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Status of Plants in Virginia

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OVERVIEW OF BOTANICAL DIVERSITY

Virginia possesses a unique and varied assemblage of plant life. There are 3,164 species, subspecies and varieties of plants in Virginia (Weakley et al. 2012). As classified by the Virginia Departmentof Conservation and Recreation's Division of Natural Heritage (DCR-DNH), they form some 94 ecological groups and 317 community types across five distinct physiographic provinces: Coastal Plain, Piedmont, Blue Ridge, Ridge and Valley, and Appalachian Plateau. The state extends 469 miles from east to west and 201 miles north to south at the widest points, enclosing 42,326 square miles of territory. This diverse range of environmental conditions supports the wide diversity of plant life found within the state. Virginia is on the northern boundary of many southern plant species and on the southern boundary of many northern plant species. This range overlap combined with seashore to mountain variation leads to one of the richer diversities of plant life within the continental United States.

Virginia was the source of some of the earlier plant collections by European botanists (Berkeley and Berkeley 1963).Europeans started observing and documenting Virginia's flora as early as the 1500s (Hugo and Ware 2012). Over the next two centuries, there were various explorations and reports by laypersons and scientifically trained individuals. In the eighteenth century, there were significant contributions to the documentation and descriptions of plants in Virginia. In 1739 J. F. Gronovius published John Clayton's work titled *Flora Virginica* describing some 500 or so plant species (Hugo and Ware 2012). John Mitchell, James Greenway, and prominently, John Bartram wrote extensively about plants of Virginia. Later, such botanists as Andre Michaux, Asa Gray, and John Torrey published work that included plants of Virginia (Hugo and Ware 2012).

Work toward a new Flora of Virginia began in earnest in 1926 when the Virginia Academy of Science established a flora committee through the leadership of A.B. Massey of Virginia Polytechnic Institute (Hugo and Ware 2012). Through Massey's vision and the efforts of many subsequent scientists, a new Flora of Virginia was finally published in 2012 documenting 3,164 plant species, subspecies, and varieties in 189 families in the commonwealth of Virginia (Weakley et al. 2012).

The public charge to inventory and protect this wealth of plant biodiversity is given to the Office of Plant Protection within the Virginia Department of Agriculture and Consumer Services, which under the Virginia Endangered Plant and Insect Species Act has responsibility to list and protect Virginia's endangered and threatened plant species. There were 26 species listed in 2013, whereas there were 17 species listed under the federal Endangered Species Act of 1973 (Townsend 2014). The Virginia Endangered Plant and Insect Species Act also contains provisions for the recovery of endangered and threatened species in Virginia. The VDCR, DNH and the Virginia Department of

Agriculture and Consumer Services(VDACS) all work cooperatively with each other and with the US Fish and Wildlife Service to protect the natural biological diversity of Virginia.

The DNH has the charge to evaluate Natural Heritage Resources such as the habitats of rare, threatened, and endangered plant and animal species; exemplary natural communities, habitats, and ecosystems; and other natural features of the Commonwealth (Fleming and Patterson 2013). The DNH has defined 94 ecological groups and 317 community types within Virginia. Each community has been assigned a global and state conservation status rank based on the relative rarity or endangerment of the community. This is meant to provide a framework for setting conservation priorities while trying to balance economic development within the state. Of the 317 plant communities, 111 are considered to be critically imperiled (Fleming and Patterson 2013). There are 613 species included on the Rare Vascular Plant List in Virginia, and an additional 229 that are considered uncommon and placed on a Watchlist (Townsend 2014). There are also 46 species of nonvascular plants listed as Rare in Virginia (Table 1).

Endemics

Despite the overlap of northern and southern regions of plant growth in Virginia, there are some species found only in Virginia. There are five plant species endemic to Virginia: Virginia round-leaf birch (*Betula uber*), Addison's leatherflower (*Clematis addisonii*), Virginia white-haired leatherflower (*Clematis coactilis*), Millboro leatherflower (*Clematis viticaulis*), and Peter's Mountain-mallow (*Iliamna corei*).

Virginia is a land of transitions. From east to west, our lands transition from coastal plains to piedmont to mountains. Virginia is a transition zone between the northernmost range of southern species and the southernmost range of northern species. Also there is a high human impact factor within the state, disturbing many native habitats. This creates many unique habitats and may help explain why we have endemics with such limited ranges despite the great variety of plant life within the state.

Peter's Mountain mallow (*Iliamna corei*) is limited to one site on Peter's Mountain in Giles County. It is found only on shallow sandstone outcrops growing in full sunlight. It is a bushy-branched plant with erect stems and produces 15-20 pink flowers per plant. The fruit is a schizocarp (Weakley et al. 2012). Threats to this species include grazing, plant competition, shading, and fire suppression. This population is now protected by the Nature Conservancy and its partners, and is being carefully managed. The genus *Iliamna* contains only seven species and is restricted to North America.

Three of the endemics are in the genus *Clematis* in the Ranunculaceae. This genus has about 295 species distributed around the world. There are 11 species of Clematis in Virginia, including the three that are endemic to Virginia. Millboro leatherflower (*Clematis viticaulis*) is a small (2-4 dm) upright herbaceous plant with apetalous flowers with 2-4 cm styles and purplish sepals that form a bell-like structure. This species is limited to shale barrens and woodlands. Its range is restricted to Augusta, Bath, and Rockbridge counties(Weakley et al. 2012). Addison's leatherflower (*Clematis addisonii*) is limited to dolomitic outcrops in Botetourt, Montgomery, Roanoke and Rockbridge counties. The herbaceous plant can grow to 10 dm, at first erect, then

TABLE 1. Status and Rankings of Virginia Plants. [1]

Status	No. Species
Rare Vascular Plant Species	613
Uncommon Vascular Plant Species	229
Rare Nonvascular Plant Species	46
State-listed Endangered Plant Species	26
Federal-listed Endangered Plant Species	17
State Critically-Imperiled Vascular Plant Species	360
State Imperiled Vascular Plant Species	166
State Critically-Imperiled Nonvascular Plant Species	31
State Imperiled Nonvascular Plant Species	10
Highly Invasive Plant Species	32
Moderately Invasive Plant Species	32
Low Invasive Plant Species	16

[1] Data from Wilson and Tuberville (2003).

becoming procumbent. It has solitary, terminal apetalous flowers with reddish to bluish purple sepals forming a bell-shaped floral structure(Weakley et al. 2012).Virginia white-haired leatherflower (*Clematis coactilis*) occurs on shale, calcareous sandstone, dolomite, and limestone outcrops and barrens. It is restricted to mountainous counties of the Ridge and Valley region. This is a bushy herbaceous perennial growing to 2-4.5 dm with solitary, terminal apetalous flowers that have purplish sepals that appear white because they are densely plumose with white to pale-yellow hairs. The sepals form a bell-shaped floral structure (Weakley et al. 2012). All three *Clematis* species are perennials, which may aid in their survival and continuation of the populations. These species have elongated styles on numerous pistils and seeds are enclosed within achenes.

The Virginia round-leaf birch (*Betula uber*) was first collected in 1914 and described by Ashe in 1918 (Davis 2006). Subsequently, the tree was not seen again in the wild for some time and was presumed extinct (Mazzeo 1971, Smithsonian 1974).

In 1975, a small population of *B. uber* was discovered in Smythe County, Virginia (Ogle and Mazzeo 1976). Efforts were undertaken to propagate, distribute and protect individuals of this species. Trees were located at the Reynolds Homestead, the National Arboretum, and on other public and private lands (Davis 2006). In addition to traditional methods of propagation, Virginia round-leaf birch has been successfully propagated from dormant buds (Vijayakumaret al.1990) and by in vitro nodal culture (Jamison and Renfroe 1998).

The Virginia round-leaf birch is a small tree (7.6-14 m), with dark aromatic bark, and ovate or short elliptic leaves with rounded or obtuse apex, and a cordate base (Mazzeo 1971, Ogle and Mazzeo 1976). Leaf shape and fruit characteristics in *B. uber* are significantly different than *B. lenta*(Sharik and Ford 1984). Leaf shape difference is maintained in pure populations over decades (Sharik and Ford 1984). *B. uber* has a more compact crown than *B. lenta*. There are chemical distinctions between *B. uber* and *B. lenta*, such as the presence of rhododendrin (Santamour and Vettel 1978). Although wood anatomy is similar between *B. uber* and *B. lenta* (Hayden and Hayden 1984), this should not be surprising since they belong to a common clade (Li et al. 2005). Thomson et al. (2015) suggest that *B. lenta* and *B. uber* possibly have a shared ancestry.

Weakley et al. (2012) relegate Virginia round-leaf birch to varietal status as *Betula lenta* L. var. *uber* Ashe. McAllister and Ashburner (2004) question the species status of *B. uber* based upon variability of leaf traits in a small population of presumptive selfed seedlings. However, other authorities still recognize this as a distinct species of birch including the *Flora of North America* (Furlow 1997). Although several investigations have employed molecular data approaches to resolve the phylogenetic relationships of the birches (Jarvinen et al. 2004, Li et al. 2005, Schenk et al. 2008), only one included *B. uber*, which separated it from *B. lenta* based on sequences of the internal transcribed spacer region of nuclear ribosomal DNA (Li et al. 2005). Mazzeo (1971) recognized *B. uber* as a valid species. Ogle and Mazzeo (1976) noted significant differences among *B. uber*, *B. lenta*, and *B. alleghaniensis* in the field. An examination of trees in the area revealed no apparent hybrids, and as a population, *B. uber* showed a strong uniformity.

Davis (2006) reviewed previous studies of *B. uber* and *B. lenta* and indicated that hybridization studies used to delineate traits were based upon plants that had been growing in close proximity. It is well established and widely recognized that birches readily hybridize (Woodworth 1929, Johnsson 1945, Elkington 1968, Guerriero et al. 1970,Sharik and Barnes 1971, Barnes et al. 1974, Eriksson and Jonsson 1986,Wilsey et al. 1998, Palme et al. 2004). Therefore, it calls to question whether studies of presumptive *B. uber* individuals are truly *B. uber* or whether they may have been introgressed with *B. lenta*. Ogle (2003) recommends that direct DNA testing be performed on the known populations of *B. uber* and *B. lenta* to help resolve the status of this species.

Virginia round-leaf birch is protected by the Endangered Species Act. Following its rediscovery, it was classified as endangered. Recovery efforts toward this species have resulted in a sufficient number of breeding populations such that the status of this species subsequently has been changed from endangered to threatened (USFWS 1994).

HISTORICAL PERSPECTIVE

Around 5000 BC, the general flora of modern eastern U.S. became established. Over the next couple of thousand years, the eastern US experienced a general warming trend. Oak species became prevalent in southwestern Virginia between 3000-2500 BC. In addition, chestnut and hickory trees became an important part of the mixture of trees during this period of warming and dry climate. Oaks, chestnuts, and hickory trees helped support the indigenous people, who began to migrate seasonally into this area around 3000 BC. By 1000 BC, Amerindians began to settle in the eastern US and began a culture of autonomous populations that lasted until European contact (Sarvis 2011).

Around 500 BC to 900 AD, maize was introduced into Virginia by native Americans and populations started to become more settled and less nomadic. During this time, there is evidence of tree girdling and slash-and-burn techniques being introduced. Distinct natural zones developed in Virginia based on geographical variations between the Coastal Plains, Piedmont, Ridge and Valley region, and Appalachian plateau. Indigenous populations developed distinctive cultures reflecting the unique natural resources by which they were surrounded. Improved strains of corn became prevalent and the cultivation of beans began. Southwestern Virginia was a region in which numerous native populations overlapped for hunting purposes, but was not heavily populated. Sioux, Shawnees, Delawares, Catawbas and Tuscaroras all spent time in Virginia hunting, harvesting, and living in transient camps. Native American populations had a fairly minimal impact on the flora of southwestern Virginia (Sarvis 2011).The Cherokee and Shawnee tribes hunted throughout southwestern Virginia. The Cherokee became well-acquainted with the plants not only as a source of food, but also for medicinal uses (Hamel and Chiltosky 1975).

Contact with Europeans during the 1600s led to changes in Amerindian populations. European demand for pelts and hides led to overhunting of deer and other animal populations, and the introduction of European diseases such as smallpox decimated Amerindian populations in the eastern US. By the mid-to-late 1700s, Europeans settlers had made their way into the western areas of Virginia (Sarvis 2011). As Europeans occupied the valley areas of Virginia, they cleared the land, introduced domesticated livestock and began the cultivation of corn, wheat, rye, and oats. Europeans brought potatoes, peach and apple trees, and many other species. They introduced new forage grasses to support their introduced livestock. Kentucky bluegrass, timothy, redtop, white clover and other species were introduced, along with pastoral species such as daisies, yarrow, dandelion, buttercup, garlic mustard, and other species. As native flowering plants became scarcer and forests were cleared, native bee species decreased and Europeans introduced the European honeybee. Tobacco (*Nicotianatabacum*) was introduced as an agricultural commodity and had a major impact on land-clearing and farming in Virginia.

Bottomlands were cleared for growing corn, wheat, rye, and oats. Hillsides were cleared for grazing. Wildlife populations were drastically altered, which had an impact on plant growth and forest regeneration. Forests continued to provide non-timber products such as ginseng, galax, elderberry flowers, polkberries, buck vine, lobelia, moss and cherry bark (Sarvis 2011).

During the 18th and 19th centuries, much of the forest, especially in the bottomlands was cleared. Charcoal production for furnaces and forges led to major tree cutting, as one charcoal iron furnace could consume wood from an acre of land per day. Wythe and Carroll Counties had extensive forest clearing in support of charcoal iron production. Salt production in Saltville consumed about six cubic feet of hardwood per bushel of salt produced by boiling off brine, with peak production of around 4 million bushels per year. In 1880, Virginia salt production consumed about 550,000 cords of wood. Development of railroads and industrial logging led to much more extensive deforestation into the mountainous areas of the state (Sarvis 2011).

Land clearing and overcultivation led to erosion. Growth of cities and industry after the Civil War led to heavy demand for coal, timber and tannin. Development of the railroad industry in the mid-1800s increased timbering and mining in the mountain regions of the state. Stream siltation and flooding increased, causing loss of life and property. Fire also destroyed much of the cutover forest land. The Massanutten range was largely denuded of trees between 1850 and 1880, then experienced many fires that burned over the remaining trees and killed off regeneration (Satterthwaite 1993).

From around 1890 to 1920, industrial timbering and railroad construction led to massive deforestation along the Appalachians. Only the drastic drop in timber prices associated with the Great Depression slowed the deforestation. The U.S. Department of Agriculture established the Division of Forestry in 1881, which became the U.S. Forest Service in 1905 under President Theodore Roosevelt, with Gifford Pinchot becoming the first Chief. The Forest Reserve Act of 1891, along with the Weeks Act of 1911 laid the foundation for the federal government to acquire land and hold it in the public trust to protect watersheds and maintain navigable waters by conserving forest land. The Weeks Act created a National Forest Reservation Commission. During the first several decades of the twentieth century, major land purchases were made from private individuals, corporations, and state governments. Such purchases in Virginia led to the formation of the Jefferson National Forest and the George Washington National Forest (Sarvis 2011).

Three northern Virginia purchase units (Potomac, Massanutten Mountain, and Natural Bridge Purchase Units) were combined in 1917 to form the Shenandoah National Forest. In 1932, the forest was renamed George Washington National Forest to avoid confusion with Shenandoah National Park, also located in Virginia (Satterthwaite 1993).

In the Depression era, the Civilian Conservation Corps (CCC) engaged in replanting forests and began an important program of fire protection. Indigenous Americans used fire in the forest, but not in the way that wild fires decimated the cutover lands following the industrial period of deforestation and land abandonment. Forest protection from fire became an important strategy of the Forest Service, and the CCC provided road and trail construction along with building fire lookout towers, and even engaged in fire suppression (Sarvis 2011).

The national consciousness regarding fire in the forest had been irrevocably altered by events such as the Big Burn that occurred in 1910. Following an extensive drought, fires started in Idaho and spread into Montana and Washington, burning an area the size of the state of Connecticut in 48 hours. Fire swept across the northern Rockies and made its own weather system, racing along until more than three million acres burned and one billion dollars worth of wood was consumed (Egan 2009). Rain and snow of late August finally extinguished the fire. Soot darkened sunsets in Boston, and covered snow in Greenland.

The massive destruction, loss of human life, loss of towns and property, and loss of natural resources of the forest had a profound effect on the perspective of the Forest Service in shaping their view of forest fire suppression. Gifford Pinchot, first Chief Forester of the U.S. Forest Service, regarded loss of forest to fire as a waste of natural resources and understood forest fires to be "wholly within the control of men" (Pinchot 1967). For the newly formed U.S. Forest Service, fire prevention became a top priority that would be maintained for decades.

Workers in the CCC were used to replant forests. White pine was one of the species extensively planted in Virginia. It was during the 1930s that the Chestnut blight was decimating populations in Virginia, and that white pine blister rust started spreading through the Appalachians. Control measures were taken including eradicating the rust's alternate hosts, currant and gooseberry plants, within 900 feet of white pines. As severe as the blister rust epidemic was, it paled in comparison to the devastation caused by the Chestnut blight. In some areas, chestnuts constituted 60-90 % of the standing trees. The forest composition was radically altered by these diseases (Sarvis 2011).

As agriculture and forestry advanced over the decades, they had a major financial impact on the economy of Virginia. In 2006, agriculture-related industries generated over $55 billion and produced 357,100 jobs, while forestry generated over $23 billion and produced 144,400 jobs (Rephan 2008). In 2011, agriculture-related industries generated over $52 billion and produced 310,900 jobs. Forestry generated some $17 billion and produced some 103,800 jobs (Rephan 2013). About 62% of Virginia's forest land is in private hands, held by over 373,600 forest landowners (VDOF 2014a). Corporate forest holdings account for 19% of Virginia's forests, with the forest products industry holding only about 1% (186,700 acres).

Virginia has lost over 500,000 acres of forest land since 1977 (VDOF 2014a). Most of the forests in Virginia are composed of upland hardwood species (61%) and oak-pine mixtures (11%). Pine plantations form 13% of Virginia's forest lands, with 7% of the lands covered in natural pine stands. One of our more valuable pine species, longleaf pine (*Pinus palustris* P. Miller) was decimated by human harvesting following European colonization. Between 1500 and 1850, the longleaf pine population lost over 1 million acres. Today, there are fewer than 200 native longleaf pine trees left in Virginia, but the Department of Forestry has initiated a program to search for seed sources similar to our native populations and start replanting this species back into its native range (VDOF 2014a, VDOF 2014b).

Great Dismal Swamp

One of the unique features of the Coastal Plain is the Great Dismal Swamp located in southeastern Virginia and northeastern North Carolina. The swamp covers about 104,000 ha, and bears the scars of heavy human disturbance (Levy 1991, Whitehead 1972). The Great Dismal Swamp covers some 750 to 1000 square miles of land, about 40% of which lies within Virginia (Davis 1962). Pollen analysis reveals that this land mass has developed through various developmental changes in composition, first supporting a pine-spruce forest, later replaced by a beech-hemlock-birch forest, replaced by an oak-hickory forest, and finally developing into the cypress-gum assemblage some 3,500 years age. The cypress-gum community consisted largely of cypress (*Taxodium distichum*), black gum (*Nyssa sylvatica*), tupelo gum (*Nyssa aquatica*), Atlantic white cedar (*Chamaecyparis thyoides*), red maple (*Acer rubrum*), Carolina ash (*Fraxinuscaroliniana*), loblolly pine (*Pinus taeda*), pond pine (*Pinus serontina*), willow oak (*Quercus phellos*), sweet gum (*Liquidambar styraciflua*), tuliptree (*Liriodendron tulipifera*), holly (*Ilex opaca*) and other species (Whitehead 1972).

Indigenous Americans occupied this area from as early as 12,000 years ago, and were present as the land transitioned into the marsh and swamp. The area was most heavily occupied from about 9,000 to 3,500 years ago, with humans living in and around the area for hunting, fishing, and foraging. Palynological research indicates that maize was present in the swamp about 3.000 years ago, suggesting that native Americans were already cultivating corn in this area (Bradley 2013). European colonization of Virginia and North Carolina would drastically alter the nature of the Great Dismal Swamp. The development of settlements in the Norfolk area in the 1620s and around Suffolk in the 1630s-1640s brought European settlers in close contact with the swamp.

George Washington and a group of other land speculators formed the Dismal Swamp Land Company and in the early 1760s got permission from the Virginia General Assembly to drain and farm 40,000 acres located in the Virginia portion of the swamp. This turned out not to be an easy task. Despite some limited production of rice and corn, Washington and others lost interest in the venture and shortly after the War of 1812, they turned their interests elsewhere (Bradley 2013). The soil beneath the swamp is not suitable for cultivation and is probably what has spared the total clearing of the swamp during historical times (Davis 1962).

The Great Dismal Swamp provided wood for Colonial America. Pine, maple, juniper and cypress provided wood for fencing, buckets, barrels, framing, siding, and shingles. Wood was also used in shipbuilding and charcoal production. Naval stores, pitch, turpentine, and tar were produced from pines from the Great Dismal Swamp (Davis 1962). Despite the lack of agricultural success, the Great Dismal Swamp was heavily logged with most of the cypress being removed for the production of shingles. Canals were cut through the swamp to facilitate transport of logs and in the 1830s to drain lands to enable companies to bring in railroads, establishing more logging camps within the swamp (Bradley 2013). Logging continued at a more advanced rate into the 20th century, with most of the land being privately owned. In the 1900s, the emphasis shifted to logging Atlantic white cedar. Logging of cedar in the Great Dismal Swamp was especially heavy during the first and second World Wars. During World War I, over 20 million board feet (b.f.) of cedar was removed from the swamp per year for several years, and production peaked at 5 million b.f. during World War II (Ward 1989).

In 1973, the Great Dismal Swamp was designated a wildlife refuge, becoming federally protected and managed. Efforts are underway to restore Atlantic white cedar through reforestation. Current research shows that rooted cuttings of Atlantic white cedar grow best at intermediate elevations. On high mounds, Atlantic white cedar may have difficulty with competition from plants such as sweet pepperbush (*Clethra alnifolia*), and may have increased mortality in low sites associated with deep pools (Brown and Atkinson 1999). Studies following cedar regeneration after the forest destruction caused by Hurricane Isabel in 2003 demonstrated that natural disturbances can lead to compositional changes in the forest and depression of cedar regeneration. Without salvage logging, an increase in red maple growth occurred, whereas on salvage logged plots, cedar seedling regeneration constituted the majority of seedlings present (Belcher et al. 2006).

Not all of the Great Dismal Swamp property is contained within the Refuge. Some success has been achieved in protecting more of the land. In 2007, Ecosystem Investment Partners (EIP), a private equity firm, acquired 1,030 acres within the acquisition boundary of the Great Dismal Swamp National Wildlife Refuge in southeastern Virginia. The property had been used as farmland previously. EIP is selling endangered species mitigation credits, and once all the credits are sold, EIP plans to transfer the property to either the Wildlife Refuge or to another private landowner who would be bound to conservations easements (EIP 2010).

PHYSIOGRAPHIC DISTRIBUTION OF PLANTS

Virginia's plant communities are a reflection of the physiographic properties of the state. These communities are described in detail by the Natural Heritage Program (Wilson and Tuberville 2003). The Cumberland Mountains in the southwestern portion of Virginia are characterized by a mixed mesophytic forest with various oak and hickory species, along with beech, sugar maple, eastern hemlock, yellow poplar, birches and other tree species. There are 82 rare species within this province (Table 2). The Ridge and Valley and Allegheny Mountain Provinces contain many oak species (chestnut, scarlet, white, black, and northern red), along with various hickories. At higher elevations, birches and sugar maple are present, and red spruce is found at the highest elevations. Beech and cherry are also mixed in the higher Allegheny Mountains. There are also small communities of red spruce-hemlock swamps and bogs. There are 503 rare species within this area. The Northern Blue Ridge Physiographic Province has a mixed oak and oak-hickory forest cover that includes yellow poplar and supports 130 rare species. The Southern Blue Ridge Physiographic Province has many communities including mixed oaks, oak-hickory, northern hardwood forests, relict stands of red spruce and fraser fir, and rare wetlands such as The Glades near Galax. Within this province, there are 136 rare species.

TABLE. 2. Distribution of rare plant species across Virginia physiographic provinces.[1]

Province	No. Rare plant species
Cumberland Mountains	82
Ridge and Valley/Allegheny Mountains	503
Northern Blue Ridge	130
Southern Blue Ridge	136
Northern Piedmont	108
Southern Piedmont	147
Northern Coastal Plain	125
Southern Coastal Plain	174
Outer Coastal Plain	190

[1] Data from Townsend (2014).

The Northern Piedmont Physiographic Province contains mixed oak forests and mixed hardwood forests with oaks, beech, yellow poplar, hickories, and ash. There are 108 rare species located in this province. The large Southern Piedmont Physiographic Province contains mixed oak and mixed hardwood forests. Additional species that appear in these forests include yellow poplar, sweetgum, Virginia pine, and loblolly pine. There are 147 rare species found within this province (Wilson and Tuberville 2003).

The Northern Coastal Plain Physiographic Province reflects a history of land clearing for agriculture, and repeated forest harvests. The forests here consist of secondary mixed oak and mixed hardwood forests including oak, beech and yellow poplars. Pines, especially planted loblolly, are prevalent. Mountain-laurel establishes dense undergrowths in areas. There are 125 rare species present. The Southern Coastal Plain Physiographic Province historically contained longleaf pine and pond pine, both fire-dependent species. There were also beech, oaks and hickories in ravines, and baldcypress and tupelos in swampy bottomlands. Loblolly is the most common pine today due to replanting practices. There are 174 rare species located within this province. The Outer Coastal Plain Physiographic Province covers the eastern shore and the peninsula off the coast of Virginia. Maritime upland forests are present and include loblolly pine and live oak. Special features of this province include Atlantic white cedar swamps, coastal dunes, pocosins, and other rare communities. Within this province there are 190 rare plant species (Wilson and Tuberville 2003).

PLANTS IN AGRICULTURE

Virginia has a long history of agriculture. Farmland covers about 32% of Virginia, amounting to some 7.9 million acres (VDACS 2015a). Agriculture contributes about $52 billion to the economy each year and provides over 300,000 jobs (Rephann 2013). The allocation of farmland to various crops fluctuates with commodity prices and subsidies. Between 2006 and 2011 vegetable production declined, but soybean, corn and wheat production increased (Rephann 2013). The top fourteen commodities (Table 3) generated cash receipts of about $1.4 billion(VDACS 2013). Soybeans, corn and wheat are the commodities covering the most acreage of farmland (Table 4).

Over 1.4 million acres were dedicated to forage and silage in 2012. Over 3 million acres were used for pasture, with another 434,000 acres of pastured woodland. Over 2 million acres of farmland were wooded (NASS 2014b). In 1997, Virginia had 28,806 acres in orchards. Orchard acreage fell to 26,354 acres in 2002 and to 19,114 in 2012 (NASS 2014a). Over 300 acres of apple orchards went out of production between 2013 and 2014.

Virginia is becoming well-known as a wine producing state. In 2013, Virginia produced 4,942 tons of grapes from Vinifera grapes, 412 tons of grapes from American grapes, and 1,507 tons from hybrid grapes (Virginia Wine Marketing Office 2014). The top producing counties are Loudoun with 1,046 tons, Orange with 1,042 tons, and Albemarle at 1,013 tons. There were 3,089 acres in vineyards in 2013.

Well-managed agricultural systems provide soil retention, food production, carbon sequestration, and aesthetics. Agricultural ecosystems rely on other ecosystems for pollination services. Agricultural mismanagement can have adverse effects on surrounding ecosystems through soil erosion and deposition, stream siltation, pesticide runoff, fertilizer runoff, fecal contamination, and production of volatile organic compounds (Dale and Polasky 2007).

THREATS TO PLANT BIODIVERSITY

Plant biodiversity in Virginia faces a number of threats (Table 5). Non-native or exotic plants can invade, outcompete, and/or inhibit native plant populations. Diseases and insects have had and continue to have major impacts on entire ecosystems. The presence of browsers along with the loss of pollinators and animal dispersers has an impact on plant populations. Finally, forest mismanagement and land development greatly affect plant biodiversity.

Exotic Plants

Non-native plants have been a part of the Virginia landscape since European populations arrived on America's shores. Many non-native plants have escaped cultivation and become naturalized. Non-native species were brought to America as crops, culinary herbs, medicinal plants, and ornamentals. Unfortunately, some introduced species have become competitors with native species, displacing native populations, and altering ecosystems. Invasive species displace native species not just

TABLE 3. Economic value of leading commodities in Virginia during 2011 and 2013.[1]

Commodity	Cash Value (millions)	
	2011	2013
soy beans	302	284
greenhouse/nursery	272	263
grain corn	212	171
hay	123	124
winter wheat	109	120
tobacco	109	113
cotton	69	65
tomatoes	62	37
apples	54	33
peanuts	24	31
potatoes	15	15
cottonseed	12	12
barley	12	9
grapes	11	10
Total	1,386	1,287

[1] Data from VDACS (2013, 2015a).

by offering competition, but also by inhibiting growth of mycorrhizal fungi that are important to the growth of native species (Callaway et al. 2008).

Over 2500 non-native species have become naturalized in the U.S. (Mack 2003). As early as the establishment of the Plymouth Colony in 1620, European species were introduced into the eastern seaboard of America. By 1671, accounts indicated that many of our common exotic weed species had escaped and were well established outside of cultivation. Over the next 350 years, ornamental species became the largest

TABLE 4. Crop acreage and yields of selected crops in Virginia in 2012[1]

Crop	Acres	Yield	Units
soybeans	578,852	22,680,879	bushels
grain corn	338,132	33,984,647	bushels
grain wheat	241,979	14,804,947	bushels
winter grain wheat	240,208	14,701,510	bushels
barley	37,023	2,905,047	bushels
grain rye	4,291	157,851	bushels
grain sorghum	4,043	258,000	bushels
oats	3,456	238,928	bushels
spring grain wheat	1,771	103,437	bushels
Cotton	89,072	191,513	bales
tobacco	22,982	53,179,801	pounds
peanuts	20,208	81,182,563	pounds
potatoes	5,423	1,350,000	cwt

[1] Data from USDA NASS 2012 Census of Agriculture.

category of imported species contributing to the host of exotic species in the United States. Asa Gray, visiting Winchester at the northern end of the Shenandoah Valley in June 1841 noted fields overrun with viper's bugloss (*Echium vulgare*). Japanese honeysuckle (*Lonicera japonica*) was documented in the wild by 1860 (Mack 2003).

Japanese honeysuckle is a climbing vine that will shade out the canopy of trees and kill them from above ground competition or by girdling. It also makes tree crowns more susceptible to snow and ice damage by increasing the weight load on the crowns during snow and ice events. In addition to above ground competition, below ground competition with trees by honeysuckle has a greater effect on tree growth than native vine species such as Virginia creeper (*Parthenocissus quinquefolia*) (Dillenburg et al. 1993). Japanese honeysuckle reduces pine seedling growth by interference and light competition, and litter from Japanese honeysuckle, whether on top of the soil, or incorporated, also reduces growth of pine seedlings, indicating a possible allelopathic

TABLE 5. Major threats to plant biodiversity in Virginia.

Exotic Plants	Pathogens	Insects	Other
Japanese honeysuckle	Chestnut blight	Gypsy moth	White-tailed deer
Amur honeysuckle	Dogwood anthracnose	Hemlock woolly adelgid	Loss of pollinators
Garlic mustard	Dutch elm disease	Emerald ash borer	Loss of fruit dispersers
Japanese stiltgrass	Thousand cankers disease	Kudzu bug	Loss of seed dispersers
Multiflora rose			Forest mismanagement
Japanese barberry			Fire exclusion
Phragmites			Land development
Russian olive			Climate change
Autumn olive			
Tree of heaven			

component (Skulman et al. 2004). Removal of Japanese honeysuckle vines from sweetgum (*Liquidambar styraciflua*) boles and branches results in increased stem diameter (Whigham 1984). However, removal of vines from trees and ground results in an even larger effect, indicating that competition for soil factors and/or allelopathy plays a role in the deleterious effects of vine growth on mature trees.

Japanese honeysuckle is dispersed by birds (Naumann and Young 2007), and is an invasive species from the mountains of Virginia to the coastal forests and the maritime forests of the barrier islands (DCR 2009, Naumann and Young 2007). *Lonicera japonica* has lower herbivory rates in the southeastern United States than the native species *Lonicera sempervirens*, giving Japanese honeysuckle a distinct competitive advantage compared to native species (Schierenbeck et al. 1994).

The exotic, invasive shrub bush honeysuckle or Amur honeysuckle (*Lonicera maackii*) reduces growth and fecundity in three native plants, *Allium burdickii*, *Thalictrum thalictroides*, and *Viola pubescens* (Miller and Gorchov 2004). Extracts of leaves and roots of Amur honeysuckle inhibit germination of seeds of *Impatiens capensis*, *Alliaria petiolata*, and *Arabidopsis thaliana* without any evidence of autotoxicity (Dorning and Cipollini 2006). Both species richness and abundance are reduced under crowns of *L. maackii* (Collier et al. 2002). In addition to deleterious effects on the herbaceous layer, bush honeysuckle increases mortality of native tree seedlings including *Acer saccharum*, *Fraxinus americana*, *Quercus rubra*, and *Prunus serotina* (Gorchov and Trisel 2003). Bush honeysuckle does not have a strong seed dormancy and, following dispersal, can have seedling establishment under various light conditions throughout fragmented forests (Luken and Goessling 1995). *L. maackii* can be controlled by stem injection with herbicide on larger stems and by cut and painting with herbicide on smaller stems (Hartman and McCarthy 2004).

Garlic mustard (*Allaria petiolata*) was brought to America by Europeans as a culinary herb. It has since become a widespread and aggressive invasive species

(Eschtruth and Battles 2009). Garlic mustard inhibits mycorrhizal fungi and lowers the viability and infectivity of arbuscular mycorrhizae spores. Garlic mustard also alters the bacterial communities in American soils, but not in European soils where it is native. Furthermore, garlic mustard, by affecting mycorrhizal fungi, also decreased emergence, growth, and survival of mycorrhizal-dependent plants (Callaway et al. 2008). Canopy disturbance is not an important factor in garlic mustard invasions (Eschtruth and Battles 2009). Such effects can explain why invasive species become so successful in exotic locations. Invasive species can interact with communities in complex fashions, which can lead to more profound interactions than might initially seem likely.

Japanese stiltgrass (*Microstegium vimineum*) is another widespread aggressive invasive species. Japanese stiltgrass is able to invade relatively undisturbed, late-successional forests. Canopy disturbance is not necessary for this annual grass to invade due to an extremely plastic response to shade (Eschtruth and Battles 2009). Japanese stiltgrass is more competitive in high light conditions such as along roadsides, whereas populations growing in low light conditions such as in forest interiors produce less biomass, fewer flowers and set fewer seeds (Huebner 2010). Two-year-old seedlings of northern red oak (*Quercus rubra*), sugar maple (*Acer saccharum*), and yellow poplar (*Liriodendron tulipifera*) are able to escape competition from Japanese stiltgrass, although sugar maple and yellow poplar survival is reduced. This would suggest that planting woody seedlings over top of Japanese stiltgrass may be useful in planning forest regeneration (Beasley and McCarthy 2011).

In the more mesic environment of the Allegheny Plateau and the more xeric environment of the Ridge and Valley Province, Japanese stiltgrass has been shown to grow slowly in forest interiors, and can become established, even without disturbance (Huebner 2010). This demonstrates the necessity of maintaining a healthy tree seedling population to help reduce *Microstegium* establishment inside forest stands. *Microstegium* grows along roadsides in a competitive manner (Huebner 2010).

Multiflora rose (*Rosa multiflora*Thunb.) was introduced into the U.S. by 1886 and was planted as a hedgerow in the 1930s. Multiflora rose has become invasive and is associated with disturbed sites. It can tolerate soils with low fertility, but grows best in fertile soils, invading pastures and forests (Huebner et al 2014). Multiflora rose propagates locally by vegetative propagation and is widely dispersed by bird species (Naumann and Young 2007).

Japanese barberry (*Berberis thunbergii*) is an exotic invasive plant species that is widespread and aggressive. Barberry is more dependent upon canopy disturbance than some other invasive species. Species richness does not appear to strongly affect invasion by exotic plants (Eschtruth and Battles 2009) such as barberry and others.

Phragmites australis is an invasive species that spreads vigorously in wetlands. Created wetlands in eastern Virginia are often heavily infested with *Phragmites*. These created wetlands often have shallow sediment thickness that favors the spread of *Phragmites*. This suggests that disturbance of existing wetlands and the creation of new wetlands as mitigation efforts may favor invasive species over native species (Pyke and Havens 1999).

Natural wetlands are being replaced by constructed wetlands in Virginia as a mitigation effort by industry, developers, and agricultural ventures. An extensive survey of constructed wetlands in Virginia has revealed that 80% of these wetlands are colonized by the invasive species *Phragmites australis*. Aggressive species of *Typha* were also present. *Phragmites* displaces native plant species and could overrun the constructed wetlands before the mid-21st century (Havens et al. 1997).

In a study of all constructed wetlands over an acre in size in the coastal plain of Virginia, 73% were found to be colonized by *Phragmites australis* (Havens et al. 1997). *Phragmites* appears to be limited by extreme nutrient deficiency or high salinity. *Phragmites* spreads by rhizomes to develop a pattern of circular patches. Growth rates indicate that *Phragmites* could dominate constructed wetlands within 40 years. Thus wetlands constructed for mitigation would not be representative of wetlands that had been destroyed (Havens et al. 1997).

A number of exotic tree species have become established outside of cultivation in Virginia. *Paulownia tomentosa* (Thumb.) Steud. is an Asian tree species that has become naturalized in the US, but is not invasive. It produces small populations that arise primarily from large-scale disturbances (Williams 1993), but is not considered particularly invasive. *Pyrus calleryana* is an escaped tree species that has spread from cultivation. This species hybridizes with other *Pyrus* species and has been spreading since the 1950s inside the United States. *Pyrus calleryana* is considered invasive. There are few, if any, natural controls for this species (Vincent 2005).Russian olive (*Elaeagnus angustifolia* L.) is spreading and forming extensive populations that are displacing native species, and is especially a problem in riparian zones and wetlands (Stohlgren 2003). Autumn olive (*Elaeagnus umbellata*Thunb.) tends to form dense monotypic stands. It can grow in shade as an understory tree or can colonize fields and disturbed sites. It produces prolific fruit and is readily dispersed by birds. It grows rapidly, survives in poor soil and resprouts following cutting or burning.

Ailanthus altissima, commonly known as tree-of-heaven, is an invasive woody species widely distributed in the state from the mountains (Kowarik 1995) to the shore (Naumann and Young 2007). This tree has been shown to differentially suppress native herbaceous species while allowing growth of non-native herbaceous species (Small et al. 2010). This may well encourage the spread of other non-native species along with *Alilanthus*. *Ailanthus* produces allelopathic compounds such as ailanthone (Heisey 1996), which exhibits strong phytotoxic properties. Phytotoxins are produced in *Ailanthus* leaves and stems, and accumulate in soil surrounding the trees (Lawrence et al. 1991). The combination of allelopathy and competition with native plants makes *Ailanthus* a problematic species. In addition, *Ailanthus* reproduces sexually and asexually, and rapidly becomes invasive to native plant populations.

Ailanthus establishes large clonal populations by root suckering (Kowarik 1995), and is also wind-dispersed by producing very large numbers of samaras (Naumann and Young 2007). Seeds are dispersed greater distances into open fields than into closed forest canopies (Landenberger et al. 2005). *Ailanthus* samaras are also well adapted for water dispersal and exhibit high percentages of samaras capable of floating for weeks. Germination of seeds can be as high as 87% after three days in water (Kowarik and

Saumel 2008). Stem fragments from first and second year shoots can also produce shoots, set roots and establish new plants after being carried by water for up to 10 days (Kowarik and Saumel 2008). Genetic investigations reveal that *Ailanthus* has a high level of sexual reproduction, and long distance dispersal, especially along road and railway corridors (Aldrich et al. 2010). Dispersal by wind, water, and clonal propagation makes *Ailanthus* a highly invasive species.

Exotic tree species are interfering with efforts to reestablish forests in Virginia. During the first decade of the 21st century, about 1,000 miles of riparian forests were established by the Virginia Department of Forestry and the USDA Natural Resources Conservation Service to protect water quality and enhance wildlife habitat through the CREP program (Bradburn et al. 2010). Tree survival was high in the Coastal Plain (97.6%), high in the Piedmont (90.5%), but lower in the Ridge and Valley region (68.4%). The Piedmont riparian zones contained 29 planted species, but also had 40 naturally regenerated species. Many of the naturally regenerated species were trees with lightweight seeds such as red maple, yellow poplar, boxelder, and green ash.

The Ridge and Valley plantings included 31 species, and 27 additional species regenerated naturally. Unfortunately, 43.5% of the natural regeneration consisted of tree-of-heaven and autumn olive, both invasive exotic species. Natural regeneration exceeded planted trees on each site, and herbivory control increased survival at each site. It is recommended that planting densities be increased in the Ridge and Valley region, along with aggressive control of invasive species (Bradburn et al. 2010).

All fragmented forests in the Shenandoah Valley that were sampled had exotic species present, and in some sites exotics appeared to be inhibiting regeneration of more valuable native species (Siderhurst et al. 2012). Increased forest fragmentation leads to increased edge effects including deeper penetration into the forest by exotic species (Fraver 1994).

Invasive species can profoundly affect plant-pollinator networks. Invasive plants often act as pollination super generalists, potentially drawing pollinators away from plant species with which the pollinator may have mutualistic interactions (Bartomeus et al. 2008). Invasive plants are generally more resistant to herbivory compared to native plants. Native birds, reptiles, and small mammals do not use invasive plants for cover or nesting sites as frequently as native plants. Invasive plants also alter ecosystem processes (Bell et al. 2003) and the effect of invasive species on ecosystem services is vastly underestimated (Funk et al. 2014).

Environmental conditions that promote native species richness also promote exotic species richness. Exotic species that have niche requirements different from native flora can colonize sites with little resistance from native species, and perhaps cause few effects on native species (Gilbert and Lechowicz 2005). Exotic species can become abundant and persistent because of a lack of natural enemies, development of new associations within the ecosystem, artificial and/or disturbed habitats that provide favorable ecosystems, and adaptability and success of alien species. Exotic species are costing billions of dollars per year in crop and forest losses and in attempts to control the pests (Pimentel et al. 2005).

Pathogens

Chestnut Blight

Chestnut blight is caused by the fungus *Cryphonectria parasitica* (formerly *Endothia parasitica*). The blight was most likely introduced into the United States by an infected Japanese chestnut (*Castanea crenata*) that had been imported into New York. In 1904, the chief forester for the New York Zoological Park noticed an American chestnut (*Castanea dentata*) was dying, soon followed by several more. Various states attempted preventive and treatment measures, but all proved unsuccessful (Hepting 1974).Studies of chestnut mortality in the Southern Appalachians indicate that as the blight proceeded through our region during 1928-1938 the death of the chestnuts, comprising some 20% of the forest stands, resulted in a 27% release of understory saplings (Lorimer 1980). This release had a major impact on the ecological succession and change in forest composition in the Appalachian forests. By 1940, mature chestnuts had been killed off throughout their range (Freinkel 2007).

The American chestnut was an extremely important component of the eastern forest. The chestnut tree supported populations of bears, elks, deer, squirrels, raccoons, mice, wild turkeys, passenger pigeons, and the indigenous human populations such as the Cherokee and Iroquois. The tree also provided nectar for honey production. Chestnuts comprised some 25-30% of the trees in the forest. Commercially, chestnut was used for telephone and telegraph poles, mine supports, framing lumber and shingles for housing, furniture of all sorts from cradles to coffins, fiber for pulp and paper production, and tannins for leather production (Freinkel 2007).

In 1902, the U.S. Geological Survey warned President Theodore Roosevelt that industrial loggers were inflicting serious changes on the Appalachian forests. During 1909, four billion board feet of hardwood lumber was cut from Maryland to Georgia from the mountain forests. Over 600 million board feet of chestnut was cut each year, not including poles, posts, and cordage wood. The U.S. Agricultural Census of 1910 recorded that Patrick County and four surrounding counties produced 360,000 pounds of chestnuts, about half of Virginia's crop that year (Freinkel 2007).

Soon after those bountiful harvests of the early 1900s, the blight arrived in Virginia. A survey in 1914 revealed infections in 18 of 95 counties in Virginia. By 1915, the USDA Bureau of Forest Pathology concluded "the chestnut stand of the southern Appalachians was doomed." By 1925, chestnut blight was documented in North Carolina and was spreading westward. Within 25 years, the fungus had covered the southern Appalachians, affecting 33 million acres. By mid-century, the chestnut was largely exterminated (Freinkel 2007) as timber and mast trees.

There are still abundant chestnut sprouts throughout the southern Appalachian region (Paillet 2002). Chestnut is still surviving as sprouts off of established root systems from trees that originated before the blight. However, these trees usually do not reach sufficient size to bear fruit before they are stricken with the blight. The niche and habitat distribution of the American chestnut has also been altered from the original distribution, shifting to drier sites on southern and western slopes (Burke 2012).

Arthur Graves, a Yale botanist, began experiments to hybridize American chestnut with Asian species to find a blight-resistant hybrid. The USDA took over hybridization efforts for about three decades, then largely gave up their efforts. Philip Rutter and Charles Burnham wanted to backcross chestnut hybrids to produce a chestnut that would be 15/16 American chestnut, yet carry the resistance of a Chinese chestnut. In 1983, the American Chestnut Foundation was established to preserve and restore the American chestnut through funding a scientific breeding program and related research. The American Chestnut Foundation maintains a research farm in Meadowview, Virginia, managed by Fred Hebard. Breeding research continues, as well as research in genetic-engineering of resistance genes into chestnut, although the latter is making little progress. In 1985, the American Chestnut Cooperators' Foundation was established and developed a research program based on interbreeding within the species to develop resistant trees (Freinkel 2007). Whatever the future of the American chestnut might be, we are not likely to see this tree as a significant forest tree any time in the next few generations.

Dogwood Anthracnose

Dogwood anthracnose (*Discula destructiva* Red) was noted on the east coast in the early 1980s. This fungal disease causes necrotic lesions on the leaves and leads to twig dieback and eventual tree mortality (Daughtrey and Hibben 1994). The disease has spread all along the east coast from Massachusetts to Alabama. Monitoring of disease incidence in the Great Smoky Mountain national park between 1988 and 1991 revealed an increase of plots with severe epidemics of 638% with tree mortality in 41% of the plots. The native flowering dogwood (*Cornus florida*) on the east coast and the Pacific dogwood (*Cornus nutallii*) on the west coast are highly susceptible to the disease. The non-native Kousa dogwood (*Cornus kousa*) also hosts the disease, but has fewer symptoms (Daughtrey and Hibben 1994).

Dogwood mortality within the Cumberland Plateau has eliminated seedling and sapling size classes completely within cove areas and is likely to be eliminated from upland forest ridges. Lack of fruit production resulted in negligible additions to the population (Hiers and Evans 1997). With the loss of the high fat fruits of dogwood, fall-migrating birds are consuming more fruits and dispersing more seeds of blackgum (*Nyssa sylvatica*) and spicebush (*Lindera benzoin*). Dogwoods are also important calcium reservoirs within the ecosystem and loss of dogwoods can have serious implications for passerine bird egg production dependent upon invertebrates that get their calcium from dogwood leaves and litter. Calcium leaching is also accelerated by acid rain prevalent along the east coast (Hiers and Evans 1997). This loss of dogwoods represents not only a loss of nesting sites and food source, but also has impacts on nutrient cycling in the ecosystem.

Dutch Elm Disease

First discovered in Ohio in 1930, Dutch elm disease was introduced into the United States on imported logs. The fungus is spread by elm bark beetles. Elm trees of all ages and all species are susceptible (Boyce 1961). The first pandemic of elms was caused by *Ophiostoma ulmi*. In the mid-1900s, a new, more virulent species, *O. novo-ulmi* emerged (Santini and Faccoli 2014). The disease was rapidly spread by the small elm bark beetle (*Scolytus multistriatus*). The beetle infects elms when feeding at crotches of young twigs, introducing the fungus into the tree's vascular tissues. Dutch elm disease can also be transmitted from tree to tree through root grafts (Santini and Faccoli 2014). This disease has been devastating to the native elm population. Research on Dutch elm disease has declined dramatically over the last few decades, although modern genomic approaches may open new avenues to understanding and dealing with the elm-fungus-beetle pathosystem (Bernier et al. 2013).

Thousand Cankers Disease

In 2011, thousand cankers disease was discovered on black walnut (*Juglans nigra*) trees growing around the Richmond area, in Fairfax, and Prince William counties (VDOF 2014b). This disease is caused by a fungus, *Geosmithia morbida*, which is spread by the walnut twig beetle (*Pityophthorus juglandis*). Currently, 10 counties and six municipalities have been quarantined to limit the spread of the beetle and the disease (VDOF 2014b). There are no currently available controls for this disease. The disease has been present in the western United States in Colorado, Idaho, Oregon, Utah, and Washington. The first observation of the disease in the eastern US was in Tennessee. Now, it is present in Virginia and Pennsylvania as well. If this disease is not contained, there will be an enormous ecological and economic impact (Randolph et al. 2013).

Insects

Gypsy moth

Oaks comprise a large percentage of forest trees in Virginia with various species distributed from the mountains to the seashore. Anything that can seriously affect oaks can have a devastating effect on Virginia woodlands. As such, gypsy moth (*Lymantra dispar*) has proven to be a serious threat. Gypsy moth was introduced into the US at Medford, Massachusetts, in 1869, in an ill-fated attempt to use the moths for silk production (McManus 2007). Some moths escaped into the wild and established an enduring population in Massachusetts, spreading into Maine, New Hampshire, and Rhode Island over the next 25 years. Gypsy moths reached Virginia in 1984, resulting in our first noticeable defoliations. By 2000, over 71,000 acres had been defoliated in Virginia (Roberts 2001). If unchecked, populations of gypsy moth will spread at a rate of about 13 miles per year (USFS 2007).

Initial infestations generally result in 15-35% tree mortality, but can result in mortality as high as 75% (Roberts 2001). Gypsy moth larvae can defoliate trees, leaving them susceptible to attack by secondary agents that result in tree mortality. Oaks (*Quercus*) are a preferred food source, but gypsy moths will also feed on birches (*Betula*), sweetgum (*Liquidambar*), poplars (*Populus*), willows (Salix), basswood (*Tilia*), hornbeam (*Carpinus*), hophornbeam (*Ostrya*), witch-hazel (*Hamamelis*), hazelnut (*Corylus*), and hawthorne (*Crataegus*) (Davidson et al. 1999). In all, gypsy moths can feed on more than 500 species of plants. As the moths are tannin-adapted, they are able to preferentially feed on oaks that are resistant to feeding by other insect larvae. Oaks with higher levels of carbohydrates and proteins are fed on preferentially, despite high concentrations of tannins (Foss and Rieske 2003).

Gypsy moths preferentially feed on oaks over other tree species. Foliar characteristics such as tannin, carbohydrate, and nitrogen content, leaf toughness, and density have no clear relationship to feeding preference. One study (Foss and Rieske 2003) demonstrated that black, burr, cherry bark, and northern red oaks are most preferred by gypsy moth larvae, while pin, swamp white, white, and willow oak are least preferred. Despite the feeding preference, larvae fed on pin oak grow and develop most rapidly. As a tannin-adapted species, gypsy moth can readily consume oak foliage with high tannin concentrations such as burr oak and pin oak (Foss and Rieske 2003).

Another study (Campbell and Sloan 1977) recorded defoliation of white and black oak tends to be heavy, while scarlet oak defoliation is moderate and red oak defoliation tends to be low. Heavily defoliated oaks require about 10 years to restore pre-attack foliage levels. When defoliation occurs on white pine and red maple, the red maple trees are more likely to die. Defoliation tends to be most severe in the first year of the outbreak, and within species, some trees were more consistently defoliated than others, indicating that individual genetic differences among trees can be important in feeding preferences (Campbell and Sloan 1977).

Control of the spread of gypsy moth was attempted by developing barrier zones in the 1920s and 1930s, but limited funds hampered these efforts. In the 1940s and 1950s, spraying of DDT was employed in selected areas, but was abandoned due to detrimental environmental effects. Carbaryl became the treatment of choice for a number of years until better methods of control were developed. Research efforts in the 1970s led to integrated pest management approaches in an attempt to contain the threat, while allowing differences in regional approaches. In 1972, *Bacillus thuringiensis kurstaki* was introduced for control of gypsy moths. In 1976, diflubenzuron was added to the arsenal, followed by the introduction of Gypchek (gypsy moth nucleopolyhedrosis virus) in 1978, and the use of synthetic pheromone flakes to disrupt mating in the following year 1979 (McManus 2007). Disparlure (cis-7,8-epoxy-2-methyloctadecane) is an important part of the surveillance of gypsy moth spread and the key to the Slow-the-Spread program coordinated through the USDA (Tobin et al. 2012). Gypsy moth females cannot fly and this pheromone is used in baited traps to disrupt mating with males. Dispalure can remain in the environment for a short time after removal of the dispensers, but is primarily emitted from the dispenser for up to two years (Onufrieva et al. 2013).

In Virginia, acres defoliated by gypsy moth increased steadily from 374 acres in 1984 to a peak of 748,000 acres in 1992. Gypsy moth populations show natural fluctuations and defoliation fell to 452,475 acres in 1994, followed by a resurgence in 1995 to 850,000 acres defoliated. Then the population crashed in 1996 and remained at insignificant levels until a population explosion in 2000 resulting in 71,122 acres defoliated. By 2000, a total of 4,428,412 acres had been defoliated in Virginia (Roberts 2001).

Between 1996 and 2000, a couple of factors seems to have been important in suppressing the gypsy moth population. Two pathogens of gypsy moth were found to be present at the sites of the population crash. Nuclear polyhedral virus (LdNPV) and *Entomophaga maimaiga* were found in the local environments (Hajek et al. 1996, Webb et al. 2003). *E. maimaiga* had been introduced into the US years before as a control attempt. The persistence and slow spread of this fungus makes it effective in helping to prevent outbreaks of gypsy moth, whereas the LdPNV seems to be more active when gypsy moth populations reach high densities (Elkinton et al. 1991).

Gypsy moth nuclear polyhedrosis virus (LdNPV) causes epizootics in gypsy moth. The virus particularly affects later instars. Rainfall can distribute LdNPV in tree crowns, washing the virus out of upper branches and distributing the virus to lower branches (D'Amico and Elkinton 1995). Heavy rains can wash the virus from tree bark and solar radiation can inactivate the virus (Podgwaite et al. 1979). Different textures of tree bark can affect the persistence of the virus on the tree bole. LdNPV persists at high concentrations in forest litter and soil (Podgwaite et al. 1979).

LdNPV is spread through the forest through a variety of animals passing the polyhedral inclusion bodies through their alimentary tracts. Animals distributing the virus include the white-footed mouse, short-tailed shrew, southern flying squirrel, opossum, raccoon, house finch, redwing blackbird, and mourning dove (Lautenschlager and Podgwaite 1979).

Entomophaga maimaiga is a pathogenic fungus causing epizootics in gypsy moths. This fungus produces resting spores (azygospores) and airborne conidia that spread the fungus from dead larvae to living larvae. Spores can travel several kilometers during storms with strong winds (Weseloh 2003). Older larvae (fifth or sixth instars) typically produce more resting spores than conidia, and pupae are not infected, but, if infected as larvae, can produce some spores upon their death (Weseloh 2003). Resting spores can germinate during the larval stage of the moth life cycle. Infection of larvae is most likely a function of rainfall and soil moisture when spores are present. Newly hatched larvae are most susceptible to infection from ground spores, whereas most later-stage infections are probably due to secondary infection via conidia (Weseloh and Andreadis 1992).

Beyond LdNPV and *E. maimaiga*, other agents within the forest help control gypsy moth populations. Deer mice, shrews, and birds predate the gypsy moths and help reduce their numbers when populations are scarce. Forest ants are also important in the control of gypsy moth larvae (Weseloh 1994). Finally, temperature is another factor in the survival of gypsy moths. Temperatures can be too high for larval development and pupation and might limit expansion into warmer zones (Tobin et al. 2014). Warmer temperatures associated with the Coastal Plain and Piedmont regions of Virginia have limited the growth of populations of gypsy moth (Tobin et al. 2014). Gypsy moth outbreaks affect bird populations by disrupting or, in some cases, creating nesting sites. These outbreak populations are usually not long-term, but suggest that non-pesticide control measures are best for managing gypsy moth infestations from an ecosystem perspective (Gale et al. 2001).

Hemlock woolly adelgid

Hemlock woolly adelgid (*Adelges tsugae*) is native to Asia. This destructive insect was introduced into the eastern U.S. from southern Japan (Havill et al. 2006) and first detected on eastern hemlock (*Tsuga canadensis*) in Richmond, VA in the 1950s (Souto et al. 1996). Hemlock stands provide an essential habitat for birds, mammals, amphibians, reptiles, some fish, and a number of invertebrates. Decline of hemlock forests will have effects on avian species that are dependent on these habitats for breeding. Several bird species are hemlock obligates. Specialized obligate species will be most affected by loss of hemlocks. One study estimated that hemlock loss would adversely affect 3600 bird pairs in the Delaware Water Gap National Recreation Area, which covers 1130 ha. This would extrapolate to millions of pairs of breeding birds being affected by hemlock loss in the northeastern US alone (Ross et al. 2004). The loss of hemlock stands has a great ecological impact (Degraaf and ChadWick 1987, Quimby 1995, Ross et al. 2004).

Trees infested with woolly adelgid exhibit reduced growth and needle loss. Infestations can kill a mature tree within three to four years (McClure 1991). Aphids in general tend to feed on the phloem tissues in their host plants, but the woolly adelgid feeds on xylem parenchyma in xylem rays, which serve as a transport canal between phloem and pith, and serve as nutrient storage cells. Multiple nymphs insert their stylets at the base of a needle penetrating the petiole and feed off the nutrients of the hemlock (Young et al. 1995).

There are no known natural predators of the woolly adelgid in the eastern U.S., but populations may be limited by weather. Woolly adelgid is a cool-weather species (Day and Salom 2010), and warmer coastal temperatures may limit the spread in the eastern part of the state. Winter temperatures below -20C cause significant reductions in woolly adelgid populations, but these temperatures are not likely to limit Virginia populations due to their rare occurrence.

Mortality in infested hemlock stands is higher in understory trees than overstory trees. Intermediate and sub-dominant trees die out first. Replacements for canopy trees are severely limited due to understory mortality. Thus, as the overstory trees eventually die off, the entire stand of hemlock is lost (Krapfl et al. 2011). The ecological impact of the loss of the hemlock forests may equal the losses resulting from the forest devastation caused by the chestnut blight (Krapfl et al. 2011).

In a study following eastern hemlock stands over nine years (Eschtruth, et al. 2006), 25% of the hemlocks were either dead or in severe decline. Understory light increased significantly and litter cover decreased. There was a shift in angiosperm woody species including increases in tulip tree (*Liriodendron tulipifera* L.), black gum (*Nyssa sylvatica* Marsh.), red maple (*Acer rubrum*), and birches (*Betula* spp.). Most disturbingly, 35% of sample plots contained at least one invasive plant species and 5% contained two or more species. Loss of hemlock stands may dramatically increase the spread of invasive species (Eschtruth, et al. 2006).

The loss of eastern hemlocks may leave *Rhododendron maximum* to define future successional patterns in the Appalachian Mountains due to the high density of this shrubby species in the understory (Krapfl et al. 2011). It was predicted that hemlock stand mortality would result in increased stream flow because hemlocks are such an important tree component in cove and riparian habitats (Ford and Vose 2007). However, a study of water relations in North Carolina found a significant decrease in water flow from a watershed that suffered loss of hemlock following infestation with hemlock woolly adelgid (Brantley et al. 2015). Following hemlock decline, *Rhododendron maximum* became more prevalent, and this combined with species such as *Acer rubrum*, *Betula lenta*, and *Liriodendron tulipifera* with higher transpiration rates than hemlock resulted in water flow reduction.

Hemlock woolly adelgid infestations alter the energy and organic and inorganic nitrogen fluxes in hemlock stands (Stadler et al. 2006). Nitrification rates increase in infested forest stands, and nitrate leaching into waterways can be a problem (Jenkins et al. 1999). Death of hemlock trees also affects carbon cycling from foliage drop, and loss of fine roots further affects nutrient cycling (Nuckolls et al. 2009). Further changes in nutrient and water cycling would be expected to occur with the successional stands that follow the hemlock mortality.

Adelgids are dispersed in the egg and crawler stages by wind, birds, and deer (McClure 1990). This dispersal can be very long range during bird migration periods. Control of infestations in individual trees can be carried out by systemic insecticide applications or applications of dormant oils (Dilling et al. 2009). However, control over forested regions has proven problematical, and may be best addressed by biological control. Biological control has been attempted by releasing *Sasajiscymnus tsugae*, *Scymnus sinuanodulus*, and *Laricobius nigrinus*, but these have not proven effective (Vieira et al. 2011). *Laricobius niginus* released in field tests in Virginia was shown to persist over at least two generations, but adelgid populations were maintained even as the predator population increased in density (Lamb et al. 2006). Another species, *Laricobius osakensis* is highly specific to predating *A. tsugae*. This predatory insect is active in winter, making it synchronous with woolly adelgid activity, while other prey species are dormant. *L. osakensis* appears to have great potential for controlling hemlock woolly adelgid with minimal risks to other populations (Vieira et al. 2011).

Emerald Ash Borer

There are at least 16 species of ash native to the United States. White ash (*Fraxinus americana*) is of economic importance and is used in manufacturing tool handles, baseball bats, flooring, and furniture (MacFarlane and Meyer 2005). Members of the various species of ash cover a wide range of ecological habitats from dry uplands to wet lowlands, and inhabit a variety of soil types (MacFarlane and Meyer 2005). Ash trees have also been widely used in urban areas as a preferred shade tree. Green ash (*Fraxinus pennsylvanica*) is tolerant of salt, high pH, and drought stress, and has been used to replace American elms that were lost to Dutch elm disease. It is expected that emerald ash borer (*Agrilus planipennis*) will lead to widespread loss of ash trees from forest ecosystems and from urban environments (MacFarlane and Meyer 2005).

Emerald ash borer was introduced into the US from Asia, probably in the 1990s, and was first identified near Detroit, Michigan in 2002 following local ash decline and mortality. Infested trees are killed from larva girdling branches and the trunk by feeding in galleries in the phloem and cambium (Herms and McCullough 2014).Between 2002 and 2007, over 20 million ash trees were killed by the borer (Poland 2007) and the infestation was spreading at a rate of 10.6 km/yr (Smitley et al. 2008). It is estimated that the cost of treatment, removal and replacement of ash trees between 2009 and 2019 will reach $10.7 billion (Kovacs et al. 2010).

Following unsuccessful attempts to quarantine the beetle by removal of ashes in the border area, searches began for natural controls by predators, pathogens, and parasitoids. Woodpeckers feed on the larvae (Lindell et al. 2008), and at least one native parasitoid, *Atanycolus cappaerti*, attacks the larvae, but at a fairly low rate (Herms and McCullough 2014). Other native parasitoids include *Balcha indica*, *Eupelmus pini*, and *Dolichomitus vitticrus*, along with a species of *Orthizema* and one species of *Cubocephalus*, but none of these appear to be very effective (Duan et al. 2009). An egg parasitoid, *Oobiusagrili*, native to northern China was introduced into the U.S. for biological control of emerald ash borer. This insect overwinters successfully in the U.S. (Duan et al. 2012) but seems to have had limited success thus far. Two larval parasitioids, *Tetrastichus planipennisi*, and *Spathiusagrili*, may hold more promise, and are still being evaluated (Herms and McCullough 2014).

The fungus *Beauveria bassiana* has also been tested as a biological control measure and may be beneficial in the fight against emerald ash borer (Liu and Bauer 2008). However, in a controlled study comparing the effects of host tree defense, disease, predation, and parasitism, all control measures proved to be relatively ineffective (Duan et al. 2010).

Imidacloprid is a recommended insecticide for treating emerald ash borer, for soil injection, soil drenches, basal trunk sprays, and trunk injections by professional applicators and as a soil drench by homeowners. Application times are mid-spring to late spring or mid-fall (Herms et al. 2014). Trees injected with imidacloprid had larval densities reduced from 82-96 % (McCullough et al. 2010). Injected imidacloprid is translocated mainly through the xylem, and tends to become concentrated in the leaves (Mota-Sanchez et al. 2009). Imidacloprid treatments must be repeated annually (McCullough et al. 2011). Imidacloprid use has serious effects on non-target species as will be discussed elsewhere in this text.

Emamectin benzoate has also proven an effective insecticide with multi-year protection (McCullough et al. 2011, Smitley et al. 2010). Emamectin benzoate provides better protection from ash borer larvae than imidacloprid (Smitley et al. 2010). Emamectin disrupts nerve signal transmission and is used against lepidopterous pests (Zhao et al. 2006). Although this insecticide is very effective against the emerald ash borer, one has to be concerned about the effects on non-target species.

Kudzu bug

One of the emerging threats to agriculture in Virginia is the approach of the Kudzu bug (*Megacopta cribraria*). First noticed in northern Georgia, near Atlanta in 2009, this insect quickly spread from nine counties in Georgia across seven states in only three years. By 2012, the kudzu bug was present in two southern Virginia counties (Ruberson et al. 2013). The kudzu bug is so-named because of its close association with kudzu (*Pueraria montana* var. *lobata*), on which it preferentially feeds. Because kudzu is a severely invasive plant species, the appearance of the kudzu bug might seem like a benefit, and indeed the insect can reduce kudzu biomass production by about a third in its first year of infestation (Zhang et al. 2012), but the bad news is that the kudzu bug will also attack crop plants including soy beans (*Glycine max*), snap beans (*Phaseolus vulgaris*), and cotton (*Gossypium hirsutum*) (Ruberson et al. 2013). Soybeans and cotton were among the top10 agricultural plant commodities for Virginia, generating $284 million and $77 million, respectively, in cash receipts in 2013 (VDACS 2015a). The kudzu bug has been documented to cause up to 50% reduction in soybean yields (Wang et al. 1996).

The climate of the southeastern U.S. is ideal for the growth and reproduction of this insect which has no natural enemies here. It is anticipated that this pest will continue to spread rapidly. Human contact with this insect (especially in the nymph stage) can cause skin rashes. The insects are attracted to light-colored structures, and invade homes and other structures and can congregate in large numbers. Control can be accomplished by wide-spectrum insecticides such as organophosphates and pyrethroids, but organophosphates are particularly dangerous around humans and pets. Biological control may be a better route to pursue. Kudzu bug is the only plataspid species in North America, so finding a parasitoid species that feeds exclusively on kudzu bug would be beneficial without harming native species of insects. *Paratelenomus saccharalis* (Dodd) has been proposed as a biological control agent because it is highly host-specific, its ecology is well understood, and it has a wide geographic distribution. *Paratelenomus saccharalis* only attacks several species in the family Plataspidae, and given the geographic distribution, finding one or more populations that can survive in the US should be likely (Ruberson et al. 2013).

Browsers

White-tailed Deer

Tree regeneration in Virginia has been negatively affected by white-tailed deer (*Odocoileus virginianus*) browse. Deer reduce growth and survival of seedlings and saplings. Deer intensely browse American beech (*Fagus grandifolia*), black cherry (*Prunus serotina*), blackgum (*Nyssa sylvatica*), flowering dogwood (*Cornus florida*), ironwood (*Carpinus caroliniana*), and ash species (*Fraxinus* spp.) (Carter and Fredericksen 2007).

Deer have a pronounced effect on the forest ecosystem. Red oak (*Quercus rubra*) regeneration is very strongly limited by deer. Eastern hemlock (*Tsuga canadensis*) and northern white-cedar (*Thuja occidentalis*) are important winter food sources for deer. The largest species component of forests is the herbaceous species of plants, often by a factor of 10:1 herbaceous to tree species. Changes in plant species diversity by deer affects insect diversity through alterations in the food web. Heavy browsing pressure by deer often favors graminoids and ferns. This reduces pollen and nectar availability and can decrease invertebrate diversity (Rooney and Waller 2003). High deer densities produce a threat to biological diversity.

Browse-tolerant tree species tend to have higher lignin contents, and with heavy herbivory, decomposition and mineralization rates can be altered, affecting soil fertility in the forest (Rooney and Waller 2003).Browsing by deer, even at densities as low as four deer/km^2, negatively influences woody vegetation height and species richness. After 20 years of excluding deer on forest plots, seedling heights were 2.25 times higher and stem count was 4.1 times greater than outside the exclosure (McGarvey et al. 2013).Deer browsing can lead to impoverishment of the herbaceous layer. In a study spanning 26 years, deer in the southern Appalachians caused the disappearance of 46 herbaceous species, while species richness increased by 106% and cover increased by 183% in reference plots without chronic herbivory. Chronically browsed areas tended to become more homogeneous in species composition as they decreased in species diversity (Thiemann et al. 2009).

Loss of Pollinators and Fruit/Seed Dispersers

Insects are important plant pollinators and are an important source of food for many bird species. In turn, many bird species are important in fruit and seed dispersal, as well as acting as pollinators themselves. The increasing use of neonicotinoid insecticides to treat farm and forest pests is posing a grave threat to the plant community by reducing or removing components of the ecosystem that provide vital ecosystem services.

Neonicotinoids are systemic insecticides that are relatively long-lived, water-soluble, and can accumulate in soils and move into surface and ground waters and thus have the potential to affect many organisms. They can affect non-target invertebrates and can cause prey-base collapses, which can subsequently affect avian populations. Farmland bird populations show negative or lower growth rates in response to higher concentrations of neonicotinoids. Neonicotinoids can cause cascading trophic effects in the ecosystem (Hallmann et al. 2014).

Imidacloprid in leaves can adversely affect non-target leaf-shredding invertebrates. Toxic effects of leaf material inhibit feeding. Neonicotinoids adversely affect leaf litter breakdown, organic-matter processing, nutrient cycling, and detritus-based food webs. Riparian forest corridors are especially vulnerable where leaf litter inputs are important drivers of the aquatic ecosystems. Alternatives to neonicotinoids should be considered for the control of invasive forest insects (Kreutzweiser et al. 2009). Imidacloprid binds the acetylcholine receptor of insects, resulting in movement coordination problems, trembling, and tumbling (Suchail et al. 2000).

Honeybees that are important crop pollinators and our source of commercial honey are especially sensitive to the effects of neonicotinoid pesticides such as imidacloprid and related compounds. Imidacloprid is a potent insecticide to honeybees and should not be applied during flowering times. Neonicotinoids can enter honeybee hives through contaminated nectar, pollen and water. As imidacloprid is metabolized, some of its metabolites show toxicity levels close to the parent compound. Acute exposure to imidacloprid or its metabolites produce symptoms of neurotoxicity. Low dose chronic exposure of honeybees to imidacloprid and its metabolites are all toxic (Suchail et al. 2001).

Honeybees exposed to sub-lethal concentrations of imidacloprid show significant increases in *Nosema* infections. Gut parasites *Nosemaapis* and *Nosemaceranae* were significantly higher in the guts of honeybees that were exposed to even low, sub-lethal concentrations of imidacloprid (Pettis et al. 2012).

Sublethal exposure of honey bees to neonicotinoids (imidacloprid and clothianidin) leads to colony collapse. The effect is worsened by severity of winter weather. Members of colonies exposed to neonicotinoids fail to resume brood rearing, even into warm weather. Bees also abandon hives during winter, atypical of non-treated populations (Lu et al. 2014).

Bee diversity benefits pollination services by increasing fruit and seed set, increasing pollination stability, and enhancing efficiency of pollinators within the community (Rogers et al. 2014). A mixture of pollinator generalists and specialists best supports plant communities. Overuse and indiscriminate use of pesticides, and especially the neonicotinoid insecticides pose an imminent threat to the diversity of our plant communities.

Forest Mismanagement

Non-industrial private forestland (NIPF) is often harvested by “high-grading” in Virginia. The lack of a market for low-grade logs and pulpwood leads to selective harvesting of the best stems. High-grading leaves trees with defective stems and non-merchantable species of trees to regenerate the forest. Very little silviculture is practiced on NIPF in Virginia. About 77% of Virginia’s forest land is in private hands, held by over 300,000 forest landowners. As of 2003, only 3% of family forest owners in the southern U.S. had a written management plan (Butler and Leatherberry 2004).

Consider the effects of population mismanagement on eastern redcedar (*Juniperus virginiana*) in Virginia. Eastern redcedar, has been subjected to over 300 years of negative genetic selection. High-grading, or removal of the superior members (most commercial) of the species, leaves behind the genetically inferior members of the species to repopulate and produce a new generation that is less merchantable than the prior generation. This practice has resulted in the successive lowering of the quality of eastern redcedar to the point that there is virtually no market for Virginia eastern redcedar, which now lacks the quality heartwood that is essential to its use in chests, paneling, and other uses.

We need to guard against repeating this scenario with other commercial species such as oaks, tuliptree, and other commercial species. It may be that their longer generation time has slowed the genetic degradation suffered by eastern redcedar.

Fire Exclusion

It may seem strange to think that fire exclusion may be a threat to native biodiversity, but fire has been a large part of Virginia’s history. Indigenous populations and European settlers used fire to clear land and to control brush growth. Consequently, fire-adapted species became established in Virginia. Also, by excluding small, regular fires, fuel accumulation can lead to more intense wildfires that are more destructive to native vegetation. Forest Service, Park Service, and Nature heritage personnel are currently using controlled burns to reduce forest litter and to maintain fire-dependent species.

Threats to Agriculture

Herbicide resistance

Herbicide resistance is now increasing at a rate comparable to the rise of insecticide resistance and fungicide resistance in the past (Holt 1992). Weeds had developed resistance to fifteen different classes of herbicides by 1989 (Holt 1992). Chickweed (*Stellaria media*) that is resistant to chlorsulfuron has shown cross-resistance to imidazolinone herbicides as well (Hall and Devine 1990). As one example, Asiatic dayflower (*Commelina communis*) has become a serious weed problem in soybean fields and is showing resistance to glyphosate (Ulloa and Owen 2009). Asiatic dayflower, or slender dayflower has become a problem weed along the east coast. *Commelina communis* is an animal-dispersed invasive plant species (Naumann and Young 2007). Seeds persist in the soil bank and can germinate at very high rates after four years.The rapid adoption of glyphosate-resistant crops such as corn, soybean, canola and cotton has increased the selection pressure for herbicide resistant weed species.

Land Development

Agriculture and forestry are facing considerable pressure from population increase and land development. The population of Virginia increased by about 14.4% (890,000 people) between 1990 and 2000. This increase is faster than the national average. Within the state, the largest growth rates are seen in Northern Virginia, Hampton Roads, and around Richmond. Loudoun County grew the fastest of any county during this period. Northern Virginia added 435,320 people between 1990 and 2000, pushing growth into Spotsylvania, Stafford, Caroline, and King George counties. Richmond, during this time, added 130,872 persons increasing population pressure in Chesterfield, Henrico, Hanover, and Powhatan counties. In Hampton Roads, population increased by 120,377 adding to James City County and York County (Pollard 2007).

Virginia's farmlands, natural areas, and open spaces are being lost to development, which is occurring even faster than population growth. In 15 years between 1982 and 1997, 784,500 acres were developed, and development is increasing at an increasing rate. Furthermore, 31% of the 343,500 acres developed between 1992 and 1997 was prime farmland. Between 2007 and 2010, over 79,500 acres of land in Virginia were lost to development (UDA 2013). If trends continue, several counties stand to lose all their farmland. A similar impact is occurring on forestry, with 650,000 acres of forest land lost to development between 1992 and 2001 (Pollard 2007). During the 15 years between 1997 and 2012, the number of farms decreased by 3,336.

Virginia is also continuing to pave more and more land. As miles of roads increase, so does the amount of travel by car and transport by truck, with a concomitant increase in carbon emissions. Virginia is making large contributions to carbon dioxide emissions, showing a 34% increase between 1990 and 2004, the ninth highest of any state during that time. Transportation is the leading source of carbon dioxide emissions (Pollard 2007). Rather than looking at mass transit solutions and moving freight from roads to rail, Virginia continues to pursue a policy of building more roads, consuming more land, leading to more damage to forests, farmland, and wildlife communities. Road expansion has not relieved congestion or time lost in traffic delays, but has spurred more development, population increase, and more congestion (Pollard 2007). The economic model of endless growth is as much of a myth as the idea of endless natural resources. We must shift from a model of endless economic growth to one of economic and environmental sustainability (Meadows et al. 1992, Ekins 1993, Hofkes 1996, Giddings et al. 2002). No longer can expansion be considered the solution to societal problems, and no longer can environmental degradation be considered an externality without economic cost. The future of human society depends upon finding a balance between economic welfare and environmental sustainability.

Climate Change and Agroforestry

The increase in atmospheric carbon dioxide and other greenhouse gases is causing a rise in average global temperature (Mann et al. 1998, Karl and Trenberth 2003). We can expect warmer summers, milder winters, and more weather extremes to occur in Virginia in the future. Future temperature increases may reduce wildflower reproduction and crop yields because of effects on pollen viability. Changes in plant phenology may lead to loss of pollinators and birds and pollen for reproduction and fruit/seed dispersal.

The effects of climate change are likely to be numerous and problematic for farming in the southeastern United States (Asseng et al. 2013). Increased summer heat stress is likely to reduce crop productivity. Flowering and seed set are particularly vulnerable to heat stress, especially if combined with drought. Increased diurnal temperatures have resulted in reduced yields of rice and corn in several agricultural regions around the world. Increased temperatures will also result in increased water demand by plants due to increased transpiration. If water needs are not met, yields will decrease. Increased seasonal temperatures will result in an advanced phenology. Warmer temperatures in winter months may reduce fruit set on crops with a chilling requirement such as blueberry and peach (Asseng et al. 2013). In addition, higher temperatures inhibit photosynthesis and carbon uptake in plants (Zinn et al. 2010).

Pollen is particularly sensitive to temperature changes. Various cultivars of corn (Herrerro and Johnson 1980), wheat (Dawson and Wardlaw 1989), and grain sorghum (Prasad et al. 2006) all showed loss of pollen viability at elevated temperatures. Even at elevated carbon dioxide levels, seed set and yield was reduced in sorghum at elevated temperatures (Prasad et al. 2006). In addition to these grain species, cotton is sensitive to elevated temperatures and shows reduced pollen germination (Song et al. 2014), reduced pollen tube growth (Snider et al. 2011a, Snider et al. 2011b), and reduced boll retention and development (Reddy et al. 1992). Peanuts have reduced pollen production, pollen viability, and reduced fruit set at higher temperatures (Prasad et al. 1999). Finally, vegetable crops such as tomatoes (Abdul-Baki and Stommel 1995), and beans (Halterlein et al. 1980) also show reduced pollen germination and elongation at higher temperatures. For fruit and grain production, anything that interferes with plant reproduction decreases yield.

Weed management is likely to become more problematical because of the benefits of warmer temperatures and increased carbon dioxide availability for these weed species. While more atmospheric carbon dioxide may increase rates of photosynthesis up to a point, it does not necessarily correlate to an increase in crop yield. Increases in biomass with increased carbon dioxide availability depends upon water availability and soil nutrient content. Other factors that may limit increases in crop yield include increased predation by pests and competition from weeds (Asseng et al. 2013).

Virginia is highly susceptible to damage from hurricanes, storms, and tidal surges, which are likely to intensify with forecast climate change. Increases in storm intensity is likely to increase damage to most agricultural systems, with wind damage causing long lasting damage to perennial crops. Coastal areas may be affected by rising sea levels leading to salt water intrusion into ground water and increased salinity in coastal rivers affecting irrigation efforts from surface waters (Asseng et al. 2013).

Climate change is posing a threat to the future of eastern forests. Warmer annual temperatures lead to relaxation of range constraints for insects such as the hemlock woolly adelgid, and increases survival and fecundity of the insect. We are likely to see increases in mortality of oaks from forest tent caterpillar (*Malacosoma disstria*) (Asseng et al. 2013).

High elevation forests of the Appalachians are particularly susceptible to changes from a warming climate. A 3C increase in July temperatures would raise climate-elevation bands by about 480m. Mid-elevation cove forests support a diversity of fire-intolerant tree species, ephemeral spring wildflowers, and populations of amphibians that would be substantially changed in an adverse manner by warming and precipitation variability, both of which have been documented since the early 1980s. Increases in intensity of hurricanes as predicted with climate change could cause catastrophic changes to Virginia woodlands. Even a Category 2 storm such as Hurricane Isabel in 2003 damaged many canopy trees in a maturing hardwood forest in the Coastal Plain (McNulty et al. 2013).

CONSERVATION OF NATURAL RESOURCES

There are numerous agencies and organizations that help protect and preserve Virginia's plant life. These include federal and state level public agencies and private or non-governmental agencies working at the state, national, or international levels.

State

Within the Virginia Department of Conservation and Recreation, the Virginia Natural Heritage Program is charged with preserving the diversity of biological resources. The Natural Heritage Program was founded in 1986 and helps establish conservation priorities, and develop management plans for natural communities and rare species (Wilson and Tuberville 2003).

Virginia Land Conservation Foundation

The Virginia Land Conservation Foundation (VLCF) was established in 1999 by the General Assembly and Governor of Virginia. The Foundation has members representing each congressional district. Members are appointed by the governor, the senate and the House of Delegates. In addition, the Secretary of Agriculture and Forestry is also a member. The VLCF is chaired by the Secretary of Natural Resources. The General Assembly funds the VLCF to conserve open spaces and parks, natural areas, historic areas, and forest and farmland. Monies are made available through grants awarded by the Foundation and often involve matching funds from other sources. In addition to purchasing land, funds may be used to establish permanent conservation easements. Grant applications are considered from local governments, state agencies, and qualified nonprofit groups. As of 2012, VLCF grants have helped to protect over 45,500 acres in 130 separate projects (Virginia Land Conservation Foundation 2015).

Virginia Outdoors Foundation

The Virginia Outdoors Foundation (VOF) was established by the Virginia General Assembly in 1966 to promote the preservation of open-space lands. The Foundation uses private gifts (money, securities, land, or other properties) to preserve the natural, scenic, historic, scientific, open-space, and recreational lands of Virginia. The VOF administers the Open Space Lands Preservation Trust Fund. The VOF holds conservation easements, restricting certain types of development on lands in perpetuity. Through the VOF, over 750,000 acres in Virginia has been protected (Virginia Outdoors Foundation 2014).

Virginia Department of Forestry

The Virginia Department of Forestry (VDOF) manages 24 state forests occupying 68,626 acres of forest land. The VDOF is charged with protecting forest resources from fire, managing the Commonwealth's forest resources, protecting our water resources, conserving the forest land, and managing the forests and state tree nurseries. Nurseries provide seedlings for timber stand establishment, provide pulpwood crops, provide trees for Christmas tree plantations, enhance wildlife habitat, stabilize stream banks, and improve watersheds (VDOF 2014a).

Office of Farmland Preservation

The Virginia Department of Agriculture and Consumer Services administers the Office of Farmland Preservation, which helps localities to obtain agricultural conservation easements, helps develop farmland preservation policies at state and local levels, and helps to educate citizens about the importance of farmland preservation. Importantly, they also operate a program that helps connect potential farmers with retiring farmers so that farmland can be kept in the hands of farmers through the Farm Link Program (VDACS 2015b).

Virginia Department of Conservation and Recreation

The Virginia Department of Conservation and Recreation (VDCR) manages and protects the state parks. It also identifies, inventories and protects rare plants, animals, and communities. There are currently 36 state parks, five undeveloped parks, and 61 natural areas and preserves covering over 126,000 acres under the VDCR jurisdiction (VDCR 2015).

State Natural Area Preserve System

Natural area preserves are established through a legal deed that protects the area in perpetuity by limiting activities on the land to those appropriate and compatible with protection goals for that site. Preserves may be either public or private lands. Currently there are 36 dedicated preserves covering 27,899 acres, and protecting 151 rare plant species (Wilson and Tuberville 2003).

Open Space Recreation and Conservation Fund

The Open Space Recreation and Conservation Fund receives funds monies voluntary contributions designated from state income tax refunds. The funds are used to acquire land for recreation, to preserve natural areas, and to improve state parks. The Fund also provides grant opportunities on a matching basis to localities for recreation projects. The Fund is administered through the Department of Conservation and Recreation (Open Space Recreation and Conservation Fund 2015).

Federal

The Federal Government owns more than 2.3 million acres in Virginia, including national forests, national parks, wildlife refuges, and military bases. The U.S. Forest

Service has 1,785,663 acres of public forest in Virginia. The U.S. Department of Interior National Parks Service has 299,642 acres of parkland in Virginia. The U.S. Department of Interior Fish and Wildlife Service has 128,310 acres in Virginia in wildlife refuges. The U.S. Fish and Wildlife Service has enforcement authority for the federal Endangered Species Act. At the state level, authority for enforcement is located in the Virginia Department of Agriculture for threatened and endangered plant species.

Private and Non-governmental organizations

Nature Conservancy

The Nature Conservancy, founded in 1951, is a private, nonprofit organization that protects biodiversity through acquisition of unique and sensitive habitats for direct management or through transfer to public agencies who take over the protection and management function, usually in cooperation with the Conservancy and/or other environmental groups. The Nature Conservancy has chapters in every state within the United States and is also active internationally. The Nature Conservancy holds 86,000 acres in protection from development. In Virginia the Nature Conservancy has helped to protect over 340,000 acres of land, and maintains 16 preserves open to the public and an additional four preserves that are protected and not open to the public (Nature Conservancy 2015a, 2015b).

Virginia Native Plant Society

The Virginia Native Plant Society is a nonprofit organization dedicated to the protection and preservation of the native plants of Virginia and their habitats. Their goals are to slow the conversion of natural landscape to built and planted landscape areas and to reduce damage to natural ecosystems. The Society provides information about conserving and growing native plants, among other activities. They address plant conservation issues at the state level as well as those in particular communities and regions. The Society organizes local chapters that take the lead in identifying and addressing local concerns (Virginia Native Plant Society 2009).

The Land Trust of Virginia

The Land Trust of Virginia is a nonprofit organization that partners with private landowners to establish conservation easements. Land protected through this program remains in private ownership, and can be sold or passed to heirs (Land Trust of Virginia 2015).

The American Farmland Trust

The American Farmland Trust is a national organization founded in 1980 and is dedicated to protecting farmland and ranchland and to promoting sound farming practices. It operates a national level farmer-to-farmer land exchange program and promote sound agricultural policy development (American Farmland Trust 2015).

NATURE AND HUMAN HEALTH

So, what is the value of conserving our green spaces, our forests, fields, and natural communities? Why should we support conservation efforts at the local, state, and national levels? As a species that evolved outdoors, humans have a visceral connection with nature for nurture and well-being. Numerous scientific studies have demonstrated the connection between nature and human health (Bowler, et al. 2010). The relationship between nature and human health and well-being is clearly established and increasingly supported by scientific research. Spending time in natural settings has been shown to reduce childhood obesity and improve mental health (McCurdy et al. 2010). Outdoor activity reduces the incidence of myopia in children (Rose et al. 2008). Numerous studies demonstrate the beneficial effects of outdoor experiences on improving concentration and reducing hyperactivity in children (Faber Taylor et al. 2001, Kuo and Faber Taylor 2004, Faber Taylor and Kuo 2009, van den Berg and van den Berg 2010, Sahoo and Senapati 2014).

A longitudinal public health study conducted in areas where emerald ash borer killed millions of ash trees examined human mortality caused by cardiovascular and lower-respiratory-tract illness (Donovan et al. 2013). As the ash borer infestation increased, and tree mortality increased, there were an additional 6,113 human deaths related to respiratory illness and an additional 15,080 human deaths related to cardiovascular disease over what would have been expected without the tree loss. Exposure to natural landscapes also can lower heart rate and blood pressure, and enhance immune system defenses such as natural killer cells that help protect against cancer (Laumann et al. 2003, Hartig et al. 2003, Li et al. 2008). Compared to urban settings, people in natural settings are happier and have lowered anger and less aggression (Hartig et al. 1991). Measures of psychological well-being in humans had positive associations with species richness in greenspaces, with species richness of plants giving the strongest benefits (Fuller et al. 2007). The greater the plant biodiversity in one's environment, the greater are the positive benefits.

Virginia has a great diversity of plant life including many rare species distributed across the state. This richness of plant biodiversity supports the animal biodiversity found in Virginia.

Biodiversity is the single best promoter of ecological stability. From plants, we obtain food, medicines, fibers, various wood products, chemical feedstocks, and many other items that are essential to our civilization. By protecting and sustaining our plant communities, we provide not only economic security, but undergird human health and well-being. The value of maintaining natural spaces for humans to enjoy is inestimable. We must face the challenges of invasive plants, insects, diseases, land development, and climate change if we are to maintain the world as we know it for our children and future generations. Only by protecting and sustaining Virginia's natural resources can we sustain ourselves.

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