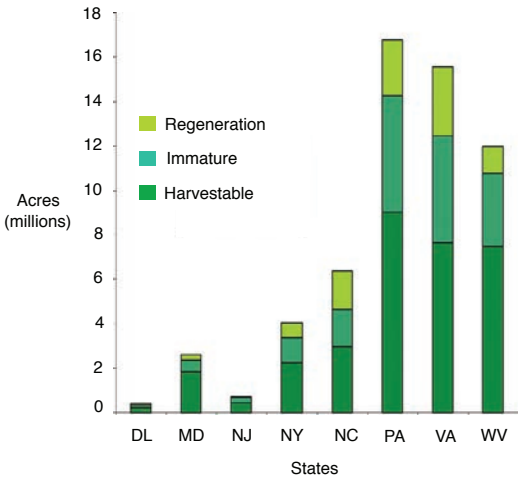


Figure 115—Forest stand-size class distribution by MAIA region states. Source: USDA Forest Service, Forest Inventory and Analysis program's Eastwide Database; (http://www.srs.fs.fed.us/sustain/data/fia/eastwideguide.htm).

5.1- to 10-inches; 10.1- to 15-inches; 15.1- to 20-inches; and > 20-inches d.b.h.), and the relative importance (rank) of each in each class. About 75 percent of all trees were in the 1- to 5-inch class, but accounted only for about 18 percent of the total biomass (fig 116).

Because smaller diameter trees have relatively little biomass (total weight of stems, branches, and foliage), this is to be expected. The most biomass (about 55 percent) was concentrated in the 5.1- to 10-inch and 10.1- to 15-inch classes. A uniform distribution of the five diameter classes we examined, and possibly more reflective of the stability of tree distribution in the MAIA region, would contain 20 percent of the total biomass in each class. In fact, however, total tree biomass in the MAIA region was found to be higher in trees 5.1- to 15-inches d.b.h. Trees > 20-inches d.b.h. represented only 10 percent of the total biomass (fig. 116).

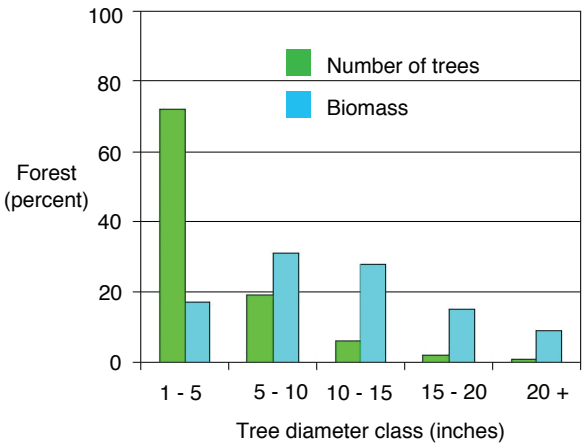


Figure 116—Distribution of numbers of trees and biomass in five size-classes in the MAIA region. Source: USDA Forest Service, Forest Inventory and Analysis program's Eastwide Database; (http://www.srs.fs.fed.us/sustain/data/fia/eastwideguide.htm).

The absence of very large trees and other old growth forest conditions, which were common before European settlement, indicated that forests in the MAIA region were still recovering from Colonial and subsequent deforestation, and were still maturing toward a more balanced distribution of biomass per size class. These forests, however, are no longer the young, reestablishing forests they were in the early 1900s. After a hundred years of growth under relatively undisturbed conditions, e.g., little harvest, urbanization, and fragmentation, some forests are en route to becoming the magnificent, cathedral-like forests found in some parts of the MAIA region in early Colonial times. It will likely take another 100 years or more to reach that maturity, but in the interim Americans will enjoy Eastern forests evolving toward a stature that has been missing for 300 years or more.

The spatial distribution of this structural diversity within counties in the MAIA region, based on the average of the five size classes in ranges of averages, shown in figure116, is given in fig. 117.

The highest size class average (greater than 4.2 classes or 4.2 to 5.0 classes) was found mostly in counties of West Virginia, northern and south-central Pennsylvania, and south-eastern New York. Most forest stands in the MAIA region contained an average of about three size classes (3.6 to 4.2 classes) of trees (fig. 117). The Piedmont area of Virginia and North Carolina typically contained forest stands with fewer size classes of trees (fewer than 3.6 classes).The even-aged structures of loblolly pine plantations and natural stands in oak–gum–cypress swamps in these areas contain fewer size classes of trees than in other parts of the MAIA region.

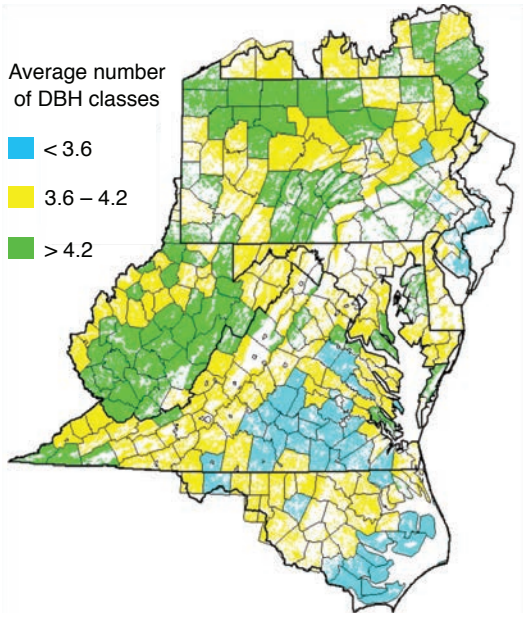


Figure 117—Three classes of tree size distribution based on the average of five d.b.h. size classes (0 to 5; 5.1 to 10; 10.1 to 15; 15.1 to 20; and greater than 20 inches) in the MAIA region. Source: USDA Forest Service, Forest Inventory and Analysis program's Eastwide Database; (http://www.srs.fs.fed.us/sustain/data/fia/eastwideguide.htm).

Chapter 23.

Tree and Lichen Diversity

William D. Smith and Kenneth W. Stolte
Southern Research Station, USDA Forest Service

Species diversity can be expressed in indices typically constructed from combinations of the number of species and relative abundance of each. Number refers to number of individual taxa at the species level, while number of individuals, size, cover, or other attributes based on number or size of individuals are used to quantify abundance. While there are numerous indices to evaluate the combination of number and abundance, two basic indices are species richness (number of species) and species abundance (number or size of individuals within a species).

Species composition and abundance change spatially with climate, soils, topography, and stand disturbance—and change temporarily as stands develop and progress naturally through seral stages. Natural differences in plant diversity caused by varying abiotic and biotic factors within ecological units are expected, so comparisons of diversity only within individual stands or ecological units over time can be indicative of forest health and sustainability. We know plant diversity is greater in warm and wet areas with long growing seasons, and less in cooler or drier areas with short growing seasons.

Tree species

Tree species diversity is an attribute important for recreational uses, wildlife habitat, aesthetic values, and timber and non-timber commodities.

Mid-Atlantic forests had more tree species than most other regions in the country. Over 60 hardwood and softwood species were found in the MAIA forests, and 25 of those each accounted for ≥ 1 percent of all trees. High species diversity over the whole region's landscape was apparent due to the common occurrence of most of these 25 species in many areas. There were more than 15 species in some stands within the region, representing more than the seven genera shown in figure 113.

The number and mixture of different tree species within forest stands showed the high diversity within some forest types. Total tree species richness was highest, from east to west, in the Middle Atlantic Coastal Plain (232A) and Atlantic Coastal Flatlands (232C); Southern Appalachian Piedmont (231A); Blue Ridge Mountains (M221D); and the Northern Ridge and Valley (M221A) ecoregion sections (Bailey 1995).

Red maple accounted for 12 percent of all trees in the MAIA region, and was the most abundant and widely occurring tree species in the forest overstory and understory because it was the most common tree species across most size classes (1- to > 15-inches d.b.h.). It was the most common species in smaller size classes (1- to 5-inch and 5.1- to 10-inches d.b.h.), and was also very common in the larger size classes (10- to 15-inch and > 15 inch d.b.h.) (table 29).

This probably was due to the high regeneration capability and strong competitive nature of red maple, especially in areas of disturbance. Red and white oaks were the second and third most common across all size classes, respectively, and red oaks were the most common in the largest size classes (10.1- to 15-inches d.b.h. and > 15-inches d.b.h.). American dogwood was the least common species across all size classes, and uncommon in larger size classes—expected, because the species is a small, understory tree. The broad representation across all size classes for all species indicates great species and structural diversity within the forests of the MAIA region.

The red oak group (mostly black oak, northern red oak, and scarlet oak) was the second most abundant tree species in the region, comprising another 12 percent of all forest trees. Red oaks also were the most abundant tree species > 10-inches d.b.h. (table 29). Red oaks occurred most often in oak–hickory and oak–pine forest types, and thus were distributed widely throughout the region.

White oak, yellow-poplar, and chestnut oak were the next most abundant species in the region, each representing 5 to 6 percent of all forest trees (table 29).These species occurred throughout the region. Chestnut oak and yellow-poplar were more common among trees > 15 inches d.b.h., but were less common among saplings 1- to 5-inches d.b.h.

Loblolly pine was the next most frequently occurring tree species in the MAIA region, and represented about 7 percent of all forest trees. Unlike the red maple and red oak groups, loblolly pine did not occur throughout the region but was most abundant in the Piedmont area of Virginia and North Carolina (fig. 111), often a result of prominence in

Table 29—Relative importance (rank) of the twenty most common tree species by abundance (number of trees) in five d.b.h. size class circa 2000. Column 1 (1 to 15+ inches d.b.h.) determined the overall rank of each species. Columns 2 to 5 are the ranks of species in other d.b.h. classes

	Diameter at breast height (inches)				
Species	1 - 15+	1 - 5	5 -10	10 -15	15+
	species rank				
Red maple	1	1	1	2	3
Red oaks[a]	2	3	2	1	1
White oak	3	9	4	3	4
Yellow-poplar	4	10	7	5	2
Chestnut oak	5	16	5	4	5
Loblolly pine	6	8	3	6	6
Sugar maple	7	2	6	7	8
Sweetgum	8	4	10	10	15
American beech	9	7	14	12	7
Black cherry	10	11	13	9	9
Blackgum	11	5	12	11	10
Hickory	12	12	8	8	13
Ash	13	14	15	13	14
Birch	14	13	9	15	16
Virginia pine	15	17	11	14	20
Hemlock	16	18	16	16	12
Black locust	17	19	17	17	17
White pine	18	22	20	18	11
Sourwood	19	15	18	27	29
Dogwood	20	6	21	32	32

[a]Mostly black oak, northern red oak, and scarlet oak
Source: USFS Forest Inventory and Analysis program; (http://www.fia.fs.fed.us).

plantations managed for timber production. Most loblolly pines were < 10-inches d.b.h. (table 29) because of the relatively high number of newly established or harvested/replanted plantations.

Sugar maple, sweetgum, American beech, and black cherry also ranked among the top ten most abundant species (table 29). Most of those species were common throughout the MAIA region, except sugar maple, which was more concentrated in maple–beech–birch forest types of cooler climates (fig. 109). In addition, sweet gum, black gum, and dogwood–most often found in the forest understory—were very common among saplings (1- to 5-inches d.b.h.).

Lichens Species

Lichens are a unique and diverse group of non-vascular plants found on many substrates, including soil, rocks, and tree branches and boles. They are a symbiosis of fungal and algal species: the fungi supply structural support, protection, and water absorption; and the algae conduct photosynthesis and supply nutrition (Stolte and others 1993). Lichens constitute an important component of a forest ecosystem, serving as fixers of atmospheric nitrogen, as food sources for a variety of animals, homes for microinvertebrates, substrates for germinating epiphytes, and as an aesthetic-visual and craft-making recreational resource. Because lichens lack an epidermis, cuticle, and stomata, they cannot control gas exchange with the atmosphere and thus are very susceptible to many air pollutants. Sulfur and nitrogen oxides, hydrogen fluoride, and metal and organic toxins are particularly harmful. Some lichen species are also susceptible to high levels of ozone, wet and dry nitrates and ammonium, and other phytotoxic substances.

Data collected on lichen communities on FHM and FIA plots are typically analyzed in two ways: (1) direct enumeration of lichen species richness at the plot-level—as a component of biological diversity, and (2) lichen community composition helps us determine air quality and climatic influences at each plot (USDA Forest Service 1995).The latter analyses required additional research into the species composition within lichen communities along known air quality and climate gradients within defined large ecological strata, such as the entire MAIA region.

While we have not yet computed air quality and climatic values for lichens in the MAIA region, patterns of species richness of lichen communities within the MAIA could suggest a link with air quality in at least part of the region. Lichen species richness in the region varied from relatively high (18 to 32 species per plot) in the southwest part of the region, to relatively low (2 to 7 and 8 to 12 species per plot) in the northern and eastern parts of the region, respectively (fig. 118). Northern sections of the MAIA region have been known to receive relatively high regional wet deposition of sulfates (fig. 18), nitrates (fig. 19), and acidic precipitation (fig. 20), and are also areas where forest cover is relatively high (figs. 12 and 13).

The geographic overlap of higher pollution deposition and lower lichen diversity in the MAIA region suggest that pollution eventually may be shown to be a causal factor of lower diversity. Investigation of reasons for the differences in species richness, particularly between the northern and southern parts of the Southern Unglaciated Allegheny Plateau (221E), Allegheny Mountains (M221B), and Northern Ridge and Valley (M221A) ecoregion sections (Bailey 1995) would be good subjects for more intensive studies. Additional analyses of lichen species distributions as they relate to stand density, disturbances, air quality, climatic gradients, and other factors in these ecoregion sections are warranted.

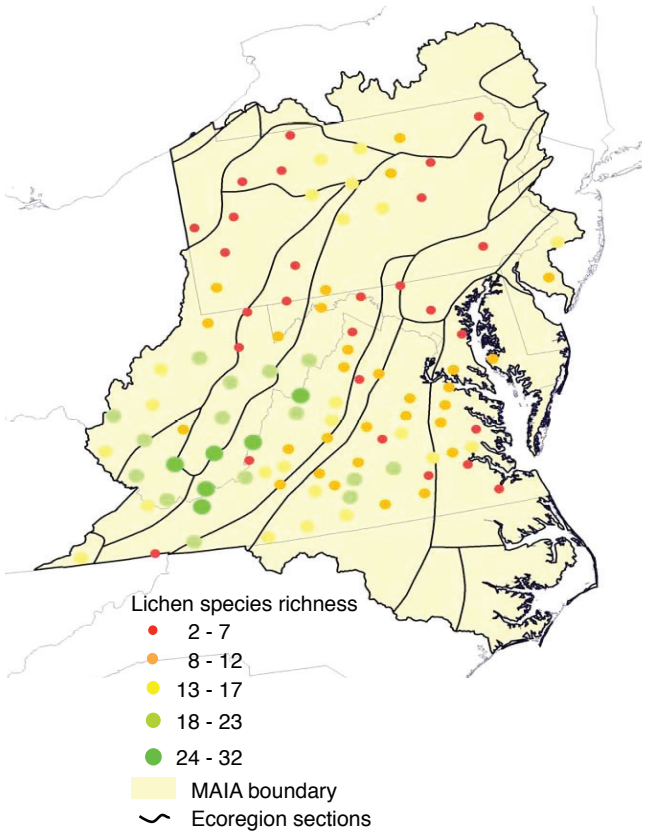


Figure 118—Lichen species richness in the MAIA region. Source: National Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

Chapter 24.

Bird Diversity

Emmanue Cam and Laura E. Jackson
Laboratoire Evolution et Diversité Biologique, Université P. Sabatier, Toulouse, France; Office of Research and Development, Research Triangle Park Laboratory, U.S. Environmental Protection Agency

The North American Breeding Bird Survey (BBS) is a large-scale, long-term national and international avian monitoring program to track the status and trends of North American bird populations. It is coordinated by the USGS Patuxent Wildlife Research Center and the Canadian Wildlife Service's National Wildlife Research Center (<http://www.pwrc.usgs.gov/BBS/about>). The primary purpose is to monitor and report on significant changes in bird populations so that causal factors can be identified and remedial actions can take place before populations reach critically low levels.

The BBS program was initially started over concerns about the effects of DDT and other pesticides on bird reproduction. It continues to be strongly supported because bird populations continue to be subjected to numerous and widespread threats, including habitat loss, habitat fragmentation, land-use changes, other chemical contaminants, exotic species, and other stressors. Each summer during bird-breeding months (e.g., June) trained participants conduct roadside surveys (each 24.5 miles long) with stops at 0.5-mile intervals and 3-minute point counts. During the counts, every bird seen or heard within a 0.25-mile radius is recorded. Surveys start one-half hour before local sunrise and take about 5 hours to complete. Over 4,100 survey routes are located across the continental U.S. and Canada. BBS data provide an index of population abundance that can be used to estimate population trends and relative abundances at various geographic scales.Trend estimates for more than 420 bird species and all raw data are currently available on the BBS web site.

In this report we analyzed BBS data on forest birds (table 30) across EPA Region III (encompassing most of the MAIA region) from 1975 and 1990 to determine if changes in species richness had occurred. This analyses was on BBS routes in HUC 8 watershed cataloging units in the MAIA region (fig. 119) (< http://water.usgs.gov/GIS/huc.html>).

In 1975, average species richness was relatively low (8.5 to 28.6 species) in much of the MAIA region, particularly the northern areas (fig. 120). Highest average species richness in 1975 (28.6 to 38.7 species) primarily occurred in West Virginia, southern coastal areas, and a few other scattered areas. The lowest average species richness that year (8.5 to 18.5 species) was found mostly in watersheds of eastern Pennsylvania.

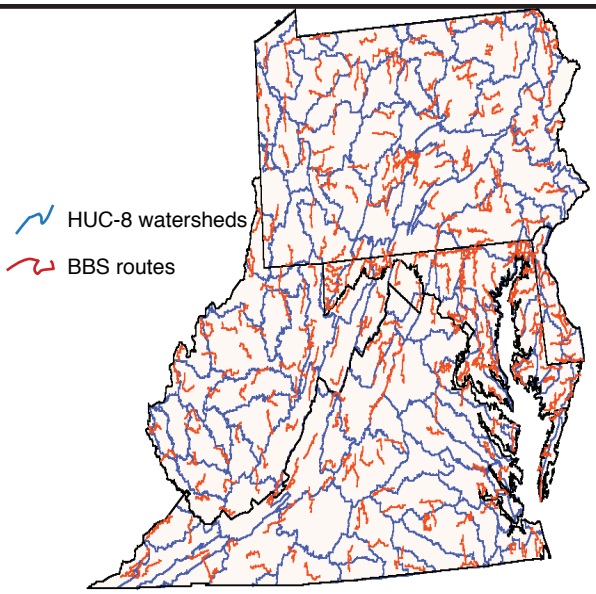


Figure 119—HUC-8 watersheds and Breeding Bird Survey routes in the MAIA region. Source: Seaber and others 1997; Sauer and others 1999; (http://www.mbr-pwrc.usgs.gov/bbs/).

In 1990 the average number of bird species in many MAIA watersheds had increased from 8.5 to 28.6 to 28.7 to 38.6 (fig. 121). The average bird species richness in two northern watersheds in Pennsylvania this same year increased from 18.6 to —more species of birds than were found anywhere in the MAIA region in 1975 or 1990. The lowest numbers of bird species, both in 1975 and in 1990, occurred primarily in eastern watersheds above the Chesapeake and Delaware Bays (figs. 120 and 121).

Because this analysis did not differentiate native from exotic species, we cannot say with any certainty whether observed increases in the number of bird species was due to increases in native bird species as a result of habitat improvement marked by the maturity of forests in these areas, or to an influx of exotic, invasive bird species.

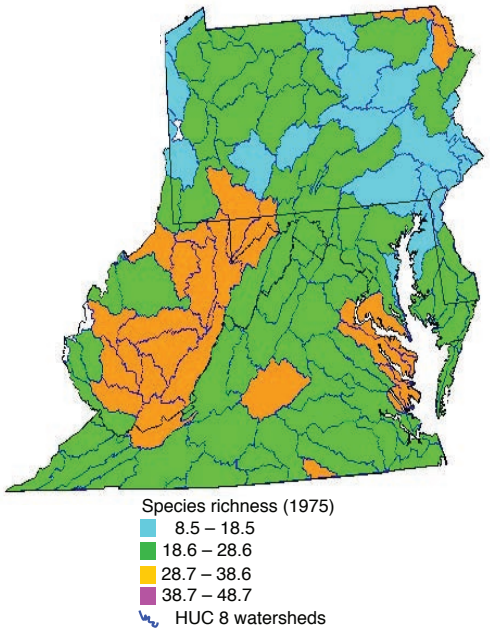


Figure 120—Species richness of forest birds in HUC-8 MAIA region watersheds in 1975 based on Breeding Bird Survey data. Source: Seaber and others 1987; Sauer and others 1999; (http://www.mbr-pwrc.usgs.gov/bbs/).

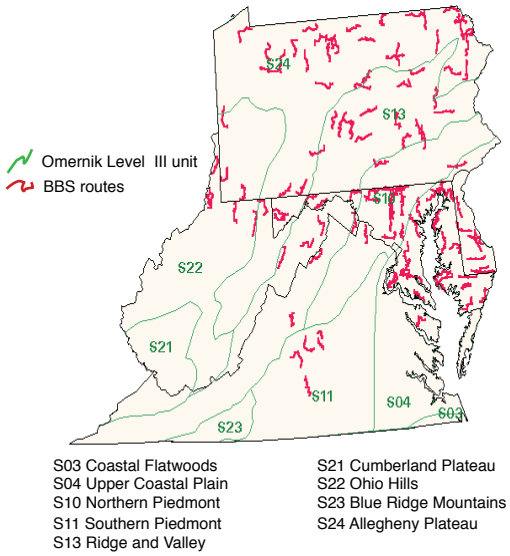


Figure 122—Physiographic strata of Omernik Level III ecological units and Breeding Bird Survey routes retained for estimating relative species richness in the MAIA region. Source: (http://nationalatlas.gov/mld/ecoomrp.html); (http://www.mbr-pwrc.usgs.gov/bbs/).

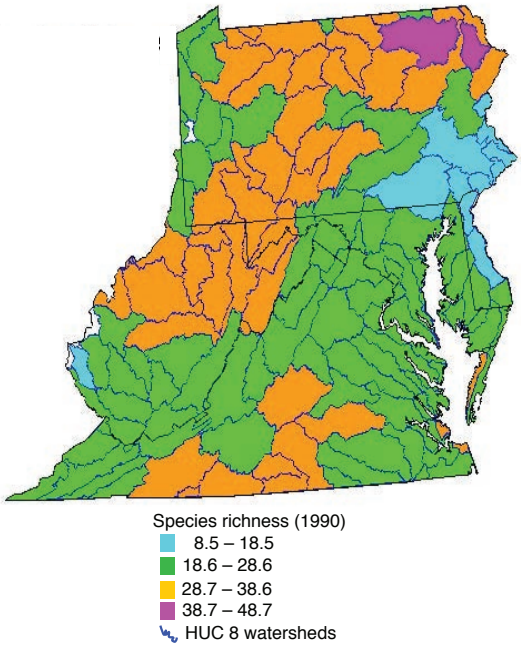


Figure 121—Species richness of forest birds in HUC-8 MAIA region watersheds in 1990 based on Breeding Bird Survey data. Source: Seaber and others 1987; Sauer and others 1999; (http://www.mbr-pwrc.usgs.gov/bbs/).

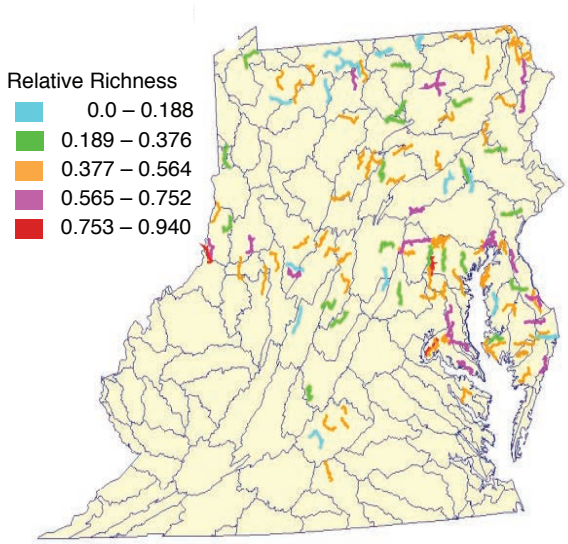


Figure 123—Relative species richness of forest birds on Breeding Bird Survey routes in HUC-8 MAIA region watersheds in 1992. Source: Seaber and others 1987; Sauer and others 1999; (http://www.mbr-pwrc.usgs.gov/bbs/).

Relative Bird Species Richness

Although species richness characterizes forest bird communities, as well as other biota, it must again be emphasized that there is no threshold for what are acceptable or desirable values for this metric. For this reason, we applied another metric for *relative richness* that basically evaluates the stability of bird communities over time. However, as with the first metric, our relative species richness evaluation did not differentiate between native and exotic bird species.

Computation of species *relative richness* required defining an historical-reference species list (table 30) and estimating the number of species in that list that were present on each target BBS route within distinct ecological strata (fig. 122) based on Omernik's Level III classification (<http://www.nationalatlas.gov/mld/ecoomrp.html>) in the 1992 assessment year (Cam and others 2000b) (Technical Appendix D).The ratio of those numbers (reference species list/observed species list from BBS surveys) corresponds to relative richness (0 to 1 ratio scale—higher numbers indicate higher relative richness, which is most desirable).

The ratio of species present on a BBS route in the past compared to species found in the same BBS surveys route for any year corresponds to *relative richness*. Bird species detected on the same specific route of interest in any year of interest that was not detected in past BBS evaluations on this same route were excluded from the analysis in order to ensure statistical independence of the numerator and denominator. We used the reference list (table 30) of all bird species detected in all years of Breeding Bird Surveys within about an 80-km radius centered on the route of interest and within the physiographic stratum in each State where the route was located (Bystrak 1981; fig. 122). Physiographic strata were spatial units harboring relatively homogeneous natural communities that were expected to have generally comparable bird communities. Because this metric is sensitive to the number of BBS routes in each strata used to develop the species pool (Cam and others 2000b), we only analyzed data from physiographic strata that contained a minimum of five BBS routes. This approached helped to avoid analytical problems linked to small sample sizes. However, this resulted in some physiographic areas not being included in analysis and mapping at watershed scales; that is, data were too sparse to allow interpolation in several watersheds.

Neither the data collected, nor any mapping of relative species richness from those data (fig. 122) indicated strong spatial patterns at survey-route scales. Nonetheless, we did find the highest relative richness values (0.55 to 0.94) on many survey routes in watersheds near the Chesapeake Bay, parts of northern Pennsylvania, and parts of western West Virginia. Some of the lowest relative richness values (0 to 0.19 and 0.19 to 0.38) were also found on BBS routes in northern Pennsylvania (fig. 123), an area where a relatively large increase in bird species is reflected in data collected from 1975 to 1990 (figs. 120 and 121). We have concluded, however, that there were no apparent relationships between increases in bird species richness from 1975 to 1990 and relative species richness evaluated in 1992.

Table 30—Forest bird species found at five or more Breeding Bird Survey routes in EPA Region III circa 2000

AOU[a]	Common name	AOU[a]	Common name
2970	Blue grouse	5708	Gray-headed junco
3000	Ruffed grouse	5750	Bachman sparrow
3100	Wild turkey	5950	Rose-breasted grosbeak
3120	Band-tailed pigeon	5960	Black-headed grosbeak
3270	Swallow-tailed kite	6070	Western tanager
3320	Sharp-shinned hawk	6080	Scarlet tanager
3330	Cooper's hawk	6090	Hepatic tanager
3340	Northern goshawk	6100	Summer tanager
3390	Red-shouldered hawk	6240	Red-eyed vireo
3430	Broad-winged hawk	6260	Philadelphia vireo
3570	Merlin	6270	Warbling vireo
3680	Barred owl	6280	Yellow-throated vireo
3730	Eastern screech-owl	6290	Solitary vireo
3732	Western screech-owl	6320	Hutton vireo
3790	Northern pygmy-owl	6360	Black-and-white warbler
3870	Yellow-billed cuckoo	6370	Protonotary warbler
3880	Black-billed cuckoo	6380	Swainson warbler
3930	Hairy woodpecker	6390	Worm-eating warbler
3940	Downy woodpecker	6470	Tennessee warbler
3950	Red-cockaded woodpecker	6480	Northern parula
3960	Ladder-backed woodpecker	6500	Cape May warbler
3970	Nuttall woodpecker	6540	Black-throated blue warbler
3990	White-headed woodpecker	6550	Myrtle warbler
4000	Black-backed woodpecker	6560	Audubon warbler
4010	Three-toed woodpecker	6570	Magnolia warbler
4020	Yellow-bellied sapsucker	6580	Cerulean warbler
4021	Red-naped sapsucker	6600	Bay-breasted warbler
4030	Red-breasted sapsucker	6610	Blackpoll warbler
4040	Williamson sapsucker	6620	Blackburnian warbler
4050	Pileated woodpecker	6630	Yellow-throated warbler
4070	Acorn woodpecker	6640	Grace warbler
4090	Red-bellied woodpecker	6670	Black-throated green warbler
4100	Golden-fronted woodpecker	6680	Townsend warbler
4160	Chuck-will-widow	6690	Hermit warbler
4170	Whip-poor-will	6710	Pine warbler
4240	Vaux swift	6740	Ovenbird
4280	Ruby-throated hummingbird	6750	Northern waterthrush
4290	Black-chinned hummingbird	6760	Louisiana waterthrush
4320	Broad-tailed hummingbird	6770	Kentucky warbler
4330	Rufous hummingbird	6840	Hooded warbler
4360	Calliope hummingbird	6860	Canada warbler
4520	Great Crested flycatcher	6870	American redstart
4530	Brown-crested flycatcher	7220	Winter wren
4590	Olive-sided flycatcher	7260	Brown creeper
4610	Eastern wood-pewee	7270	White-breasted nuthatch
4620	Western wood-pewee	7280	Red-breasted nuthatch
4630	Yellow-bellied flycatcher	7290	Brown-headed nuthatch
4640	Cordilleran flycatcher	7300	Pygmy nuthatch
4641	Pacific-slope flycatcher	7310	Tufted titmouse
4650	Acadian flycatcher	7320	Black-crested Titmouse
4670	Least flycatcher	7330	Plain titmouse
4680	Hammond's flycatcher	7350	Black-capped chickadee
4690	Dusky flycatcher	7360	Carolina chickadee
4710	Vermilion flycatcher	7380	Mountain chickadee
4780	Steller jay	7400	Boreal chickadee

[a]American Ornithologists' Union numbering system for avian taxa (http://www.aou.org/checklist/north/index.php).
Note: North American State bird list was baseline to compute relative species richness of birds in MAIA region.

INFORMATION SOURCES

LITERATURE CITED

Adams, T.O.; Hook, D.D.; Floyd, M.A. 1995. Effectiveness monitoring of silvicultural best management practices in South Carolina. Southern Journal of Applied Forestry, Vol. 19(4): 170-176

American Forests. 1999. Regional Ecosystem Analysis Chesapeake Bay Region and the Baltimore-Washington Corridor. Final Report 3/22/99 American Forests, Washington, DC.

Anagnostakis, S.L. 1995. The pathogens and pests of chestnuts. In: Advances in Botanical Research, J.H. Andrews and I. Tommerup, eds., Academic Press, New York. Vol. 21: 125-145.

Anderson, J.R.; Hardy, E.E.; Roach, J.T.; Witmer, R.E. 1976. A land use and land cover classification system for use with remote sensor data. U.S. Geological Survey Professional Paper 964.

Anonymous (Anon.). 1995a. Criteria and indicators for the conservation and sustainable management of temperate and boreal forests. In: Funston, M., USDA Forest Service (contact), Santiago Declaration [Web page]. http://www.fs.fed.us/global/pub/links/santiago.htm.

Anonymous (Anon.). 1995b. Sustaining the world's forests: the Santiago agreement. Journal of Forestry. 93(4): 18-21.

Aubertin, G.M.; Patric, J.H. 1975. Comments on water quality after clearcutting a small watershed in West Virginia. Journal of Environmental Quality. 4(2): 282-283.

Bailey, R.G. 1995. Description of the ecoregions of the United States. 2nd ed., rev. and expanded. Misc. Publ. 1391. Washington, DC: U.S. Department of Agriculture. 108 p. [With separate map at 1:7,500,000].

Bechtold, W.; LaBau, V.J.; McLain, W. [and others]. 1995. Forest mensuration. 1995 version. In: Conkling, B.L.; Tallent-Halsell, N.G., eds. Forest health monitoring field methods guide. EPA/620/R-94/027. Washington, DC: U.S. Environmental Protection Agency. 371 p. Chapter 1.

Beers, T.W. 1962. Components of forest growth, J. For. 60(4): 245-248.

Binkley, D.; Brown, T.C. 1993. Forest Practices as Nonpoint Sources of Pollution in North America. Water Resources Bulletin, Vol. 29(8): 729-740.

Birch, T.W. 1996a. Private Forest-land Owners of the United States, 1994. Resource Bulletin NE-134. Radnor, PA: USDA Forest Service, Northeastern Forest Experiment Station. 183 p.

Birch, T.W. 1996b. Private Forest-land Owners of the Southern United States, 1994. Resour. Bull. NE-138. Radnor, PA: USDA Forest Service, Northeastern Forest Experiment Station. 293 p.

Birch, T.W. 1996c. Private Forest-land Owners of the Northern United States, 1994. Resour. Bull. NE-136. Radnor, PA: USDA Forest Service, Northeastern Forest Experiment Station. 293 p.

Birdsey, R.A. 1996. Carbon storage for major forest types and regions in the conterminous United States. In: Sampson, N.R.; Hair, D., eds. Forests and global change (vol. 2). Washington, DC. American Forests: 1-25.

Blake, J.G. 1983. Trophic structure of bird communities in forest patches in east-central Illinois, Wilson Bulletin 95: 416-430.

Bormann, F.H.; Likens, G.E.; Siccama, T.G. [and others]. 1974. The export of nutrients and recovery of stable conditions following deforestation at Hubbard Brook. Ecol. Monograph. 44: 255-277.

Boulinier, T.; Nichols, J.D.; Sauer, J.R. [and others]. 1998a. Estimating species richness to make inferences in community ecology: the importance of heterogeneity in species detectability as shown from capture-recapture analyses of North American Breeding Bird Survey data. Ecology. 79: 1018-1028.

Boulinier T.; Nichols, J.D.; Hines, J.E. [and others]. 1998b. Higher temporal variability of forest breeding bird communities in fragmented landscapes. Proc. Natl. Acad. Sci. USA. 95: 7497-7501.

Brady, N.C. 1984. The nature and properties of soils, 9th edition. New York: Macmillan Publishing Company. 750 p.

Braun, E.L. 1950. Deciduous forests of eastern North America. Hafner, New York. 596 p.

Brauning, D.W. (ed.) 1992. Atlas of breeding birds in Pennsylvania. University of Pittsburgh Press, Pittsburgh, PA.

Bren, L.J. 1993. Riparian zone, stream, and floodplain issues: a review. Journal of Hydrology. 150 (2/4): 277-299.

Britton, K.O.; Berrang, P.; Mavity, E. 1996. Effects of pretreatment with simulated acid rain on the severity of dogwood anthracnose. Plant Dis. 80(6): 636-649.

Brooks, R.P.; Cole, C.A.; Wardrop, D.H. [and others]. 1996. Wetlands, wildlife, and watershed assessment techniques for evaluation and restoration: Final report for the project: Evaluating and implementing watershed approaches for protecting Pennsylvania's wetlands. Report No. 96-2, Penn State Cooperative Wetlands Center, University Park, PA. Volumes I and II.

Brooks, R.P.; Croonquist, M.J. 1990. Wetland, habitat, and trophic response guilds for wildlife species in Pennsylvania. Journal of the Pennsylvania Academy of Science. 64: 93-102.

Brush, G.S. 1984a. Stratigraphic evidence of eutrophication in an estuary. Water Resources Research. 20(5): 531-541.

Brush, G.S. 1984b. Patterns of recent sediment accumulation in Chesapeake Bay (Virginia-Maryland, USA) tributaries. In: Robbins, J.A., guest-editor. Geotechnology of recent deposits. Chemical Geology. 44: 227-242.

Brush, G.S. 1986. Geology and paleoecology of Chesapeake Bay: a long-term monitoring tool for management. Journal of the Washington Academy of Sciences. 76(3): 146-160.

Bull, K.A., Stolte, K.W., and T.J. Stohlgren. 1998. Forest Health Monitoring: Vegetation Pilot Field-Method Guide. Vegetation Diversity and Structure, Down Woody Debris, Fuel Loading. USDA Forest Service, Washington, DC. 53 p.

Burkman, W.G.; Bechtold, W.A. 2000. Has Virginia pine declined? The use of Forest Health Monitoring and other information in the determination. In: Integrated tools for natural resources inventories in the 21st century. USDA Forest Service North Central Research Station GTR (NC-212, Mark Hansen and Tom Burk, eds.).

Burkman, W.; Schomaker, M.; Stolte, K.W. [and others]. 1995. Chapter 2. Crown condition classification. 1995 version. In: Conkling, B.L.; Tallent-Halsell, N.G., eds. Forest Health Monitoring field methods guide. EPA/620/R-94/027. Washington, DC: U.S. Environmental Protection Agency. 371 p.

Burnham, K.P.; Overton, W.S. 1978. Estimation of the size of a closed population when capture probabilities vary among animals. Biometrika. 65: 625-633.

Burnham, K.P.; Overton, W.S. 1979. Robust estimation of population size when capture probabilities vary among animals. Ecology. 60: 927-936.

Burns, R.M.; Honkala, B.H. 1990. Silvics of North America. Agricultural Handbook 654. Washington, DC: U.S. Department of Agriculture. Vol. 1. 675 p.

Bystrak, D. 1981. The North American Breeding Bird Survey. In C.J. Ralph and M.J. Scott (eds.). Estimating numbers of terrestrial birds. Studies in Avian Biology 6, Cooper Ornithological Society. Lawrence, KS: pp. 34-41.

Cam, E.; Nichols, J.D.; Sauer, J.R. [and others]. 2000a. Relative species richness and community completeness: avian communities and urbanization in the Mid-Atlantic States. Ecological Applications. 10: 1196-1210.

Cam, E.; Sauer, J.R.; Nichols, J.D. [and others]. 2000b. Geographic analysis of species richness and community attributes of forest birds from survey data in the Mid-Atlantic Integrated Assessment region. Environmental Monitoring and Assessment. 63: 81-94.

Christensen, R. 1991. Linear models for multivariate time-series and spatial data. Springer-Verlag, New York. 371 p.

Cleland, D.T.; Avers, P.E.; McNab, W.H. [and others]. 1997. National Hierarchical Framework of Ecological Units. Published in, Boyce, M.S.; Haney, A., eds. 1997. Ecosystem Management Applications for Sustainable Forest and Wildlife Resources. Yale University Press, New Haven, CT: pp. 181-200.

Community Resources. 1999. Urban Non-timber Forest Products. (Internet www.community resources.org/ntfp.htm).

Conkling, B.L.; Coulston, J.W.; Ambrose, M.J. (eds.). 2005. Forest Health Monitoring: 2001 National Technical Report. Gen. Tech. Rep. SRS-81. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 204 p.

Conner, E.F.; Simberloff. 1978. Species number and compositional similarity of the Galapagos flora and Avifauna. Ecological Monographs. 48: 219-248.

Cooper, S.R. 1995. Chesapeake Bay watershed historical land use: impact on water quality and diatom communities. Ecological Applications. 5(3): 703-723.

Corbett, E.; Lynch, J.; Sopper, W. 1978. Timber harvesting practices and water quality in the Eastern United States. Journal of Forestry. 76: 484-488.

Crane, R.G.; Hewitson, B.C. 1998. Doubled CO_2 precipitation change for the Susquehanna basin: down-scaling from the GENESIS general circulation model. International Journal of Climatology. 18: 65-76.

Cumming, A.B.; Galvin, M.; Rabaglia, R. [and others]. 2001. Forest Health Monitoring protocol applied to roadside trees in Maryland. Journal of Arboriculture. 27(3): 126-437.

Dahlgren, R.A.; Driscoll, C.T. 1994. The effects of whole-tree clear-cutting on soil processes at the Hubbard Brook Experimental Forest, New Hampshire, USA. Plant Soil. Vol. 158(2): 239-262.

Davenport, E.L., 1990. Industrial Roundwood Products Output, 1987-1990. U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station. Resource Bulletin SE-132. 22 p.

DC Environmental Agenda 1999. http://www.foe.org/camps/reg/dcen/envagenda.pdf. Last accessed 1999.

DeGraaf, R.M.; Tilghman, N.G.; Anderson, S.H. 1985. Foraging guilds of North American birds. Environmental Management 9: 493-536.

Dietterick, B.C.; Lynch, J.A. 1989. Cumulative hydrologic effects on stormflows of successive clearcuts on a small headwater basin. In Proceedings of the Symposium on Headwaters Hydrology, American Water Resources Association: pp. 473-485.

Doran, J.W.; Coleman, D.C.; Bezdicek, D.F.; Stewart, B.A., eds. 1994. Defining soil quality for a sustainable environment. Soil Science Society of America Special Publication Number 35. Madison, WI: Soil Science Society of America, Inc. and American Society of Agronomy, Inc. 244 p.

Dwyer, J.F.; Nowak, D.J.; Noble, M.H.; Sisinni, S.M. 2000. In: Assessing our Nations' urban forests: Connecting people with ecosystems in the 21st century. General Technical Report (2000), USDA Forest Service. 488 p.

Edgmand, M.; Moowaw, R.; Olson, K. 1996. Economics and contemporary issues. 3rd ed. [location unknown]: Dryden Press. (Also http://www.nd.edu/~cwilber/econ504/504book/outln11c.html).

Edwards, P.J.; Rochenderfer, J.N.; Seegrist, D.W. 1991. Effects of forest fertilization on stream water chemistry in the Appalachians. Water Resource Bulletin. Vol. 27(2): 265-274.

Environmental Protection Agency. 2008. 1990 Clean Air Act Amendments. http://www.epa.gov/air/caa//.

Eyre, F.H., ed. 1980. Forest cover types of the United States and Canada. Washington, DC: Society of American Foresters. 148 p. [905].

Fire Science Laboratory. 1999a. Current condition classes v1.0. Missoula, Montana: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Science Laboratory. Unpublished database. On file with Fire Science Laboratory.

Fire Science Laboratory. 1999b. Historic natural fire regimes v3.0. Missoula, Montana: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Science Laboratory. Unpublished database. On file with Fire Science Laboratory.

Forman, R.T.T.; Godron, M. 1986. Landscape Ecology. John Wiley & Sons, Inc., NY. 619 p.

Forman, R.T.T.; Mellinger, A.D. 1999. Road networks and forest spatial patterns: Comparing cutting-sequence models for forestry and conservation. In Nature Conservation 5: Nature Conservation in Production Environments: Managing the Matrix. J.L. Craig, N. Mitchell, and DA. Saunders (eds.). Surrey Beatty, Chipping Norton, Australia: pp. 71-80.

Freemark, K.; Collins, B. 1992. Landscape ecology of birds breeding in temperate forest fragments. Pages 443-454 in J. M. Hagan III and D.W. Johnston, eds. Ecology and conservation of neotropical migrant land birds. Smithsonian Institution Press, Washington, D.C.

Gilbert, F.S; Lee, S.B. 1980. Two perils of plant population dynamics. Oecologia. 46: 283-284.

Gregoire, T.G.; Schabenberger, O.; Barret, J.P. 1995. Linear modeling of irregularly spaced, unbalanced, longitudinal data from permanent-plot measurements. Canadian Journal of Forest Research. 25: 137-156.

Hansen, B.G.; Luppold, W.G.; Thomas, R.E. 1991. Revised data on U.S. hardwood lumber export volumes to Europe, 1981 to 1990. Northern Journal of Applied Forestry. 8(4): 156-160.

Hansen, M.H.; Frieswyk, T.; Glover, J.F.; Kelly, J.F. 1992. The Eastwide forest inventory data base: user's manual. General Technical Report, NC-151. St. Paul, MN. U.S. Department of Agriculture, USDA Forest Service, North Central Forest Experiment Station. 46 p.

Harrison, H.H. 1975. A field guide to the birds' nests: United States east of the Mississippi River. Houghton Mifflin Co., Boston, MA. 257 p.

Haynes, R.W. 1990. An analysis of the timber situation in the United States: 1989-2040. General Technical Report RM-199. Fort Collins, CO. U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 286 p.

Haynes, R.W.; Adams, D.M.; Mills, J.R. 1995. The 1993 RPA timber assessment update. General Technical Report RM-259. Fort Collins, CO. U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 66 p.

Herkert, J.R. 1994. The effect of habitat fragmentation on mid-western grassland bird communities. Ecological Applications. 4: 461-471.

Herlihy, A.T.; Larsen, D.P.; Paulsen, S.G.; Urquhart, N.S.; Rosenbaum, B.J. 2000. Designing a spatially balanced, randomized, site selection process for regional stream surveys: The EMAP Mid-Atlantic pilot study. Environmental Monitoring and Assessment [special issue] 63(1): 95-113.

Herwitz, S.R.; Wunderlin, R.P.; Hansen, B.P. 1996. Species turnover on a protected subtropical barrier island: a long-term study. Journal of Biogeography. 23: 705-715.

Hibbert, A.R. 1967. Forest treatment effects on water yield. In: Proceedings of a International Symposium on Forest Hydrology. W.E. Sopper and H.W. Lull., eds. Pergamon Press, Oxford, England: pp. 527-544.

Hines, J.E.; Boulinier, T.; Nichols, J.D. [and others]. 1999. COMDYN: Software to study the dynamics of animal communities using a capture-recapture approach. Bird Study: Vol. 46, supplement: S209-217.

Hoover, M.D. 1945. Effects of removal of forest vegetation upon water yields. Amer. Geophys. Union Trans. Part 6 (1944): 969-977.

Hornbeck, J.; Pierce, B.; Federer, C. 1978. Streamflow changes after forest clearing in New England. Water Resources Research. Vol. 6 (4): 1124-1132.

Houston, D.R. 1981. Stress triggered tree diseases, the diebacks and declines. NE-IF-41-81. Radnor, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 36 p.

Hudson, B.D. 1992. The Soil Survey as Paradigm-based Science. Soil Sci. Soc. Am. J. 56: 836-841.

Hurd, M.K.; Perry, S.A.; Perry, W.B. 1996. Non-target effects of a test application of diflubenzuron to the forest canopy on stream macroinvertebrates. Environmental Toxicology and Chemistry. 15(8): 1344-1351.

Hutto, R.L.; Pletschet, S.M.; Hendricks, P. 1986. A fixed-radius point count method for nonbreeding and breeding season use. Auk. 103: 593-602.

Hynes, H.B.N. 1975: The stream and its valley. Verh. Internat. Verein. Limnol., 19: 1-5.

Ice, G.G.; Stuart, G.W.; Waide, J.B. [and others]. 1997. 25 years of the Clean Water Act. How clean are forest practices? Journal of Forestry. Vol. 95(7): 9-13.

IPCC (Intergovernmental Panel on Climate Change) 2001: Climate Change 2001: The Scientific Basis. Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change [Houghton, J.T., and others. (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA. 881 p.

Isaaks, E.H.; Srivastava, R.M. 1989. An introduction to applied geostatistics. Oxford University Press, New York. 561p.

Iverson, L.R.; Prasad, A.M. 1998. Predicting abundance of 80 tree species following climate change in the eastern United States. Ecological Monographs. 69(4): 465-485.

Jenkins, G.S; Barron, E.J. 1997. Global climate model and coupled regional climate model simulations over the eastern United States: GENESIS and RegCM2 simulations. Global and Planetary Change. 15: 3-32.

Johnson, T.G. 1994. Virginia's Timber Industry: An Assessment of Timber Product Output and Use, 1992. U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station. 32 p.

Jones, K.B., Riitters, K.H.; Wickham, J.D. [and others]. 1997. An ecological assessment of the United States Mid-Atlantic Region. United States Environmental Protection Agency, Office of Research and Development, Washington, DC 20460.

Kareira, P.; Wennergren, U. 1995. Connecting landscape patterns to ecosystem and population processes. Nature. 373: 299-302.

Karr, J.R.; Dudley, D.R. 1981. Ecological perspective on water quality goals. Environmental Management. 5: 55-68.

Karr, J.R.; Schlosser, I.J. 1978. Water resources and the land-water interface. Science. 201(21): 229-234.

Kartesz, J.T.; Kartesz, R. 1980. A Synonymized Checklist of the Vascular Flora of the United States, Canada, and Greenland. Volume II. The Biota of North America. University of North Carolina Press. 500 p.

Kimmins, J.P. 1987. Forest Ecology. New York, New York: Macmillan Publishing Company. 531 p.

Kochenderfer, J.N.; Helvey, J.D. 1987. Using gravel to reduce soil losses from minimum-standard forest roads. Journal of Soil and Water Conservation. 42(1): 46-50.

Kuchler, A.W. 1964. Potential natural vegetation of the conterminous United States. American Geological Society Special Publication 36. 116 p. + map.

Lal, R.; Kimble, J.; Levine, E.; Whitman, C. 1995. World soils and greenhouse effect: an overview. in Advances in Soil Science: Soils and global change. R. Lal, J. Kimble, E. Levine, and B.A. Stewart, eds. CRC Lewis Publishers: Boca Raton, FL. 440 p.

Lee, J.S; Mulkey, T.J.; Evans, M.L. 1983. Gravity-induced polar transport of calcium across root tips of maize. Plant Physiology. 73: 874-876.

Lefohn, A.S.; Foley, J.K. 1992. NCLAN results and their application to the standard-setting process: protecting vegetation from surface ozone exposures. Journal of Air and Waste Management Association. 42(8): 1046-1052.

Lefohn, A.S.; Runeckles, V.C. 1987. Establishing a standard to protect vegetation-ozone exposure/dose consideration. Atmospheric Environment. 21: 561-568.

Lenat, D.R.; Penrose, D.L. 1996. History of the EPT taxa richness metric. Bulletin of the North American Benthological Society. Summer, 1996; Vol. 13(2).

Lewis, T.E; Conkling, B.L., eds. 1994. Forest Health Monitoring Southeast Loblolly/Shortleaf Pine Demonstration Interim Report. EPA/620/R-94/006. U.S. Environmental Protection Agency, Washington, DC. 327 p.

Likens, G.E.; Bormann, F.H.; Johnson, N.M. [and others]. 1970. Effects of forest cutting and herbicide treatment on nutrient budgets in the Hubbard Brook Watershed. Ecosystem Ecological Management. 40: 23-47.

Likens, G.E.; Bormann, F.H.; Pierce, R.S. [and others]. 1977. Biogeochemistry of a Forested Ecosystem. Springer-Verlag. NY. 146 p.

Littell, R.C.; Milliken, G.A.; Stroup, W.W.; Wolfinger, R.D. 1996. SAS system for mixed models. Cary, NC: SAS Institute Inc. 633 p.

Loeb, R.B. 1987. Pre-European settlement forest composition in east New Jersey and southeastern New York. The American Midland Naturalist. 188(2): 414-423.

Loveland, T.R., Shaw, D.M. 1996. Multi-resolution land characterization: building collaborative partnerships. Pages 83-89 in J.M. Scott, T. Tear, and F. Davis, eds. 1996. Gap Analysis: A landscape approach to biodiversity planning. Proceedings of the ASPRS/Gap Symposium, Charlotte, NC, National Biological Survey, Moscow, Idaho.

Luppold, W.G. 1993. Decade of change in the hardwood industry. In: Proceedings of the Twenty-first Annual Hardwood Symposium of the Hardwood Research Council, Cashiers, NC. 1994 May 24-26: 11-24.

Luppold, W.G. 1994. The U.S. hardwood log export situation: What is the problem? Forest Products Journal. 44(9): 63-67.

Luppold, W.G. 1995. Structural Changes in the Central Appalachian Hardwood Sawmilling Industry. Wood and Fiber Science. 28(3): 346-355. 1996.

Luppold, W.G.; Baumgras, J.E. 1996. Relationship between hardwood lumber and sawlog prices: a case study of Ohio, 1975-1994. Forest Products Journal. 46(10): 35-40.

Luppold, W.G. 1997. Regional Examination of Red Oak Lumber Price Trends. Northern Journal of Applied Forestry. 14(4): 173-177.

Lynch, J.A.; Corbett, E.S. 1990. Evaluation of Best Management Practices for Controlling Nonpoint Pollution from Silvicultural Operations. Water Resource Bulletin. Vol. 26(1): 41-52.

Lynch, J.A.; Corbett, E.S.; Mussallem, K. 1985. Best management practices for controlling nonpoint-source pollution on forested watersheds. Journal of Soil and Water Conservation. 40(1): 164-167.

Lynch, J.A.; Sopper, W.E.; Corbett, E.S. 1975. Clearcutting in Pennsylvania. University Park, PA, School of Forest Resources, College of Agriculture, The Pennsylvania State University. 81 p.

Lynch, J.A.; Bowersox, V.C.; Grimm, J.W. 1996. Trends in Precipitation Chemistry in the United States, 1983-1994: An Analysis of the Effects in 1995 of Phase I of the Clean Air Act Amendments of 1990, Title IV. U.S. Geological Survey, USGS 96-0346, Washington, DC, June 1996.

MacCleery, D.W. 1992. American forests: a history of resiliency and recovery. Washington, DC: U.S. Department of Agriculture, Forest Service. 58 p.

Mann, M.E.; Bradley, R.S.; Hughes, M.K. 1998. Global-scale temperature patterns and climate forcing over the past six centuries. Nature. 392: 779-787.

Mann, M.E.; Bradley, R.S.; Hughes, M.K. 1999. Northern hemisphere temperatures during the past millennium: inferences, uncertainties, and limitations. Geophysical Research Letters. 26: 759-762.

Manning, W.J., and W.A. Feder. 1980. Biomonitoring air pollutants with plants, Applied Science Publ. Ltd., London. 142 p.

Manuwal, D.A.; Carey, A.B. 1991. Methods for measuring populations of small, diurnal forest birds. Gen. Tech. Rep. PNW-GTR-278. U. S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, OR.

MARA (Mid-Atlantic Regional Assessment). 2000. Preparing for a changing climate: the potential consequences of climate variability and change. Mid-Atlantic Overview. A Report of the Mid-Atlantic regional assessment team for the U.S. Global Change Research Program. http://www.essc.psu.edu/mara/.

Marquis D.A. 1981. Effect of deer browsing on timber production in Allegheny hardwood forests of northwestern Pennsylvania. NE-475, United States Forest Service, Broomall, Pennsylvania. 32 p.

Martin, C.W.; Hornbeck, J.W. 1994. Sediment yields from undisturbed and intensively harvested northern hardwood forests. Proceedings of Hardwood Symposium, Hardwood Research Council Vol. 22: 69-79.

Mayack, D.T.; Bush, P.B.; Neary, D.G.; Douglas, J.T. 1982. Impact of hexazinone on invertebrates after application to forested watersheds. Arch. Environ. Contam. Toxicol. 11: 209-217.

McClimans, R.J. 1980. Best management practices for forestry activities. Symposium on Watershed Management. Boise, Idaho, July 21-23, 1980. [Place of publication unknown]: [Publisher unknown]; Vol. 2: 694-705.

McCune, B. 1992. Lichen Communities. Training Document. Forest Health Monitoring Western Pilot. Field Guide. 75 p. On file with FHM program at USDA Forest Service, Southern Research Station, 3041 E. Cornwallis Road, Research Triangle Park, North Carolina, 27709.

McCune, B.; Dey, J.; Peck, J. [and others]. 1997a. Repeatability of community data; species richness versus gradient scores in large-scale lichen studies. Bryologist. 100: 40-46.

McCune, B.; Dey, J.; Peck, J. [and others]. 1997b. Regional gradients in lichen communities of the southeast United States. Bryologist. 100: 145-158.

McKenney-Easterling, M; DeWalle, D.R.; Iverson; L.R. [and others] 2000. The potential impacts of climate change and variability on forests and forestry in the Mid-Atlantic region. Clim. Res. 14: 195–206.

McNab, W.H.; Avers, P.E., comps. 1994. Ecological subregions of the United States: Section descriptions. Administrative Publication WO-WSA-5. Washington, DC: U.S. Department of Agriculture, Forest Service. 267 p.

McPherson, G. 1993. Monitoring urban forest health. Environmental Monitoring and Assessment. 26: 165-174.

Mercer, E.; Murthy (Aruna), P.B. 2000. Assessing the Impacts of Forests on Human Welfare: Preliminary Results from the Mid-Atlantic Integrated Assessment. Environmental Monitoring and Assessment. 63(1): 43-63.

Meyer. J.L.; Tate, C.M. 1983. The effect of watershed disturbance on dissolved organic carbon dynamics of a stream. Ecology. Vol. 64(1): 33-44.

Mielke, M.E. 1999. Forest Health Monitoring Damage Indicator Report. Forest Health Monitoring Program, FHM Management Team, May 1999. http://fhm.fs.fed.us/.

Mielke, M.; Burkman, W.; Cox, S. [and others]. 1995. Chapter 3. Damage and catastrophic mortality assessment.1995 version. In: Conkling, B.L.; Tallent-Halsell, N.G., eds. Forest Health Monitoring field methods guide. EPA/620/R-94/027. Washington, DC: U.S. Environmental Protection Agency. 371 p.

Miller, H.M. 1986. Transforming a "Splendid and delightsome land": colonists and ecological change in the Chesapeake 1607-1820. Journal of the Washington Academy of Sciences. Vol. 76(3): 173.

Miller, J.N.; Brooks, R.P.; Croonquist, M.J. 1997. Effects of landscape pattern on biotic communities. Landscape Ecology. 12: 137-153.

Miller, P.R.; Stolte, K.W.; Duriscoe, D.M.; Pronos, J. 1996. Evaluating ozone air pollution effects on pines in the western United States. United States Department of Agriculture, Forest Service. Pacific Southwest Research Station. General Technical Report PSW-GTR-155. 79 p.

NADP (National Atmospheric Deposition Program). 1998. http://nadp.sws.uiuc.edu/.

NAPAP (National Acid Precipitation Assessment Program). 1991. Changes in forest health and productivity in the United States and Canada. Vol. III, Report 16. Washington, DC: National Acid Precipitation Assessment Program. 186 p.

National Assessment of Climate Change. 2000. U.S. national assessment of the potential consequences of climate variability and change. http://www.nacc.usgrp.gov.

NCASI (National Council of the Paper Industry for Air and Stream Improvement, Inc.). 1994. Forests as Nonpoint Sources of Pollution, and Effectiveness of Best Management Practices. Technical Bulletin No. 672. Corvallis, OR.

Neary, D.G.; Bush, P.B.; Michael, J.L. 1993. Fate, Dissipation and Environmental Effects of Pesticides in Southern Forests: A Review of a Decade of Research Progress. Environmental Toxicology and Chemistry. Vol. 12: 411-428.

New Jersey Department of Environmental Protection. 1996. Trees in Crisis. A Statewide Assessment of New Jersey's Street Trees. Draft Summary Report. New Jersey Department of Environmental Protection. New Jersey Forest Service.

Nichols, J.D.; Boulinier, T.; Hines, J.E. [and others]. 1998a. Estimating rates of local extinction, colonization and turnover in animal communities. Ecological Applications. 8: 1213-1225.

Nichols, J.D.; Boulinier, T.; Hines, J.E. [and others]. 1998b. Inference methods for spatial variation in species richness and community composition when not all species are detected. Conservation Biology. 12: 1390-1398.

Nichols, J.D.; Conroy, M.J. 1996. Estimation of species richness. Pages 226-234 in D.E. Wilson, F.R. Cole, J.D. Nichols, R. Rudran, and M. Foster (eds.). Measuring and monitoring biological diversity. Standard methods for mammals. Smithsonian Institution Press, Washington, DC.

Norris, L.A.; Montgomery, M.L.; Loper, B.R.; Kochenderfer, J.N. 1984. Movement and persistence of 2,4,5-trichlorophenoxyacetic acid in a forest watershed in the eastern United States. Environmental Toxicology and Chemistry. Vol. 3: 535-549.

Norris, V. 1993. The use of buffer zones to protect water quality: a review. Water Resource Management. Vol. 7(4): 257-272.

Nowak, D.J. 1994a. Air Pollution Removal by Chicago's Urban Forest, pp. 63-81. In Chicago's Urban Forest Ecosystem: Results of the Chicago Urban Forest Climate Project. Gen. Tech. Rpt. NE-186.

Nowak, D.J. 1994b. Urban forest structure: The state of Chicago's forest, pp. 3-18. In Chicago's Urban Forest Ecosystem: Results of the Chicago Urban Forest Climate Project. USDA Forest Service Gen. Tech. Rpt. NE-186.

Nowacki, G.J.; Abrams, M.D. 1992. Community, edaphic, and historical analysis of mixed oak forests of the Ridge and Valley Province in central Pennsylvania. Canadian Journal of Forest Research. (22): 790-800.

O' Connell, T.J.; Jackson, L.E.; Brooks, R.P. 1998. A bird community index of biotic integrity for the Mid-Atlantic Highlands. Environmental Monitoring and Assessment. 51: 145-156.

O' Connell, T.J.; Jackson, L.E.; Brooks, R.P. 2000. Bird guilds as indicators of ecological condition in the central Appalachians. Ecological Applications. 10(6): 1706-1721.

Omernik, J.M. 1977. Non-point source-stream nutrient level relationships: a nationwide study. EPA 600/3 77 105. Corvallis Environmental Research Laboratory. US Environmental Protection Agency, Corvallis, OR.

Orwig, D.A.; Abrams, M.D. 1994. Land-use history (1720-1992), composition, and dynamics of oak-pine forests within the Piedmont and Coastal Plain of northern Virginia. Canadian Journal of Forest Research. (24): 1216-1225.

Otis, D.L.; Burnham, K.P.; White, G.C.; Anderson, D.R. 1978. Statistical inference from capture data on closed animal populations. Wildlife Monographs. 62 p.

Pardo, L.H.; Driscoll, C.T.; Likens, G.E. 1995. Patterns of nitrate loss from a chronosequence of clear-cut watersheds. Water, Air, Soil Pollution. 85: 1659-1664.

Patric, J.H. 1976. Soil erosion in the Eastern forest. Journal of Forestry. 74(10): 671-677.

Patric, J.H. 1978. Harvesting effects on soil and water in the eastern hardwood forest. Southern Journal of Applied Forestry. 2(3): 66-73.

Patric, J.H.; Aubertin, G.M. 1977. Long-term effects of repeated logging on an Appalachian stream. Journal of Forestry. 75: 492-494.

Patric, J.; Reinhart, K. 1971. Hydrologic Effects of Deforesting Two Mountain Watersheds in West Virginia. Water Resources Research. 5(5): 1182-1188.

Pollock, K.H. 1982. A capture-recapture sampling design robust for unequal catchability. Journal of Wildlife Management. 46: 252-257.

Pollock, K.H.; Nichols, J.D.; Brownie, C.; Hines, J.E. 1990. Statistical inference for capture-recapture experiments. Wildlife Monographs. 62 p.

Pollock, K.H.; Otto, M.C. 1983. Robust estimation of population size in closed animal populations from capture-recapture experiments. Biometrics. 39: 1035-1049.

Prasad, A.M.; Iverson, L.R. 2000. A climate change atlas for 80 forest tree species of the eastern United States. http://www.fs.fed.us/ne/delaware/atlas. USDA Forest Service, Northeastern Research Station, Delaware, Ohio.

Prestemon, J.P.; Pye, J.M. 2000. A technique for merging areas in Timber Mart-South data. Southern Journal of Applied Forestry. 24(4): 219-229.

Preston, F.W. 1979. The invisible birds. Ecology. 60: 451-454.

Pritchett, W.L.; Fisher, R.F. 1987. Properties and Management of Forest Soils. John Wiley & Sons: New York, New York. 500 p.

Pye, J. 1999. Mill Locations. http://www.rtp.srs.fs.fed.us/econ/data/mills/mills.htm. USDA Forest Service, Southern Research Station, Economics Research Work Unit #4851, Research Triangle Park, NC.

Ralph, C.J.; Geupel, G.R.; Pyle, P. [and others]. 1993. Handbook of field methods for monitoring land birds. Gen. Tech. Rep. PSW-GTR-144. Pacific Southwest Research Station, Forest Service, U.S. Department of Agriculture, Albany, CA. 41 p.

Rapport, D.J.; Reigier, H.A.; Hutchinson, T.C. 1985. Ecosystem behavior under stress. American Naturalist 125: 617-640.

Reinhart, K.G.; Eschner, A.R.; Trimble, Jr., G.R. 1963. Effect on streamflow of four forest practices in the mountains of West Virginia. USDA Forest Serv. Res. Pap. NE-1. Upper Darby, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station, 79 p.

Rexstad, E.; Burnham, K.P. 1991. User's guide for interactive program CAPTURE. Abundance estimation of closed animal populations. Colorado State University, Fort Collins, Colorado. 33 p.

Richardson, C.J.; McCarthy, E.J. 1994. Effect of land development and forest management on hydrologic response in southeastern coastal wetlands: a review. Wetlands. 14(1): 56-71.

Ricklefs, R.E. 1979. Ecology, 2nd ed. New York: Chiron Press, Inc. 966 p.

Riekerk, H. 1983. Impacts of Silviculture on Flatwoods Runoff, Water Quality, and Nutrient Budgets. Water Resources Bulletin. 19(1): 73-80.

Riitters, K.H. 1999. USGS Biological Resources Division FTP access to United States Land Cover Pattern Maps. FTP site: http://152.1.67.36/us_maps.html. Accessed May13, 1999.

Riitters, K.H.; Wickham, J.D.; Jones, K.B.; Vogelman, J.E. 2000. National land-cover pattern data. Ecology. 81:604.

Robbins, C.S.; Dawson, D.K.; Dowell, B.A. 1989. Habitat area requirements of breeding forest birds of the middle Atlantic states. Wildlife Monographs. Number 103.

Roberts, T.H. 1987. Construction of guilds for habitat assessment. Environmental Management. 11: 473-477.

Robinson, G.K. 1991. That BLUP is a good thing: The estimation of random effects. Statistical Science. 6: 15-51.

Salisbury, F.B.; Ross, C.W. 1978. Plant Physiology 2nd edition. Wadsworth Publishing Company, Inc. Belmont California. 966 p.

SAMAB (Southern Appalachian Man and the Biosphere). 1996. The Southern Appalachian Assessment Atmospheric Technical Report. Report 3 of 5. Atlanta: U.S. Department of Agriculture, Forest Service, Southern Region. 83 p.

Sanders, R.A. 1986. Urban vegetation impacts on the hydrology of Dayton, Ohio. Urban Ecol. 9: 361-376.

Santner, S.J.; Brauning, D.W.; Schwalbe, G.; Schwalbe, P.W. 1992. Annotated list of the birds of Pennsylvania. Pennsylvania Biological Survey No.4, Ornithological Technical Committee.

Sargent, C.S. 1884. Report on the forests of North America. U.S. Department of the Interior, Census Office. Gov't Print. Office, Washington DC. 612 p.

SAS. 1999. SAS Institute Inc., SAS OnlineDoc®, Version 8, Cary, NC: SAS Institute Inc., 1999. http://www.tau.ac.il/cc/pages/docs/sas8/common/images/copyrite.htm.

Sauer, J.R., J.E. Hines, and J. Fallon. 1999. The North American Breeding Bird Survey, Results and Analysis 1966-1999. USGS Patuxent Wildlife Research Center, Laurel, MD (http://www.mbr-pwrc.usgs.gov/bbs/).

Schlarbaum, S.E., F. Hebard, P.C. Spaine and J.C. Kamalay. 1997. Three American tragedies: chestnut blight, butternut canker, and Dutch elm disease. Proc. Exotic Pest of Eastern Forests, April 8-10, 1997, Nashville, TN. pp. 45-54.

Schlesinger, W.H. 1995. An overview of the carbon cycle. in Advances in Soil Science: Soils and global change. R. Lal, J. Kimble, E. Levine, and B.A. Stewart, eds. CRC Lewis Publishers: Boca Raton, FL. 440 p.

Schreuder, H.T.; Gregoire, T.G.; Wood, G.B. 1993. Sampling Methods for Multi-Resource Forest Inventory. John Wiley & Sons, Inc., NY. 446 p.

Scott, C.T.; Bechtold, W.A. 1995. Techniques and computations for mapping plot clusters that straddle stand boundaries. For. Sci. Monogr., Vol. 31: 46-61.

Seaber, P.R.; Kapinos, F.P.; Knapp, G.L. 1987. Hydrologic unit maps: U.S. Geological Survey Water-Supply Paper 2294. Second Printing. 63 p. (http://www.nationalatlas.gov/mld/hucs00m.html).

Shepard, J.P. 1994. Effects of Forest Management on Surface Water Quality in Wetland Forests. Wetlands. Vol. 14(1): 18-26.

Smith, E.R.; Tran, L.T.; O'Neill, R.V. 2003. Regional Vulnerability Assessment of the Mid-Atlantic Region: Evaluation of Integration Methods and Assessment Results. U.S. Environmental Protection Agency, Washington D.C., EPA/600/R-03/082. Online access at (http://www.epa.gov/reva/docs/vulnerable.pdf). 83 p.

Smith, W.D.; Conkling, B.L. 2000. Analyzing forest health data. Gen. Tech. Rep. SRS-077. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 33 p.

Smith, W.D.; Gumpertz, M.L.; Catts, G.C. 1996. An analysis of the precision of change estimation of four alternative sampling designs for forest health monitoring. Forest Health Monitoring Technical Report Series (10/96). Research Triangle Park, NC; U.S. Department of Agriculture, Forest Service, Southern Research Station. 25 p.

Smith, W.H. 1974. Air pollution - Effects on the structure and function of the temperate forest ecosystem. Environmental Pollution. 6: 111-129.

Society of American Foresters (SAF). 1998. Forest fragmentation in the Chesapeake Bay watershed: what are its ecological, economic, and policy/law impacts? Draft working paper. A Professional Round Table Series, 12-16, January, 1998, Society of American Foresters.

Soil Conservation Service. 1975. Soil taxonomy. Agriculture Handbook No. 436. U.S. Department of Agriculture Soil Conservation Service: Washington, D.C. 754 p.

Stein, J. (ed.). 1982. The Random House College Dictionary-Revised Edition. New York, NY 10022. 1568 p.

Stevens, Jr., D.L. 1997. Variable density grid based sampling designs for continuous spatial populations. Environmetrics. 8(3): 167 95.

Stohlgren, T.J.; Binkley, D.; Chong, G.W. [and others]. 1999. Exotic plant species invade hot spots of native plant diversity. Ecological Monographs. 69(1): 25-46.

Stolte, K.W. 1997. 1996 National Technical Report on Forest Health. Admin. Rep. FS-605. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 47 p.

Stolte, K.; Mangis, D.; Doty, R.; Tonnessen, K. 1993. Lichens as bioindicators of air quality. General Technical Report RM-224. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 131 p.

Stolte, K.W.; Coulston, J.W.; Ambrose, M.A.; Riitters, K.R.; Conkling, B.L.; Smith, W.D. 2005. 2000 FHM National Technical Report. U.S. Department of Agriculture, Forest Service, Southern Research Station, General Technical Report. On file with FHM program at USDA Forest Service, Southern Research Station, 3041 E. Cornwallis Road, Research Triangle Park, North Carolina, 27709.

Stout, B.M., III; Benfield, E.F.; Webster, J.R. 1993. Effects of forest disturbance on shredder production in southern Appalachian headwater streams. Freshwater Biology. Vol. 29(1): 56-69.

Stoyenoff, J.; Witter, J.; Leutscher, B. 1997. Is the forest healthy: Mid-Atlantic Region. University of Michigan, School of Natural Resources, Lansing, Michigan.

Strahler, A.N. 1964. Geology - Part II. Handbook of Applied Hydrology. McGraw-Hill Book Company, NY. 1418 p.

Taylor, A.F.; Wiley, A.; Kuo, F.E.; Sullivan, W.C. 1998. Growing up in the inner city: Green spaces as places to grow. Environment and Behavior. 30(1): 3-27.

Terborgh, J. 1989. Where have all the birds gone? Princeton University Press, Princeton, NJ. 228 p.

Tilman, D.; May, R.; Lehman, C; Nowak, M. 1994. Habitat destruction and the extinction debt. Nature. 371: 65-66.

Tomlinson, F.L. 1990. Effects of acid deposition on the forest of Europe and North America. CRC Press, Inc. Boca Raton, Florida. 281 p.

Trimble, S.W. 1974. Man-induced erosion on the southern Piedmont: 1700-1970. Soil Conservation Society of America. 180 p.

Trumbull V.L., E.J. Zielinski, and E.C. Aharrah. 1989. The impact of deer browsing on the Allegheny forest type. Nor. J. Appl. For. 6: 162-165.

Tubbs, C.H.; Houston, D.R. 1990. American Beech *Fagus grandifolia* Ehrh. In: Silvics of North America Volume 2, Hardwoods. R.M. Burns and B.H. Honkala, Tech. Coords. USDA Forest Service, Washington, DC, Agric. Handbook. No. 654: 325-332.

Turner, M.G. 1989. Landscape ecology: the effect of pattern on process. Ann. Rev. Ecol. Syst. 20: 171-197.

Urquhart, N.S.; Overton, W.S.; Birkes, D.S. 1993. Comparing sampling designs for monitoring ecological status and trends: Impact of temporal patterns. In Statistics for the Environment. John Wiley & Sons Ltd: 71-86.

USDA (U.S. Department of Agriculture). 2000. *Summary Report: 1997 National Resources Inventory (revised December 2000),* Natural Resources Conservation Service, Washington, DC, and Statistical Laboratory, Iowa State University, Ames, Iowa, 89 pages. (http://www.nrcs.usda.gov/technical/NRI/1997/summary_report/).

U.S. Department of Agriculture, Forest Service (USDAFS). 1995. Forest Health Monitoring 1995 Field Methods Guide. USDA Forest Service, National Forest Health Monitoring Program, Research Triangle Park, NC 27709. 371 p.

U.S. Department of Agriculture, Forest Service (USDAFS). 1998a. Forest Health Monitoring 1998 Field Methods Guide. USDA Forest Service, National Forest Health Monitoring Program, Research Triangle Park, NC 27709.

U.S. Department of Agriculture, Forest Service (USDAFS). 1998b. Forest Health Protection, Gypsy Moth Digest. (http://na.fs.fed.us/fhp/gm/maps_charts/index.shtm).

U.S. Department of Commerce, Bureau of Census (USDC-BC). 1994. Geographic areas reference manual. U.S. Department of Commerce: Washington, DC.

USDL-BLS (U.S. Department of Labor, Bureau of Labor Statistics). 1989. *Employment Data Under the New Standard Industrial Classification,* First Quarter 1988, Report 772, October 1989.

USDL-BLS (U.S. Department of Labor, Bureau of Labor Statistics). 1996. *Employment and Wages, Annual Averages,* 1995, Bulletin 2483, December 1996.

U.S. Geological Survey (USGS). 1974-1987. State Hydrologic Unit Map (for each State): U.S. Geological Survey, Reston, VA. (http://water.usgs.gov/GIS/huc.html).

Valenti, Michael. 1998. Delaware Street Tree Assessment. Delaware Forest Health Highlights. Delaware Forest Service, Dover, Delaware.

Van Deusen P.C. 1989. Multiple occasion partial replacement sampling for growth components. For. Sci. 35: 388-400.

Vose, J.M.; Swank, W.T. 1990. Chapter 24. A conceptual model of forest growth emphasizing stand leaf area. In: Dixon, R.K.; Meldahl, R.S.; Ruark, G.A.; Warren, W.G., eds. Process modeling of forest growth responses to environmental stress. Portland, OR. Timber Press. pp. 278-287.

Vose J.M. and Allen H.L. 1988. Leaf area, stemwood growth and nutrition relationships in loblolly pine. For. Sci. 34: 546–563.

Wallace, J.B.; Webster, J.R.; Cuffney, T.F. 1982. Stream detritus dynamics: Regulation by invertebrate consumers. Oecologia. 53: 197-200.

Warrington, G.E.; Knapp, K.L.; Klock, G.O. [and others]. 1980. In: USDA Forest Service. An approach to water resources evaluation of non-point silvicultural sources (a procedural handbook). EPA/600/8-80-012. Athens, GA: U.S. Environmental Protection Agency. Environmental Research Laboratory. 81 p.

Wear, D. 1996. The timber economy of the Southern Appalachians. The Southern Appalachian Assessment. Social/ Cultural/ Economic Technical Report. 219 p.

Wear, D.N.; Foreman, J.M.; Sheffeld, R.M. 1999. The effects of population growth on timber management and inventories in Virginia. Forest Ecology and Management. 118: 107-115.

Weinstein, D.A.; Beloin, R.M. 1990. Evaluating the effects of pollutants on integrated tree processes: A model of carbon, water and nutrient balances. In: Dixon, R.K.; Meldahl, R.S.; Ruark, G.A.; Warren, W.G., eds. Process modeling of forest growth responses to environmental stress. Portland, OR. Timber Press. pp. 313-323.

Wharton, E.H.; Nevel, R.L. 1990. The Timber Industries of Delaware, 1985. Resour. Bull. NE 118. Broomall, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 28 p.

Wharton, E.H.; Bearer, J.L. 1994. The Timber Industries of Pennsylvania, 1988. Resour. Bull. NE 130. Broomall, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 40 p.

White, G.C.; Anderson, D.R.; Burnham, K.P.; Otis, D.L. 1982. Capture-recapture removal methods for sampling closed populations. Los Alamos National Laboratory Publication LA-8787-NERP.

Whitney, G.G. 1990. The history and status of the hemlock-hardwood forests of the Allegheny Plateau. Journal of Ecology. (78): 443-458.

Wickham, J.D.; Norton, D.J. 1994. Mapping and analyzing landscape patterns. Landscape Ecology. 9: 7-23.

Widmann R.H. 1987. Pulpwood Production in the Northeast—1987. Resour. Bull. NE 111. Broomall, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 24 p.

Widmann R.H. 1988. Pulpwood Production in the Northeast—1988. Resour. Bull. NE 116. Broomall, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 26 p.

Widmann R.H. 1989. Pulpwood Production in the Northeast—1989. Resour. Bull. NE 119. Broomall, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 26 p.

Widmann R.H. 1990. Pulpwood Production in the Northeast—1990. Resour. Bull. NE 123. Broomall, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 27 p.

Widmann R.H. 1996. Pulpwood Production in the Northeast:1991, 1992, and 1993. Resour. Bull. NE 133. Broomall, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 24 p.

Widmann R.H.; Murriner, E.C. 1990. West Virginia Timber Products Output -1987. Resour. Bull. NE 115. Broomall, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 20 p.

Wigley, T.B.; Roberts, T.H. 1994. A Review of Wildlife Changes in Southern Bottomland Hardwoods due to Forest Management Practices. Wetlands. Vol. 14(1): 41-48.

Wood, P.; Armitage, P. 1997. Biological effects of fine sediment in the lotic environment. Environmental Management. 21(2): 203-217.

Wood, P.J., and Armitage, P.D. 1999. Sediment deposition in a small lowland stream—management implications. Regul. Rivers: Res. Mgmt. 15: 199–210.

List of Figure Captions

Figure 1—The Mid-Atlantic Integrated Assessment (MAIA) region includes all of Maryland, Virginia, West Virginia, Pennsylvania, Delaware, District of Columbia, and whole or partial counties in northeast North Carolina (~47), western New Jersey (~12), and southeastern New York (~25).Source: Environmental Protection Agency Region III; (http://www.epa.gov/region03/index.htm).

Figure 2—Conceptual model of relationships among stressors, forest ecosystems, and healthy and sustainable forests. New stressors, or exacerbated endemic stressors (top bar), interact with forest ecosystem components and processes (middle oval) in one or more primary functional areas organized by ecological criteria of the Montreal Process Criteria and Indicators. Forest ecosystems respond to new stressors in a sustainable or non-sustainable manner (bottom two ovals) depending on above factors.
Source: Stolte 1997.

Figure 3—Classifications of forestland and non-forestland types and areas by MAIA region states in 1990s. Entire Washington D.C. area is considered urban and therefore not forestland.Source: USDA 2000; (http://www.nrcs.usda.gov/technical/NRI/).

Figure 4—Ownership of forestlands in the MAIA region. Forestlands in Natural Resource Inventory are similar to Forest Inventory and Analysis's timberland classification of forestlands. Source: USDA 2000; (http://www.nrcs.usda.gov/technical/NRI/). Birch 1996a.

Figure 5—Bailey's ecoregion sections within the MAIA region. The U.S. Forest Service's Forest Health Monitoring, Forest Inventory and Analyses, and other program data were frequently averaged at these spatial scales for comparison of forests within the MAIA region. Source: Bailey 1995; USDA Forest Service; (http://www.fs.fed.us/land/ecosysmgmt/index.html).

Figure 6— Major 4-Digit Hydrologic Unit Code (HUC4) watersheds in the MAIA region. Some data from the U.S. Forest Service's Forest Health Monitoring program was averaged at HUC4 spatial scales to assess differences with ecoregion section values and provide potential for comparisons with water condition. Source: United States Department of Interior's Geological Survey; (http://water.usgs.gov/GIS/huc.html).

Figure 7—Major roads in the MAIA region, indicating major sources for fragmenting forest ecosystems, introduction of exotic invasive species, and other urban-wildland interface issues. Source: Environmental Systems Research Institute, Inc, Data & Maps; (http://esri.com/).

Figure 8—Metropolitan and urban areas in the MAIA region. Source: Environmental Protection Agency Region III; (http://www.epa.gov/region03/index.htm).

Figure 9—Classification of all lands in the MAIA region. Woodland is defined as forest land with less than 10 percent stocked with timberland species and with at least 10 percent crown cover in timberland and woodland species. Reserved land is defined as land that is withdrawn from all timber utilization by a public agency or by law. Source: US Forest Service Forest Inventory and Analysis Eastwide Database; (http://www.fia.fs.fed.us/).

Figure 10—Non-federally owned urban and rural land that is classified as forestland in each State and in the entire MAIA region in 1992. Some counties in New Jersey (~12 counties), North Carolina (~47 counties), and New York (~25 counties) included in MAIA region. Source: USDA 2000; (http://www.nrcs.usda.gov/technical/NRI/).

Figure 11—Acres of primary land use types five States contained entirely in the MAIA region in 1997 and the total for all five States. Total agriculture is the combination of cropland and pastureland. Total area is total surface area including land and water cover. Source: USDA 2000; (http://www.nrcs.usda.gov/technical/NRI/).

Figure 12—Forest cover within 1457 acre landscape units in the MAIA region ca. 1990. The fine-scale (0.22 acres per pixel) land cover map from EPA's MRLC was generalized to show the proportion of forest cover within the 1457 acre units. Areas with more than 90 percent forest cover are highly interior forested landscapes; in contrast to highly fragmented units with less than 10 percent forest cover. A 60 percent cover threshold separates interior forests from more fragmented forests. Source: Loveland and Shaw 1996; Riitters and others 2000; U.S. Environmental Protection Agency's early 1990s Multi-Resolution Landscape Consortium (MRLC); (http://www.epa.gov/mrlc/).

Figure 13—Landscape Pattern Types (LPT) in the MAIA region ca. 1990. The fine-scale (0.22 acres per pixel) land cover map from EPA's MRLC indicated the relative proportions of forest, agriculture, and developed cover types within the 1457 acre units; LPTs were labeled by the relative amount of three land cover types within the surrounding 1457 acre units. The 19 LPT categories of MRLC were condensed into 7 LPT aggregates to simplify regional patterns. Source: Wickham and Norton 1994; Loveland and Shaw 1996; Jones and others 1997; Riitters and others 2000; U.S. Environmental Protection Agency's early 1990s Multi-Resolution Landscape Consortium (MRLC); (http://www.epa.gov/mrlc/).

Figure 14—Total tree cover in metropolitan areas (MA) and urban areas (UA) in MAIA region states and the U.S. in 1990's. U.S. area includes District of Columbia but not Alaska or Hawaii. Source: Dwyer and others 2000.

Figure 15—Trees per person (per capita) in metropolitan areas (UA) and urban areas (UA) in MAIA region states and the U.S. in 1990's. U.S. includes District of Columbia but not Alaska or Hawaii. Source: Dwyer and others 2000.

Figure 16—Average annual ozone concentrations (W126 index values) in the MAIA region for 1993 to1996 (colored polygons). Legend also shows ozone concentration values for other parts of U.S. for comparison. Closed circles are average ozone injury index values recorded at FHM ozone biomonitoring sites around the same time. Source: Ozone data from EPA database (http://www.epa.gov/); ozone bioindicator response data from USDA Forest Service National Forest Health Monitoring program; (http://fhm.fs.fed.us/).

Figure 17— Potential biomass loss in black cherry in 1990 where the distribution of this species overlapped areas with phytotoxic ozone exposures in the MAIA region. Source: North Carolina State University 1996; (http://www.ncsu.edu/).

Figure 18—Average annual wet deposition of sulfate in MAIA region from 1979 to 1995. Data from this16-year period were averaged and interpolated at 5-km grid scale to estimate average annual wet deposition at ecoregion section spatial scales. Source: 1996 National Atmospheric Deposition Program's National Trends Network; (http://www.nadp.sws.uiuc.edu/). 1996 Canadian Air and Precipitation Monitoring; (http://www.msc-smc.ec.gc.ca/capmon/index_e.cfm).

Figure 19—Average annual wet deposition of nitrate in MAIA region from 1979 to 1995. Data from this16-year period were averaged and interpolated at 5-km grid scale to estimate average annual wet deposition at ecoregion section scales. Source: 1996 National Atmospheric Deposition Program's National Trends Network; (http://www.nadp.sws.uiuc.edu/). 1996 Canadian Air and Precipitation Monitoring; (http://www.msc-smc.ec.gc.ca/capmon/index_e.cfm).

Figure 20—Average annual precipitation pH in the MAIA region from 1979 to 1995. Data from this 16-year period were averaged and interpolated at 5-km grid scale to estimate average annual wet deposition at ecoregion section scales. Source: 1996 National Atmospheric Deposition Program's National Trends Network; (http://nadp.sws.uiuc.edu/). 1996 Canadian Air and Precipitation Monitoring; (http://www.msc-smc.ec.gc.ca/capmon/index_e.cfm).

Figure 21—Historic fire regimes constructed from tree ring analyses and fire scars in the MAIA region. Source: USDA Forest Service; (http://www.fs.fed.us/fire/fuelman/index.htm).

Figure 22— Condition classes of changes in current fire regimes from historic fire regimes in forested areas in the MAIA region. Higher numbered condition classes indicate increasing amounts of silvicultural treatments would be needed to restore historic fire regimes. Source: USDS Forest Service; (http://www.fs.fed.us/fire/fuelman/index.htm).

Figure 23— Dominant forest types under circa 2000 climate regimes, and potential forest type distributions under Canadian Climate Center and Hadley doubled CO_2 equilibrium climate scenarios. Source: Mid-Atlantic Regional Assessment (MARA) Team; (http://www.nhbs.com/mid_atlantic_regional_assessment_of_climate_change). Pennsylvania State University; (http://www.psu.edu/).

Figure 24—Gypsy moth infested areas in MAIA counties through 1998 with gypsy moth caterpillar photograph (inset). Source: USDA Forest Service, Forest Health Protection; (http://www.fs.fed.us/foresthealth/).

Figure 25—Hemlock woolly adelgid (HWA) infested areas in MAIA counties through 1997. Photographic insets show hemlock trees killed by HWA, and biological control ladybird beetle feeding on HWA. Source: USDA Forest Service, Forest Health Protection; (http://www.fs.fed.us/foresthealth/).

Figure 26—Southern pine beetle outbreak infestations in MAIA counties over 30 year period (ca. 1968 to1998). Photographic insets show an example of resulting tree damage and mortality, and beetle in a carved gallery. Source: USDA Forest Service, Forest Health Protection; (http://www.fs.fed.us/foresthealth/).

Figure 27—Butternut species range and documented butternut canker distribution. Butternut canker is thought to be distributed throughout the range of butternut but is often not observed because of the scarce and sporadic distribution of Butternut trees. Photographic inset shows an example of a sparse tree crown, the obvious typical first sign of infection. Source: USDA Forest Service, Forest Health Protection; (http://www.fs.fed.us/foresthealth/).

Figure 28—Beech bark disease is a disease-complex of American beech trees attacked by the beech scale insect that creates entry points for invasion by two primary killing fungi: Nectria coccinea var. faginata and N. galligena. The map shows incidence by MAIA counties through 1998. Inset shows an example of thinning tree crowns symptomatic of established beech bark disease infection centers. Note: American beech is generally found throughout the MAIA region. Source: USDA Forest Service, Forest Health Protection; (http://www.fs.fed.us/foresthealth/).

Figure 29—Range of dogwood species and distribution of dogwood anthracnose in MAIA counties through 1998. Inset shows an example of leaf blotching indicative of dogwood anthracnose infection. Source: USDA Forest Service, Forest Health Protection; (http://www.fs.fed.us/foresthealth/).

Figure 30—Exotic invasive plant species by county in the MAIA region compiled from herbaria and other data sources. Source: Biota of North America Program; (http://www.bonap.org/).

Figure 31—Population density in MAIA region in 1990. Source: U.S. Census Bureau; (http://www.census.gov/population/www/).

Figure 32—Change in population density 1950 to 1990. Source: U.S. Census Bureau; (http://www.census.gov/population/www/).

Figure 33—Change in population density in MAIA region 1970 to 1990. Source: U.S. Census Bureau; (http://www.census.gov/population/www/).

Figure 34—Forest fragmentation in the MAIA region1990 to 1993. Note: Changes in forest fragmentation were not analyzed due to lack of regional remote sensing data. Source: Riitters 1999.

Figure 35—Non-federal rural land cover area in MAIA region in 1982, 1987, and 1992. Source: USDA 2000; (http://www.wa.nrcs.usda.gov/technical/NRI/).

Figure 36—Land cover share of total non-federal rural land in MAIA region in 1982, 1987, and 1992. Source: USDA 2000; (http://www.wa.nrcs.usda.gov/technical/NRI/).

Figure 37—Land cover in five MAIA region states in 1997. Total agriculture is the combination of crop lands and pasture lands. Source: USDA 2000; (http://www.wa.nrcs.usda.gov/technical/NRI/).

Figure 38—Land cover in five MAIA region states in 1982, 1987, 1992, and 1997. Source: USDA 2000; (http://www.wa.nrcs.usda.gov/technical/NRI/).

Figure 39—Urbanization changes in primary land cover types in five MAIA region states 1982 to 1997. Source: USDA 2000; (http://www.wa.nrcs.usda.gov/technical/NRI/).

Figure 40—Differences in land cover types for five MAIA region states 1982 to 1997. Total agriculture is the combination of crop and pasture lands. Source: USDA 2000; (http://www.wa.nrcs.usda.gov/technical/NRI/).

Figure 41—Differences in major land cover types in five MAIA region states 1982 to 1997. Source: USDA 2000; (http://www.wa.nrcs.usda.gov/technical/NRI/).

Figure 42—Contribution of land cover types to rural lands in five MAIA region states 1982 to 1997. Source: USDA 2000; (http://www.wa.nrcs.usda.gov/technical/NRI/).

Figure 43—Change in number of acres per owner of forest land by State and MAIA region 1978 to 1994. Source: Birch 1996a, Birch 1996b, Birch 1996c.

Figure 44—Change in acres per owner share of forest land by State and MAIA region 1978 to 1994. Source: Birch 1996a, Birch 1996b, Birch 1996c.

Figure 45—Breeding-bird sample site locations within the Mid-Atlantic highlands area (MAHA). Source: O'Connell and others 1998, 2000.

Figure 46—Distribution of 126 MAHA sample sites in five condition categories determined by the Bird Community Index (BCI). Error bars represent 95 percent confidence interval estimate for the percentage of land area in MAHA supporting these five categories. Source: O'Connell and others 2000.

Figure 47—Spatial patterns of BCI scores across MAHA. Note: data from individual sites are not statistically representative of subregional conditions. Source: O'Connell and others 2000.

Figure 48—Bird community index (BCI) scores along a gradient of forest cover. Source: O'Connell and others 2000.

Figure 49—Mean and standard deviation of percent forested, agricultural/herbaceous, and residential/commercial land cover of sites in five condition categories determined by BCI scores. Source: O'Connell and others 2000.

Figure 50—Land-cover configuration of representative sites in each condition category determined by BCI scores. Source: O'Connell and others 2000.

Figure 51—Ecoregion sections and analytical approaches used were based on amount of data available only within the MAIA region. Some ecoregion section averages (light green color) were based on plot data in the same ecoregion section that was both within and external to the MAIA region. The plot value was the actual value if the plot was measured in 1998, and an estimated value based on previous plot measurements otherwise. Source: National Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

Figure 52—Average percent foliar transparency of hardwood trees in 1998 by ecoregion section in the MAIA region (colored polygons), derived from the average foliar transparency of hardwood crowns at each FHM plot (solid black dots) in each ecoregion section in 1998. The plot value was the actual value if the plot was measured in 1998, and an estimated value based on previous plot measurements otherwise. Note legend also gives average percent foliar transparency for hardwood trees in other ecoregion sections in the U.S. in 1998 for comparison to MAIA region. Data collected 1991-1998 in Delaware, Maryland, New Jersey, and Virginia; in Pennsylvania in 1995 and 1998; in North Carolina in 1998. Source: National Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

Figure 53—Average annual change in percent foliar transparency of hardwood tree for the period of record for each State by ecoregion section (colored polygons), derived from the average foliar transparency of hardwood crowns at each FHM plot (solid black dots) in each ecoregion section in 1998. The plot value was the actual value if the plot was measured in 1998, and an estimated value based on previous plot measurements otherwise. Note legend also shows annual percent change in hardwood foliar transparency for ecoregion sections outside of the MAIA region for perspective. Data collected 1991-1998 in Delaware, Maryland, New Jersey, and Virginia; in Pennsylvania in 1995 and 1998; in North Carolina in 1998.
Source: National Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

Figure 54—Average percent crown dieback of hardwood trees in 1998 by ecoregion section (colored polygons) in the MAIA region derived from the average crown dieback of hardwood crowns at each FHM plot (solid black dots) in each ecoregion section in 1998. The plot value was the actual value if the plot was measured in 1998, and an estimated value based on previous plot measurements otherwise. Note legend also gives average percent crown dieback for hardwood trees in other ecoregion sections in the U.S. in 1998 for comparison to MAIA region. Data collected 1991-1998 in Delaware, Maryland, New Jersey, and Virginia; in Pennsylvania in 1995 and 1998; in North Carolina in 1998. Source: National Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

Figure 55—Average annual change in percent dieback of hardwood trees for the period of record for each State by ecoregion section (colored polygons), derived from the average foliar dieback of hardwood crowns in each FHM plot (solid black dots) in each ecoregion section in 1998. The plot value was the actual value if the plot was measured in 1998, and an estimated value based on previous plot measurements otherwise. Note legend also shows annual percent change in hardwood dieback for ecoregion sections outside of the MAIA region for perspective. Data collected 1991-1998 in Delaware, Maryland, New Jersey, and Virginia; in Pennsylvania in 1995 and 1998; in North Carolina in 1998. Source: National Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

Figure 56—Average percent foliar transparency of softwood trees in 1998 by ecoregion section (colored polygons). The black circles show the average softwood foliar transparency at each FHM plot in 1998; the plot value is the actual value if the plot was measured in 1998 and an estimated value based on previous measurements otherwise. Data collected 1991-1998 in Delaware, Maryland, New Jersey, and Virginia; in Pennsylvania in 1995 and 1998; in North Carolina in 1998. Source: National Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

Figure 57—Average annual change in percent foliar transparency of softwood trees for the period of record in each State by ecoregion sections (colored polygons). The black circles show the average foliar transparency of softwood crowns at each FHM plot in 1998. Data collected 1991-1998 in Delaware, Maryland, New Jersey, and Virginia; in Pennsylvania in 1995 and 1998; in North Carolina in 1998. Source: National Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

Figure 58—Average percent crown dieback of softwood trees in 1998 by ecoregion section (colored polygons). The black circles show the average softwood crown dieback at each FHM plot in 1998; the plot value is the actual value if the plot was measured in 1998 and an estimated value based on previous measurements otherwise. Data collected 1991-1998 in Delaware, Maryland, New Jersey, and Virginia; in Pennsylvania in 1995 and 1998; in North Carolina in 1998. Source: National Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

Figure 59—Average annual change in percent dieback of softwood trees for the period of record in each State by ecoregion sections (colored polygons). The black circles show the average dieback of softwood crowns at each FHM plot in 1998. Data collected 1991-1998 in Delaware, Maryland, New Jersey, and Virginia; in Pennsylvania in 1995 and 1998; in North Carolina in 1998. Source: National Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

Figure 60—Average percent foliar transparency of hardwood trees in 1998 by major HUC4 watershed (colored polygons). The black circles show the average foliar transparency of hardwood crowns at each FHM plot in 1998; the plot value is the actual value if the plot was measured in 1998 and an estimated value based on previous measurements otherwise. Data collected 1991-1998 in Delaware, Maryland, New Jersey, and Virginia; in Pennsylvania in 1995 and 1998; in North Carolina in 1998. Source: Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

Figure 61—Average percent crown dieback of hardwood trees in 1998 by major HUC4 watersheds (colored polygons). The black circles show the average hardwood crown dieback at each FHM plot in 1998; the plot value is the actual value if the plot was measured in 1998 and an estimated value based on previous measurements otherwise. Data collected 1991-1998 in Delaware, Maryland, New Jersey, and Virginia; in Pennsylvania in 1995 and 1998; in North Carolina in 1998. Source: National Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

Figure 62—Average percent foliar transparency of softwood trees in 1998 by majorHUC4 watershed (colored polygons). The black circles show the average softwood foliar transparency at each FHM plot in 1998; the plot value is the actual value if the plot was measured in 1998 and an estimated value based on previous measurements otherwise. Data collected 1991-1998 in Delaware, Maryland, New Jersey, and Virginia; in Pennsylvania in 1995 and 1998; in North Carolina in 1998. Source: National Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

Figure 63—Average percent crown dieback of softwood trees in 1998 by major HUC4 watershed (colored polygons). The black circles show the average softwood crown dieback at each FHM plot in 1998; the plot value is the actual value if the plot was measured in 1998 and an estimated value based on previous measurements otherwise. Data collected 1991-1998 in Delaware, Maryland, New Jersey, and Virginia; in Pennsylvania in 1995 and 1998; in North Carolina in 1998. Source: National Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

Figure 64—Average percentage of hardwood trees per plot that had any significant damage to the roots, trunk, bole, or crown by major HUC4watershed (colored polygons). The closed circles indicate the Damage Severity Index (based on the type of damage, severity, and location on the tree) value of hardwood trees on each FHM plot. Data collected 1991-1998 in Delaware, Maryland, New Jersey, and Virginia; in Pennsylvania in 1995 and 1998; in North Carolina in 1998. Source: National Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

Figure 65—Average percentage of softwood trees per plot that had any significant damage to the roots, trunk, bole, or crown by major HUC4 watershed (colored polygons). The black circles indicate the Damage Severity Index (based on the type of damage, severity, and location on the tree) value of softwood trees on each FHM plot. Data collected 1991-1998 in Delaware, Maryland, New Jersey, and Virginia; in Pennsylvania in 1995 and 1998; in North Carolina in 1998. Source: National Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

Figure 66—Tree mortality volume by ecoregion section expressed as the ratio of annual mortality volume to annual growth volume (colored polygons). Mortality ratio volumes of 1.0 indicate that there was no net gain in tree volume on the plot. The closed black circles represent plot-level values of the ratio of the average diameter of trees that died to the average diameter of the surviving trees on each plot (DDLD ratios). DDLD ratios of 1.0 indicate that on average the trees that died were as large as the surviving trees. Data collected 1991-1998 in Delaware, Maryland, New Jersey, and Virginia; in Pennsylvania in 1995 and 1998. Source: National Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

Figure 67—Tree mortality by major HUC4 watershed expressed as the ratio of annual mortality volume to annual growth volume (colored polygons). The black circles represent the ratio of the average diameter of trees that died to the average diameter of the surviving trees on each plot. Data collected 1991-1998 in Delaware, Maryland, New Jersey, and Virginia; in Pennsylvania in 1995 and 1998. Source: Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

Figure 68—Forest stand productivity in MAIA counties as average net annual growth of trees (gross volume growth minus mortality volume) in cubic feet per acre per year circa 2000. Source: Forest Inventory and Analysis program data; (http://fia.fs.fed.us/).

Figure 69—Forest stand volume in MAIA counties as average tree volume in cubic feet per acre prior to 2000. Source: Forest Inventory and Analysis program data; (http://fia.fs.fed.us/).

Figure 70—Age of forest stands in MAIA counties based on average size of trees. Ages relate to industry standards for potential tree harvest, and do not reflect ecological maturation. Immature = stands with the majority of trees 5 through 10 inches diameter at breast height (d.b.h.); mixed = stands with about half the trees 5 to 10 inches d.b.h. and half the trees greater than 10 inches d.b.h.; and harvestable = stands with the majority of trees greater than 10 inches d.b.h. Source: Forest Inventory and Analysis program data; (http://fia.fs.fed.us/).

Figure 71—Forest stand density in MAIA region counties, in basal area feet squared per acre (ft^2 / ac) per unit area circa 2000. Source: Forest Inventory and Analysis program data; (http://fia.fs.fed.us/).

Figure 72—Volume of annual hardwood production in the MAIA region in decades 1970s and 1980s. Source: Forest Inventory and Analysis program data; (http://fia.fs.fed.us/).

Figure 73—Volume of annual softwood production in the MAIA region in decades 1970s and 1980s. Source: Forest Inventory and Analysis program data; (http://fia.fs.fed.us/).

Figure 74—Share by State of total timber production in the MAIA region in decades 1970s and 1980s. Source: Forest Inventory and Analysis program data; (http://fia.fs.fed.us/).

Figure 75—Real prices of delivered black cherry sawlogs in Pennsylvania 1987 to 1997. Low, medium, and high grades correspond to USFS grades F3, F2, and F1, respectively. Source: Pennsylvania Woodlands, Timber Market Report, quarterly reports; (http://www.sfr.psu.edu/TMR/TMR.htm).

Figure 76—Real prices of delivered red oak sawlogs in Pennsylvania by low, medium, and high grades from 1987 to1997, corresponding to USFS grades F3, F2, and F1, respectively. Source: Pennsylvania Woodlands, Timber Market Report, quarterly reports; (http://www.sfr.psu.edu/TMR/TMR.htm).

Figure 77—Real prices of delivered white oak sawlogs in Pennsylvania 1987 to 1997, by low, medium, and high grades, corresponding to USFS grades F3, F2, and F1, respectively. Source: Pennsylvania Woodlands, Timber Market Report, quarterly reports; (http://www.sfr.psu.edu/TMR/TMR.htm).

Figure 78—Real prices of delivered pine sawlogs in western and central-far eastern North Carolina 1977 to1997. Source: Timber Mart South, monthly and quarterly reports; (http://www.tmart-south.com/).

Figure 79—Real prices of delivered pine sawlogs in western and eastern Virginia 1977 to 1997. Source: Timber Mart South, monthly and quarterly reports; (http://www.tmart-south.com/).

Figure 80—Real prices of stumpage hardwood pulpwood in Pennsylvania by low, medium, and high grades, corresponding to USFS grades F3, F2, and F1, respectively, 1987 to 1997. Source: Pennsylvania Woodlands, Timber Market Report, quarterly reports; (http://www.sfr.psu.edu/TMR/TMR.htm).

Figure 81—Real prices of delivered pine pulpwood in western and central-eastern North Carolina 1977 to 1997. Source: Timber Mart South, monthly and quarterly reports; (http://www.tmart-south.com/).

Figure 82—Real prices of delivered pine pulpwood in western and eastern Virginia 1977 to 1997. Note discontinuity in years 1985 and 1986 due to unavailability of price data. Source: Timber Mart South, monthly and quarterly reports; (http://www.tmart-south.com/).

Figure 83—Sawmill locations in the MAIA region in 1990s. Source: Pye 1999.

Figure 84—Pulp mill locations in the MAIA region in 1990s. Source: Pye 1999.

Figure 85—Other mills in the MAIA region in 1990s.The Other category represents miscellaneous mills that produce specialty products like flooring, cabinets, and plywood. Source: Pye 1999.

Figure 86—Wage and salary employment (number of employees) in lumber and wood products (SIC 24), furniture and fixtures (SIC 25), and paper and allied products (SIC 26) in the MAIA region 1975 to 1995. Source: Department of Labor, unemployment insurance database, ES-202; (http://www.bls.gov/bls/blswage.htm).

Figure 87—Real wages and salaries (1982=100) for lumber and wood products (SIC 24), furniture and fixtures (SIC 25), and paper and allied products (SIC 26) in the MAIA region 1975 to 1995. Source: Department of Labor, unemployment insurance database, ES-202; (http://www.bls.gov/bls/blswage.htm).

Figure 88—Real wages per job in lumber and wood products (SIC 24), furniture and fixtures (SIC 25), paper and allied products (SIC 26), and all sectors of the MAIA economy 1975 to 1995. Source: Department of Labor, unemployment insurance database, ES-202; (http://www.bls.gov/bls/blswage.htm).

Figure 89—Black bear population and harvest in MAIA region states 1975 to 1993. Estimates include counties outside of the MAIA region for New York, New Jersey, and North Carolina. Source: Curtis H. Flather, personal communication; (http://www.fs.fed.us/rm/analytics/staff/flather.html).

Figure 90—Gray squirrel population and harvest in MAIA region states 1975 to 1993. Estimates include counties outside of the MAIA region for New York, New Jersey, and North Carolina. Source: Curtis H. Flather, personal communication; (http://www.fs.fed.us/rm/analytics/staff/flather.html).

Figure 91—White-tailed deer population and harvest in MAIA region states 1975 to 1993. Estimates include counties outside of the MAIA region for New York, New Jersey, and North Carolina. Source: Curtis H. Flather, personal communication; (http://www.fs.fed.us/rm/analytics/staff/flather.html).

Figure 92—Wild turkey population and harvest in MAIA region states 1975 to 1993. Estimates include counties outside of the MAIA region for New York, New Jersey, and North Carolina. Source: Curtis H. Flather, personal communication; (http://www.fs.fed.us/rm/analytics/staff/flather.html).

Figure 93—Concentration of exchangeable calcium in soil surface mineral layers (0 to 4 inches) on FHM plots in the MAIA region in 1998. Source: National Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

Figure 94—Concentration of exchangeable magnesium in soil surface mineral layers (0 to 4 inches) on FHM plots in the MAIA region in 1998. Source: National Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

Figure 95—Soil pH values in surface mineral layers (0 to 4 inches) on FHM plots in the MAIA region in1998. Source: National Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

Figure 96—Soil pH values for the subsurface mineral layer (4.1 to 8 inches) on FHM plots in the MAIA region in1998. Source: National Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

Figure 97—Interactive effects of forest management practices on processes affecting stream quality.

Figure 98—Cumulative effects of forest management on a stream and associated watershed. Changes in arrow sizes reflect changes in the magnitude of the process before and after timber harvest.

Figure 99—The Mid-Atlantic Highlands Area is circumscribed by the dark border. About 360 stream reaches, representing over 110,000 stream kilometers in the area, were sampled (dots). Source: U.S. EPA; (http://www.epa.gov/reva/vulnerability/).

Figure 100—Land use as a function of stream size or order in Mid-Atlantic Highlands Area region. Source: U.S. EPA; (http://www.epa.gov/reva/vulnerability/).

Figure 101—Median Ephemeroptera, Plecoptera, and Trichoptera (EPT) scores by stream order. EPT scores are based on type and number of these three groups of stream insects. Source: U.S. EPA; (http://www.epa.gov/reva/vulnerability/).

Figure 102—Mid-Atlantic Highlands Area stream miles with sport fish, fish, or no fish. Note that low numbers of fish does not necessarily mean a stream is degraded. Source: U.S. EPA; (http://www.epa.gov/reva/vulnerability/).

Figure 103—Total carbon by weight in the litter samples collected from forested Forest Health Monitoring plots in the MAIA region in 1998. Source: National Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

Figure 104—Total carbon by weight in the mineral surface horizon (0 to 4 inches) collected below the litter floor on forested Forest Health Monitoring plots in the MAIA region in 1998. Source: National Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

Figure 105—Total carbon by weight in the mineral subsurface horizon (4.1 to 8 inches) on forested Forest Health Monitoring plots in the MAIA region in 1998. Source: National Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

Figure 106—Forest stand distribution in the MAIA region. Maple-beech-birch trees are also known as Northern Hardwoods. Source: USDA Forest Service, Forest Inventory and Analysis program's Eastwide Database; (http://www.srs.fs.fed.us/sustain/data/fia/eastwideguide.htm).

Figure 107—Forest cover type by MAIA region state. Source: USDA Forest Service, Forest Inventory and Analysis program's Eastwide Database; (http://www.srs.fs.fed.us/sustain/data/fia/eastwideguide.htm).

Figure 108—Oak-hickory forest-type distribution in the MAIA region. Source: USDA Forest Service, Forest Inventory and Analysis program's Eastwide Database; (http://www.srs.fs.fed.us/sustain/data/fia/eastwideguide.htm).

Figure 109—Maple-beech-birch forest-type distribution in the MAIA region. Source: USDA Forest Service, Forest Inventory and Analysis program's Eastwide Database; (http://www.srs.fs.fed.us/sustain/data/fia/eastwideguide.htm).

Figure 110—Other Conifer forest-type distribution in the MAIA region. Source: USDA Forest Service, Forest Inventory and Analysis program's Eastwide Database; (http://www.srs.fs.fed.us/sustain/data/fia/eastwideguide.htm).

Figure 111—Loblolly-shortleaf pine forest-type distribution in the MAIA region. Source: USDA Forest Service, Forest Inventory and Analysis program's Eastwide Database; (http://www.srs.fs.fed.us/sustain/data/fia/eastwideguide.htm).

Figure 112—Oak-gum-cypress forest-type distribution in the MAIA region. Source: USDA Forest Service, Forest Inventory and Analysis program's Eastwide Database; (http://www.srs.fs.fed.us/sustain/data/fia/eastwideguide.htm).

Figure 113—Distribution of tree genera in the MAIA region. Source: USDA Forest Service, Forest Inventory and Analysis program's Eastwide Database; (http://www.srs.fs.fed.us/sustain/data/fia/eastwideguide.htm).

Figure 114—Forest stand-size class distribution in the MAIA region based on three commercial classes: harvestable (sawtimber-sized stands where average tree is greater than 5 inches d.b.h.); immature (pole-sized stands where average tree is a sapling less than 5 inches d.b.h.); and regeneration (stands where average tree saplings are 1- to 4.9 inches d.b.h.). Size classes reflect commercial standards for harvesting and not ecological maturity. Source: USDA Forest Service, Forest Inventory and Analysis program's Eastwide Database; (http://www.srs.fs.fed.us/sustain/data/fia/eastwideguide.htm).

Figure 115—Forest stand-size class distribution by MAIA region states. Source: USDA Forest Service, Forest Inventory and Analysis program's Eastwide Database; (http://www.srs.fs.fed.us/sustain/data/fia/eastwideguide.htm).

Figure 116—Distribution of numbers of trees and biomass in five size-classes in the MAIA region. Source: USDA Forest Service, Forest Inventory and Analysis program's Eastwide Database; (http://www.srs.fs.fed.us/sustain/data/fia/eastwideguide.htm).

Figure 117—Three classes of tree size distribution based on the average of five d.b.h. size classes (0 to 5; 5.1 to 10; 10.1 to 15; 15.1 to 20; and greater than 20 inches) in the MAIA region. Source: USDA Forest Service, Forest Inventory and Analysis program's Eastwide Database; (http://www.srs.fs.fed.us/sustain/data/fia/eastwideguide.htm).

Figure 118—Lichen species richness in the MAIA region. Source: National Forest Health Monitoring program data 1991 to 1998; (http://fhm.fs.fed.us/).

Figure 119—HUC-8 watersheds and Breeding Bird Survey routes in the MAIA region. Source: Seaber and others 1987; Sauer and others 1999; (http://www.mbr-pwrc.usgs.gov/bbs/).

Figure 120—Species richness of forest birds in HUC-8 MAIA region watersheds in 1975 based on Breeding Bird Survey data. Source: Seaber and others 1987; Sauer and others 1999; (http://www.mbr-pwrc.usgs.gov/bbs/).

Figure 121—Species richness of forest birds in HUC-8 MAIA region watersheds in 1990 based on Breeding Bird Survey data. Source: Seaber and others 1987; Sauer and others 1999; (http://www.mbr-pwrc.usgs.gov/bbs/).

Figure 122—Physiographic strata of Omernik Level III ecological units and Breeding Bird Survey routes retained for estimating relative species richness in the MAIA region. Source: (http://nationalatlas.gov/mld/ecoomrp.html); (http://www.mbr-pwrc.usgs.gov/bbs/).

Figure 123—Relative species richness of forest birds on Breeding Bird Survey routes in HUC-8 MAIA region watersheds in 1992. Source: Seaber and others 1987; Sauer and others 1999; (http://www.mbr-pwrc.usgs.gov/bbs/).

List of Table Captions

GLOSSARY

Abiotic—Non-living, non-biological, non-organic; physical. In ecological context, the physical and chemical attributes or factors that influence biological components and processes in an ecosystem.

Accessible forest land—Forestland that is within the population of interest, is accessible, can be sampled safely, and meets the definition of forestland (see Forest land).

Acre—Unit of land containing 43,560 square feet, or 0.4 hectare of area.

Afforestation—Return of forests in an area where the preceding vegetation or land use was not forest.

Age at breast height—Number of annual growth rings between the bark and the center of the tree at 4.5 feet above the root collar on the bole of a tree. Only relevant for trees that produce one distinctive (visible) growth ring per year.

Age class—An interval, commonly 10 or 20 years, into which the age range of a tree stand is divided for classification or analytical use. Also pertains to the trees included in such an interval. For example, trees ranging in age from 21 to 40 years fall into a 30-year age class, the rounded midpoint of the interval from 21 to 40 years.

Agricultural land—Land managed for crops, pasture, or other agricultural uses.

Artificial regeneration—Renewal of the forest by direct planting or seeding; establishing a new stand of trees by planting seeds or seedlings by hand or machine.

Aspect—Compass direction that a slope faces. Also called exposure. Typically measured with a compass.

Azimuth—Horizontal angle to an object; measured in degrees clockwise from north. Typically measured with a compass.

Barren land—Areas of very limited plant life. Examples are mudflows, talus slopes, beaches, dunes, dry salt-flats, bare rock.

Basal area—Cross-sectional area of the stem of a plant in a stand, generally expressed as square units per unit area. For trees measured at 4.5 feet (1.37 m) above ground; for shrubs and forbs measured at the root crown.

Base saturation—Measure of the extent to which soil minerals and organic matter are chemically saturated with exchangeable cations (e.g., potassium, magnesium, and calcium) other than hydrogen and aluminum. Expressed as a percentage of cation exchange capacity (CEC).

Baseline—Reference line of sight located and measured on both an aerial photo and the ground. Also used analytically as a term denoting reference condition, state, or value of a biological system.

Bioindicator species—Any plant or animal that responds to a force in a way that is distinctive or unique for that force, and the response to the force can be qualitatively and quantitatively measured. Examples are plants that respond to ambient ozone air pollution with distinct visible foliar symptoms that are easy to diagnose from other forces acting on the plants.

Biomass—Total biologically-generated material in a forest. Refers to both plants (above and below-ground parts) and animals. All of the organic (carbon-based) material in a given area.

Biosphere—Interacting world of all living creatures with the non-living physical and chemical factors.

Biotic—Biological or living components or processes in an ecosystem.

Blow down— Knocked-down by the wind. Also see Windfall.

BLUP—Best Linear Unbiased Predictors. BLUPs are "best" because they have the minimum mean square error; "linear" because they are linear functions of the data; "unbiased" because the average value of the estimate is equal to the average value of the quantity being estimated; and "predictors" because they predict random effects.

Board foot—Volume of wood that is 1 foot wide, 1 foot long and 1 inch thick, equal to 1/12 of a cubic foot of wood.

Bole—Woody trunk or main stem of a tree.

Bottomland forest—Lands typically under water part of the year, e.g., swamps, river bottoms, piedmont bottomlands, or cypress strands or domes.

Breast height—Distance of 4.5 feet above the ground (near *breast height* on a person about 6 feet tall). Refers to point on bole where diameter measurements and tree cores are taken on tree species.

Bulk density—Mass of soil per unit volume. A measure of the ratio of pore space to solid materials in a given soil. Expressed in units of grams per cubic cm of oven dry soil.

Bureau Of Land Management (BLM)—Land administered by the U.S. Department of Interior's Bureau of Land Management.

Canker—Localized injury response (swelling, decay) on stem, branch, or root; often caused by pathogens or insects.

Canopy—Above-ground cover of branches, foliage, seeds/cones formed collectively by adjacent, over-lapping tree crowns. The portion of sky not visible because of trees or tall shrubs.

Canopy closure—Percentage of ground area covered by the vertically projected cross-sections of tree crowns, including branches, foliage, reproductive structures, and upper tree bole. Same as canopy cover; the inverse of canopy opening.

Canopy cover—Proportion of ground, usually expressed as a percentage that is occupied by the perpendicular projection down on it of the aerial parts of the vegetation (trees, shrubs, herbs, vines, etc.) under consideration. The percentage of shade covering the ground within a defined area if the sun was directly overhead.

Canopy density—Density of the canopy either directly overhead or obliquely at different angles depending on desired information. The additive density when considering multiple layers of branches and leaves often exceeds 100% compared to ground area underneath. Related to Leaf Area Index, but includes all parts of tree crown (e.g., upper stem, branches, foliage, cones).

Canopy opening—Percentage of ground area not covered by the vertically projected cross-sections of tree crowns. Areas open to sky and allowing overhead sunlight to hit the ground. The inverse of Canopy Closure.

Carrying-capacity—Refers to animal populations in any ecosystem and indicates the maximum population that can be supported by the resources available in that ecosystem.

Cation—Positively charged ion. In general, mono-valent (e.g., potassium=K^{+}) and di-valent cations (e.g., calcium^{++}, magnesium^{++}) are important plant nutrients, and some tri-valent cations (e.g., aluminum^{+++}) are toxic.

Cation Exchange Capacity (CEC)—Sum total of the exchangeable cations that a soil can adsorb at a specific soil pH. Expressed in units of centimoles of positive charge per kilogram of soil.

Climax—In plant ecology refers to a final stage in succession that is composed of species that are continually self-replacing. A community that is stable with respect to species composition, although relative abundances might fluctuate.

Climax species—One or more plant species that replaces itself over time, and is part of a final seral stage, or climax stage, of a community. Climax species are not replaced by other species, as long as the site is free from major non-historic disturbances.

Coarse woody debris (CWD)—Dead pieces of wood including downed, dead tree and shrub boles, large limbs, and other woody pieces that are severed from their original source of growth or are leaning more than 45 degrees from vertical. Generally pieces of wood > 3.0 inches in diameter.

Codominant tree—Trees with crowns at the general level of the crown canopy. Crowns receive full light from above but little direct sunlight from the sides. Usually they have medium-sized crowns and are somewhat crowded from the sides and affected by adjacent trees. In stagnated stands, co-dominant trees have small-sized crowns and are crowded on the sides. See Crown Class.

Colonization—Introduction of species into an area where historically that species was absent from the evolutionary processes that led to the existing species composition.

Compacted Trail—Soil compaction results from many passes of heavy machinery or vehicles, grazing animals, or human use (e.g., foot trails). Compacted soils have elevated bulk density, low water penetrability, hinder growth of roots and plants on the surface, and are vulnerable to erosion.

Condition—Area of relatively homogeneous vegetation, geological form, aquatic system, human-created structure, management type, etc. Used in fixed-area plot measurements to scale-up for population estimates.

Conifer—Cone-bearing trees, mostly evergreens, with needle or scale-like leaves belonging to the botanical division Coniferophyta. Also referred to generically as Softwoods.

Conk—Visible fruiting body of a wood-destroying fungus, usually indicating rot in the underlying wood.

Contiguous forestland—Forested areas at least 120 feet (36.6 m) wide according to USFS FIA definitions. Boundaries are non-forested areas at least 120 feet wide, and are not defined by ownership, forest type, or age class. Clearcuts may not have tree cover but are still considered forest if intent is tree regrowth.

Cooperage—Barrels of selected wood species, sometimes burned on inside, used to age and flavor alcoholic beverages.

Cord—Stack of wood equivalent to 128 cubic feet of wood and the spaces in-between; standard dimensions are 4 by 4 by 8 feet.

Cover type—Designation based upon the plant species forming the plurality of composition within a given area (e.g., Oak-Hickory).

Crook—Abrupt bend in a tree or log.

Cropland—Land under cultivation within the past 24 months, including orchards and land in soil-improving crops, but excluding land cultivated in developing improved pasture. In FIA, to qualify in this or other non-forest category, the area must be at least 1 acre in size and at least 120 feet in width.

Crown—Part of a tree or woody plant bearing live branches with foliage or foliage sprouting from the tree bole.

Crown class—Classification of trees based on dominance in relation to adjacent trees in the stand as indicated by crown development and amount of light received from above and the sides. Crown classes recognized by FIA include Open Grown, Dominant Trees, Codominant Trees, Intermediate Trees, and Overtopped (sometimes called Suppressed Trees).

Crown cover—Percentage of the ground surface covered by a vertical projection of crowns from above.

Crown width—Maximum horizontal span of the crown of a tree or shrub. Often averaged by measuring the maximum width and perpendicular to maximum width on asymmetrical crowns.

Cull tree—Live trees that are unsuitable for the production of roundwood products, now or prospectively. See Rough Trees and Rotten Trees.

Cut—Volume of trees cut between time t and time t+1, where t is the initial measurement and t+1 is the terminal inventory. This is a component of change that is usually expressed in terms of growing-stock or all-live volume. Trees felled or killed in conjunction with a harvest or silvicultural operation (whether they are utilized or not) are included, but trees on land diverted from forest to nonforest (diversions) are excluded.

Decay class—Stage of decay of standing dead snag or large down woody materials or debris. Typically a five class system where decay class 1 = small dead twigs and/or foliage is still there; decay class 5=raised area of broken chunks of dead wood where no specific parts (bole, branches, branch stubs, etc.) can be distinguished.

Defoliators (defoliated)—An insect that feeds upon and removes foliage from plants. In addition, chemicals or other means used to remove foliage.

Delivered log price—Price for logs (see Logs) when delivered to mills.

Deterministic—Every event is causally determined by an unbroken chain of prior events. Fully governed by causal laws resulting in only one possible state at any point in time.

Developed land—Developed land has been permanently removed from the natural resource land base (land or water). Developed lands include: (a) large tracts of urban and built-up land; (b) small tracts of built-up land, less than 10 acres in size; and (c) land outside of these built-up areas that is in roads, railroads, and associated right-of-ways.

Diameter at breast height (DBH, d.b.h., or dbh)—Point where the diameter of the tree stem or bole is measured, located at 4.5 feet above the ground *(breast height)* on the uphill side of a tree. The point of diameter measurement may vary on abnormally formed trees.

Discoloration—In FIA refers to loss of green pigment in foliage with the subsequent manifestation of red, yellow, brown, or other colors. Can be natural (e.g., Autumn) or caused by insects, pathogens, damages, etc.

Dominant species—Species with the highest percent of cover, usually in the uppermost dominant layer of trees. Refers to being floristically dominant or the most important in terms of biomass, density, height, coverage, etc.

Dominant tree—Trees with crowns extending above the general level of the crown canopy and receiving full light from above and partly from the sides. These trees are taller than the average trees in the stand and their crowns are well developed, but they could be somewhat crowded on the sides. Crown form or shape appears to be mostly free of influence from neighboring trees. See Crown Class.

Down woody material (DWM)—Coarse and fine woody pieces of trees and shrubs that have been uprooted (no longer supporting growth) or severed from the tree bole or root system, not self-supporting, and lying on the ground. Also called Down Woody Debris (DWD).

Down woody debris (DWD)—See Down Woody Material (DWM).

Dry weight—Oven or air dried weight of a material (organic or physical).

Duff—Soil layer dominated by organic material derived from the decomposition of plant and animal litter (see Litter Layer) and deposited on either an organic or a mineral surface. This layer has undergone sufficient decomposition that the source of this material (e.g., individual plant parts) can no longer be identified.

Ecotone—Transition area between 2 distinct communities possessing physical and biological attributes of both communities and sometime containing other species not found in either community. Often areas of high biological diversity.

Edge effect—Biological and physical effects associated where forested lands join other land use types (e.g., agriculture). These effects typically include increased light and temperature; introduction of invasive plant and animal species; and disturbances associated with a variety of human activities (e.g., dogs chasing wildlife; fireworks starting fires; etc.).

Endemic—Characteristic of a particular region or locality. Indigenous. Common in the endemic area but not found in other places.

Ericaceous—Generic term referring to the Ericaceae family of plants, also called heaths. Sometimes used also as a generic term for plants or foliage indicative of hot, dry climates and acidic soils.

Erosion—Loosening or wearing away of soil or rock surfaces by running water, wind, ice or other physical agents.

Erosion, rill—Type of erosion process in which numerous small channels of only several centimeters are formed.

Erosion, sheet—The removal of a relatively uniform layer of soil from the land surface by runoff water or wind.

Farm—Lands on which agricultural operations are conducted and from which $1,000 or more in agricultural products were sold during the year.

Federal Information Processing Standard (FIPS)—A unique code identifying U.S. States and counties (or units in Alaska).

Federal land—Ownership class of public lands owned by the U.S. Government.

Ferns and allies—Vascular plants that reproduce by spores. Include ferns, horsetails, clubmosses, spikemosses and quillworts.

Fiberboard—Fiberboard is a type of engineered wood product that is made out of wood fibers. Types of fiberboard (in order of increasing density) include particle board, medium-density fiberboard, and hardboard. Medium-density fiberboard (MDF) is often used in the furniture industry.

Fine woody debris (FWD)—Dead branches, twigs, wood splinters 0.1 to 2.9 inches in diameter. Smaller (< 3 inches) fractions of down woody material or debris.

Floc—Floc is a flake(s) of precipitate from a solution; the precipitate forms as floc or flakes at a concentration generally below the solubility limit of the solution; fine particulates that clump together. Floc is the result of the process of contact and adhesion whereby the particles in a dispersion solution form larger-size clusters.

Forb—Often broad-leaved herbaceous plants (often annuals) as distinguished from grasses, shrubs, and trees.

Forest floor—Entire thickness of organic material overlying the mineral soil, consisting of the litter, duff, and humus.

Forest industry land—Ownership class of private lands owned by a company or an individual(s) operating a primary wood-processing plant.

Forest land (FIA and FHM programs)—Land that is at least 10 percent stocked by forest trees of any size, or land formerly having such tree cover, and is not currently developed for a nonforest use. The minimum area for classification as forestland is one acre. Roadside, streamside, and shelterbelt strips of timber must have a crown width at least 120 feet wide to qualify as forest land. Unimproved roads and trails, streams and other bodies of water, or natural clearings in forested areas shall be classified as forest, if less than 120 feet in width or 1 acre in size. Grazed woodlands, reverting fields, and pastures that are not actively maintained are included if the above size qualifications are satisfied. (Also see definitions of nonforest land). Forest land is divided into timberland, reserved timberland, and woodland by FIA.

Forest land (NRI)—Land cover/use category that is at least 10 percent stocked (areal canopy cover of leaves and branches of 25 percent or greater of area) by single-stemmed woody species of any size that will be at least 4 meters tall at maturity. Also included is land-bearing evidence of natural regeneration of tree cover (cut over forest or abandoned farmland) and not currently developed for nonforest use. The minimum area for classification as forest area is 1 acre, and the area must be at least 100 feet wide.

Forest trees—Woody perennial plants having a well developed stem and usually more than 12 feet height at maturity. In FIA, all trees in the species list of the FIA manual (www.fia.fs.fed.us).

Forest type—Classification of forestland based upon and named for the tree species that forms the majority of live tree stocking. A forest type classification for a field location indicates the predominant live-tree species cover; hardwoods and softwoods are grouped to determine predominant group; and Forest Type is selected from the predominant group.

Forest type group—Combination of forest types that share closely associated species or site requirements.

Forked trees—A tree with piths that fork so that the new fork is greater than 1/3 the diameter of the main stem and branches out from the main steam at an angle of 45 degrees or less.

Foot slope—Topographic or physiographic position on a hill or mountain that is the initial area rising above flatter lands. The initial slope encountered when climbing a hill or mountain.

Fragmentation—Breaking into smaller pieces. In forestry, the process of reducing larger, contiguous blocks of forest into smaller, more isolated patches of forest.

Fuelwood—More typically called firewood. Generally refers to wood from trees often burned in fireplaces, stoves, etc. for heating. The best firewood produces relatively high BTU output per unit mass of wood.

Furnishing applications—Products that emphasis the beauty of wood. Sometimes called appearance applications. Examples are cabinets, flooring, furniture, etc.

Generalist—Biological term that refers to a species that has broad requirements for food, habitat, environmental conditions, etc. Opposite of a Specialist.

Grade—Reference to sawlog quality. Low (F1 in USFS), Medium (F2 in USFS), and High (F3 in USFS).

Graminoid—Grasses and grass-like plants, including sedges and rushes.

Grassland—Areas dominated by upland grasses and forbs. In rare cases, herbaceous cover is less than 25 percent, but exceeds the combined cover of the woody species present. These areas are usually not intensely managed, but are often utilized for grazing.

Growing stock trees—All live trees 5.0 inches DBH or larger (now or prospectively), except rough and rotten trees, from a timber perspective (paper, wood, etc.).

Growing stock volume—Net volume in cubic feet of live sawtimber and poletimber growing stock trees from 1-foot stump to a minimum 4.0-inch top of central stem. Net volume equals gross volume less deductions for rotten and missing bole sections.

Growth—Increase in diameter, basal area, height, and volume of individual trees, tree-stands, or other species over a given time period. Sometimes called increment.

Guild—Group of species that have similar requirements for habitat, food, reproduction, behavior, or other characteristics.

Hardwood—Tree species belonging to the botanical subdivision Angiospermae, class Dicotyledonous, usually broad-leaved and deciduous.

Hectare—A metric unit of area of 10,000 square meters and equal to 2.47 acres.

Herbaceous—Seed-producing annual, biennial, or perennial plant that does not develop persistent woody tissue, and dies down at the end of a growing season.

Humus—Soil layer dominated by organic material derived from the decomposition of plant and animal litter and deposited on either an organic or a mineral surface. This layer is distinguished from the litter and duff layers in that the latter is rough looking material, with coarse pieces still visible. Humus is more uniform in appearance (a dark, spongy, jelly-like substance) and amorphous (without any determinate shape or character) in structure.

Importance—In biodiversity denotes high abundance or cover of a species or species group.

Improved road—Road of any width maintained periodically, as evidenced by pavement, gravel, grading, ditching, and other improvements.

Indicator—Measurement or estimation of a component or process that is relatively easy to obtain under field conditions that can be qualitatively and/or quantitatively clearly related to a more complex system of components or processes. For example, the amount of tree foliage is an indicator of the potential for a tree to fix carbon, grow, survive, and reproduce.

Industrial applications—Products based on the strength and durability of wood. Examples are railroad ties, pallets, boxes, etc.

Integrity—Supporting and maintaining a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of other natural habitats in the same or similar region.

Intermediate tree—Trees that are shorter than dominants and co-dominant, but crowns extend into the canopy of co-dominant and dominant trees. They receive little direct light from above and none from the sides. Intermediate trees usually have small crowns and are very crowded from the sides. See Crown Class.

Intolerant—Refers to plants relatively incapable of developing and growing normally in the shade of, and in competition with, other trees.

Lake—Natural inland body of water, fresh or salt, extending over 4.5 acres or more and occupying a basin or hollow on the earth's surface, which may or may not have a current or single direction of flow.

Land area—In FIA, the area of dry land and land temporarily or partly covered by water, such as marches, swamps, and river flood plains; streams, sloughs, estuaries, and canals less than 200 feet wide; and lakes, reservoirs, and ponds less than 4.5 acres in area.

Land cover—Term that includes categories of what is currently on a unit of land. Land cover is the vegetation or other material that covers the land surface (e.g., rock, water, buildings). Often, but not always, related to land use.

Land use—Land use is the type of human activity (use) intended for the land (e.g., agriculture, urban, forest); it is often, but not always, related to land cover.

Leaf—Foliage of a plant (e.g., leaves, needles, fronds). Typically the main photosynthetic organ of a plant.

Leaf Area Index (LAI)—Amount of leaf surface area (one side only) over an area of ground directly below. Since leaves are often overlapping, the amount of leaf coverage per unit of ground area often exceeds 100 percent (LAI value of 1); thus, LAI values are often greater than 1.

Lichen—Organism generally recognized as a single plant that consists of a fungus and an algae or cyanobacterium living in symbiotic association. The fungi supply support to the organism, and the alga or cyanobacterium supply food and energy via photosynthesis.

Lichen plot—In FIA lichen plot is a circular area, total 0.935 acre with a 120-foot radius centered on subplot 1, and excluding the areas of the 4 subplots. Small amount of each lichen species is collected for post-field identification.

Litter—Undecomposed or only partially decomposed organic material that can be readily identified as whole or part of leaves, branches, twigs, etc.

Live tree—All living trees. All size classes, all tree classes, and both commercial and noncommercial species are included. See FIA field manual for list of tree species (www.fia.fs.fed.us).

Log—Segment of a tree 8 feet or longer. Minimum length of tree suitable for harvesting for processing into lumber, veneer, or other wood products.

Logging—Felling and removal of trees for producing commercial timber products or other silvicultural needs.

Macrolichen—Lichen species that is leafy, tufted, or hanging, and is easily separated from its substrate, as opposed to crustose lichens.

Macrophytic—Visible to the unassisted eye; as opposed to microscopic.

Maintained road—Any road, hard topped or other surfaces, that is plowed or graded periodically and capable of use by a large vehicle.

Margin—In economics, a margin is a set of constraints conceptualized as a border. A marginal change is the change associated with a relaxation or tightening of constraints — either change of the constraints, or a change in response to this change of the constraints.

Marsh—Low, wet areas characterized by heavy growth of weeds and grasses and an absence of trees because the soil is too wet too often to support trees.

Merchantable sawtimber top—In FIA, the point on the bole of sawtimber trees above which a sawlog cannot be produced. Minimum merchantable top is 7.0 inches DOB for softwoods and 9.0 inches DOB for hardwoods.

Mesic—Moist or wet. Typically used to refer to sites or species associated with high moisture in soils and high moisture requirements for species.

Metric to English conversions—1 inch (in) = 2.54 centimeter (cm); 1 cm = 0.3937 inch; 1 foot (ft) = 12 inches = 30.48 cm; 3 feet = 0.9144 m; 1 meter (m) = 39.37 inches; 1 m = 3.2808 feet; 1 mile (5280 feet) = 1.6093 Kilometer (km); 1 km = .62137 mile; 1 acre = 0.405 hectare (ha); 1 hectare = 2.47 acres; 1 acre = 43,560 square feet; 1 hectare = 10,000 square meters; 1 square foot (sq ft) = .0929 square meter (sq m); I sq m = 10.76 sq ft; 1 cubic foot (cf) = 0.028317 cubic meter; 1 cubic meter = 35.315 cf.

Malathion—Malathion is an organophosphate parasympathomimetic which binds irreversibly to cholinesterase. Malathion is an insecticide of relatively low human toxicity.

Mineral soil—Soil consisting predominantly of products derived from the weathering of rocks (e.g., sands, silts, and clays). Upper layers contain minute particles of organic material that leaches in from the duff or humus layers.

Mortality—Death of an organism. Typically, tree mortality is of greatest concern because trees provide the structural and ecological framework in forest ecosystems. Primary interests are causal agents, timing, number or volume, and what effects if any on the system.

Mosses—Plants with leafy green shoots that lack complex vascular systems and roots, often growing in tufts or clusters on the ground, decaying wood or on rocks.

Mottling—Splotches or blotches of different colors or shades of colors interspersed with the dominant matrix color. In some soils, this may be evidence of compaction.

Multi-brood—Species (e.g., birds) that produce more then one group of offspring in a season.

Municipal land—Land owned by municipalities or land leased by them for more than 50 years.

National Forest land—Ownership class of Federal lands, designated by Executive order or stature, as National Forests or purchase units, and other lands under the administration of the Forest Service including Experimental Forests and Bankhead-Jones Title III lands.

Natural Resource Conservation Service (NRCS) plants—The USDA Natural Resource Conservation Service's Plants Database provides standardized information about plants. In FIA, source for PLANT codes used to describe species identified by the P3 Vegetation Indicator. NRCS also provides Natural Resource Inventory (NRI) data on land use in U.S.

Net change—In FIA, Net Change = Net Growth – Removals.

Net growth—In FIA, Net Growth = Gross Growth – Mortality.

Net volume—In FIA, gross volume less deductions for sound and rotten defects. Net volume is gross cubic-foot volume less deductions for rotten and missing bole sections on poletimber and sawtimber trees.

Niche—Space or habitat available for use by an existing or new species of plant or animal. Niches are produced by environmental conditions and the complex interactions of physical environments with existing biological species.

Nonforest land—In FIA, land that does not support or has never supported, forests, and lands formerly forested where use for timber management is precluded by development for other uses. Includes areas used for crops, improved pasture, residential areas, city parks, improved roads of any width and adjoining rights-of-way, power line clearings of any width, and noncensus water. If intermingled in forest areas, unimproved roads and nonforest strips must be more than 120 feet wide, and clearings, etc., more than 1 acre in size, to qualify as nonforest land.

Nonstocked area—In FIA, forest land with less than 20 percent tree-crown cover or less than 10 percent stocked with growing-stock trees.

Nonstockable—Areas of forest land that are not capable of supporting trees because of the presence of rock, water, etc.

Non-stocked stands—Forested lands less than 10 percent stocked with live trees.

Northern Hardwoods—Hardwood tree species common to Northern forests, also called the Maple-Beech-Birch forest type named for the common occurrence of these genera.

Old growth stands—In FIA, stands that conform to the definitions developed by Forest Service Regions for the major forest type groups. Typically stands that have trees greater than 50% of maximum age; multiple structural layers; large snags and down wood; high biological diversity, and relatively undisturbed by human activity for long periods of time.

Old growth trees—In FIA, trees that meet regional definitions. Typically trees greater then 50% maximum age and large size (relative to species type).

Open grown trees—Trees with crowns that received full sunlight from above and from all sides throughout most of its life, particularly during its early developmental period. See Crown Class.

Orchards—Areas dominated by fruit or nut trees planted on a regular and generally consistent row and plant spacing. Stands are planted to produce a fruit or nut crop. Examples include areas used for the production of apples, peaches, oranges, pecans, walnuts, cherries, and bananas.

Organic soil—Soils within organic horizon that is greater than 8 inches in thickness. These soils are prevalent in wetland areas such as bogs and marshes, and may be frequently found in certain regions of the country.

Other farmland—Farmland not classified elsewhere, including farmsteads, barns, etc.

Other Federal lands—Lands administered by Federal agencies and not reported separately. These may include wildlife refuges administered by the Fish and Wildlife Service, U.S. Department of Interior, and military reservations administered by the Department of Defense.

Other forest lands—In FIA, forest land other than timberland and reserved timberland. It includes available and reserved low-productivity forestland, which is incapable of producing 20 cubic feet of growing stock per acre annually under natural conditions because of adverse site conditions such as sterile soil, dry climate, poor drainage, high elevation, steepness, or rockiness.

Other private lands—Lands in private ownership and not reported separately. These may include coal companies, land trusts, and other nonindustrial private landowners.

Other public lands—Public land other than National Forests. See Other Federal Lands.

Other removals—Un-utilized growing-stock volume that is cut or otherwise killed during cultural operations, such as timber-stand improvements, or during forestland clearing operations.

Overstory trees—Trees that form the uppermost canopy layer in a forest. The layer of the highest trees in the forest.

Overtopped—Trees with crowns entirely below the general level of the crown canopy; these trees receive no direct sunlight either from above or the sides. See Crown Class.

Owner group—In FIA, a variable combining owner classes into the following groups: Forest Service, Other Federal Agency, State and Local Government, and Private. Differing categories of Owner Groups on a plot require different conditions.

Ozone—Chemically O_3; a regional, gaseous air pollutant produced through sunlight-driven chemical reactions of NO_2, O_2, and hydrocarbons in the atmosphere. Enters leaf stomata and causes visible, distinct foliar injury on susceptible trees, shrubs, and herbaceous species; can reduce growth, increase other stressor sensitivity; accelerate mortality. Criteria pollutant regulated by EPA.

Ozone bioindicator—Use of an ozone-susceptible plant species to assess ambient ozone effects and indicate the risk of ozone stress on ecosystem structure and function.

Ozone bioindicator site—In FIA, an open area used for ozone injury evaluations on ozone bioindicator species. The site must meet guidelines regarding size, condition, and number of ozone bioindicator plants for valid ozone injury evaluations.

Ozone biomonitoring—Using the foliar response of of ozone bioindicator species to monitor the extent and severity of ambient ozone pollution effects across a region or forest type.

Ozone-sensitive—Environmental conditions that allow the production of foliar injury symptoms on ozone-susceptible species when exposed to ambient concentrations of ozone under field *(in-situ)* conditions.

Ozone stipple—Typical ozone-specific foliar response characterized by distinct interveinal pattern of discoloration on the upper-leaf-surface of ozone-sensitive plants.

Ozone-susceptible—Anatomical, morphological, and physiological characteristics of a plant species that make it vulnerable to ambient ozone exposures if environmental conditions (e.g., drought) are not reducing sensitivity.

Parcelization—Breaking-up of single contiguous land ownerships into smaller ownership tracts or parcels. Degree of parcelization is related to land use.

Pasture—Any area that is devoted to the production of forage, native or introduced, and harvested by grazing. Often irrigated and fenced areas.

Pixel—In digital imaging, a pixel (derived from *picture element*) is the smallest piece of information in an image. Pixels are often arranged in a regular 2-dimensional grid, and represented using dots or squares. Each pixel is the smallest piece of the original image.

Poletimber stands size class—In FIA, tree stands at least 10 percent stocked with live trees of which half or more of the total stocking is in Poletimber and Sawtimber trees, and with poletimber stocking exceeding that of sawtimber.

Poletimber trees—In FIA, live softwoods (conifers) 5.0 inches to 8.9 inches in diameter at breast height (d.b.h.) and live hardwoods (deciduous) 5.0 to 10.9 inches in d.b.h.

Primary productivity—Accretion of biomass through the reduction or fixation of carbon dioxide into sugars during the process of photosynthesis. The net gain in carbon in the biota.

Private land—Ownership group that includes all Forest Industry, Non-Industrial Private, and Native American lands.

Productive reserve forest land—Forested lands that could be classified as timberland, except that they are withdrawn from timber utilization by statute or administrative regulation.

Production intensity—Ratio of average annual removals to growing stock volume is an indicator of the proportion of growing stock volume removed per year.

Productivity class—Classification of forestland in terms of potential annual cubic-foot volume growth per acre at culmination of mean annual growth in fully stocked natural stands.

Public land—An ownership group that includes all Federal, State, County, and Municipal lands.

Pulpwood—Wood intended for use as paper, cardboard, pressed-wood, etc.

Quality Assurance (QA)—In FIA, the total integrated program for ensuring that the uncertainties inherent in field data are known and do not exceed acceptable magnitudes, within a stated of confidence. Quality assurance encompasses the plans, policies, and specifications affecting the collection, processing, and reporting of data. It is the system of activities designed to provide program managers and project leaders with independent assurance that total system quality control has been effectively implemented.

Quality Control (QC)—Routine application of prescribed field and laboratory procedures (e.g., random-check cruising, periodic calibration, instrument maintenance, use of certified standards, etc.) in order to reduce random and systematic errors and ensure that data are generated within known and acceptable performance limits. Quality control also ensures the use of qualified personnel; reliable equipment and supplies; training of personnel; good field and laboratory practices; and strict adherence to standard operating procedures.

Random sample—Any method of sample selection based on the theory of probability (degree of certainty) where samples are chosen without any bias or intention. At any stage of the operation of selection, the probability of any set of units being selected must be known. It is the method that provides a measure of precision of the estimate.

Rangeland—Land dominated by natural plant cover composed principally of native or exotic grasses, forbs, or shrubs valuable for forage. Natural rangeland is unimproved, i.e., not irrigated, and has not been seeded artificially.

Reference plot—In FIA, regionally-representative plots established with known target values for measurements. These plots are generally used during training to assist in certification of crew members. They are measured by multiple crews and the results used to assess crew accuracy under known conditions, as well as to provide a measure of between crew comparability (precision). These plots can also be used to evaluate crew performance following the field season. Reference plots are not blind crew checks because they know they are being evaluated, and tend to be more careful during measurements.

Regeneration status—Stand descriptor that indicates whether a stand was naturally or artificially regenerated.

Relative richness—Number of species present when compared to baseline species list. Provides information on the stability of populations in regions over time.

Reserved land—Land reserved from wood products utilization through statute or administrative designation. Reserved land is withdrawn by administrative designation through written document(s) that carries the weight of legal authority, prohibiting the management of land for the production of wood products (not merely controlling wood harvesting methods). This authority is usually vested in a public agency, department, etc., and supersedes rights of ownership. The prohibition against management for wood products cannot be changed by decision of the land manager (management agency) or through a change in land management personnel, but is rather permanent in nature. Examples include Wilderness areas and National Parks and Monuments.

Restrictive layer—A soil condition that changes soil properties to the extent that it limits root growth because of physical (hard rock) and/or chemical (acid layer) impediments.

Richness—Number of species, genera, etc. when used in reference to biological diversity. A commonly-used measure of biodiversity.

Rotten tree—In FIA, live trees that do not meet specifications for freedom from cull, primarily because of rot.

Rough tree—In FIA, live trees that do not meet specifications for freedom from cull, primarily because of poor form, too many limbs, and splits.

Roundwood products—Logs, bolts, or other roundtimber generated from harvesting trees for industrial or consumer uses. Includes sawlogs; veneer and cooperage logs and bolts; pulpwood; fuelwood; piling; poles; posts; hewn ties; mine timbers; and various other round, split or hewn products.

Rural—Rural areas are low-density (less than 500 people per square mile) populated areas outside of urban or metropolitan areas yet distinct from unsettled areas like a wilderness. Agricultural areas are examples or rural areas.

Rutted trail—Type of compaction measured as part of the soil indicator. Ruts must be at least 2 inches deep into mineral soil or 6 inches deep from the undisturbed forest litter surface.

Sapling—Live trees 1.0 to 4.9 inches in diameter (DBH/DRC).

Sawlog— Log meeting minimum standards of diameter, length and defect, including logs at least 8 feet long, sound and straight and with a minimum diameter outside bark for softwoods of 6 inches and 8 inches for hardwoods.

Sawtimber—Sawtimber size trees are ≥ 9 inches dbh for softwoods and ≥ 11 inches d.b.h. for hardwoods.

Sawtimber stands—Stands at 10 percent stocked with live trees of which half or more of the stocking is in trees 5.0 inches d.b.h. and larger, in which the stocking of sawtimber trees is at least equal to the stocking of poletimber trees.

Sawtimber volume—Sawtimber net volume in FIA is gross board-foot volume sometimes expressed as volume in cubic feet in non-FIA publications, less deductions for rot, sweep, crook, missing bole sections, and other defects that affect the use of sawtimber trees for lumber.

Scarcity—Economics term that refers to the concept that resources are limited and societal demands can be infinite. That is, there are limitations for any resource compared to the potentially unlimited demands of society. Term is used even at some point in time when there is an abundance of a resource compared to the demand upon those resources.

Secondary productivity—Reproduction and growth of non-photosynthetic species by consumption of carbon fixed by plants in primary productivity. Production and growth of organisms (heterotrophs) that rely on photosynthetic organisms (autotrophs) for food and energy.

Seedling—Live trees smaller than 1.0 inch DBH or DRC that are at least 6 inches in height for softwoods and 12-inches in height for hardwoods.

Senescence—Dying or dead due to completion of a cycle, sometimes accelerated by stressors. Natural process caused by diverse agents that occurs over variable periods based on nature and magnitude of causal agents.

Seral—Refers to an intermediate stage (or sere) that is part of an ecological succession of a community advancing towards some climax state.

Seston—Particulate matter suspended in bodies of water such as lakes and seas. It includes all particulates, such as plankton, organic detritus, and inorganic material.

Shrub—Woody plants greater than19 inches in heights that generally have several erect, spreading, or prostrate stems, and have a bushy appearance. If the life form cannot be easily determined, woody plants greater than 19 inches in height, but less than 16.5 feet in height, are considered shrubs.

Shrub land—Land class defined by areas dominated by vegetation generally greater than 19 inches tall with individuals or clumps separate to interlocking. Shrub canopy cover is generally greater than 25% while tree cover is generally less than 25%. In rare cases, shrub cover exceeds the tree, dwarf shrub, herb, non-vascular plant cover and is less than 25% cover.

Silvicultural—Science of controlling the establishment, growth, composition, health, and quality of forests to meet diverse needs and values of the many landowners, societies and cultures over the parts the globe that are covered by dry land.

Site class—Classification of forest lands in terms of inherent capacity to grow trees in fully stocked natural stands. For example, trees in poor site classes stands grow 0 to 20 ft^3 per acre per year; in high site classes grow 85 to 120+ ft^3 per acre per year.

Site index—Average total height that dominant and co-dominant trees in fully-stocked, even-aged stands will obtain at key ages (usually 25 or 50 years).

Site preparation—Removal or deadening of unwanted vegetation prior to natural or artificial regeneration of trees; includes prescribed burning, use of herbicides, disking, and other mechanical means of removing vegetative cover.

Slash—Unmerchantable tree residue on the ground from logging activities or from natural breakup of trees caused by insects, disease, weather, etc. Slash includes logs, stems, heavier branch wood, stumps, etc.

Slope—Inclination (angle relative to horizontal) of the soil surface from the horizontal, measured in degrees or percents.

Snag—Standing dead tree. A snag must be at least 5.0 inches DBH/DRC and 4.5 feet tall, and have a bole that does not touch the ground. A snag is either self-supported by its roots, or supported by another tree or snag.

Softwoods—Coniferous trees, usually evergreen having needles or scale-like leaves. See Conifer.

Soil bulk density—Mass of soil per unit volume. A measure of the ratio of pore space to solid materials in a given soil. Expressed in units of grams per cubic cm of oven dry soil.

Soil compaction—Reduction in soil pore space caused by heavy equipment or by repeated passes of light equipment or animals that compress the soil and break down soil aggregates Compaction disturbs the soil structure and can cause decreased tree growth, increased water runoff, and soil erosion.

Soil nutrient status—Refers to the concentration of plant nutrients (e.g., potassium, calcium, magnesium, sodium, and phosphorus) and is a key indicator of site fertility, species composition, and forest health.

Soil texture—Relative proportions of sand, silt, and clay in soils, determined in the field by how easily, or if, it can be formed into a ball or ribbon when wetted.

Specialist—Biological term referring to a species that has specific requirements for habitat, food, environmental conditions, etc., and therefore a more limited number of suitable habitats compared to a Generalist.

Sport fish—Fish caught for recreation, pleasure, or competition. Species of fish pursued by anglers vary with geography. Some fish are sought for their value as food, and others are pursued for their fighting abilities or for the difficulty of pursuit.

Spots—at the leaf or stem level refers to relatively circular discoloration of foliage or wood. At a landscape level refers to clumps of dead trees that are readily observable from the air.

Species—Group of similar individuals having a number of correlated characteristics and sharing a common gene pool. The species is the basic unit of taxonomy on which the binomial system was established. The scientific name of a plant or animal gives the genus first and then the species as in *Abies* (genus) *grandis* (species). Species is both the singular and plural form of the word.

Species group—Collection of species used for reporting purposes, often sharing similar characteristics based on form, habitat requirements, etc.

Stand—Trees or a group of plants occupying a specific area that is uniform in species composition, age arrangement, structure and condition as to be distinguished from vegetation in adjoining areas.

Stand age—Stand descriptor that indicates the average age of the live trees not overtopped in the predominant stand size-class of a condition.

Stand origin—Classification of forest stands describing their means of origin, e.g., natural or planted.

Stand size—Classification of stands based on tree size (basal area). Stand sizes are large sawtimber, small sawtimber, poletimber, and seedling-sapling stands. If less than 10 percent stocked with live trees, the site is called nonstocked.

Stand size class—Classification of stands based on stocking within diameter class groups.

State land—Ownership class of public lands owned by States or lands leased by States for more than 50 years.

Stochastic—Subsequent states are determined both by predictable processes and by random elements. Some contend that any kind of time development (be it deterministic or essentially probabilistic) which is analyzable in terms of probability deserves the name of stochastic process.

Stocking—Degree of occupancy of land by trees, measured by basal area and/or the number of trees in a stand by size or age and spacing, compared to the basal area and/or number of trees required to utilize fully the growth potential of the land; that is, the stocking standard or reference value.

Stream—A relatively small body of running water. Stream types are ephemeral, intermittent, and perennial. In FIA, ephemeral and intermittent streams are classified as land.

Stump—Woody base of a tree remaining in contact with the soil after the trunk or main stem has been severed at a point below 4.5 feet above ground height (measured on the uphill side).

Stumpage price—Price put on trees while still standing in the forest.

Subplot—In FIA, a circular area with a fixed horizontal radius of 24.0 feet primarily used to sample, in a non-destructive way, trees, saplings, and seedlings. In FHM additionally used to sample understory vegetation, down wood, and fuel loading.

Suppressed—Trees whose crowns seldom if ever receive direct sunlight from above or from the sides. See Overtopped.

Suppression—Process whereby certain trees, shrubs, etc., in a community become weakened and/or stunted, essentially due to light and water competition from surrounding trees, shrubs, etc., in the immediate environment (natural suppression). Suppression may also be the result of human intervention (e.g., selective lopping, girdling, cutting back) or selective browsing by animals (artificial suppression).

Survivor growth—In FIA, the growth on trees tallied at time t that survives until time t+1, where t is the initial inventory of a measurement cycle and t+1 is the terminal inventory. This is a component of change that is usually expressed in terms of growing-stock or all-live volume.

Thinning—Removal of saplings or trees to increase productivity of remaining trees, or to reduce fuel-loading and fire risk.

Timberland—Forestland that is producing or capable of producing in excess of 20 cubic feet per acre per year of wood at culmination of mean annual increment (MAI) or growth. Timberlands exclude reserved lands.

Topography—Relief or terrain, the three-dimensional type dimensions of the surface, and the identification of specific landforms. Also known as geomorphometry.

Total length—Total length of the tree, recorded to the nearest 1.0 ft from ground level to the tip of the apical meristem. For trees growing on slopes, measured on the uphill side of the trees. If a tree has a broken or missing top, the total length is estimated to what the length would be if there were no missing or broken top. Forked trees should be measured separately, the same as unforked trees.

Transition zone—Area where a distinct boundary between two or more different conditions (e.g., 2 different forest types) cannot be determined.

Tree—Woody perennial plant, typically large, with a single well-defined stem with a somewhat definite crown; sometimes defined as attaining a minimum diameter of 5 inches and a minimum height of 15 ft at maturity.

Tree class—Classification system based on a tree's physical characteristics, and used to classify all live timber species as sound, rough, or rotten trees, and dead timber species as either hard or soft.

Tree grades—Classification of trees based on external characteristics as indicators of quality or value.

Understory—Forest vegetation growing under the overstory vegetation, usually trees.

Upper bole—Part of a tree bole from a point where the diameter outside bark is 4.0 inches to the tip or to the point where the central stem is no longer distinguishable. Excludes all foliage and branches.

Upper stem portion—Part of the stem or fork of sawtimber trees above the sawlog top to a minimum top diameter of 4.0 inches outside bark or to the point where the central stem breaks into limbs.

Urban land—Also see Developed Land. Land that has been permanently removed from the natural resource base (land or water). Developed lands include: (a) large tracts of urban and built-up land; (b) small tracts of built-up land, less than 10 acres in size; and (c) land outside of the built-up areas that is in roads, railroads, and associated rights-of-way.

Urban forest land—Land that meets the minimum requirements for forestland, based upon crown cover or tree stocking, but falling within census tracts with population densities are greater than 500 people per square mile.

Vegetated—Condition in which herbaceous plants, shrubs, trees, and nonvascular plants make up more than 5 percent of the total surface cover.

Vegetation profile—Vertical section of vegetation within designated height strata to describe relative percent occupancy by various life forms, such as herbs, vines, shrubs, and trees. In FIA, typical strata are 0-2 ft., 2-6 ft., 6-16 ft., and > 16 ft.

Veneer—Thin slices of wood, usually thinner than 3 mm (1/8 inch), that are typically glued onto core panels (often wood, particle board or medium density fiberboard) to produce flat panels such as doors, tops and panels for cabinets, parquet floors and parts of furniture.

W126 ozone index—An ozone exposure index that gives heavier analytical *weighting* to high ozone exposures (> 100 ppb) that is added to the cumulative lower, chronic exposures. The index proposed by EPA for the 2008 Ozone Criteria Documents of ozone effects on native plant communities.

Water—Streams, sloughs, estuaries, and canals more than 200 feet (60 meters) wide; and lakes, reservoirs, and ponds more than 4.5 acres (1.8 hectares) in area. See related definition for Census Water.

Water erosion prediction project (WEPP)—Model used to predict erosion losses on forested lands, based on estimated rainfall amounts, soil texture, and other factors.

Windfall—Tree or trees felled by wind. Also known as Blow Down.

Witness tree—Tree used by surveyors and others to mark the location of a survey corner; the tree is located near the survey corner and is inscribed with survey data. Also known as a bearing tree. In long-term monitoring, refers to trees used to locate or reference the center of a plot or some other point.

Woodland—Forests where the mean annual growth is less than 20 cubic feet per acre per year.

Young growth stands—Stands in which 50 percent or more of the stand is occupied by trees that do not qualify as old-growth by FIA Regional definitions. For example, stands where more than 50 percent of trees are less than half the dbh attained at maturity.

Sources: Many forest-specific definitions were from the USFS's Forest Inventory and Analysis (FIA) program (circa 2003). The FIA glossary can be found at (http://www.fia.fs.fed.us/).

Other sources include:

The Random House College Dictionary. Revised Edition 1982. Random House Incorporated. New York, NY 10022. 1568 pps.

Steen, E.B. 1971. Dictionary of Biology. Barnes and Noble Books. U.S.A. 630 pps.

Wikipedia (http://en.wikipedia.org/wiki/)

USDA 2000, Natural Resource Inventory (http://www.fs.fed.us/foresthealth/).

ACRONYMNS

Acronym	Definition
BAF	Basal Area Factor
BLM	U.S. Department of the Interior's Bureau of Land Management
CEC	Cation Exchange Capacity.
Cf	Cubic foot
cm	Centimeter
CWD	Coarse Woody Debris
CWM	Coarse Woody Material
DBH	
d.b.h.	Diameter at Breast Height
DIB	
d.i.b.	Diameter Inside Bark
DOB	
d.o.b.	Diameter Outside Bark
DRC	Diameter At Root Collar
DWD	Down Woody Debris
DWM	Down Woody Material
Ecec	Effective Cation Exchange Capacity
FHM	USFS's Forest Health Monitoring
FIA	USFS's Forest Inventory and Analysis
FIPS	Federal Information Processing Standard
FSVeg	Forest Service Vegetation database
FWD	Fine Woody Debris
FWM	Fine Woody Material
GPS	Global Positioning System
ha	Hectare
in	Inch
km	Kilometer
m	Meter
mmcf	Million cubic feet (wood volume)
mmbf	Million board feet
MQO	Measurement Quality Objective
NFS	USFS's National Forest System
NRCS Plants	Natural Resource Conservation Service's Plants Database
PI	Photo Interpretation.
QA	Quality Assurance
QC	Quality Control
ROW	Rights-Of-Way
RUSLE	Revised Universal Soil Loss Equation
USFS	United States Forest Service
Wepp	Water Erosion Prediction Project

TECHNICAL APPENDICES

Technical Appendix A—Introduction

S. Fulton[1], E. Mercer[2], K.W. Stolte[2]
[1]Region 4, U.S. Environmental Protection Agency
[2]Southern Research Station, USDA Forest Service

Our analyses of land-use status and trends were conducted using databases of the USDA Forest Service Forest Inventory and Analysis (FIA) and Forest Health Monitoring (FHM) programs, and the USDA Natural Resources Conservation Service (NRC), National Resources Inventory (NRI) program. The Forest Service conducts periodic forest inventories in every State in the U.S. to create FIA datasets include extensive data on forest resources at the county, plot, and tree level (Hansen and others 1992), often at the resolution of 1 plot every 6,000 acres. NRI data, which is produced every 5 years by the NRC, includes data on land cover and use, soil erosion, prime farmland, wetlands, and other natural resource characteristics of all non-Federal, rural land in the U.S. (USDA 2000). We analyzed NRI data on land types and use, and changes in use, using summary data from the NRI web site (http://www.nrcs.usda.gov/technical/NRI/). Land-cover types were analyzed for all States in the MAIA region, including counties of southeast NY, western NJ, and northeastern NC, were from 1982, 1987, and 1992. Slight changes in NRI estimation protocols and the availability of 1997 data resulted in additional evaluations of change in cover types (water, Federal land, crop land, pasture land, developed land, forest land, and total rural land) for the five States entirely within the MAIA region (MD, WV, VA, DE, and PA) in 1997. Thus these evaluations covered the NRI reporting periods of 1982, 1987, 1992, and 1997. For some analyses crop land and pasture land were also combined as total agricultural land.

Forest Ownership

We examined forest ownership patterns using data from national landowner surveys conducted by the USDA Economic Research Service, the USDA Forest Service, the USDA National Resources Conservation Service, and the National Association of State Foresters (Birch 1996a, 1996b, 1996c).

Population/Demographics

U.S. Census data from 1970, 1980, and 1990 was used to generate population trends that were an element of the MAIA region (http://www.census.gov/).

Forest Fragmentation and Extent

We determined the types and distribution of Landscape Pattern Types (LPTs) in the MAIA region (ca. 1990-1993) using the following methods. A spatial filter was applied to an MRLC generated from Landsat TM data of that time frame. The filter was an 81-pixel moving window that produced a new value based on surrounding pixels in the window (Riitters 1999). The filter was used to produce a new map that generalized fine-scale (7-ha) detailed land cover information into 590-ha LPTs—areas of land characterized by a single or combination of land use/cover type(s). We calculated percent forest cover as the number of pixels classified as forest within the window, divided by the total number of pixels in the window. The bottom threshold was chosen to match the MRLC threshold for non-forest (< 10-percent forest cover), and the next lowest threshold (10- to 59-percent forest cover) was chosen so that the maps of fragmentation and forest density could be compared more easily. Landscape patterns types represented three dominant land uses—forest, agriculture, and urban/developed land—as well as rural land, which was characterized by a mix of land uses, i.e., not dominated by any single use. The 19 MRLC LPT classes were combined into 7 classes based on similarities of types to simplify the information and still represent the major land use types. No analysis of change over time in landscape pattern was possible due to the unavailability of adequate regional data (Riitters 1999).

Technical Appendix B—FHM and FIA Data Analyses

W.D. Smith[1], M.J. Ambrose[2], and J.W. Coulston[1]
[1]Southern Research Station, USDA Forest Service;
[2]North Carolina State University, Department of Forestry

Forest Health Monitoring Indicators

We analyzed FHM data collected through May 1998 for the period 1991 through 1997. The FHM program provides, on a State-by-State basis and among groups of States sampled in the same subsequent years, measurable changes in forest health from a baseline year (the year of first sampling). We evaluated changes in indicator values over different time periods for the following groups of States: Delaware, Maryland, New Jersey, and Virginia (1991 through 1997) and West Virginia (1995 through 1997). Plots were first established in North and South Carolina and Pennsylvania in the summer of 1998; plot data for those States were not available for analyses in this report.

In addition, we evaluated defoliation, mortality, and risk of mortality using non-plot ground-and-aerial survey data (FHM Detection Monitoring; www.fhm.fs.fed.us) to address select forest health issues in other MAIA States where FHM's Detection Monitoring ground plot component had not been implemented as of 1997.

Condition Class on FHM Subplots

The status and change in forest land cover and land use based on FHM plot data used condition-class information collected as part of the Mensuration Indicator (USDA Forest Service 1995). For each of the four subplots in each FHM plot, the proportion of each subplot in distinctly different land-use classes was quantified and used to estimate land use classes at the plot and population levels (Scott and Bechtold 1995). All plots were proportioned as either all or mostly in one land-use class, or in combinations thereof, such as forest/urban and forest/agricultural. Our analysis was done for Eastern plots in MAIA region that had been measured in 1995. Status was based on percentage of each FHM plot assigned to each of 11 land-use classes. We classified all plots as forest, woodland, forest/agriculture, forest/urban, open, range, agriculture, idle farmland, urban, water or wetland. Our evaluations paid particular attention to land-use classes that were primarily forested but also had varying percentages of other land use. The urban land class denoted an urban-and-other development condition, e.g., paved or dirt roads, buildings. Change in land use was evaluated by comparing Measurement Type 1 (MT1—initial plot establishment and measurement) land-use percentages with MT3 (all subsequent re-measurement of the same plot) values (http://www.fia.fs.fed.us). In 1997 we analyzed urban land use class and other land-use classes to evaluate the spatial pattern of forest ecosystems within the MAIA region.

Forest Biodiversity

Tree Species Richness

Tree species richness is the total number of species of trees (> 5 inches d.b.h.) and understory saplings (1 to 5 inches d.b.h.) and seedlings (< 1.0 inch d.b.h. and > 10 inches in height) on each plot. For example, a plot with16 species in the overstory and 9 species in the understory, where 5 of the species occurred in both the understory and overstory, would have a richness of 20 unique species per plot. We evaluated tree species diversity during the initial evaluation of each plot; therefore, the record of diversity evaluations is for the period from 1994 through 1997.

Lichen Species Richness

The use of woody macrophytic lichens to evaluate additional measures of plant biodiversity and also as indicators of air quality and climatic conditions was developed by the FHM program in 1992 (McCune 1992), and is now implemented nationally by the FIA program (www.fia.fs.fed.us). This nationally-standardized method provides comparable data on lichen communities in all forests throughout the U.S. Lichens are a component of plant biological diversity, and additionally can provide information on air quality and climate regimes when lichen community analyses gradients are developed in specific regions.

Lichen community field procedures include collecting macrophytic lichens from woody substrates on the FHM or FIA plots. Field crews are trained to collect a sample of each species found and to record the relative abundance of each. Lichen species are subsequently identified by lichen taxonomic experts in the autumn following summer sample collection.

Gradient analyses studies in ecological strata are necessary to compute air quality and climatic scores for the species found in lichen communities on each plot (McCune and others, 1997a and 1997b). For example, air quality gradient analyses are based on evaluation of lichen communities along known sulfur and nitrogen air pollution gradients where climatic values are constant, and climatic gradients are based on evaluation of lichen communities along known climatic gradients where air quality values are mostly constant. However, these gradient analyses were not available for the Mid-Atlantic region for this report. Thus this report only contains species diversity for lichens in some areas of the MAIA region. However, even basic evaluations of lichen communities identifies areas where lichen species are common and areas where they are sparse, and thus aids in the overall evaluation of plant biodiversity in the MAIA region.

Forest Productivity

Tree Volume

Annual change in volume growth of trees on FHM plots, expressed in pounds per acre, was analyzed by comparing the tree-by-tree change in volume (where the measure of diameter at breast height [d.b.h.] and estimated tree height were used to calculate volume) plus any in-growth trees recorded as part of MT3 measurements. For States where all trees were not measured in MT1, the volumes of trees not measured were estimated using Best Linear Unbiased

Predictors (see below: Estimating Current Values in Non-Measured Years).

Forest Game Species Population and Harvest Trends

Estimates of species populations and harvests were compiled primarily from data provided by cooperating State and Federal agencies. Source documents and databases will be discussed for each species category section separately because data sources varied by categories of species. Because we used a diversity of data sources, data quality varied greatly. In some cases, national inventories were designed to provide statistically-based estimates from which strong inferences on population size and trend could be made at State, regional, and national scales. In other cases, estimates were based on the best judgments of wildlife professionals; in those cases, the professionals emphasize that attention should focus on the direction of the trend rather than the magnitude of estimates.

Big Game Species

We compiled data on big game populations and harvests primarily from cooperating State wildlife agencies. Because of the management and economic importance of big game, harvest statistics from State agencies were much more complete than population statistics. We sent questionnaires developed cooperatively by the USDA Forest Service and Natural Resources Conservation Service to State wildlife agency offices through the International Association of Fish and Wildlife Agencies. All States cooperated and responded to our request for information. The absence of data from certain States resulted from variation in the distribution of species or the lack of data for certain years. We included only those States that provided estimates for 1975 to 1990 (in 5-year intervals), and 1993 in the trend analysis. Population and harvest data were sufficient to analyze short-term trends for white-tailed deer, black bear, and wild turkey.

Small Game Species

Population and harvest statistics from cooperating State wildlife agencies were similarly compiled to evaluate small game populations and harvest. Questionnaires developed cooperatively by the USDA Forest Service and Natural Resources Conservation Service were sent to State wildlife agency offices through the International Association of Fish and Wildlife Agencies. In addition, State agency estimates of upland game bird populations with relative abundance data from the North American Breeding Bird Survey were also evaluated.

There is some variation among State wildlife agencies about which species are managed as small game. We reviewed population and harvest statistics for quail, ring-necked pheasant, grouse, rabbit, hare, and squirrel, and in cases where State data were not distinguishable to the species level, we analyzed trends for species groups that are taxonomically or ecologically similar. Some results of these analyses are given in the Productivity section (Chapter 16) of this report.

Forest Vitality

Insects and Pathogens

The National Forest Health Monitoring program evaluates the effects of insects, infectious and noninfectious diseases caused by pathogens, and other agents in two ways. First, they used data collected by the closely-affiliated Forest Inventory and Analysis (FIA) program of effects of insects and diseases on trees on FIA Phase 3 plots; these plots are often the original FHM Detection Monitoring plots. Secondly, FHM conducts annual aerial and ground surveys to detect and map forest insects and disease damage and mortality, as well as damage and mortality from storms and other large-scale agents.. This data is now routinely analyzed to produce maps of areas of forest stress, and to produce risk maps for specific insects and diseases. For more information, see USDA Forest Service, Forest Inventory and Analysis Eastwide Data (Hansen and others, 1992; (http://www.srs.fs.fed.us/sustain/data/fia/eastwideguide). Also see Gypsy Moth Digest at (http://fhpr8.srs.fs.fed.us/wv/gmdigest/gmdigest.htm/) (USDA Forest Service 1998b).

Invasive Species

The FHM program initiated pilot studies in 1994 and 1995 to develop nationally-standardized methods for the evaluation of species composition and abundance of all vascular understory plants (both native and exotic invasives) to complement the existing tree species evaluations which include both native and exotic species. Bull and others (1998) further improved on the methods developed in the first pilot studies by including full subplot surveys of both native and exotic plant species to capture the more infrequently-occurring species. These latter methods are currently in use in some parts of the U.S., although still not formally adopted nationally as a core Phase 3 indicator by the FIA program. This current FIA vegetation indicator (www.fia.fs.fed.us) identifies and evaluates abundance (cover) of all species in a vertical profile of the forest stand that includes the herbaceous stratum (0- to 2-ft. tall), shrub stratum (2- to 6-ft. tall), large shrub/sapling/small tree

stratum (6- to 16-ft. tall), and the overstory tree stratum (> 16-ft. tall). These four strata were evaluated in three 1-m^2 quadrates on each of the four FHM/FIA subplots, for a total of 12 quadrates. However, little information from the 1994-1995 or 1998-2000 pilot studies was available for the MAIA region for this report.

Data on invasive plants in many States in the MAIA region was obtained by the Biota of North America Program (BONAP: http://www.bonap.org/synth.html) from herbaria in each State, literature reviews, and data already existing in their BONAP database (Kartesz and Kartesz, 1980). Data were not obtained for Maryland and Pennsylvania counties, and some counties in New York, for various reasons including availability of State-level herbaria data. The available data were combined to produce county-level estimates of the number and abundance of exotic invasive species in these areas. The invasive species abundance were compared to native species abundance to produce county-level data on the percentage of the total flora represented by exotic invasive species. County-level data were then entered into ArcView™ to produce county-level distributions of the relative abundance of exotic invasive plant species in representative MAIA areas.

Storms

Impacts from severe storms were evaluated on ground plots through analysis of the FHM Damage Indicator. The Damage Severity Index (DSI) for storm damage was calculated based on broken roots, wounds on the bole, and broken tree branches, which are indicators of high winds and/or severe ice or snow storms (Mielke and others 1995). Damage indices were calculated for individual trees based on the change in severity from the initial damage recorded to the most recent damage recorded, then averaged at the plot level for softwoods and hardwoods. The damage index ranged in value from 0 to 100 for each damage type, and values over 15 were considered to be serious impacts to the growth, reproduction, and/or survival of the affected trees.

Ozone Bioindicator Plants

We used levels of W126 (Lefohn and Runeckles 1987) ozone exposure indices based on and similar to those used in the Southern Appalachian Assessment Report 3 (SAMAB 1996) to assess the average ozone exposures for the MAIA region from 1993 through 1996. The W126 indices were separated into 4 classes based on the relationships developed in fumigation and field studies on the response of native plants to ozone exposures. Thus Level 1 exposures (5.9 to 23.8 W126 values) were thought to be sufficiently phytotoxic to injure Eastern native plant species that are known to be highly susceptible to ozone exposures (e.g., black cherry), and Level 3 exposures (> 66.6 W126 values) are believed to be sufficiently phytotoxic to injure Eastern native plant species affected at lower concentrations and those known to be more tolerant of exposure to ozone (e.g., red oak). We also used the same ozone-susceptible native plant species (SAMAB 1996) in our evaluations of ozone bioindicator response to ozone exposures.

Severity of ozone injury on bioindicator plants is based on the average number of plants per bioindicator species injured, average number of leaves injured per plant, and the average severity of injury on foliage of each plant. These values are combined into an Ozone Bioindicator Response index that ranges from 0 to 25 or greater, with values of > 0 to 4.9 being minor injury and values > 25 being severe injury that is likely to be associated with significant effects at the stand level.

We used GIS spatial overlays to compare W126 ozone exposures with severity of injury recorded on bioindicator plants at both on-frame (within 3 miles of FHM or FIA plot centers) and off-frame (> 3 miles from plot center) plots in Great Lakes and Northeastern states (areas with pronounced gradients of ozone exposures) for the years 1994 through 1996 to assess the relationships between ozone exposures and bioindicator plant responses. While there were some anomalies, plots with the highest ozone injury severity generally were found within the highest W126 ozone exposure polygons. In the MAIA region, polygons depicting areas of moderate W126 ozone exposures were populated by FHM ozone indicator plots with low to high ozone injury severity values—these variations in ozone exposure-response are believed to be cause by the modifying effects of different environmental conditions found in areas with a lot of topographical variations.

Forest Fragmentation

We applied a spatial filter to a Multi-Resolution Land Characteristics (http://www.mrlc.gov/glovis.php) land cover map generated from Landsat Thermal Mapping satellite data from ca. 1990-1993. The filter produced a new map that generalized fine-scale (7-ha) detailed land cover information into 590-ha units. The filter was an 81-pixel by 81-pixel moving window that assigned the middle pixel of the original land cover classification a new value based on surrounding pixels in the window (Riitters 1999). Fragmentation was calculated as "connectivity," or the number of "forest-to-forest" pixel edges (the edge between two adjacent pixels classified as forest) divided

by the number of single "forest" pixel edges (the edge between two adjacent pixels, only one of which is classified as forest) in the window. In regions of low fragmentation—or "connectivity"—forest pixels were more likely to be adjacent to other forest pixels than in regions with high fragmentation. In other words, the greater the level of fragmentation, the further the distance between forest pixels, and the lower the level of connectivity between forest pixels.

Crown Dieback and Transparency

Crown condition is an important indicator of individual tree and forest stand health. For many tree species, crown condition has been directly linked to tree growth, mortality, and reproductive success. Five crown indicators have been developed by the FHM program and are now implemented nationally by the FIA program. The crown indicators have been used to detect various states of crown decline resulting from natural and anthropogenic stresses (Burkman and others 1995; Stolte and others 2005). For this report, we evaluated the dieback of sun-exposed portions of the tree crown and the transparency (to sunlight) of the foliage of the whole tree crown.

The FHM program recorded crown dieback as the percent mortality (in 5 percent classes) of the terminal portion of branches < 1-inch (2.54 centimeters) diameter and in the upper, sun-exposed portion of the crown ($0 \leq$ crown dieback ≤ 100) (Burkman and others 1995). Foliar transparency was recorded as the percent of sky visible through the live, normally foliated portion of the crown. An ocular estimate is made of transparency to the nearest 5 percent. Data analysis was conducted by hardwoods and softwoods separately, as a stratification of tree species into broad species groups.

The FHM program used a Rotating Panel with Annual Overlap sampling design to evaluate status and changes in crown indicators (Smith and others, 1996). The rotating panel sample allows partial sampling of all sample plots with a random, smaller resample of the previous year's partial sample. Each partial sample is a panel, and the prior-year resample is an overlap. The panels are equally spaced within the sampled population. In the early FHM program all four panels (all plots) were measured in the first year. In the following years, field crews sampled 1/4 of the total plots (a panel), and 1/3 of the previous year's panel (1/12 of total population) for a total of a 1/3 of the plots each year (1/4 current, 1/12 previous year). Currently in FIA no overlaps are sampled, but the rotating panel sample, typically consisting of 5 to 7 panels, is still used (www.fia.fs.fed.us).

In the analyses we aggregated FHM plot data spatially using Bailey's ecoregion section (Bailey 1995) or a major Hydrological Unit Code (HUC) watershed (http://water.usgs.gov/GIS/huc.html). The minimum level of analysis in this report was the mean plot value of each variable and/or indicator by ecoregion section or watershed (Smith and others 1996).

Tree Damage

The DSI score is based on three variables: the type of damage, the location of damage on the tree, and the severity of damage (Mielke and others, 1995). Tables were developed that listed DSI scores to each damage occurrence based on these three variables (www.fia.fs.fed.us). The index value associated with each particular combination of damage type, location, and severity was determined following several workshops of Federal, State, and university experts in forest pathology and entomology[12]. Up to three damages per tree can be scored. The scale runs from 0 to a theoretical maximum of 300, where zero indicates no damage above the minimum threshold recorded, and 300 indicates three of the most serious damages in the worst locations and of maximum severity. Generally, a high damage index value indicates multiple damages, severe types of damage, and/or extensive damage with the damages occurring near the base of the tree.

The damage index (DI) for a plot is computed as:

$$DI_{plot} = \frac{1}{n}\sum_{i=1}^{n}\sum_{j=1}^{3} f(d_{ij}, l_{ij}, s_{ij}) = \frac{1}{n}\sum_{i=1}^{n} DSI_i$$

where:
d_{ij} = damage type (1 to 3 per tree),
l_{ij} = location of damage (1 to 3 per tree),
s_{ij} = severity of damage (1 to 3 per tree),
n = number of trees per plot, and
f (d,l,s) = the severity index value for each damage to the tree found in the appropriate look-up table.
DSI_i = damage severity index for tree i

Our analysis of tree damage used only the most recent measurement of each forested plot through 1998. The damage severity index was calculated for softwood and hardwood tree species as the average DSI for each plot in the MAIA watersheds, and the average percent of trees damaged for all plots at the watershed-scale (figs. 64 and 65).

[12]Personal communication. 2001. Manfred Mielke, USDA Forest Service, Northeastern Area State and Private Forestry, 1992 Folwell Avenue, St. Paul, MN, 55108

Tree MortalityVolume

Annual tree mortality, in terms of wood volume per acre, is based on trees that have died since the initial plot establishment. Trees that were dead at any initial sampling were not included in the analysis because of the difficulty in determining the year of mortality and thus calculating any annual mortality rate. Because different forest types grow at different rates, a simple measure of mortality volume is not a good measure of forest health on a regional or national basis. A more standardized and regionally-representative mortality indicator is the ratio of annual mortality volume to gross volume growth of surviving trees (MRATIO). An MRATIO value greater than 1.0 indicates that mortality is exceeding growth and that live standing volume is actually decreasing over time. The MRATIO can be large if an over-mature forest is senescing and losing a cohort of older trees. If forests are not naturally senescing, a high MRATIO (greater than 0.6) may indicate high mortality due to some acute cause (insects or pathogens) or generally deteriorating forest health conditions.

We used the following procedure to determine the MRATIO for each ecoregion section and watershed. We estimated mortality volume for each plot using the measured diameters of trees that had died, height estimates based on site trees measured on each plot, and published volume equations (Schreuder and others 1993). Similarly, we estimated live tree volume for each plot. Using the same GLS modeling procedure (Schreuder and others 1993; Urquhart and others 1993; Gregorie and others 1995) as we used for the analysis of crown indicators for each ecoregion section and major watershed, we independently averaged annual mortality volume and gross growth volume (change in live tree volume).The MRATIO for each ecoregion section or watershed was the ratio of average mortality volume to average gross growth volume.

Tree Mortality Diameter Ratio

Where mortality had occurred we also calculated the ratio of the average dead tree diameter (DD) to the average live tree (LD) diameter (DDLD ratio), which indicated the size of those trees lost to mortality relative to the surviving trees in the stand. Low (much less than 1) DDLD ratios usually indicated that observed mortality comprised mostly smaller trees as part of the natural "self-thinning" development of the forest. This competition-induced mortality is typical of young, vigorous stands. High ratios (much greater than1) indicated mortality was occurring in larger trees. Such mortality usually is associated with senescence or some external factors such as insects or disease (Smith and Conkling, 2000).

Carbon Sequestration in Trees

For the MAIA region as a whole, we estimated the average annual rate of carbon sequestration in woody biomass, expressed as pounds per acre per year. We determined carbon sequestration rates using tree volume data from FHM plots and estimates of other carbon (below ground, down woody debris) from published relationships for species groups (Birdsey 1996). For each FHM plot we estimated the carbon sequestered by determining the biomass of living boles and the roots of all trees and saplings, then subtracting the biomass of dead trees and approximately half the biomass of trees that would be harvested over the same time period. Our analysis assumed that approximately half of the harvested tree biomass will be sequestered for decades as wood products (Birdsey 1996). Trees that had died but were not salvaged were expected to release their carbon to the atmosphere within a significantly shorter timeframe.

The rate of carbon sequestration in the MAIA region could be estimated by modeling stored carbon–using the same GLS modeling procedure we had used for analysis of crown indicators. The sequestration rate was given by the average annual change in stored carbon given by the model.

Soil Conservation

Soil sampling protocols for the data presented are published in the 1998 FHM Field Methods Guides (USDA Forest Service 1998a). The forest floor includes the soil litter and duff layers and is collected within a 12-inch circular frame, an area approximately 113 in^2. The bottom of the litter layer is identified as the depth at which plant parts, such as needles or leaves, were no longer distinguishable due to decomposition. The surface soil was collected as the layer of soil directly underneath the forest floor sample. We collected 2 subsurface mineral soil samples: one at 0 to 4 inches below the top of the surface of the mineral soil, and a second at 4.1 to 8 inches below that. All samples were oven-dried at 190 °F prior to analysis, and not air-dried, which may have affected some of the chemical analyses discussed in the sections on soil chemistry.

Statistical Analysis

Generalized Least Squares Models

Using a generalized least squares (GLS) model, we analyzed several indicators (productivity, crown dieback and transparency, mortality, and carbon sequestration rate). Exchangeable calcium, magnesium, potassium, and sodium were analyzed by ammonium acetate extraction at pH 7, and

total nitrogen and phosphorus were analyzed by the Bray I method (www.fia.fs.fed.us). In this approach, the population level current mean value and annual change are estimated from linear mixed models for repeated measurements. This approach is discussed thoroughly by Van Deusen (1989), Urquhart and others (1993), and Gregoire and others (1995). In particular, Van Deusen demonstrated that the GLS approach using a mixed estimator extended to estimating compatible components of growth to those presented by Beers (1962); it used all the data, generalized for any number of remeasurements, and was easily extended to estimate quantities other than current volume and growth. In addition, Gregoire and others (1995) demonstrated that the procedure was particularly useful in unbalanced designs where all plots have not been measured at the same time intervals.

The analysis for change is based on the general linear model:

$$y_{ij} = \beta_0 + \beta_1 \left(t_j - t_0\right) + \eta_i + \varepsilon_{ij} \qquad (1)$$

where:

y_{ij} = the value of the indicator on plot *i* at time *j,*
$_0$ = estimated mean of the value of all plots at year 0.
$_1$ = estimated change in y over time,
t_0 = time of initial measurement,
t_j = time of measurement *j,*
$_i$ = plot effect (spatial) variability, and
$_{ij}$ = within-plot (temporal) variability;

Some of the between- and within-plot variability is due to measurement error. Measurement error δ is assumed to be normally distributed with a mean = 0 and variance = σ^2. This assumption is critical to detecting change; but the requirement can be relaxed if it can be assumed that a nonzero measurement error (bias) does not change over time. For example, if the error in measurement is of a consistent direction and magnitude, the measurement of change is minimally affected by the measurement error. Because our analysis method did not partition measurement error from random variation, all standard error, probability estimates, and R^2 statistics reflect both sources of error.

Estimating status and change for a region

The initial value, βo, and the annual change β1, are estimated for each region of interest, e.g. ecoregion section (Bailey 1995), with SAS PROC MIXED (SAS 1999) using empirical generalized least squares (Littell and others 1996). The models are termed mixed: they contain both fixed and random variables. Random implies that the observations are a random sample of all possible response levels in the population. In the case of FHM and FIA data, the random effects are the individual plots and the fixed effect is time. Specific estimation procedures are presented in greater detail in Smith and Conkling (2000).

The prediction equation is:

$$\hat{y}_j = \beta_0 + \beta_1 \left(t_j - t_0\right)$$

The change is β1 and the current status for the region is $\hat{y}_j$.

Estimating the current values of individual plots in non-measured years

The parameter estimates resulting from the previous models can be used to predict the plot values for years in which a particular plot was not measured. This is particularly useful if spatially displaying all plot values for a single point in time. As more mechanistic models are developed, the procedure can also be used to develop predictive models for future years based on current conditions.

These predicted values are referred to as Best Linear Unbiased Predictors (BLUPs). BLUPs are "best" because they have the minimum mean square error; "linear" because they are linear functions of the data; "unbiased" because the average value of the estimate is equal to the average value of the quantity being estimated; and "predictors" because they predict random effects (Robinson 1991). In this report, BLUPs are used to predict the value of particular plot attributes, such as transparency and volume from a population of random effects. This procedure maximizes the efficiency of unbalanced designs, such as those where not all samples are measured every year (Gregoire and others 1995). BLUPs are commonly used in quantitative genetics, statistical quality control, time series, and geostatistics (Christensen 1991; Robinson 1991).

Given model 1 above, the BLUP for predicting the value of plot i at time k is:

$$blup(y_{ik}) = \underbrace{\hat{y}_{ik}}_{mean} + \underbrace{\frac{n_i\sigma_\rho^2}{\sigma_\varepsilon^2 + n_i\sigma_\rho^2}}_{weight}\underbrace{\left(\bar{y}_i - \frac{1}{n_i}\sum_{j=1}^{n_i}\hat{y}_{ij}\right)}_{mean\ deaviation} = \hat{y}_{ik} + \frac{n_i\sigma_\rho^2}{\sigma_\varepsilon^2 + n_i\sigma_\rho^2}\left(\frac{1}{n_i}\sum_{j=1}^{n_i}(y_{ij} - \hat{y}_{ij})\right)$$

where:

y_{ik} = the value of plot i at time k,
$\hat{y}_{ik}$ = the fitted value for plot i at time k, that is, the expected value of all plots within an ecoregion.
n_i = the number of measurements on plot i,
y_{ij} = the value of plot i at time j.
$\bar{y}_i$ = the mean of all measurements of plot i,
σ_p^2 = the between-plot variance, and
σ_ε^2 = the residual-within plot (temporal) variance.

The BLUP consists of the mean value of all plots within the group measured at time k plus the mean deviation of the predicted values of plot i from the actual value in the years the plot was measured multiplied by a weighting factor. The weight term reflects the number of times the plot was measured and the plot and residual variance.

The weight increases as the number of measurements increases and/or as the correlation over time increases, reflecting the statistical confidence in the estimate. If the estimates were based on very few measurements, or the correlation over time was small, the weight would approach zero and the best estimate of the plot value would be the mean of the population. The procedure can be better understood by examining a few simple numerical examples:

A plot was measured in year 1 and 4, and we need an estimate of the plot value at year 5.

Assuming the model is $\hat{y}_j = 10 + 2(t_j - t_o)$, the estimate for year 0 is 10 and the change over the 5-year interval is 2 units per year, then the mean value of all plots in the region is 10 + 2(5) or 20. If, in addition, the weight were 0.7, and the observed and predicted values for the plot were:

Year	0	1	2	3	4	5	Mean
Observed	.	9	.	.	13	.	11
Fitted	.	12	.	.	18	.	15

then the average deviation from the observed value is 11 - 15 = - 4.0; that is, in the years when plot i was measured, its average was 4.0 units less than the mean of the fitted values. Therefore, the BLUP for year 5 is 20 + weight (-4.0). Given a weight of 0.7, the best estimate of the value of plot i in year 5 is 20 + 0.7(-4.0) = 17.2.

The behavior of this estimate may be understood better by considering some other possible conditions:

1. When predicting the value of a plot that has never been measured, the mean deviation is zero and the best estimate is the mean of all plots in the group (20).

2. If the value of the plot in the first measurement was 5 greater than the mean and at the second measurement the value was 5 less than the mean, then the mean deviation would be 0 and the best estimate for year 5 would, again, be 20, which is the mean estimate of all plots in the group. The mean deviation of 0.0 indicates that the within-plot variability is probably due to measurement error or seasonal variability, in contrast to the initial example where the plot was consistently lower (-4.0) than the mean of all plots.

3. If the correlation over time was 0.2 instead of 0.7, the weight would be approximately 0.2 instead of 0.8. This would indicate that there is a high degree of within-plot variability due to measurement error or seasonal variability, and the best estimate is 20 + 0.2(-4.0) = 19.2.

Technical Appendix C—Socio-Economic Analyses

E. Mercer and A. Murthy
USDA Forest Service, Southern Research Station
US-Global Financial Services, American Express

Market Benefits/Forest Products Benefits

To quantify the economic importance of forest-based industries to people in the MAIA region, we examined the employment and income generated by the following Standard Industry Classification (SIC) sectors: lumber and wood products (SIC 24); paper and allied products (SIC 26); and furniture and fixtures (SIC 25). Data were derived from the Bureau of Labor Statistics, ES-202 database (USDL-BLS, 1989 and 1996). The income from all wages and salaries was corrected for inflation using the GDP price deflator[13] and expressed in terms of buying power in 1982 dollars. To estimate the rate of change in employment and income in the forest products sectors between 1975 and1995, we used a linear regression equation in the following form:

$$Y = b_0 + b_1 X + \varepsilon$$

where:

Y = natural logarithm of employment or real wages
X = year
b_0 = regression coefficient
b_1 = regression coefficient

[13] The GDP deflator, a price index for all final goods and services, is a weighted average of the prices of all final goods and services produced in the economy (Edgmand and others 1996).

Technical Appendix D—Bird Habitat Condition and Diversity

E. Cam and L.E. Jackson
Laboratoire Evolution et Diversité Biologique; Université P. Sabatier; Toulouse, France
U.S. Environmental Protection Agency, Office of Research and Development, Research Triangle Park Laboratory

Analysis of Ecological Condition using the Bird Community Index (Chapter 12)

The bird community index (BCI) is an ecological condition indicator based on songbird community composition (O'Connell and others, 1998 and 2000). Developed as part of the EPA's Environmental Monitoring and Assessment Program (EMAP) (http://www.epa.gov/emap/), the BCI is intended to facilitate assessments of ecological condition at the regional and national scales. The EPA conducted research throughout the Mid-Atlantic highlands area (MAHA) (Smith and others 2003) that involved sampling of multiple natural resources as well as urban areas and *ecotones* (www.epa.gov/reva/vulnerability/). The design, therefore, captured ecologically significant landscape features that contributed to overall ecological condition, but were often typically omitted from traditional field monitoring. This was the first attempt to develop and apply a field indicator of ecological condition across the full extent of an EMAP reporting region without stratifying by resource type (e.g., forests, streams, wetlands).

Composed of multiple biological metrics, the BCI ranked bird communities at sample sites according to the proportional representation of 16 behavioral and physiological response *guilds*. Guilds are groups of bird species that were selected as indicators of structural, functional, and compositional ecosystem elements. For example, bark-probing insectivores indicate the presence of dead and decaying trees (a structural system property) and the presence of secondary and tertiary consumers in the food chain (a functional system property). Exotic bird species, as another example, indicate that compositional changes in species diversity have taken place in the system. Relative proportions of specialist (restrictive or limited living requirements) and generalist (broad or expansive living requirements) guilds determine the integrity of the larger system. Integrity refers to the capability of supporting and maintaining *a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that found in similar undisturbed habitats in the region* (Karr and Dudley 1981). This concept provides a system-specific framework with which to rank species assemblage data on a qualitative scale.

Researchers at the Pennsylvania State Cooperative Wetlands Center (http://www.wetlands.psu.edu/) and EPA's Region III Office of Research and Development (http://www.epa.gov/ORD/) created the BCI from 34 reference sites in central Pennsylvania that represented a gradient of ecosystem conditions from nearly pristine to severely degraded. The reference sites were independently ranked according to (1) a previously developed human disturbance gradient (Brooks and others 1996); and (2) bird guild representation (i.e., the BCI). Upon a satisfactory demonstration that the BCI could discriminate among categories of ecosystem condition identified in a human-disturbance gradient, we applied the BCI to independent samples on 126 sites across MAHA. Sites were selected using EMAP's probability based sampling design, and therefore represented the total land area in the region. To verify the BCI's discriminatory properties, the BCI assessment was compared to independent gradients of landscape disturbance applied to both the 34 reference sites and the 126 MAHA sites.

Sampling

To develop the BCI we collected data at reference sites comprising a variable number of plots (3 to 11) placed every 50 to 200 m along a transect of up to 2-km long. Each sample site on the EMAP grid comprised five plots spaced every 200 m along a randomly oriented 1-km transect. At each plot, we sampled songbirds with a 10-minute, 30 m-radius point count between sunrise and 10:00 a.m. EDT (Hutto and others 1986; Manuwal and Carey 1991; Ralph and others 1993). For these analyses we used a total species list compiled from unlimited radius point counts at each of the five plots. Sampling took place within the "safe dates" for breeding birds, so we assumed that any birds detected were resident at each site throughout the breeding season (Brauning 1992).

At each bird sampling plot we sampled a suite of vegetation variables to characterize the local habitat. We recorded the percentage herbaceous cover of graminoids, forbs, mosses, and ferns in three, 5-m radius circular subplots located 15 m from plot center at 120º, 240º, and 360º degrees. We also recorded the percent cover of shrubs in the subplots in vertical strata at 0.00 to 0.50 m, 0.051 to 2.00 m, and 2.01 to 5.00 m, as well as the percent canopy cover of overstory trees. From plot center we used an angle gauge to sample

trees > 10-cm d.b.h. All live trees were identified by species and the d.b.h. recorded for both live trees and snags. In addition, at each plot we recorded canopy height, slope, and aspect, and assigned an Anderson Land Use Code (Anderson and others 1976).

To characterize the local landscape configuration, we obtained aerial photographs of the circular area bisected by each transect. For the probability based sites this resulted in a circular site (i.e., a landscape circle) with a 0.5-km radius covering an area of approximately 79 ha. We interpreted the photographs and polygons of six cover types, digitizing them using a GIS program; and we then entered the imaging into a modified version of the spatial analysis package SPAN (Miller and others 1997). The SPAN output provided information on landscape diversity, dominance, and contagion; the amount of edge between cover types, as well as the aerial coverage within the circular sites of urban development, agricultural land, forest, woody shrubs, open water, and barren land.

We collected data for this project at several spatial scales. We assessed vegetation structure and composition in three 79-m^2 subplots and summarized these data at the plot scale (roughly 700 m^2 or 0.07 ha). Several of the vegetation variables, e.g., canopy height, number of conifer stems > 10-cm d.b.h., were further compiled and expressed as a site-level variable (i.e., 79 ha). Bird data collected at the plot level were summarized as a total species pool for the entire site. We also analyzed landscape configuration with an aerial view of the entire 79-ha site. We intended the BCI assessment to be applied at the ecoregion scale which, in the case of the Mid-Atlantic highlands, included an area > 150,000 km^2.

Landscape Attributes

We used the coverage developed by Jones and others (1997) for metrics describing the state of forests at the watershed scale (i.e., the proportion of forest land: FOR%; the level of fragmentation of forests in watersheds: FORFRAG; the proportion of edge habitat: EDGE7; the proportion of interior forest habitat: INT7). Each watershed included several survey routes. It was likely that characteristics of forest habitat varied within watersheds, and heterogeneity in landscape characteristics within spatial units masked the influence of forest attributes on bird communities. So, we addressed the influence of forest characteristics at both watershed spatial scale and at a finer scale—a buffer 0.8-km wide along each BBS route. We used Jones and others (1997) coverage to compute the proportion of forestland within each buffer. This approach permitted assessment of forest condition in the immediate neighborhood of each survey route.

Statistical Approach

We used generalized linear models to test for the effect(s) of landscape characteristics on species richness and relative species richness. As relative richness varies between 0 and 1, these variables had to be transformed prior to fitting models (Cam and others 2000b). The landscape variables were also transformed in order to better approximate the assumption of normality (Boulinier and others 1998b, Cam and others 2000b). The variables used were: (1) the proportion of forest land, (2) the level of fragmentation, (3) the proportion of edge habitat, and (4) the proportion of interior habitat (these variables are defined by Jones and others (1997). We used a backward variable-selection procedure.

For analyses conducted at the route level, we tested for the effect of the proportion of forest habitat within the buffer while accounting for possible variation in species richness among physiographic strata (i.e., we performed an analysis of covariance). In the initial model we allowed the influence of the proportion of forest land to vary according to stratum (i.e., the initial model included an interaction between stratum and the landscape variable). For analyses of the influence of forest fragmentation on relative species richness, we excluded data from areas with a lower density of survey routes: data were too sparse in several watersheds to investigate the influence of forest characteristics at that level. As a result, we used the values corresponding to buffers along the routes.

Mapping of Community Attributes

We mapped most of the community attributes presented in this report at the HUC 8 (1:2,000,000) watershed scale. We used coverage developed by Jones and others (1997) to define watershed boundaries. We estimated watershed means using inverse distancing (Isaaks and Srivastava 1989) to interpolate mean abundance for a systematically located grid of points (see Cam and others 2000a for details). Then we averaged values from points in each watershed. The basic data correspond to one value per route; and we used these values to specify the data range and the five classes of equal width that were used in maps (figs. 120 and 121). If the same attribute were represented for two years, e.g., 1975 and 1990—we used values from 1990 to specify the

classes. Because all the BBS routes were not run in 1975 or 1990, we selected those that were run at least once within a temporal window of 5 years centered on 1975 or 1990, respectively, and retained data from the years closest to the earlier or later years.

BCI Development

We built the BCI with data on all the *Passeriformes* (perching birds), *Piciformes* (woodpeckers), *Cuculiformes* (Cuckoos), and *Columbiformes* (doves) that we had documented in MAHA from 1994 to 1996 (112 total species). We assigned birds to behavioral and physiological response guilds based on a literature review (Harrison 1975, Blake 1983, DeGraaf and others 1985, Roberts 1987, Brooks and Croonquist 1990, Freemark and Collins 1992, Santner and others 1992). From preliminary analyses of 32 guilds, we ultimately included the 16 guilds in 8 guild categories. We considered several factors (e.g., high correlation with other guilds, predictable response to land-cover change) in determining the final list of guilds to be included in BCI development.

Because we selected guilds specifically to reflect different aspects of the life history traits of each species, species may belong to several guilds, simultaneously. Guild assignments within each of the eight guild categories, however, were mutually exclusive, so species belonged to no more than eight guilds. For example, in the Migratory category, species are classified as either residents or temperate migrants (for statistical reasons we excluded tropical migrants). Also, guild assignments applied only to breeding season life history traits. For example, we considered the Eastern kingbird to be an insectivore, even though this species subsists largely on fruit in its wintering range (Terborgh 1989). O'Connell and others (2000) contains full technical descriptions of analyses used to develop and test the BCI as an indicator of ecological condition, as well as those used to associate BCI scores with landscape pattern.

Richness Analysis of Breeding Bird Survey Data (Chapter 24-Biodiversity)

The Breeding Bird Survey (BBS) is a long-term monitoring program begun in 1966 and continuing to yield data on species diversity of birds in the U.S.. The data are collected once a year (generally during June) on more than 4,000 permanent survey routes located along secondary roads in the U.S. Trained observers start 0.5 hour before sunrise and travel routes, each 24.5 miles long with 50 stops at 0.5-mile intervals where 3-minute point counts are conducted. During the counts, every bird seen or heard within a 0.25-mile radius is recorded.. In our analyses (Cam and others 2000a, 2000b), we focused on species of birds that breed primarily in forests (c.f. Boulinier and others 1998b; also see table 30 for species list). BBS routes retained for analysis in watersheds in the MAIA region are shown in fig. 122.

Estimation of species richness in forest bird communities

The distinctive feature of our analyses—based on BBS data—is that we were able to explicitly address sampling artifacts common in studies based on enumeration of species present in a given location. Often species are unreported during field observations, and consequently the number of species counted is necessarily less than the true number of species in the area of interest (Herwitz and others 1996, Nichols and Conroy 1996, Nichols and others 1998b). However, the number of species observed can be used to estimate species richness on the condition that all species present in an area are detected during sampling sessions (Boulinier and others 1998a, Burnham and Overton 1979, Conner and Simberloff 1978, Gilbert and Lee 1980, Herwitz and others 1996, Nichols and Conroy 1996, Nichols and others 1998a and 1998b, Preston 1979).

Appropriate statistical inference procedures were used to compensate for species known to be in an area yet might be missed in a survey, since previous BBS work indicated that not all species are detected in field sampling (Boulinier and others 1998a). We therefore used a recently developed approach to estimating species richness and related parameters (e.g., relative species richness), which does not assume that all species are detected during sampling efforts (Boulinier and others 1998a, Nichols and Conroy 1996, Nichols and others 1998a and 1998b).

Species richness

We took advantage of the replication of sampling effort along BBS routes (i.e., at each stop) to estimate species richness using capture-recapture methods for closed animal populations (e.g., see Boulinier and others 1998a, Burnham and Overton 1978 and 1979, Nichols and Conroy 1996, Nichols and others 1998a and 1998b, Otis and others 1978, Pollock 1982, Pollock and Otto 1983, Pollock and others 1990, Rexstad and Burnham 1991, White and others 1982). Replicate counts along each route can be used to document a detection history for each species, which can then be used to estimate both the probability of detecting species and the number of species (see Boulinier and others 1998a and

Nichols and others 1998a and 1998b for explanations of the statistical approach and its application to BBS data). We used the program COMDYN (Hines and others 1999) to estimate species richness. Following the suggestion of Boulinier and others (1998a), we used a model that permits variation in detection probabilities among species in the community (Otis and others 1978, White and others 1982). We also acknowledge other scientists who contributed to the analyses and graphics associated with bird species richness evaluations including John Sauer, Jim Nichols, Ian Thomas and Jim Hines of the USGS, Patuxent Wildlife Research Center, Laurel, MD (http://www.pwrc.usgs.gov/).

Relative species richness

Computation of relative species richness requires the definition of a reference species list (or species pool), and an estimation of the number of species in that list present on a BBS route in a given year (Cam and others 2000b). The ratio of species present on a BBS route in the past compared to species found in a BBS surveys by route in any given year corresponds to relative richness. The species detected only on the specific route of interest in the year of interest, and not detected in past BBS evaluations on the same route, must be excluded in order to ensure statistical independence of the numerator and denominator. We developed a reference list (table 30) of all the species detected within an 80-km radius centered on the route of interest and constrained to within the State and physiographic stratum to which the route belonged (Bystrak 1981; fig. 122) for all years of the Breeding Bird Surveys. Physiographic strata are spatial units harboring relatively homogeneous natural communities that could be expected to have generally consistent bird communities. Because this metric is sensitive to the number of BBS routes used to specify the species pool (Cam and others 2000b), we only retained data from physiographic strata that had a minimum of 5 BBS routes in the area used to specify the pool, a measure designed to avoid problems linked to small sample sizes. Exclusion of data from areas with a lower density of routes precluded mapping at the watershed scale, that is, data were too sparse to allow interpolation in several watersheds. Consequently, the results were only presented in a route-level map (fig. 123).

Technical Appendix E—Forest Management Practices and Stream Quality

K. Thornton
FTN Associates

This section provides additional information on the assessment of effects of forest management and harvesting practices on stream quality, and the contrast of these effects with stream quality in forests that have strictly implemented best management practices found in Chapter 18 (Aquatic Systems)

Hydrologic Modifications

There is evidence that in the Eastern U.S. responsibly-managed timber harvest does not cause significant adverse effects on forest soil productivity or forest water quality (Binkley and Brown 1993, McClimans 1980, Patric 1976 and 1978). However, careless logging, grazing, burning, and recreational uses can cause damages to both soil productivity and associated water quality.

Streamflow and stormflow discharges increased following timber harvesting due to loss or decreases in interception, evapotranspiration, and soil moisture storage (Dietterick and Lynch 1989, Hibbert 1967, Hornbeck and others 1978, Reinhart and others 1963). Increases in streamflow typically are proportional to the intensity of harvesting, e.g., clearcutting entire stands caused maximum changes in watershed and stream hydrology (Patric 1978). In poorly drained soils, a decrease in evapotranspiration also can cause a rise in water tables (Shepard 1994). Logging roads have been shown to augment stormflow; however, dirt logging roads within the MAIA region forests probably were too few and scattered to aggravate flooding on a regional scale.

Two experimental watersheds at Hubbard-Brooke (Hornbeck and others 1978; Dahlgren and Driscoll 1994) in New Hampshire were clearcut and then sprayed with herbicide to prevent regrowth. Streamflow increased at an average of 12.2 acre inches for the first 2 years after harvesting as a result of nearly eliminating evapotranspiration and reducing canopy interception losses. Increases were noted during the growing season and during summer low flow periods. This treatment, although severe, produced over 35 million gallons of water over the 3-year post-treatment period at a final cost of 39 cents per thousand gallons. Annual increases of 13.5 and 10.8 acres inches recorded at these watersheds compared with experiments conducted by Reinhart and others (1963), and Patric (1978), and Hibbert (1967) at Fernow Experimental Forest in West Virginia, and Coweeta Hydrologic Laboratory in North Carolina,

respectively. Reinhart and others (1963) noted 5 acre inches in streamflow increase the first year after cutting 80 percent of the area in West Virginia and Patric (1978) noted 14.8 acre inches and 11.4 acre inches of stormflow increased after clearing and herbicide treating for 3 years following cutting. Hibbert (1967) observed 11.3 to 16.1 acre inches of annual increases following complete timber cutting (Hornbeck and others 1978).

Hydrologic changes can occur in ecosystems when water storage and release patterns are affected by land development activities. In North Carolina, forest thinning decreased evapotranspiration and canopy interception, doubling drainage loss. Simulations suggest that clearcutting can increase annual runoff over natural undrained wetlands by as much as 13 percent (Richardson and McCarthy 1994). In Pennsylvania, clearcutting was shown to significantly increase water yield; however, these water yields returned to preharvesting levels within 4 years following cutting as a result of rapid regrowth (Lynch and others 1975).

The location of timber harvesting within a watershed also influenced hydrologic changes observed in the stream. Two mountain watersheds at the Fernow Experimental Forest in West Virginia were deforested and hydrologic effects were evaluated (Patric and Reinhart 1971). The upper half of one watershed (29 acres) and the lower half of another watershed (27 acres) were deforested in three stages that resulted in the removal of all vegetation > 1 inch diameter. Intensive spraying of 2,4,5-trichlorophenoxyacetic acid (2,4,5 T) kept areas free of vegetation. Complete deforestation was concluded 3 years following partial cutting. Results showed an increase in streamflow of an average of 6 inches for partial deforestation and 10 inches for complete deforestation (Patric and Reinhart 1971). Although substantial increases in seasonal and annual streamflow were observed in both watersheds, streamflow and peak storm discharge were greater in the watershed where the lower half was deforested first. The entire channel area on this watershed was barren early in the treatment, as opposed to the watershed where the upper half was cut first, which remained forested. Soil moisture remained at high levels, resulting in less water storage capacity for this watershed, which, in turn, contributed to greater streamflow and peak discharge.

Watershed experiments have shown that annual streamflow increases were proportional to the extent of forest cut (Hibbert 1967). Deforestation also increased maximum turbidities, but these were not high enough to cause concern. In Pennsylvania, the effect of successive clearcuts and herbicide treatment on forested watersheds was studied on a 106-acre watershed. First-year effects were observed after three successive areas were clearcut (21, 27, and 42 acres, respectively) within the watershed. Although increases in total stormflow volumes were not found to be directly proportional, the size of clearcut was important in determining the magnitude of increased stormflow. Significant increases in stormflow and peakflows were observed during the first growing season following each successive clearcut. Again, increases were attributed to reductions in interception, evapotranspiration, and soil moisture storage opportunities (Dietterick and Lynch 1989). In north central West Virginia, small increases in streamflow were observed following diameter-limited cutting at the Fernow Experimental Forest. Water quality and turbidity were unaffected except in areas of poorly located or poorly managed logging roads (Patric and Aubertin 1977).

Forest management practices can cause hydrologic changes in streams by changing the input of woody debris (Bren 1993). Woody debris traps and holds sediment in streams and provides habitat and food for certain biota. Deforestation, by decreasing the availability of large woody debris input to streams, disrupts habitat and food sources for stream-dependent species. Research in the MAIA region on hydrologic changes of streams associated with forestry management practices, primarily clearcutting, also addressed changes in annual and seasonal water yields. In North Carolina, clearcutting caused an annual water yield increase of 16 inches (Hoover 1945), and in Pennsylvania a 5.75 inch increase was measured after cutting 110 acres (Lynch and Corbett 1990). Additional studies in Pennsylvania showed that a 21 acre clearcut produced an increase of 2.8 inches of stormflow the first year (Lynch and others 1975).

Forest management practices such as clearcutting (partial or complete) and treatment with herbicide increased stream flow and water yield through the loss of evapotranspiration and canopy interception. Changes in woody debris also affected stream flow as well as biodiversity of the stream (Bren 1994). Pennsylvania and West Virginia studies showed increases as low as 2.8, 5.0, and 5.75 inches of streamflow, while greater increases of 10 to 16 inches were found in New Hampshire, North Carolina, and some West Virginia studies. Although there are changes in forest hydrology during and immediately following timber harvesting, these changes diminish quickly over time: within 4 to 5 years following harvesting, flows approached pre-harvest volumes (Binkley and Brown 1993).

Erosion and Sedimentation

Forest management practices, particularly the construction of forest roads, can impact the streams receiving runoff from affected watersheds. Fine sediment contributions to streams in forested watersheds resulted from poorly designed roads and ditches; from increased peak flows, where cutbank and slope failure, debris flows, stream erosion, and channel scour will occur; as well as diversion of streams at haul-road crossings (NCASI 1994, Wood and Armitage 1997).

Sediment production generated by storm flow from poorly designed logging roads was the most common cause of water quality degradation in forested watersheds (Binkley and Brown 1993, Corbett and others 1978, Riekerk 1983). Timber harvesting and site preparation, combined with drainage from secondary ditches, skidding, and planting resulted in large increases of suspended sediment loads into streams (Shepard 1994).

Research indicated that wood products can be harvested with adequate protection of soil and water (Patric 1978). Erosion can be reduced significantly through reduced disturbance of streamside soil and vegetation. Forest cover provided the best protection against soil erosion, while careless logging, grazing, burning, and recreation accelerated it.

Erosion rates and sediment yields from undisturbed forests in New England were among the lowest in the country (Patric 1976, Martin and Hornbeck 1994), mostly because New England mineral soils are well-drained, coarse-textured sandy loams with high infiltration capacities. Even so, sedimentation and turbidity resulting from logging were the major forms of water quality degradation in New England forest streams (Martin and Hornbeck 1994). Research showed that such effects could be reduced by careful management of forest land. Controlled experiments in sections in North Carolina—where there was no road building or logging—showed no discernible change in forest stream turbidity following timber harvest (Hibbert 1967). Also, logging on well-located and carefully managed skid roads caused only a brief increase in sedimentation in the North Carolina experiments. In West Virginia, major soil losses occurred on heavily used logging roads, but only minor soil losses were found on roads that were carefully managed there; erosion rates declined to negligible levels within 2 years of logging (Patric 1976).

Logging at the Fernow Experimental Forest caused little effect on water quality except for turbidity (Aubertin and Patric 1975). Careless logging-road construction caused minor increases in stream turbidity; however, during logging on a 38.3 acre watershed in 1957, stream turbidity increased greatly, to an average of 897 Jackson Turbidity Units (JTU)—probably as a result of locating a heavily used skid trail too near the channel. Turbidity levels decreased to pre-logging values of 2 JTUs within 2 years of the 1957 cutting (Patric and Aubertin 1977). Cutting was repeated in that watershed between 1972 and 1975. Turbidity then increased only slightly from a maximum of 10 JTUs before cutting to a maximum of 14 JTUs during cutting. The drinking water standards of 5 JTUs recommended by the U.S. Public Health Service in 1962 were exceeded only occasionally during stormflow (Patric and Aubertin 1977).

Drainage waters from logged forests in South Carolina monitored to quantify suspended sediments were found to be significantly greater in new, secondary ditches than in natural streams draining from an undisturbed hardwood stand (Shepard 1994). Concentrations of suspended sediment in the secondary ditch averaged 16.4 mg per L compared to 2.5 mg per L for the undisturbed hardwood forest. Timber harvesting and site preparation, when combined with the effects of drainage, logging, planting and skidding, resulted in large suspended sediment concentrations.

Clearcutting within watersheds increased stormflow—particularly during peak storm discharges—and increased stream velocities that resulted in extensive channel scouring, bank erosion, and bank slumping (Dietterick and Lynch 1989). The average annual suspended sediment concentration in a clearcut Pennsylvania watershed was 78 mg per L compared to an average annual suspended sediment concentration of 5.2 mg per L in the control watershed (Dietterick and Lynch 1989).

In West Virginia, the complete deforestation of two mountain watersheds described in the previous section increased maximum turbidity of stormflow to 130 and 55 ppm for the upper and lower watershed halves, respectively, compared to a maximum turbidity of 7 ppm for the same year for the control watershed (Patric and Reinhart 1971). Pre-treatment turbidity levels for the upper and lower half of the watershed were 16 and 29 ppm respectively. The turbidity level for the control watershed for that year was 6 ppm. At the Coweeta Hydrologic Laboratory in North Carolina, three logging roads were built in a watershed that subsequently was clearcut and cable-logged (Meyer and Tate 1983). Results from the 59-ha clearcut watershed were compared to those from an undisturbed 61-ha watershed. Several changes were apparent in the second order stream following the disturbances, including high inputs of sediment during road construction. Higher concentrations of both inorganic and

organic matter were observed in the clearcut watershed than in the undisturbed watershed. Accumulation of sediment in the pond was 13 percent higher than sediment accumulation prior to cutting.

The effects of forestry management practices on sedimentation processes vary in relation to climate, topography, geology, and soils. Timber harvesting alone has minimal effects on sedimentation in streams; however, when harvesting is combined with careless logging practices and improper road construction, sedimentation impacts to streams are accelerated. The bare soil exposed during road construction is the major contributor to increased sedimentation of streams (Corbett and others 1978).

Soil erosion from logging roads is a major concern in terms of the impacts forestry has on streams. Data on soil erosion from the central Appalachians were gathered from graveled and un-graveled sections of a minimum-standard road and from a higher standard graveled road built 50 years prior. Soil losses from un-graveled sections were significantly higher than losses from graveled sections. In southern North Carolina, soil losses from a bare roadbed were 90 tons/acre compared with 15.6 tons/acre from a graveled roadbed. Large amounts of rainfall within the 2 months following construction may have contributed to the high losses. On a West Virginia ungraveled logging road, 6 inches of rain within 3 hours created a 6 ft deep gully (Kochenderfer and Helvey 1987).

Experiments at Hubbard Brook Experimental Forest, NH, and Leading Ridge, PA, watersheds demonstrated that only minor increases in stream turbidity occurred with implementation of best management practices (BMPs). These BMPs included: specifying a maximum steepness for truck roads; installing water control devices; using buffer and filter strips; properly locating roads, landings, and stream crossings; and reducing specified wet weather operations (Binkley and Brown 1993, Lynch and Corbett 1990, Martin and Hornbeck 1994). In northern New Hampshire, there was a skid road culvert failure in a watershed that had been subjected to a whole-tree harvest. This incident caused peak stream turbidity values to occur after harvesting (Martin and Hornbeck 1994). At Leading Ridge, an increase of 5.9 mg per L for the harvested watershed was compared to an increase of 1.7 mg per L for the control following the use of BMPs. A detailed list of all BMPs used in the Leading Ridge watershed experiment is provided in Lynch and Corbett (1990).

Neary and others (1993) reported that applying a hexazione for site preparation on a mixed hardwood-pine stand reduced sediment yields to streams. The reported levels of 170 kg per ha were relatively close to the 100 kg per ha of natural sediment yields from undisturbed forests. These results were compared to first year sediment yields of 8,000 to 15,000 kg per ha produced by mechanical site preparation. If stream channel alteration occurs during harvesting, longer-term effects on stream quality can result, even though watershed erosion might subside in 2 to 5 years with vegetation regrowth.

Habitat Alteration

Both riparian and instream habitat can be degraded by poor timber practices. In Pennsylvania, Dietterick and Lynch (1989) found that successive clearcuts in a 42-ha watershed resulted in increased frequency and magnitude of channel scouring, bank erosion, and bank slumping. Removing the streamside, or riparian, vegetation during harvest operations can result in increased sediment and slash delivery to the stream, increased stream temperatures and nutrient concentrations, and decreased stream quality. Vegetated buffer zones, or riparian zones, not only protect the stream ecosystem but also ensure water quality in adjacent water bodies such as lakes, reservoirs, streams, and rivers. By supporting grasses, trees and shrubs, these buffer zones help spread incoming and overland channelized flow, thereby reducing the velocity, increasing the infiltration, and reducing the depth of water on the surface (Norris 1993). Such vegetated strips also buffer the stream from siltation, slash and debris accumulation, and excessive increases in temperature (Corbett and others 1978). Using BMPs to protect and maintain riparian buffer zones significantly reduces those problems (Adams and others 1995, Ice and others 1997).

Vegetated buffer zones have been shown to reduce nutrient and sediment levels in runoff from forestry practices. A 15-year study (Karr and Schlosser 1978) showed that suspended sediment levels increased by 200 percent in unmitigated clearcutting, while clearcutting with buffer strips in place showed a sediment increase of only 50 percent. Buffer strips have been shown to maintain or reduce water temperature increases following timber harvest. A 40 foot-wide buffer strip on each side of a mountain stream in North Carolina caused a substantial drop in water temperatures shortly after the stream entered the buffer strip. In West Virginia, a buffer strip 33- to 66-feet wide produced similar results. Temperatures averaged 11 degrees warmer in watersheds without a buffer strip (Corbett and others 1978).

A study conducted by Lynch and others (1985) evaluated streamside buffer zones as a component of BMPs

implemented in forestry operations. Samples were collected weekly for 3 years prior to cutting, and for 2.5 years afterwards, on three forested watersheds with areas from 43 to 123 ha. The study found that the buffer zones efficiently controlled sediment and turbidity levels during and following timber harvesting (Lynch and others 1985). Suspended sediment increased in two forest catchments, a 71-ha clearcut and a 304-ha area logged with buffer strips in place. However, increases in suspended sediments were less in the area with buffer strips.

Logging activities have been shown to affect streams by increasing sediment input, elevating water temperatures, and altering leaf detritus and woody debris (Adams and others 1995). BMPs are designed to protect water quality in streams draining harvested forests by reducing erosion and sediment delivery, preventing the introduction of slash to streams, managing riparian zones for shade and other streamside functions, and controlling chemical runoff (Ice and others 1997).

A Pennsylvania study evaluated the use of BMPs to prevent substantial impacts on water quality. The BMPs used comprised: harvesting only 43 percent of the watershed, retaining 30-meter-wide buffer strips, locating skid trails and roads in advance, and rehabilitating all roads and trails after logging. Sediment concentrations for the first year after logging, after harvest operators had followed the BMPs, were 1.7 mg per L for the control watershed, compared to 5.9 mg per L for the harvested watershed (Lynch and Corbett 1990).

In a separate 2-year study in central Pennsylvania that involved commercial clearcutting on three watersheds ranging from 43 ha to 123 ha, BMPs appeared to reduce impacts of sedimentation and turbidity (Lynch and others 1985). The BMPs used on this clearcut were designed to reduce, restrict, or eliminate sources of sediments—which have been attributed to logging roads, skid trails, and log landings. Turbidity levels were observed to increase in minor amounts following clearcutting.

Another study, at the Leading Ridge Experimental Watershed Research Unit, located in the Ridge and Valley Province of central Pennsylvania, evaluated the effectiveness of BMPs in controlling non-point-source pollution during and following commercial forest harvesting. Experimental watershed sizes were 303 acres for the undisturbed (control) watershed and 257 acres for the commercial clearcut, of which 110 acres were cut. Water quality and quantity data were collected from those watersheds 3 years prior to harvest and 11 years afterwards. BMPs included but were not limited to: leaving a protective buffer strip on each side of all streams, properly retiring (putting to bed) all logging roads, and prohibiting logging. The BMPs were very effective in controlling non-point-source pollution from the clearcut watershed. Although increases in nitrate and potassium concentrations, and temperature and turbidity levels were observed at statistically significant levels during the first 2 years following harvest, only turbidity levels exceeded drinking water standards (Lynch and Corbett 1990).

Chemical Contributions

Chemicals regularly are used in managed forests for fertilization, the control of nuisance insects such as the Gypsy moth, and site preparation for timber harvesting or regeneration. Chemical contamination of forest-drained streams occurs in watersheds following the application of herbicides, pesticides, and fertilizers. Chemicals can enter streams and alter stream water chemistry following application, although concentrations usually are negligible in the short term (Binkley and Brown 1993). Forest managers were concerned that accelerated loss of nutrients following timber harvesting might lessen stream water quality and result in accelerated eutrophication (diminishing of dissolved oxygen) (Corbett and others 1978). Increased chemical contributions to streams during and following harvesting have been recorded for nutrients (particularly nitrogen), cations and anions (Ca, K, Na, Mg, SO_4, Cl), and herbicides or insecticides used to control weeds or insects following harvest (Corbett and others 1978, Dahlgren and Driscoll 1994, Edwards and others 1991, Norris and others 1984).

A fertilization study conducted on forested watersheds ranging from 2.6 ha to 3.8 ha in the Fernow Experimental Forest, WV, showed stream nitrate concentrations 18 times higher, and Ca and Mg concentrations three times higher, than under untreated conditions, following application of fertilizer at rates comparable to commercial forest management practices (Edwards and others 1991).

A related experiment conducted at Fernow Experimental Forest also evaluated nitrate and phosphate concentrations in stream water following fertilization experiments. In the first study, the watershed was treated with 225 kg N per ha after clearcutting. Results reported peak nitrate concentrations of 16 mg per L following fertilization. The second study examined the effects of nitrate and phosphate fertilization in stream water chemistry and nutrient yields. Results from this study showed an increase of nitrate above the 10 mg per L drinking-water standard for 3 weeks following application (Binkley and Brown 1993).

In 1998, the EPA halted use of the herbicide 2, 4, 5 T, used extensively from 1940 through 1978 to control vegetation for site preparation, timber stand improvement, and right-of-way management. A study was conducted at the Fernow Experimental Forest on a 22-ha watershed to develop a better understanding of the movement and persistence of 2, 4, 5 T in the forest environment, from canopy to forest floor to discharge into streams. Results showed that levels of 2, 4, 5 T in vegetation declined over time, with a 99.9 percent decline 1 year after application. The rate of loss from the forest floor was 50 percent in the first week following application, but declined over time, exhibiting a nonlinear decline due to the continued addition of 2, 4, 5 T from precipitation (rinsing leaves) and fresh-fall litter. The concentration of 2, 4, 5-T in the soil at depths of 0 to 15 cm decreased by 90 percent in 1 month, then remained relatively constant. The failure to detect 2, 4, 5 T in stream water more than 13 days after application indicated that there was little potential for long-term stream contamination from use of this herbicide n managed forests (Norris and others 1984).

Studies conducted on forest cutting have reported increases in nitrate concentrations in stream water following harvest operations; however, most of the increases did not breach the drinking water standard of 10-mg N per L (Binkley and Brown 1993). Clearcutting of mixed northern hardwood forests in the Northeast has been identified as a major cause of stream acidification and nutrient loss from forest ecosystems (Dahlgren and Driscoll 1994), with significant nutrient leaching shown to occur when all vegetation has been killed by herbicides (Corbett and others 1978).

The chemical response to clearcutting was studied on a whole-tree clearcut watershed, and an uncut reference watershed, at the Hubbard Brook Experimental Forest in New Hampshire. Effects were greatest during the second year after harvest, declining to reference levels within 5 years. The chemical response included increased acidification and elevated concentrations of potentially toxic Al in stream water (Dahlgren and Driscoll 1994).

Four experimental watersheds in New Hampshire exhibited similar species composition prior to cutting; afterwards they exhibited different species composition and nutrient concentrations. The watersheds, which were 27-, 21-, 12-, and 78-years-old, were separately clearcut, strip cut, whole-tree harvested, and logged corresponding to the age of included trees. All of the watersheds showed elevated losses of nitrate and nutrient cations during the first few years following clearcutting (Pardo and others 1995).

The effects of harvesting on watersheds at the Hubbard Brook Experimental Forest in New Hampshire included substantial nitrate increases after strip-cut harvesting. These studies involved normal forestry practices, sans herbicide use to inhibit regrowth. Although results did show pulses that exceeded the drinking water standard for nitrate, the average annual nitrate concentration never exceeded that level (Binkley and Brown 1993).

A North Carolina plantation was thinned and then fertilized with diammonium phosphate and urea (167 kg N ha^{-1}, and 28 kg P ha^{-1}, respectively). Results showed elevated levels of several forms of phosphate and nitrogen in the water draining these watersheds. Concentrations of these nutrients had all dropped to pre-treatment levels 3 weeks following application (Shepard 1994). While there is an increase in stream chemical concentrations for many constituents following timber harvesting, these increases generally return to pre-harvesting concentrations within a few years (Binkley and Brown 1993).

Altered Stream Biology

Forest management practices can result in the removal or addition of woody debris, altered leaf litter input, increased stream scour, increased chemical concentrations, and sediment delivery into stream spawning gravel beds—any of which can alter the abundance and composition of the stream biota (Binkley and Brown 1993, Neary and others 1993, Wigley and Roberts 1994). Forested stream ecosystems depend on terrestrial vegetation as a primary energy source. Aquatic invertebrates and vertebrates depend on organic input into streams for energy, habitat, and food. Forest management practices such as clearcutting can reduce the amount of leaf litter found in streams, which affects leaf-shredding insects. Leaf-shredding insects play an important role in stream ecosystems by making fine organic material available to downstream consumers. Reducing the leaf shredding insect assemblages reduces available seston—minute organic or inorganic matter—for the entire stream community (Stout and others 1993).

In a Pennsylvania study that involved successive clearcuts in a 106-acre watershed, Dietterick and Lynch (1989) found that stormflow (increased frequency and magnitude of water resulting from a precipitation event) caused considerable channel scouring, bank erosion, and slumping of the bank. A substantial increase in stream turbidity and suspended sediment affects a wide variety of the aquatic organisms that inhabit such streams.

195

Forest application of chemicals (i.e., pesticides, insecticides, herbicides) is an integral part of intensive forestry management practices. Chemicals are used on managed forests for the purposes of producing improved seed, improving quality and vigor of nursery seedlings, reducing weed composition, controlling nuisance insects, and increasing tree growth (Neary and others 1993). This publication described how those chemicals caused detrimental effects in forest stream water quality, changes of nutrient concentrations of soil and water, and the biodiversity of stream inhabitants.

In the Mid-Atlantic States the forest pesticide diflubenzuron (DFB) was commonly used to control gypsy moth. In a West Virginia study, four watersheds (two treated and two reference), were monitored 3 years prior to application and 1 year after application of DFB. Watershed sizes ranged from 14.2 ha to 41.0 ha, and elevations from 650 to 870 m. Mean densities of mayfly and stonefly shredders, as well as the densities of two predator species, decreased in the streams of treated watersheds following DFB application. Densities of these invertebrates in reference watersheds either remained constant or increased up to 100 percent (Hurd and others 1996).

The relationship between biodiversity and the effects of forestry pesticides had not been studied sufficiently, although aquatic invertebrates were recognized to be useful indicators of environmental changes (Neary and others 1993). Temporal variations in abundance and diversity of species were observed in a second-order stream below an application site for hexazinone. Reductions in species diversity were observed in summer months, during a period of drying of forest floor, after defoliation of the forest overstory and understory (Mayack and others 1982). Another study reported a reduction of insect populations in a first-order stream following the application of malathion. We also observed indirect effects after application of this chemical. A reduction in leaf breakdown and fine particulate organic matter transport were observed in treatment watersheds (Wallace and others 1982).

A paired watershed study was conducted at Coweeta Hydrologic Laboratory, NC, to assess the effects of clearcutting and road construction on stream quality. Prior to logging, the water and sediment regime and invertebrate community were similar in both watersheds. Sediment accumulation, discharge, and water temperature all increased following logging. As an indirect effect of these changes, food resources increased; subsequently, invertebrate shredder assemblages decreased, and collector-gatherer insect assemblages increased (Meyer and Tate 1983).

Increased sediment accumulation in fish spawning gravel beds reduced the diffusion of oxygen into the spawning beds, and decreased fecundity of stream fishes (Binkley and Brown 1993). Accelerated erosion and sedimentation from logging operations increased turbidity, which reduced primary biotic productivity by blocking light transmission (Corbett and others 1978).

The effect of timber harvesting followed by herbicide treatment has been shown to cause an increase in stream temperatures (Corbett and others 1978), as the low intensity, diffused light under the forest canopy is replaced by direct solar radiation. Changes in stream temperature affected fish by influencing their metabolic rate, hatching, development, and migration patterns. Certain fish species congregated in specific thermal ranges. Warming headwater streams resulted in a predominance of brown trout further upstream in former brook trout territory, and movement of warm water species into an area they had not previously inhabited (Corbett and others 1978).

Cumulative Effects

Cumulative effects of forest management practices on stream ecosystems were illustrated conceptually (fig. 98). For example, following clearcutting evapotranspiration (ET) was significantly reduced in a watershed, and resulted in increased runoff and peak flows (Q) during storm events. Increased runoff resulted in greater sediment delivery to streams, along with increased discharge of soil nutrients—both macro- and micronutrient export—from the watershed. Increased sediment delivery increased sediment accumulation in the stream. Increased stream velocities from increased discharge and peak flows resulted in both streambed and bank scour, affecting the streams' physical habitat. Greater solar insolation reached streams—particularly in smaller first- and second-order streams—because overstory vegetation had been removed. The result was increased primary productivity as a result of increased nutrient export from the watershed. Vegetative uptake of nitrate by trees was significantly diminished, which allowed nutrient concentrations to increase in streams. Increased sediment accumulation, and alteration of physical habitats, affected the biological community of stream and riparian areas. Increased slash during logging significantly altered transport of woody debris and leaf litter to the insect community, and the size distribution of organic

matter (fine versus coarse), which, in turn, altered both composition and abundance of species in the community (fig. 98). Forest management practices had significant local-scale effects on stream ecosystems. These effects could be reduced by implementing BMPs as an integral part of forest management (NCASI 1994, Norris 1993).

Technical Appendix F—Common and Binomial Names of Species

K.W. Stolte
Southern Research Station, USDA Forest Service

Plant common name	Binomial name (Authority)
alder	*Alnus* spp.
American beech	*Fagus grandifolia* (Ehrh.)
American chestnut	*Castanea dentata* (Marsh.) Borkh.
American holly	*Ilex opaca* (Aiton)
apple	*Malus* spp.
arborvitae	*Thuja occidentalis* (L.)
ash-walnut-cherry	*Fraxinus-Juglans-Prunus* spp.
ash	*Fraxinus* spp.
aspen-birch	*Populus-Betula* spp.
Atlantic white cedar	*Chamaecyparis thyoides* (L.) Britton, Sterns & Poggenb.
bald cypress	*Taxodium distichum* (L.) L.C. Rich.
balsam fir	*Abies balsamea* (L.) Mill.
basswood	*Tilia* spp.
beech	*Fagus* spp.
beech–maple	*Fagus* - *Acer* spp.
birch	*Betula* spp.
black cherry	*Prunus serotina* (Ehrh.)
black gum	*Nyssa sylvatica* (Marsh.)
black oak	*Quercus velutina* (Lam.)
black walnut	*Juglans nigra* (L.)
butternut	*Juglans cinerea* (L.)
Carolina hemlock	*Tsuga caroliniana* (Engelm.)
chenopodiaceous	Chenopodiaceae family
chestnut	*Castnea* spp.
chestnut oak	*Quercus prinus* (L.)
chestnut oak–northern red oak	*Quercus prinus* (L.) - *Quercus rubra* (L.)
dogwoods	*Cornus florida* (L.); *Cornus sericea* ssp. *sericea* (L.)
eastern cottonwood	*Populus deltoides* (Bartr. ex Marsh.)
eastern hemlock	*Tsuga canadensis* (L.) Carr.
elm	*Ulmus* spp.
elm-ash	*Ulmus-Fraxinus* spp.
ericaceous	Ericaceae family
fir	*Abies* spp.
flowering pear	*Pyrus calleryana* (Decne.)
ginseng	*Panax quinquefolius* (L.)
gray birch	*Betula populifolia* (Marsh.)
green ash	*Fraxinus pennsylvanica* Marsh.
hawthorn	*Cratageus* spp.
hemlock	*Tsuga* spp.
hemlock-white pine	*Tsuga* spp.-*Pinus strobus* (L.)
hickory	*Carya* spp.
holly	*Ilex* spp.
honey locust	*Gleditsia triacanthos* (L.)
juniper	*Juniperus* spp.
linden	*Tilia* spp.
loblolly pine	*Pinus taeda* (L.)
loblolly pine-shortleaf pine	*Pinus taeda* (L.) - *Pinus echinata* (Mill.)

Plant common name	Binomial name (Authority)
locust	*Robinia* spp. & *Gleditsia* spp.
magnolia	*Magnolia* spp.
maple	*Acer* spp.
maple-beech-birch (also northern hardwoods)	*Acer-Fagus-Alnus* spp.
mountain laurel	*Kalmia latifolia* (L.)
northern hardwoods (also maple-beech-birch)	*Acer-Fagus-Alnus* spp.
Norway maple	*Acer platanoides* (L.)
northern red oak	*Quercus rubra* (L.)
oak	*Quercus* spp.
oak-chestnut	*Quercus-Castenea* spp.
oak-gum-cypress	*Quercus-Nyssa-Taxodium* spp.
oak-hickory	*Quercus-Carya* spp.
oak–hickory–pine	Quercus-Carya-Pinus spp.
pine	*Pinus* spp.
pitch pine	*Pinus rigida* (Mill.)
pond pine	*Pinus serotina* (Michx.)
ponderosa pine	*Pinus ponderosa* (L.)
poplar	*Populus* spp.
Jeffrey pine	*Pinus Jeffreyi* (Balf.)
red cedar	*Juniperus virginiana* (L.)
red maple	*Acer rubrum* (L.)
red oak	*Quercus rubra* (L.)
red oaks	*Quercus rubra* (L.)-*Quercus falcata* (Michx.)
red pine	*Pinus resinosa* (Aiton)
red spruce	*Picea rubens* (Sarg.)
rhododendron	*Rhododendron* spp.
scarlet oak	*Quercus coccinea* (Muenchh.)
shagbark hickory	*Carya ovata* (Mill.) K. Koch
shortleaf pine	*Pinus echinata* (Mill.)
slash pine	*Pinus elliottii* (Engelm.)
sourwood	*Oxydendrum arboretum* (L.)
southern red oak	*Quercus falcata* (Michx.)
spruce	*Picea* spp.
spruce-fir	*Picea-Fagus* spp.
sugar maple	*Acer saccharum* (Marsh.)
sycamore	*Platanus occidentalis* (L.)
swamp tupelo	*Nyssa biflora* (Walt.)
sweet birch	*Betula lenta* (L.)
sweet birch-chestnut oak	*Betula lenta* (L.)-*Quercus prinus* (L.)
sweet gum	*Liquidambar styraciflua* (L.)
table mountain pine	*Pinus pungens* (Lamb.)
tobacco	*Nicotiana* spp.
tulip poplar (also yellow poplar)	*Liriodendron tulipifera* (L.)
Virginia pine	*Pinus virginiana* (P. Mill.)
walnut	*Juglans* spp.

Plant common name	Binomial name (Authority)
white ash	*Fraxinus americana* (L.)
white birch	*Betula platyphylla* (Sukatschev)

white oak	*Quercus alba* (L.)
white oaks	*Quercus* spp.
white pine	*Pinus strobus* (L.)
willow	*Salix* spp.
yellow birch	*Betula alleghaniensis* (Britt.)
yellow poplar	*Liriodendron tulipifera* (L.)

Exotic plant common name	**Binomial name (Authority)**
Queen Anne's lace	*Daucus carota* (L.)
red clover	*Trifolium pratense* (L.)
narrowleaf plaintain	*Plantago lanceolata* (L.)
ox-eye daisy	*Leucanthemum vulgare* (Lam.)
sheep sorrel	*Rumex acetosella* (L.)
barnyard grass	*Echinochloa crus-galli* (L.) Beauv.
white clover	*Trifolium repens* (L.)
yellow sweet clover	*Melilotus officinalis* (L.) Lam.
woolly mullein	*Verbascum thapsus* (L.)
asiatic day flower	*Commelina communis* (L.)

Insect common name	**Binomial name (Authority)**
beech scale insect	*Cryptococcus fagisuga* (Lind.)
EPT	*Ephemeroptera/Plecopteran/Trichoptera* (orders)
hemlock woolly adelgid	*Adelges tsugae* (Annand)
gypsy moth	*Lymantria dispar* (L.)
southern pine beetle	*Dendroctonus frontalis* (Zimmermann)
two-lined chestnut borer	*Agrilus bilineatus* (Weber)

Fungal common name	**Binomial name (Authority)**
armillaria	*Armillaria mellea* [Vahl (P. Karst.)]
beech bark disease	*Nectria coccinea* [Pers. (Fr.)]
Butternut canker	*Sirococcus clavigignenti-juglandacearum* (Nair, Kost. & Kuntz)
chestnut blight	*Cryphonectria parasitica* [Murrill (M.E. Barr)]
dogwood anthracnose	*Discula destructiva* (Redlin)
Dutch elm disease	*Ophiostoma ulmi* [Buisman (Nannf.)]
Entomophaga	*Entomophaga maimaiga* (Hum., Shim.& Sop)
fusiform rust	*Cronartium quercuum* (Berk.) Miyabe ex Shiraif. ssp. *fusiforme* (Hedge. & N. Hunt)
nectaria	*Nectria coccinea* var. *faginata* (Lohman, Watson, and Ayers); *Nectaria galligena* (Bres.)
mushrooms (edible)	*Basidiomycetes* spp.
white pine blister rust	*Cronartium ribicola* (J. C. Fisch.)

Wildlife game common name	**Binomial name (Authority)**
black bear	*Ursus americanus* (Pallas)
bobwhite quail	*Colinus virginianus* (L.)
eastern cottontail rabbit	*Sylvilagus floridanus* (J.A. Allen)
gray squirrel	*Sciurus carolinensis* (Gmelin)
white-tail deer	*Odocoileus virginianus* (Zimmerman)
wild turkey	*Meleagris gallopavo* (L.)

AOU[a]	Bird common name	Binomial name (Authority)
2970	Blue Grouse	*Dendragapus obscurus* (Say, 1823)
3000	Ruffed Grouse	*Bonasa umbellus* (Linnaeus, 1766)
3100	Wild Turkey	*Meleagris gallopavo* (Linnaeus, 1758)
3120	Band-tailed Pigeon	*Patagioenas fasciata* (Say, 1823)
3270	Swallow-tailed Kite	*Elanoides forficatus* (Linnaeus, 1758)
3320	Sharp-shinned Hawk	*Accipiter striatus* (Vieillot, 1808)
3330	Cooper's Hawk	*Accipiter cooperii* (Bonaparte, 1828)
3340	Northern Goshawk	*Accipiter gentiles* (Linnaeus, 1758)
3390	Red-shouldered Hawk	*Buteo lineatus* (J. F. Gmelin, 1788)
3430	Broad-winged Hawk	*Buteo platypterus* (Vieillot, 1823)
3570	Merlin	*Falco columbarius* (Linnaeus, 1758)
3680	Barred Owl	*Strix varia* (Barton, 1799)
3730	Eastern Screech-owl	*Megascops asio* (Linnaeus, 1758)
3732	Western Screech-owl	*Megascops kennicottii* (Elliot, 1867)
3790	Northern Pygmy-owl	*Glaucidium gnoma* (Wagler, 1832)
3870	Yellow-billed Cuckoo	*Coccyzus americanus* (Linnaeus, 1758)
3880	Black-billed Cuckoo	*Coccyzus erythropthalmus* (A. Wilson, 1811)
3930	Hairy Woodpecker	*Picoides villosus* (Linnaeus, 1766)
3940	Downy Woodpecker	*Picoides pubescens* (Linnaeus, 1766)
3950	Red-cockaded Woodpecker	*Picoides borealis* (Vieillot, 1809)
3960	Ladder-backed Woodpecker	*Picoides scalaris* (Wagler, 1829)
3970	Nuttall's Woodpecker	*Picoides nuttallii* (Gambel, 1843)
3990	White-headed Woodpecker	*Picoides albolarvatus* (Cassin, 1850)
4000	Black-backed Woodpecker	*Picoides arcticus* (Swainson, 1832)
4010	Three-toed Woodpecker	*Picoides tridactylus* (Linnaeus, 1758)
4020	Yellow-bellied Sapsucker	*Sphyrapicus varius* (Linnaeus, 1766)
4021	Red-naped Sapsucker	*Sphyrapicus nuchalis* (S. F. Baird, 1858)
4030	Red-breasted Sapsucker	*Sphyrapicus ruber* (Gmelin, 1788)
4040	Williamson's Sapsucker	*Sphyrapicus thyroideus* (Cassin, 1852)
4050	Pileated Woodpecker	*Dryocopus pileatus* (Linnaeus, 1758)
4070	Acorn Woodpecker	*Melanerpes formicivorus* (Swainson, 1827)
4090	Red-bellied Woodpecker	*Melanerpes carolinus* (Linnaeus, 1758)
4100	Golden-fronted Woodpecker	*Melanerpes aurifrons* (Wagler, 1829)
4160	Chuck-will's-widow	*Caprimulgus carolinensis* (Gmelin, 1789)
4170	Whip-poor-will	*Caprimulgus vociferous* (A. Wilson, 1812)
4240	Vaux's Swift	*Chaetura* vauxi (J. K. Townsend, 1839)
4280	Ruby-throated Hummingbird	*Archilochus colubris* (Linnaeus, 1758)
4290	Black-chinned Hummingbird	*Archilochus alexandri* (Bourcier & Mulsant, 1846)
4320	Broad-tailed Hummingbird	*Selasphorus platycercus* (Swainson, 1827)
4330	Rufous Hummingbird	*Selasphorus rufus* (Gmelin, 1788)
4360	Calliope Hummingbird	*Stellula calliope* (Gould, 1847)
4520	Great Crested Flycatcher	*Myiarchus crinitus* (Linnaeus, 1758)
4530	Brown-crested Flycatcher	*Myiarchus tyrannulus* (Statius Muller, 1776)
4590	Olive-sided Flycatcher	*Contopus cooperi* (Nuttall, 1831)
4610	Eastern Wood-pewee	*Contopus virens* (Linnaeus, 1766)
4620	Western Wood-pewee	*Contopus sordidulus* (P. L. Sclater, 1859)
4630	Yellow-bellied Flycatcher	*Empidonax flaviventris* (W.M.Baird & S.F.Baird, 1843)
4640	Cordilleran Flycatcher	*Empidonax occidentalis* (Nelson, 1897)
4641	Pacific-slope Flycatcher	*Empidonax difficilis* (S. F. Baird, 1858)
4650	Acadian Flycatcher	*Empidonax virescens* (Vieillot, 1818)
4670	Least Flycatcher	*Empidonax minimus* (W. M. Baird & S. F. Baird, 1843)

AOU[a]	Bird common name	Binomial name (Authority)
4680	Hammond's Flycatcher	*Empidonax hammondii* (Xantus de Vesey, 1858)
4690	Dusky Flycatcher	*Empidonax oberholseri* (A. R. Phillips, 1939)
4710	Vermilion Flycatcher	*Pyrocephalus rubinus* (Boddaert, 1783)
4780	Steller's Jay	*Cyanocitta stelleri* (J. F. Gmelin, 1788)
4840	Gray Jay	*Perisoreus canadensis* (Linnaeus, 1766)
4910	Clark's Nutcracker	*Nucifraga columbiana* (Wilson, 1811)
5140	Evening Grosbeak	*Coccothraustes vespertinus* (W. Cooper, 1825)
5150	Pine Grosbeak	*Pinicola enucleator* (Linnaeus, 1758)
5170	Purple Finch	*Carpodacus purpureus* (Gmelin, 1789)
5180	Cassin's Finch	*Carpodacus cassinii* (S. F. Baird, 1854)
5210	Red Crossbill	*Loxia curvirostra* (Linnaeus, 1758)
5220	White-winged Crossbill	*Loxia leucoptera* (Gmelin, 1789)
5330	Pine Siskin	*Carduelis pinus* (A. Wilson, 1810)
5670	Slate-colored Junco	*Junco hyemalis* (Linnaeus, 1758)
5679	Oregon Junco	*Junco oreganos* (J. K. Townsend, 1837)
5708	Gray-headed Junco	*Junco caniceps* (Woodhouse, 1853)
5750	Bachman's Sparrow	*Aimophila aestivalis* (Lichtenstein, 1823)
5950	Rose-breasted Grosbeak	*Pheucticus ludovicianus* (Linnaeus, 1766)
5960	Black-headed Grosbeak	*Pheucticus melanocephalus* (Swainson, 1827)
6070	Western Tanager	*Piranga ludoviciana* (A. Wilson, 1811)
6080	Scarlet Tanager	*Piranga olivacea* (Gmelin, 1789)
6090	Hepatic Tanager	*Piranga flava* (Vieillot, 1822)
6100	Summer Tanager	*Piranga rubra* (Linnaeus, 1758)
6240	Red-eyed Vireo	*Vireo olivaceus* (Linnaeus, 1766)
6260	Philadelphia Vireo	*Vireo philadelphicus* (Cassin, 1851)
6270	Warbling Vireo	*Vireo gilvus* (Vieillot, 1808)
6280	Yellow-throated Vireo	*Vireo flavifrons* (Vieillot, 1808)
6290	Solitary Vireo	*Vireo solitarius* (Wilson, 1810)
6320	Hutton's Vireo	*Vireo huttoni* (Cassin, 1851)
6360	Black-and-white Warbler	*Mniotilta varia* (Linnaeus, 1766)
6370	Prothonotary Warbler	*Protonotaria citrea* (Boddaert, 1783)
6380	Swainson's Warbler	*Limnothlypis swainsonii* (Audubon, 1834)
6390	Worm-eating Warbler	*Helmitheros vermivorum* (Gmelin, 1789)
6470	Tennessee Warbler	*Vermivora peregrine* (A. Wilson, 1811)
6480	Northern Parula	*Parula americana* (Linnaeus, 1758)
6500	Cape May Warbler	*Dendroica tigrina* (Gmelin, 1789)
6540	Black-throated Blue Warbler	*Dendroica caerulescens* (Gmelin, 1789)
6550	Myrtle Warbler	*Dendroica coronata* (Linnaeus, 1766)
6560	Audubon's Warbler	*Dendroica coronata auduboni* (Townsend, JK, 1837)
6570	Magnolia Warbler	*Dendroica magnolia* (A. Wilson, 1811)
6580	Cerulean Warbler	*Dendroica cerulean* (A. Wilson, 1810)
6600	Bay-breasted Warbler	*Dendroica castanea* (A. Wilson, 1810)
6610	Blackpoll Warbler	*Dendroica striata* (J. R. Forster, 1772)
6620	Blackburnian Warbler	*Dendroica fusca* (Statius Muller, 1776)
6630	Yellow-throated Warbler	*Dendroica dominica* (Linnaeus, 1766)
6640	Grace's Warbler	*Dendroica graciae* (S. F. Baird, 1865)
6670	Black-throated Green Warbler	*Dendroica virens* (Gmelin, 1789)
6680	Townsend's Warbler	*Dendroica townsendi* (J. K. Townsend, 1837)
6690	Hermit Warbler	*Dendroica occidentalis* (J. K. Townsend, 1837)
6710	Pine Warbler	*Dendroica pinus* (A. Wilson, 1811)
6740	Ovenbird	*Seiurus aurocapilla* (Linnaeus, 1766)

AOU[a]	Bird common name	Binomial name (Authority)
6750	Northern Waterthrush	*Seiurus noveboracensis* (Gmelin, 1789)
6760	Louisiana Waterthrush	*Seiurus motacilla* (Vieillot, 1809)
6770	Kentucky Warbler	*Oporornis formosus* (A. Wilson, 1811)
6840	Hooded Warbler	*Wilsonia citrine* (Boddaert, 1783)
6860	Canada Warbler	*Wilsonia canadensis* (Linnaeus, 1766)
6870	American Redstart	*Setophaga ruticilla* (Linnaeus, 1758)
7220	Winter Wren	*Troglodytes troglodytes* (Linnaeus, 1758)
7260	Brown Creeper	*Certhia americana* (Bonaparte, 1838)
7270	White-breasted Nuthatch	*Sitta carolinensis* (Latham, 1790)
7280	Red-breasted Nuthatch	*Sitta canadensis* (Linnaeus, 1766)
7290	Brown-headed Nuthatch	*Sitta pusilla* (Latham, 1790)
7300	Pygmy Nuthatch	*Sitta pygmaea* (Vigors, 1839)
7310	Tufted Titmouse	*Baeolophus bicolor* (Linnaeus, 1766)
7320	Black-crested Titmouse	*Baeolophus atricristatus* (Cassin, 1850)
7330	Plain Titmouse	*Parus inornatus* (Gambel, 1845)
7350	Black-capped Chickadee	*Poecile atricapillus* (Linnaeus, 1766)
7360	Carolina Chickadee	*Poecile carolinensis* (Audubon, 1834)
7380	Mountain Chickadee	*Poecile gambeli* (Ridgway, 1886)
7400	Boreal Chickadee	*Poecile hudsonica* (J. R. Forster, 1772)
7410	Chestnut-backed Chickadee	*Poecile rufescens* (J. K. Townsend, 1837)
7480	Golden-crowned Kinglet	*Regulus satrapa* (Lichtenstein, 1823)
7510	Blue-gray Gnatcatcher	*Polioptila caerulea* (Linnaeus, 1766)
7540	Townsend's Solitaire	*Myadestes townsendi* (Audubon, 1838)
7550	Wood Thrush	*Hylocichla mustelina* (J. F. Gmelin, 1789)
7560	Veery	*Catharus fuscescens* (Stephens, 1817)
7570	Gray-cheeked Thrush	*Catharus minimus* (Lafresnaye, 1848)
7580	Swainson's Thrush	*Catharus ustulatus* (Nuttall, 1840)
7590	Hermit Thrush	*Catharus guttatus* (Pallas, 1811)
7630	Varied Thrush	*Ixoreus naevius* (J. F. Gmelin, 1789)
7670	Western Bluebird	*Sialia mexicana* (Swainson, 1832)

[a] American Ornithologist Union codes

Sources:
Plants Database (http://plants.usda.gov/);
Integrated Taxonomic Information System (ITIS) website (http://www.mbr-pwrc.usgs.gov/Infocenter/i3950id.html);
Wikispecies (http://species.wikimedia.org/wiki/Main_Page);
USDA Forest Service Forest Health Protection (http://www.fs.fed.us/foresthealth/);
Patuxent Wildlife Refuge (http://www.fws.gov/northeast/patuxent/);
Index Fungorum (http://www.indexfungorum.org/Names/Names.asp);
Google (http://www.google.com/ig?hl=en)

Stolte, Kenneth W., Conkling, Barbara L., Fulton, Stephanie and Bradley, M. Patricia. 2012. State of mid-atlantic region forests in 2000. Gen. Tech. Rep. SRS–162. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 203 p.

Wet and warm climate, mountainous topography, and deep rich soils produced one of the most magnificent and diverse temperate forests in the world. In 1650 the Mid-Atlantic forests covered 95 percent of the region, but were greatly reduced in 1900 by extensive tree harvesting, and conversion to farms and pastures. Settlement of forests also led to severe wildfires, soil erosion, and destruction of wildlife. Recovery began in the early 1900s, and later improvements in agricultural allowed millions of acres to return to forest cover. Suppression of catastrophic wildfires reduced flooding and watershed degradation, and wildlife management returned native animal and fish populations. Forest management improvements led again to productive and diverse forests in more mature stages of development. By the end of the 20th century, the Mid-Atlantic forests covered 61 percent of the land area and produced numerous products that brought social and economic benefits to people. Continuing pressures from urbanization and fragmentation; selective species harvests; air pollution; exotic invasive species; wildlife habitat loss; historic fire regime changes; stream degradation; and climate change still affect and threaten these forests, and require enlightened management and policy decisions to ensure sustainability of healthy, diverse, and productive forests.

Keywords: forest health, forest economics, indicators, stressors, sustainability, Mid-Atlantic forests, Montreal Process Criteria and Indicators

Scan this code to submit your feedback,
or go to www.srs.fs.usda.gov/pubeval

You may request additional copies of this publication by e-mail at pubrequest@fs.fed.us.

Please, limit requests to two copies per individual.

The Forest Service, United States Department of Agriculture (USDA), is dedicated to the principle of multiple use management of the Nation's forest resources for sustained yields of wood, water, forage, wildlife, and recreation. Through forestry research, cooperation with the States and private forest owners, and management of the National Forests and National Grasslands, it strives—as directed by Congress—to provide increasingly greater service to a growing Nation.

The USDA prohibits discrimination in all its programs and activities on the basis of race, color, national origin, age, disability, and where applicable, sex, marital status, familial status, parental status, religion, sexual orientation, genetic information, political beliefs, reprisal, or because all or part of an individual's income is derived from any public assistance program. (Not all prohibited bases apply to all programs.) Persons with disabilities who require alternative means for communication of program information (Braille, large print, audiotape, etc.) should contact USDA's TARGET Center at (202) 720-2600 (voice and TDD).

To file a complaint of discrimination, write to USDA, Director, Office of Civil Rights, 1400 Independence Avenue, SW, Washington, D.C. 20250-9410, or call (800) 795-3272 (voice) or (202) 720-6382 (TDD). USDA is an equal opportunity provider and employer.