REPORT OF THE UNITED STATES ON THE CRITERIA AND INDICATORS FOR THE SUSTAINABLE MANAGEMENT OF TEMPERATE AND BOREAL FORESTS OF THE UNITED STATES

CRITERION 3 – MAINTENANCE OF ECOSYSTEM HEALTH AND VITALITY

INDICATOR 15. Area and percent of forest affected by processes or agents beyond the range of historic variation (e.g., by insects, disease, competition from exotic species, fire, storm, land clearance, permanent flooding, salinization, and domestic animals.)

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TABLE OF CONTENTS

LIST OF TABLES

LIST OF ILLUSTRATIONS

LIST OF ACRONYMS AND ABBREVIATIONS

CO_2	Carbon Dioxide
ENSO	El Niño Southern Oscillation
FHTET	Forest Health Technology Enterprise Team (Forest Service)
FHM	Forest Health Monitoring
FHP	Forest Health Protection
FIA	Forest Inventory and Analysis
IPCC	Intergovernmental Panel on Climate Change
MPH	Miles Per Hour
NCDC	National Climate Data Center
NIFC	National Interagency Fire Center (Boise, ID)
NOAA	National Oceanic and Atmospheric Administration
NPV	Nucleopolyhedrosis virus
NRCS	Natural Resources Conservation Service, USDA
PDSI	Palmer Drought Severity Index
RPA	Resources Planning Act
SCS	Soil Conservation Service, USDA (now NRCS)
SOD	Sudden Oak Death
USDA	United States Department of Agriculture

EXECUTIVE SUMMARY

An analysis of Criterion 3, *Maintenance of Ecosystem Health And Vitality*, and specifically Indicator 15 of the Montreal Process of Criteria and Indicators of Sustainable Forestry —*Area and percent of forest affected by processes or agents beyond the range of historic variation (e.g., by insects, disease, competition from exotic species, fire, storm, land clearance, permanent flooding, salinization, and domestic animals)* — was conducted for temperate and boreal forests in the United States.

A key consideration in this analysis is the definition of "range of historic variation" and specifically the time period used to represent baseline conditions. It was assumed that the period 1800–1850 generally represented baseline conditions. Human activities (e.g., forest clearing, livestock grazing, introduction of invasive species) related to European settlement were already underway during this period but concentrated primarily in the eastern part of the country. The most significant influences of European settlement on America's forests, especially in the Central and Western United States generally began after 1850. These influences almost certainly exceeded anything that occurred prior to the mid-nineteenth century. As a result of the aggressive fire suppression program that began in the early 1900s, natural fire (an important factor in the dynamics of most American forest ecosystems) has been either excluded, or the fire return interval has changed significantly. Fire management has significantly changed the fuel levels, composition and stocking of many forests, and, concurrently, the frequency and intensity of fire and other disturbance events. The more recent effects to forests caused by European settlement and fire suppression support the choice of the 1800–1850 time period as representative of baseline conditions. Unfortunately, little or no metric data and only a few anecdotal records exist that document the processes or agents that affected forests during this period.

Depending on the process or agent, metric data in a standard format are available for the past 20 to 80 years. These data are more contemporary than historic and were included because they help establish reference conditions for future analyses of this type. Where available, historic data on processes and agents of national or regional importance are presented and compared with data available for the current analysis period (1996–2000). Where only modern or contemporary data were available, the approach used was to compare the status of processes and agents of national or regional importance during the current analysis period with data that extended back as far as possible (e.g., 1979 for most indigenous insects and diseases).

If data were available to conduct these two analyses, it was determined whether or not each process or agent caused effects to forests that were beyond the range of *historic* variation, (comparing 1800–1850 conditions with 1996–2000 conditions) and/or beyond the range of *recent* variation (comparing 1996–2000 conditions with comparable metric data sets from the past 20–80 years).

Processes and agents included in this analysis were climate, fire, insects, disease, and invasive plants. No suitable data were found for permanent flooding, salinization, or domestic animals. Consequently these analyses were not conducted.

During the period of this analysis, several agents and processes exceeded the known range of *historic* and/or *recent* variation:

1. The period 1997–1998 was influenced by an El Niño of historically high proportions followed by 2 years of La Niña, which resulted in lower than normal precipitation over much of the United States and is considered to be outside the range of *recent* variation.

2. An ice storm, which occurred during January 1998 in the Northeast and affected 17.5 million acres (38 percent of the region's forests), is considered to be outside the range of *recent* variation and probably beyond the range of *historic* variation.

3. Nationally the 2000 fire season, with more than 8 million acres burned, is beyond the range of *recent* variation based on one data source for the area burned by wildfires in the United States between 1960 and 2000. In the West another data source for area burned indicates that the range of *recent* variation (since 1916) was exceeded in 1996, 1998 and 2000 when wildfires in the 11 Western States burned more than 4 million acres per year.

4. Several species of indigenous insects (southern pine beetle, mountain pine beetle, spruce beetle, eastern and western spruce budworms, and Douglas-fir tussock moth) reached extremely high levels of activity between 1973 and 1996 and have declined in recent years, primarily due to decimation of susceptible host types. During the current analysis period the southern pine beetle reached outbreak proportions in several areas outside of its "normal or traditional" epidemic range. These outbreaks, which occurred in central Florida, southeastern Kentucky and southeastern Arizona, are considered to be outside the range of *recent* variation and *may also be a departure from historic variation*. The area affected by mountain pine beetle in the Western States reached a recorded high in 1981 and may be beyond the range of historic variation. In 1996, a long-standing outbreak of spruce beetle in Alaska reached a recent high that is beyond the range of *recent variation and may also be a departure from historic levels.* A major eastern spruce budworm outbreak in Maine reached a recorded high in 1978 that probably exceeded the range of *historic* variation. Similarly, in 1986 western spruce budworm reached a recorded high in the West, when more than 13 million acres were defoliated, that is probably beyond the range of *historic* variation. A recorded high for Douglas-fir tussock moth in 1973 probably exceeds the range of *historic* variation.

5. Dendrochronological studies suggest that recent western spruce budworm outbreaks throughout the Rocky Mountains and in Oregon are now more synchronous, extensive, and severe than those that occurred prior to 1850 and, therefore, are considered to be beyond the range of *historic* variation.

6. While epidemic populations of most indigenous forest insects were at lower levels during the current analysis period than during recent history represented by metric data (e.g., 1979–1995) they still caused serious regional and local damage to forests. Management actions to prevent or suppress some of these insects are underway (e.g., bark beetle species in the West). Moreover, the potential for increased insect activity is high, especially the hazard of bark beetle outbreaks in areas damaged by the fires of 2000.

7. Several species of indigenous pathogens are considered to be at levels that are causing effects beyond the range of *historic* variation. Areas infested by dwarf mistletoes, parasitic plants of western conifers, have probably increased due to fire exclusion. Fusiform rust, a disease of southern pines, has increased dramatically as a result of intensive forest management and the extensive use of plantation species susceptible to this disease. The effects of western root diseases, resulting from an increase in Douglas-fir and true firs caused by the loss of western white pine and fire exclusion, are considered to be beyond the range of *historic* variation. Oak wilt, a tree killing disease of oaks, is causing extensive losses in live oak woodlands in central Texas where it is considered to be "outside of its normal range." Also, a severe occurrence of oak decline and mortality in portions of Arkansas accompanied by an aggressive infestation of red oak borer is believed to be causing effects that are beyond the range of *historic* variation.

8. Populations of white-tailed deer in parts of the Northern RPA Region are at high levels and are causing damage to forest regeneration and understory plants. This damage is considered to be beyond the range of *recent* variation.

9. All exotic insects and diseases, diseases of unknown origin, and invasive plants considered in this analysis are believed to have been introduced or at least not established until after 1850 and therefore all their effects are beyond the range of *historic* variation. Some exotic species have had significant effects on U.S. forests, for example, chestnut blight, Dutch elm disease, white pine blister rust, and gypsy moth.

    a. The area defoliated by the exotic gypsy moth in the Eastern United States during the current analysis period (1996–2000) was well within the range of *recent* variation (1924–1995), but continued to spread south and west. In 2000, defoliation was detected for the first time in Wisconsin. Several recent exotic introductions, (e.g., European pine shoot beetle and Asian longhorned beetle) are causing severe damage and have high potential for expanding their ranges.

    b. Several diseases of unknown origin are continuing to expand their geographic ranges and cause severe damage. Dogwood anthracnose has eliminated flowering dogwood from many eastern forests and a disease known as sudden oak death, first discovered in California in 1995, is affecting an increasing area of tan oak and oak forests.

c. more than 1,400 species of exotic invasive plants are known to occur in the United States. Many affect forest ecosystems by displacing trees or understory vegetation. The ranges of many of these plants are continuously expanding throughout the United States, despite pest management measures.

d. The largest number of exotic invasive species introductions has occurred in the coastal regions of the South and Pacific Coast RPA Regions.

INTRODUCTION

The material provided in this report is intended to serve as supporting data for the 2003 Report of the United States on the Criteria and Indicators for the Sustainable Management of Temperate and Boreal Forests of the United States for:

CRITERION 3 – MAINTENANCE OF ECOSYSTEM HEALTH AND VITALITY

INDICATOR 15. Area and percent of forest affected by processes or agents beyond the range of historic variation (e.g., by insects, disease, competition from exotic species, fire, storm, land clearance, permanent flooding, salinization, and domestic animals).

RATIONALE FOR USE OF THE INDICATOR

This indicator analyses and reports the effects of processes and agents, both natural and human-induced, might have on basic ecological processes in forests. These include land conversion, harvesting, insects and diseases, natural fire cycles and floods, and the introduction of exotic species. Where these processes are altered beyond some critical threshold they may produce significant changes to the condition of the forest. By regularly examining specific indicators, it may be possible to detect deleterious changes and modify management strategies to reverse the change.

DEFINITION OF A HEALTHY, SUSTAINABLE FOREST

A concept critical to this analysis is that of a "healthy and sustainable forest." The current definition of a healthy and sustainable forest currently adopted by Forest Service is: "A condition wherein a forest has the capacity across the landscape for renewal, for recovery from a wide range of disturbances, and for retention of its ecological resiliency while meeting current and future needs of people for desired levels of values, uses, products and services."

DEFINITION OF RANGE OF HISTORIC VARIATION

The range of historic variation is defined as the range of spatial, structural, compositional, and temporal variation of ecosystem elements (plants, soils, animals) within a period specified to represent 'baseline' conditions[1].

This definition does not identify a specific time period that should be used to include or exclude historic data that could represent baseline or historic conditions. Moreover, a standard reference date or time period has not been generally agreed to by scientists or policymakers in the United States. The First Approximation Report for Criterion 3, Indicator 15, noted significant influences on America's forests by European immigrants and suggested that the time period be pre-European settlement. This report also noted that

[1] (http://www.mpci.org/meetings/tac-mexico/gloss_e.html)

the actions of indigenous people contributed to the range of historic variation (Forest Service 1997b). The Forest Service's National Coordinator for Sustainable Development suggests that the early 1800s would serve well as a time period for establishing a baseline condition [2]. For this analysis, it was assumed that the period 1800–1850 represented baseline or historic conditions.

DATA AVAILABLE TO QUANTIFY THE INDICATOR

Depending on the process or agent, a variety of data from different time periods were accessed to support this analysis. These included statistical data and narrative reports published by Forest Service and other Federal and State agencies. This analysis was severely limited by the lack of metric data available to describe conditions during 1800 to 1850. To address this limitation, some historical records (almost all anecdotal and qualitative) were included to attempt to establish an historical baseline for comparison with current forest conditions. However, as noted in the First Approximation Report, the lack of observations from these more distant time frames precludes us from knowing what the exact historical conditions and trends might have been.

Also included in this analysis are metric data from more recent time periods. For example, a relatively complete data set for major forest insects and diseases exists for the period 1979 to 2000. While some scientists argue that these data are too modern and not appropriate for establishing a historical baseline, they were included because they show some distinct trends and establish a new reference condition for future analyses of this type.

Processes and agents included in this analysis are climate (drought and storm events), fire, both indigenous and exotic insects and diseases, invasive plants, and forest area. No suitable historic or modern data were found, consequently no analyses were conducted for permanent flooding, salinization, and domestic animals. The following sections provide brief descriptions of the data sets included in this analysis.

CLIMATE

Climatic data was taken from a variety of sources including the 2001 assessment of Working Group I of the Intergovernmental Panel on Climate Change (IPCC), the National Oceanic and Atmospheric Administration (NOAA), and the National Interagency Fire Center (NIFC) in Boise, ID. Some instrumental records of temperature and drought dating back to 1861 and 1895, respectively, were included. Information on specific storm events between 1938 and 2000, which caused severe forest damage, was taken from individual reports describing these events.

FIRE

Statistical data on wildfire occurrence was accessed from a database maintained by NIFC and from a series of annual forest fire reports published by Forest Service. Data available from NIFC includes countrywide annual statistics on the number of fires and area (acres)

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[2] Personal communication, Albert Abee, Forest Service, Washington, DC.

burned from 1960 to the present and on a decadal basis from 1919 to the present. Forest Service reports available from 1938 to 1978 provide data on total acres burned by State and groups of States. Annual reports from Forest Service for the period 1963 to 1994 include only area burned on lands within national forest boundaries and were not used for this analysis.

On a regional basis, a data set containing annual statistics of area burned in 11 Western States (Rocky Mountain and Pacific Coast RPA Regions) between 1916 and 2000 was found and used in this analysis.

INSECTS AND DISEASES

Information on the status of insect and disease outbreaks was taken from early historical reports and annual national conditions reports published by Forest Service. Formal aerial and ground surveys to map the status of insect damage in United States forests began in some regions as early as 1947. Beginning in 1951, the National Office of the Forest Service began to issue annual insect conditions reports. These reports were brief, narrative descriptions of the regional status of certain insect pests and contained relatively little metric information. In 1971 forest disease conditions were added to the report. Beginning in 1977, some maps, graphics, and statistical data appeared in the reports. In 1979, the format was revised significantly and metric data on a statewide basis for a number of key insects and diseases became a regular feature of the report. Reports from 1979 to the present were used to establish a new reference condition for insects and diseases. An aerial survey database, recently developed by Forest Service, Forest Health Technology Enterprise Team (FHTET) and presently containing digital data from 1996–2000, was also used. The units of measure used to describe the status of most insects and diseases was acres infested (e.g., acres of aerially visible defoliation, acres of concentrated tree mortality) by species. In other cases, especially for some invasive insects and pathogens, status was described in terms of presence or absence of a species by county.

INVASIVE PLANTS

Databases maintained by USDA and selected literature was used to compile information on the status of invasive plants in forest ecosystems. Anecdotal records are included on the introduction of some invasive plants that date back to the early 1800s. The unit of measure used to describe the status of invasive plants was presence of absence of a given invasive plant by State.

FOREST AREA

Area of forest land, by forest type groups, used in this analysis is based on data from the 2000 Resources Planning Act (RPA) assessment and accompanying FIA data (http://fia.fs.fed.us/ library/final_rpa_tables.pdf) (Table 1). Wherever possible, data was summarized by RPA regions (Powell 1994, Figure 1).

Table 1. Forested Area in the United States By Major Forest Type Groups

Eastern Forests			Western Forests		
Forest Type	Area (thousands of acres)	Percent	Forest Type	Area (thousands of acres)	Percent
White-Red – Jack pines	11,669	3.04	Douglas-fir	41,875	11.56
Spruce-fir	17,640	4.54	Ponderosa pine	33,151	9.14
Longleaf-slash pine	13,223	3.44	Western white pine	591	0.16
Loblolly-shortleaf pine	52,530	13.66	Fir-spruce	69,686	19.22
Oak-pine	33,901	8.82	Hemlock-spruce	21,418	5.91
Oak-hickory	130,250	33.88	Larch	1,274	0.35
Oak-gum-cypress	30,285	7.88	Lodgepole pine	17,515	4.83
Elm-ash-cottonwood	13,004	3.38	Redwood	916	0.25
Maple-beech-birch	54,722	14.23	Other softwoods	75,001	20.70
Aspen-birch	17,842	4.64	Western hardwoods	42,519	11.72
Other	4,825	1.26	Piñion-juniper	49,416	13.63
Nonstocked	3,075	0.80	Chapparal	5,187	1.43
Unknown	1,640	0.43	Nonstocked	3,693	1.02
			Unknown	291	0.08
Total	**384,426**	**100.00**	Total	**362,532**	**100.00**

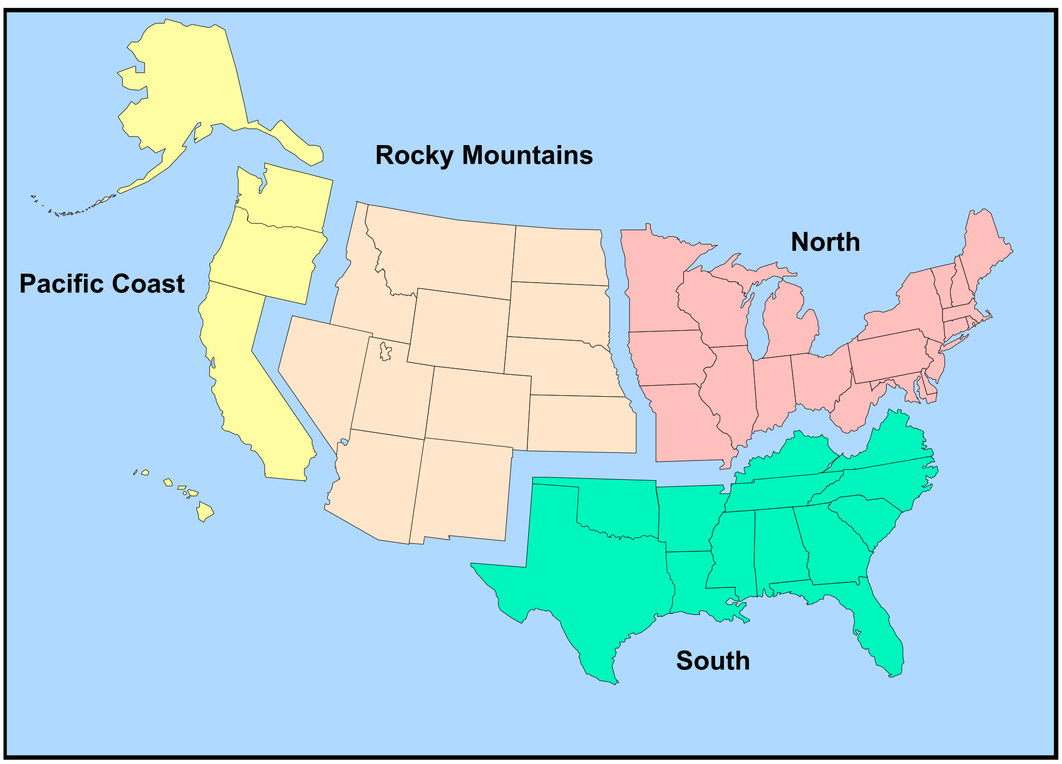

Figure 1. Resources Planning Act (RPA) Regions of the United States

APPROACH

As stated previously, the data sets and time periods covered by this analysis vary by the agent or process due to the availability of suitable date to answer the primary question posed in Indicator 15: Is the area and percent of forest affected by today's agents and processes beyond the range of historic variation?

The time period 1800 to 1850 was assumed to represent historic or baseline conditions. This is not meant to imply that the 1800 to 1850 time period represents a static or steady state condition. Events occurred during and prior to this period that caused significant changes in the distribution and condition of North America's forests. For example, a significant reduction in the population of indigenous tribes that occupied the North American continent occurred after the first Europeans arrived. This was due to exposure of indigenous tribes to several human diseases to which they had little or no resistance. According to one authority, the Indian landscape of 1492, which in the Eastern United States consisted of forest interspersed with small agricultural plots, had largely vanished by the mid eighteenth century and the landscape of 1750 was more "pristine" and heavily forested than that of 1492 (Fickle 2001).

The time period 1996 to 2000 was the period chosen to represent current conditions.

Two analytical approaches were taken in this analysis. Provided that any suitable data were available, the current (1996–2000) status of each process and agent judged to be of national or regional importance was compared to its 1800–1850 status. Based on this comparison, the study determined if the process or agent exceeded the range of ***historic*** variation.

However, because metric data does not exist for these agents and processes during the period 1800 to 1850 and because considerable metric data exists for the past 20 to 80 years, a second analytical approach was used. For this analysis, the status of the same agents and processes during the current period (1996–2000) were compared with their status in the recent past (i.e., 1979 to 1995 for most indigenous forest insects and diseases). Based on this comparison, the analysis determined if the process or agent exceeded the range of ***recent*** variation. The second analytical approach provided an additional benefit; a new reference condition was established for future analyses of this type.

Using appropriate data for each analytical approach, two primary measures were evaluated to determine whether a process or agent exceeded the ranges of historic and/or recent variation:

1. Area and percent of forest and woodland affected.

2. The spatial distribution of areas affected by the process or agent to determine if it is affecting areas beyond its traditional or historical range.

PROCESSES AND AGENTS AFFECTING THE FORESTS OF THE UNITED STATES

The temperate and boreal forests of the United States are dynamic ecosystems, subject to a number of processes and agents that can bring about sudden and drastic change. Known as "disturbance events," these processes and agents are essential to ecologically healthy forests even though in economic terms, they may be regarded as pests that are capable of causing resource damage. Disturbance events help to break down and release elements sequestered within the vegetation, and by increasing plant mortality, they facilitate succession. Disturbances initiate cycles in productivity and biodiversity and thus help to maintain diversity at all levels, (e.g., genetic, species, community, ecosystem, and landscape.) Major disturbance events in U.S. forests include insects, pathogens, herbivores, fire, and climatic events (Leopold and others 1996).

Humans have had a profound effect on the forests of the United States. Indigenous tribes used fire to clear small blocks of land for agriculture and to drive game. European settlers cleared even larger tracts of forest for agriculture and livestock grazing. It is estimated that at the beginning of European settlement, ca. 1630, the area of forest land in what became the United States was 1,045 million acres or about 46 percent of the total land area. By 1907, the area of forest land had declined to an estimated 759 million acres or 34 percent of the total land area. Forest area has remained relatively stable since 1907. In 1997, 747 million acres, or 33 percent of the land area in the United States was classified as forested. Today's forest land amounts to about 70 percent of the area that was forested in 1630. Since that time, about 297 million acres of forest have been converted to other uses, mainly agricultural. More than 75 percent of the net conversion to other uses occurred in the 19th century (Forest Service 2001).

In addition to land clearing associated with European settlement, blocks of previously continuous forest land became fragmented and interspersed with fields, pastures, villages, and cities. While this created an "edge effect," which, in certain cases, increased habitat for some species, in other cases it resulted in genetic isolation because some species were unable to disperse from one block of forest to another. A more complete discussion of forest fragmentation and its effects on biological diversity is found in the analysis of Criterion 1, Conservation of Biological Diversity: Indicator 5 – Fragmentation of Forest Types.

An aggressive forest fire management program began in the early 1900s, designed to protect human settlements and valuable forest resources. As a result, natural fire (an important factor in the dynamics of most American forest ecosystems) has been either excluded or the cycle of fire frequency has changed significantly. Fire management has significantly changed the fuel levels, composition, and stocking of many forests, and, concurrently, the frequency and intensity of fire and other disturbance events. The following paragraphs describe the effects of fire exclusion on western forest ecosystems and were taken from Brookes and others (1987).

> "Fire has historically had a strong influence on the ecology and development of most western forests. Ground fires, which occurred frequently in the drier forest types, periodically eliminated understory trees and selectively reduced the stocking of shade-tolerant-fire susceptible trees in larger size classes. Stand replacement fires often burned during abnormally dry weather or sporadically where woody debris and stand structures with variable height classes provided fuel ladders to the main canopy. The frequency and intensity of natural fires varied by forest type and site conditions, but few stands escaped fire long enough for a climax community to become established. Lighting-caused fires perpetuated a mosaic of heterogeneous seral forests that, because of their diversity, species composition, regulated density and often even-aged structure, precluded widespread outbreaks of western spruce budworm and other insects."
>
> "In western Montana, fire-history studies indicate that the time since the last burn on most forest sites now exceeds the longest fire-to-fire interval that occurred over the two centuries preceding effective fire suppression. Research studies indicate that fire frequency has been reduced by two orders of magnitude in the low-elevation mesic forests of the Bitterroot Canyons. As a result of fire exclusion, dramatic changes have taken place in the composition, structure and density of nearly all forests that were prone to periodic fire events. Open, even-aged parklike stands, composed of shade intolerant seral species have now succeeded to dense, uneven aged stands dominated by more shade tolerant climax species."

Similar records of changing forest conditions exist for the Southwestern United States (Dahms and Geils 1997). A remarkable photographic record of changes in forest conditions over a 100-year period is available from the Black Hills Region of South Dakota. This record compares photographs taken during General George Armstrong Custer's first expedition into the Black Hills in 1874 with photos taken of the same sites in 1974 (Progulske 1974).

International trade and human mobility have also had a significant effect on forests through the introduction of exotic insects, pathogens, and plants. In the absence of the factors that regulate their abundance in their native habitats, these organisms are often able to spread wherever they find suitable conditions for survival, displacing native species and causing severe damage to forests and other resources. A classic example is the introduction of the American chestnut blight, caused by the fungus *Cryphonectria parsitica* from Asia, during the early part of the 20th century. Within 50 years, this fungus destroyed the equivalent of 9 million acres of chestnut forests and changed the composition of eastern hardwood forests forever (Manion 1991). Increased international trade has increased the hazard of accidental introductions of many more invasive species. **Because most exotic invasive species now present in the United States are believed to have been introduced or at least established after 1800–1850, all of their effects on forests (by exotic insects, diseases, and plants) are considered to be beyond the range of *historic* variation.**

Woody plants in many riparian ecosystems in the Western United States are also undergoing changes due to invasion by exotic species, stress induced mortality, increases in insect and disease attack, drought, beaver, fire, climatic changes, and human activities such as agricultural development, groundwater depletion, dam construction, water diversion, gravel mining, grazing, and timber harvesting (Obedzinski and others 2001).

CLIMATE

Climate is the primary factor that influences the behavior of all processes and agents capable of causing change in the condition of temperate and boreal forests in the United States. Moreover, climate can directly influence forest processes. Climate, along with soil conditions, influences the plant associations that occupy a given site. Periodic droughts can reduce tree vigor, making them susceptible to insects, disease, and other pests. Droughts also cause a drying of grasses and woody debris, thus increasing the intensity of wildfires. Excess rainfall, on the other hand, can cause flooding and changes in soil oxygen levels. Severe windstorms, such as tornadoes, hurricanes, and straight-line winds can cause windthrow over large areas of forest. Ice storms can cause stem breakage and bending, especially of young trees.

CO_2 AND TEMPERATURE

In recent years, much concern has been expressed about global climate change due to increased levels of CO_2 and other greenhouse gases in the Earth's atmosphere, mostly from industrial sources. A recent assessment of the global climate change issue by the Intergovernmental Panel on Climate Change (IPCC) presents these hypotheses and predictions with considerably more certainty than 5 years ago. Analysis of data for the Northern Hemisphere indicates that the increase in temperature in the 20th century is likely to have been the largest of any century during the past 1,000 years (IPCC 2001). Processes and agents that affect forests can be influenced by a rapidly changing climate. Warmer global temperatures could result in redistribution of the natural ranges of plant species, rises in ocean levels, as well as changes in precipitation patterns, and frequency and intensity of storm events (IPCC 2001).

Globally, the period 1996 to 2000 was part of what was the warmest decade in the instrumental record and 1998 was the warmest year since 1861 (IPCC 2001). **Although this suggests that temperatures in U.S. forests have exceeded both the ranges of *historic* and *recent* variation, there is no data to specifically address the area and percent of forest affected.**

DROUGHT

Drought is a naturally occurring abiotic stressor to forest communities. Drought is a function of the amount of precipitation in the form of rainfall, snow, ice, and fog drip, and soil characteristics such as water holding capacity. Moderate drought stress tends to slow plant growth while severe drought stress reduces photosynthesis and growth (Kareiva and others 1993). Drought stress in forest communities also influences some insect populations. Mattson and Haack (1987) identified 10 insect families that historically reach outbreak status following drought episodes. Berryman (1973, 1982) identified drought as a cause of outbreaks for both *Scolytus ventralis* (Fir engraver beetle) and *Dendroctonus ponderosae* (Mountain pine beetle). There is also evidence that drought stress influences the uptake of ozone in plants Ozone exposure levels can be relatively high as measured by active monitors, but if plant physiological activity is reduced due to drought stress, ozone uptake and subsequent impact will be minimized.

Some examples of extended drought periods in the United States include (NOAA n.d.):

Southwestern United States (1200–1300) – An episode known as the Great Drought is believed to have brought an end to the advanced agricultural society that developed among indigenous tribes on the Colorado Plateau by the Anasazi Culture.

Southern United States (ca. 1580) – A “megadrought” is believed to have extended from California to the Carolina during the late 1500s. This drought is believed to have been responsible for the demise of the Lost Colony of Roanoke, first English colony in the Americas[3].

Midwestern United States (1932–37) – Known as the “dust bowl,” this devastating drought was in part the result of overuse of the American prairie lands. As a result, bare soil was exposed to the prairie winds and blown away. At its peak, this drought covered 70 percent of the country. It caused a massive migration of people from the Midwest to California and brought about the passage of the Soil Conservation Act, which allocated money to farmers to plant soil building crops.

The 1950’s drought (1950–57) – This drought was first felt in the Southwest in 1950 and spread to Oklahoma, Kansas, and Nebraska by 1953. By 1954 the drought encompassed a 10-State area. This drought devastated the region’s agriculture, and crop yields in some areas dropped by 50 percent.

Northeastern United States (1961–66) – This regional drought is considered to be the most severe in modern American history. It affected 14 Northeastern States (7 percent of the continental United States) and 5 million people or 28 percent of the population. Record forest fires occurred in the region in 1963.

Drought of 1987–89 – At its peak, this 3-year drought covered 36 percent of the United States at its peak and is considered to be the costliest natural disaster in U.S. history. The combined losses in energy, water, and agriculture caused by this drought were estimated

[3] http://www.unc.edu/depts/cmse/science/droughts.html

at $39 billion. The summer of 1988 is well known for the extensive forest fires that burned across Western North America, including the fires in the Greater Yellowstone Basin.

The period 1996 to 2000 was influenced by the development of one of the strongest El Niño southern oscillation (ENSO) events on record (Duffy and Bryant 1998). An ENSO is an anomalous warming of ocean waters off the Pacific Coast of North and South America. ENSO events influence weather worldwide, causing excess rainfall in some areas and drought in others. The ENSO developed in late 1996, peaked in 1997, and began to decline in 1998. The 1997–98 ENSO was followed by an equally strong La Niña, a cooling of ocean waters off the Pacific Coast of the Americas. A La Niña typically causes excessively dry conditions in areas that received excess moisture during the ENSO phase.

Two severe storms that caused extensive forest damage, a blowdown event in Colorado and an ice storm in the Northeast, have been linked to the 1997–1998 ENSO. **The ice storm in the Northeast is considered to be outside the range of *recent* variation and probably beyond the range of *historic* variation.** The La Niña event of 1998–2000 brought all time record high winter temperatures. This resulted in extensive and severe wildfires in portions of Florida during 1998 (Butry and others 2001). The summer of 2000 was the most severe fire season in the United States in 40 years. Other events and processes that may be the result of this La Niña-triggered drought include a buildup of southern pine beetle in portions of the Southeast in 2000 and a massive episode of oak mortality in the Ouachita Mountains of Arkansas. **Both the southern pine beetle and oak mortality events are considered to exceed the range of *recent* variation as well as probably the range of *historic* variation.**

The National Climate Data Center (NCDC) calculates the Palmer Drought Severity index (PDSI) monthly by climate division for the conterminous United States. The PDSI is an empirically derived index based on total rainfall, the periodicity of rainfall, and soil characteristics. The PDSI ranges from +7 to -7 where values from 0 to -0.5 are associated with normal conditions. The PDSI values from -2.0 to -3.0 are associated with moderate drought, -3.0 to -4.0 with severe drought, and less than -4.0 with extreme drought. The NCDC archive has monthly estimates of PDSI from 1895 to present (NCDC 1994).

Growing season PDSI was calculated for each climate division for each year from 1895 through 1999 using the NCDC data. For each year (1895 through 1999), the proportion of the conterminous United States under moderate, severe, or extreme drought was calculated. A spectral analysis was performed to assess whether there was some underlying frequency in growing season drought using Brocklebank and Dickey's (1986) procedure.

This procedure revealed two significant cycles, 26 years and 13 years. The 26-year cycle of growing season drought corresponds to large-scale episodes, typically 40 percent or greater of the total land area of the conterminous United States The 13-year cycle of growing season drought corresponds to smaller scale episodes of roughly 20 to 30 percent of the land area.

The frequencies of moderate, severe, and extreme drought based on the number of years of growing season droughts for the 1895 through 1999 and 1990 through 1999 time periods were calculated for each ecoregion using a weighted average.

The frequency of growing season drought from 1895 through 1999 served as an historical account or reference point for each ecoregion section and RPA region. For example, 28 years of growing season drought was recorded for Section 212G – Northern Unglaciated Allegheny Plateau in western Pennsylvania for the 1895 through 1999 time period. Conversely, Section 212C – Fundy Coastal and Interior of northeastern Maine had only 6 years of growing season drought for the 1895 through 1999 time period. These historical accounts were then put on a 10-year basis and compared to the frequency of growing season drought from 1990 through 1999 to assess deviation from historical growing season drought by ecoregion section. For example, Section 212G – Northern Unglaciated Allegheny Plateau had 27 years of growing season drought over a 105-year period. This corresponds to about 3 years of growing season drought over a 10-year period. For the 1990 through 1999 time period only 1 year of growing season drought was recorded for this ecoregion section. This implies that the 1990 through 1999 time period for this ecoregion section was not as droughty as expected based on historic records.

Overall, forests in the Eastern United States had the expected or less than expected number of growing season droughts for the 1990 through 1999 time period based on historical records. Forests in the Western United States had the expected or more than expected number of growing season droughts for the time period. In the Western United States, Section 342E – Bear Lake in southwest Wyoming, Section M262B – Southern California Mountains and Valleys, and Northwestern Basin and Range ecoregion sections had a 2-year deviation in growing season drought occurrence and consist of scattered forested areas. Section M332G – Blue Mountains was the only section with a high proportion of forested area and +2 years deviation from historic drought.

The most months of drought were experienced in the Eastern United States. Much of the Southeastern United States experienced 6–7 months of drought. Drought stress plays a major role in ecosystem dynamics including influencing insect populations and uptake of ozone in plants. The conterminous United States generally experiences large-scale drought episode on a 26-year cycle and smaller scale episodes on a 13-year cycle. Twenty-four ecoregion sections in the West had more than expected years of growing season drought for the 1990–1999 time period (+1 or +2 years drought deviation). In contrast, the East only had one ecoregion section with more years of growing season drought than expected for the 1990–1999 time period. In 1999, the East had more months of drought than the West (Figs 2-3).

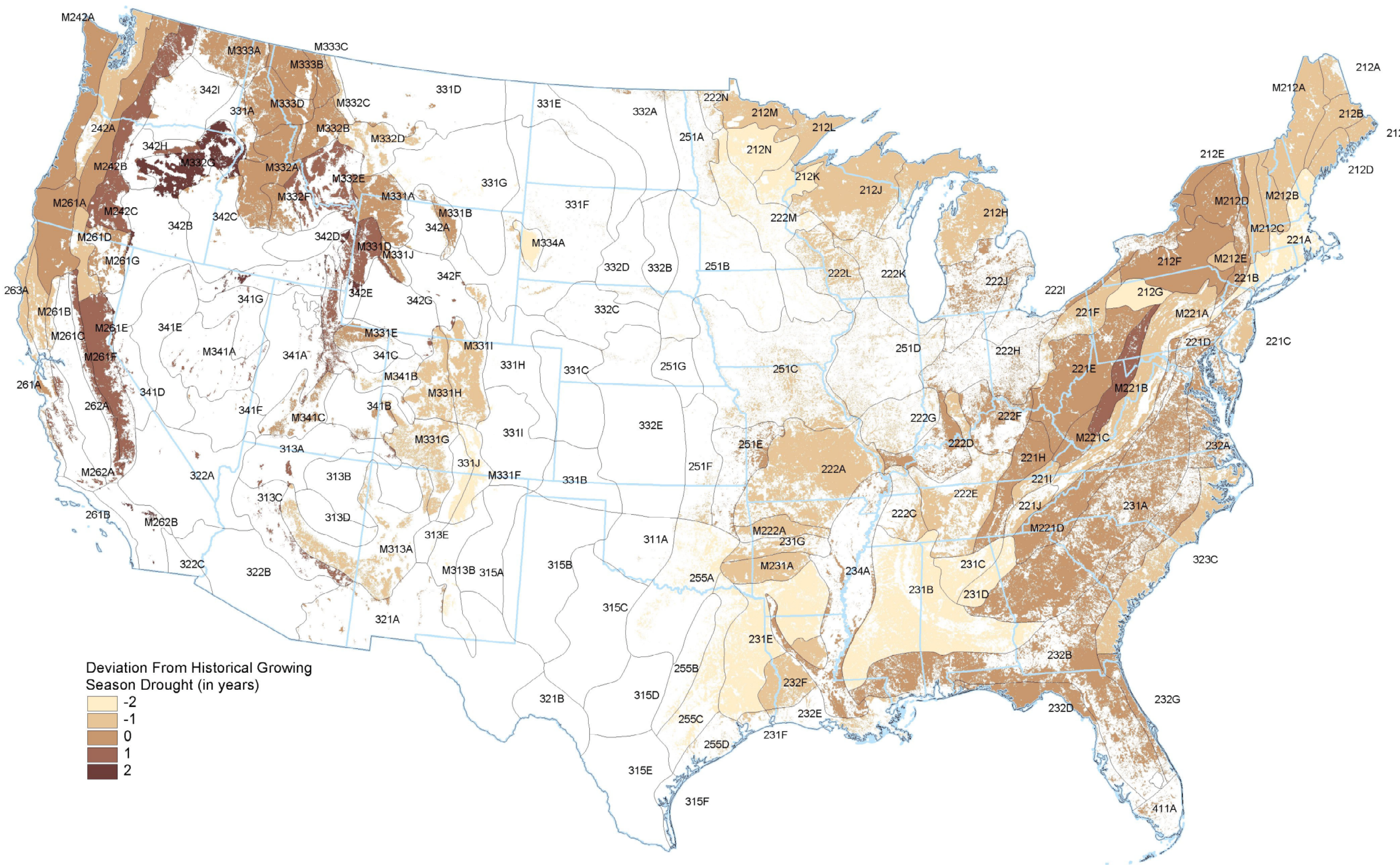

Figure 2. Deviation from historical growing season drought in years, by Bailey's ecoregion section. The frequency of growing season drought from 1895 through 1999 was the historical reference and the frequency of growing season drought from 1990 through 1999 was compared to it. See text for more information.

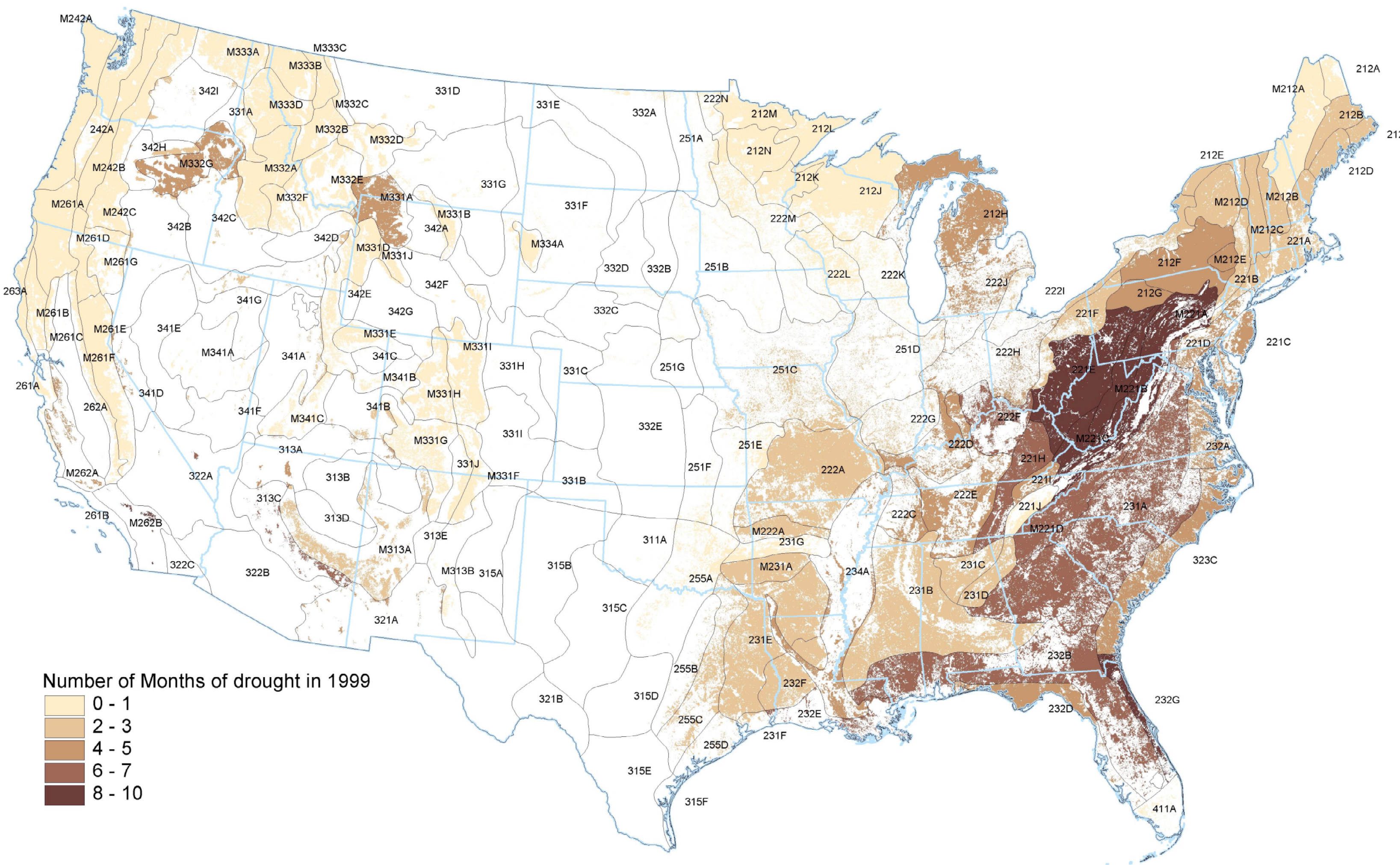

Figure 3. The number of months of moderate, extreme, or severe drought in 1999 as indicated by the Palmer Drought Severity Index

STORMS

Hurricanes, tornados, and ice storms can cause billions of dollars in damage to infrastructure and personal property, loss of human lives and damage to natural resources. Forests are especially susceptible to these events and are subject to windthrow and breakage. The following are brief anecdotal records of some major catastrophic storm events that have occurred in the United States and have had devastating effects on the forest resources of the affected regions:

1938—A major hurricane struck portions of the New England States and caused extensive damage to the region=s forest resources. Forests of eastern white pine, *Pinus strobus*, suffered particularly severe damage (Foster and Boose 1992).

1962—The "Columbus Day" storm caused severe windthrow in forests on the western slopes of the Cascades in portions of Oregon and Washington. The windthrown Douglas-fir provided a large volume of host material suitable for buildup of an outbreak of Douglas-fir beetle, *Dendroctonus pseudotsugae*.

1969—Hurricane Camille, with winds in excess of 200 miles/hour, devastated the coastal region of Mississippi, causing extensive damage to infrastructure and personal property, and loss of human lives. The hurricane damaged more than 1.9 million acres of forests and created conditions favorable for subsequent outbreaks of pine engraver beetles, *Ips* spp. (Terry and others 1969, Touliatos and Roth 1971).

1989—Hurricane Hugo struck the South Carolina coast with sustained winds of 135 miles/hour causing extensive damage to forests in 23 counties. Much of the forest resource of the Francis Marion National Forest, on the South Carolina coast, suffered heavy damage (Sheffield and Thompson 1992).

1994—In early February a severe winter storm moved from Texas and Oklahoma to the mid-Atlantic depositing major ice accumulations of 3–6 inches in northern Mississippi. Approximately 2.1 million acres of forest were affected, resulting in a loss of 16.5 percent of the hardwood volume and 15.3 percent of the softwood volume (Jacobs 1994).

During the period of this assessment (1996–2000), several significant storm events occurred, including one considered to be beyond the range of historic variation.

1996–97—An ice storm occurred in portions of eastern Washington, northern Idaho, and western Montana in November 1996. This was followed by one of the heavier snow years in several decades. The combination of ice and heavy snow caused extensive tree breakage and blowdown. Douglas-fir was most severely affected, and in 1998 an outbreak of Douglas-fir beetle, *Dendroctonus pseudotsugae*, developed in the affected area. This was the largest outbreak to have occurred in the northern Rocky Mountains since the 1950s. About 78,000 acres were affected by the outbreak in 1998 and 187,000 acres in 1999.

1997—Winds in excess of 120 miles/hour, associated with an early winter snow, blew east over the Continental Divide in northern Colorado in late October. The winds resulted in blowdown spruce and fir forests on about 20,000 acres of the western slopes of the Mt. Zirkel Range, Routt National Forest. In addition to the extensive damage caused by the high winds, the large volume of down trees posed an increased hazard of wildfire and provided breeding sites for the spruce bark beetle, *Dendroctonus rufipennis*, a major native insect of spruce forests across North America. The bark beetle outbreak materialized in 2000 (Forest Service 2001) and a portion of the blowdown area burned in 2001.

1998—A devastating ice storm struck portions of Eastern Canada and the Northeastern United States. In northern New York and portions of Vermont, New Hampshire, and Maine, more than 17 million acres, or 38 percent of the total forest area, suffered varying degrees of damage (Table 2, Figure 4). **This storm was regarded as the "ice storm of the century" and represents an incident that is beyond the range of *recent* variation for ice storm events because of the extremely large area affected.** Most severe damage occurred in broadleaf forests; especially aspen, beech, birch, black cherry, maple, white ash, and oak. The storm left a substantial volume of woody debris on the ground (Miller-Weeks et al 1999, Forest Service 2000).

Table 2. Area of Forest Land Affected by the Ice Storm of 1998, Northeastern United States

State	Area of forest damage* (thousands of acres)	Total area of forest land* (thousands of acres	Percent affected by the ice storm**
New York	4,600.0	18,581	24.7
Vermont	951.0	4,607	20.6
New Hampshire	1,055.0	4,955	21.3
Maine	11,000.0	17,711	62.1
Total	17,610.0	45,854	38.4

* Source: Miller-Weeks and others 1999
** Source: http://fia.fs.fed.us/library/final_rpa_tables.pdf

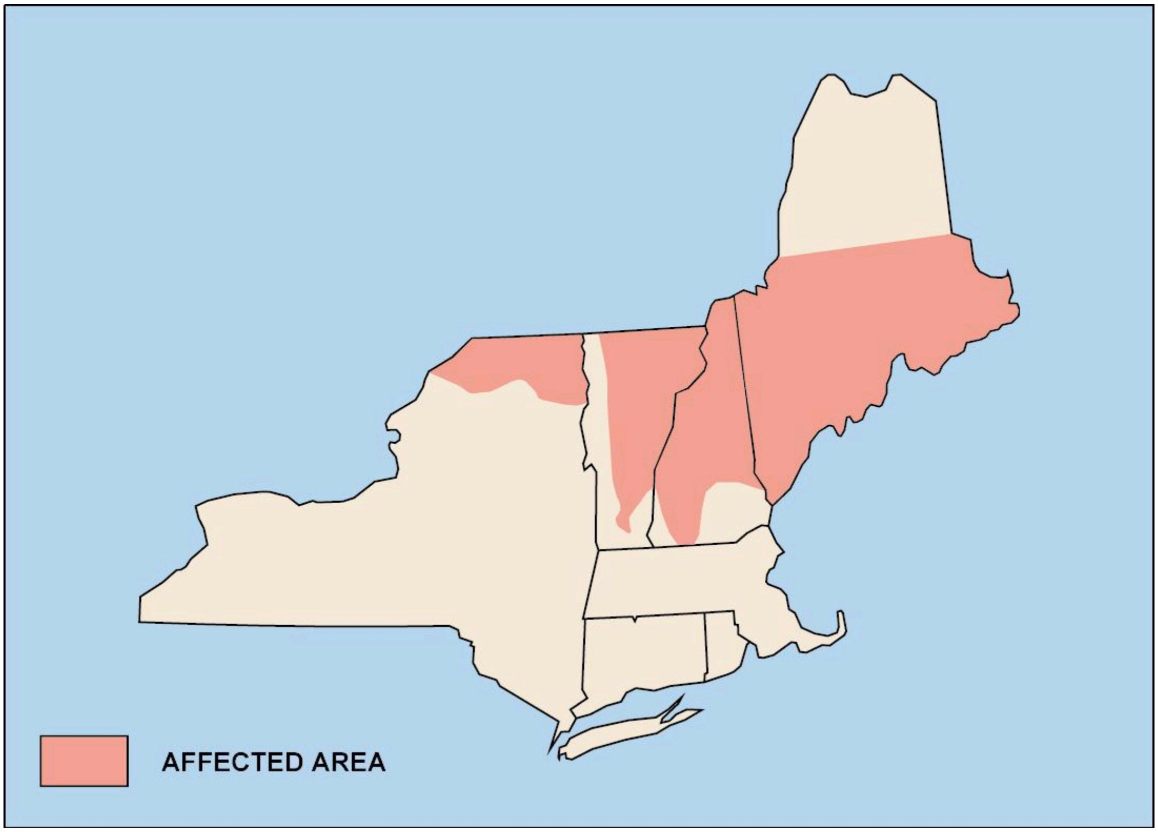

Figure 4. Areas of New York and New England affected by the ice storm of 1998 (Source: Miller-Weeks and others 1999)

1999—"Straight line" winds in excess of 90 MPH, caused by a severe thunderstorm, resulted in extensive breakage and wind-throw of pine, aspen, birch, and other species more than 300,000 acres of forest in the Superior National Forest and adjoining State and private lands in northern Minnesota. Much of the damage occurred in the Boundary Waters Canoe Area, a popular wilderness area known worldwide for canoeing and other forms of outdoor recreation.

2000—Two major ice storms occurred in portions of northeast Texas, southeast Oklahoma, and much of central and southern Arkansas. The storms occurred on December 14 and 24. One to 2 inches of ice accumulation bowed, broke, and completely uprooted trees. Hundreds of thousands of acres of young pine plantations were completely destroyed and will require replanting. Texas authorities estimated a loss of $46 million in timber values in four northeastern counties. In Oklahoma, widespread damage to trees was reported across 6 million acres in 39 counties (Forest Service 2001).

FIRE

HISTORICAL PERSPECTIVE

Fire is a major influence on the dynamics of most forest ecosystems in the United States. The frequency and occurrence of fires has been influenced by humans living in or near these forests for about 12,000 years. Brose and others (2001) present a graphic representation of fire history of Appalachian oak forests and show three profoundly different fire regimes (Figure 5). This model generally applies to all of the fire dependent forest ecosystems in the United States. During the first the period prior to European settlement, indigenous tribes used fire to prepare sites for planting; drive game; encourage fruit and berry production; create open forests, prairies, and savannas desired for early successional wildlife and to maintain a network of trails to facilitate travel. These fires were periodic, low intensity surface fires.

Following European settlement, early settlers adopted the burning practices of the indigenous people. However the low intensity fire regime was replaced by high-intensity, stand replacement fires caused by the onset of extensive logging and mining activities and the introduction of steampower for transportation and processing of raw materials. This resulted in fires of increased size and intensity, often burning over vast areas. The massive wildfires of the late 1800s and early 1900s contributed to a nationwide movement that identified fire as an undesirable, destructive force that must be controlled.

Following the massive wildfires of 1910 in the Northern Rocky Mountains, fire protection improved and eventually reduced destructive wildfires by more than 90 percent: from 20–50 million acres per year to 2–5 million acres (Frederick and Sedjo 1991, Powell and others 1994) (Table 3, Figure 6). This third phase resulted in virtual exclusion of fire from many ecosystems, causing significant changes in the character of the vegetation and fuel conditions. These forests are now susceptible to intense, stand replacement fires. Foresters and ecologists have begun to recognize the role of fire in these ecosystems and are re-introducing prescribed fire as a vegetation management tool.

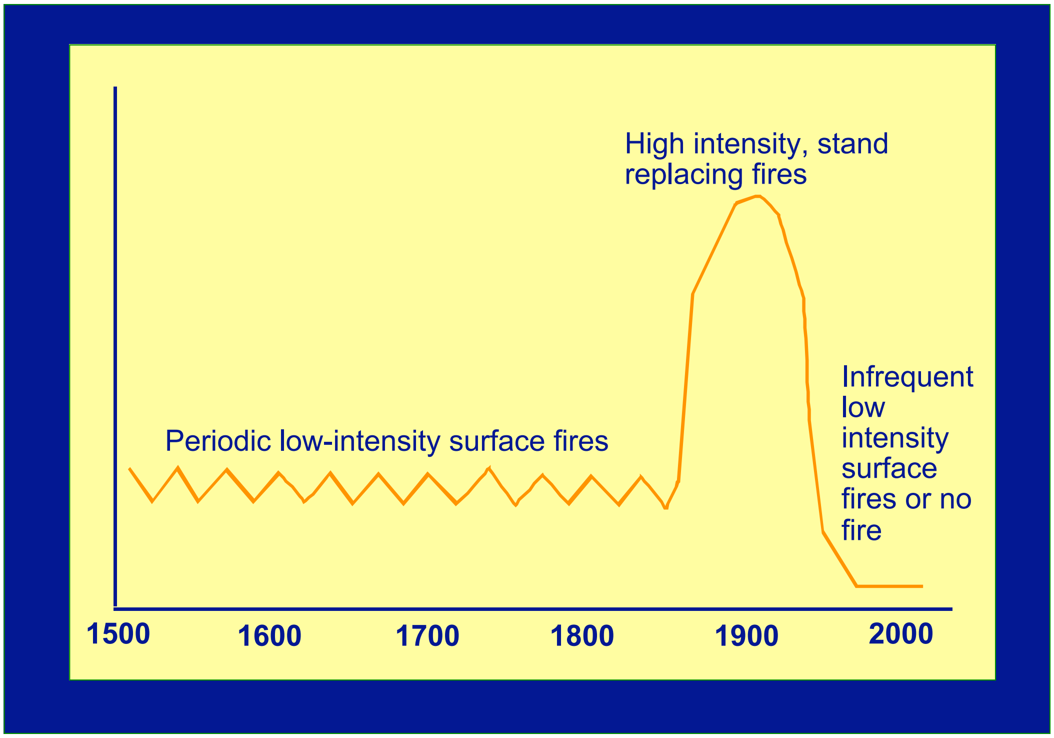

Figure 5. Conceptual model of the changes in fires regimes for the mixed-oak forests of the Appalachian Mountains since 1500. This model applies to most of North America's fire dependent forest ecosystems (Redrawn from Brose and others 2001)

Table 3. Average Number of Fires and Acres Burned Per Decade, 1919–1999*

Dates	Average Number of Fires	Average Acres Burned
1919–1929	97,599	26,004,567
1930–1939	167,277	39,143,195
1940–1949	162,050	22,919,898
1950–1959	125,948	9,415,796
1960–1969	119,772	4,571,255
1970–1979	155,112	3,194,421
1980–1989	163,329	4,236,229
1990–1999	106,306	3,647,597

* Source: NIFC, Boise, ID

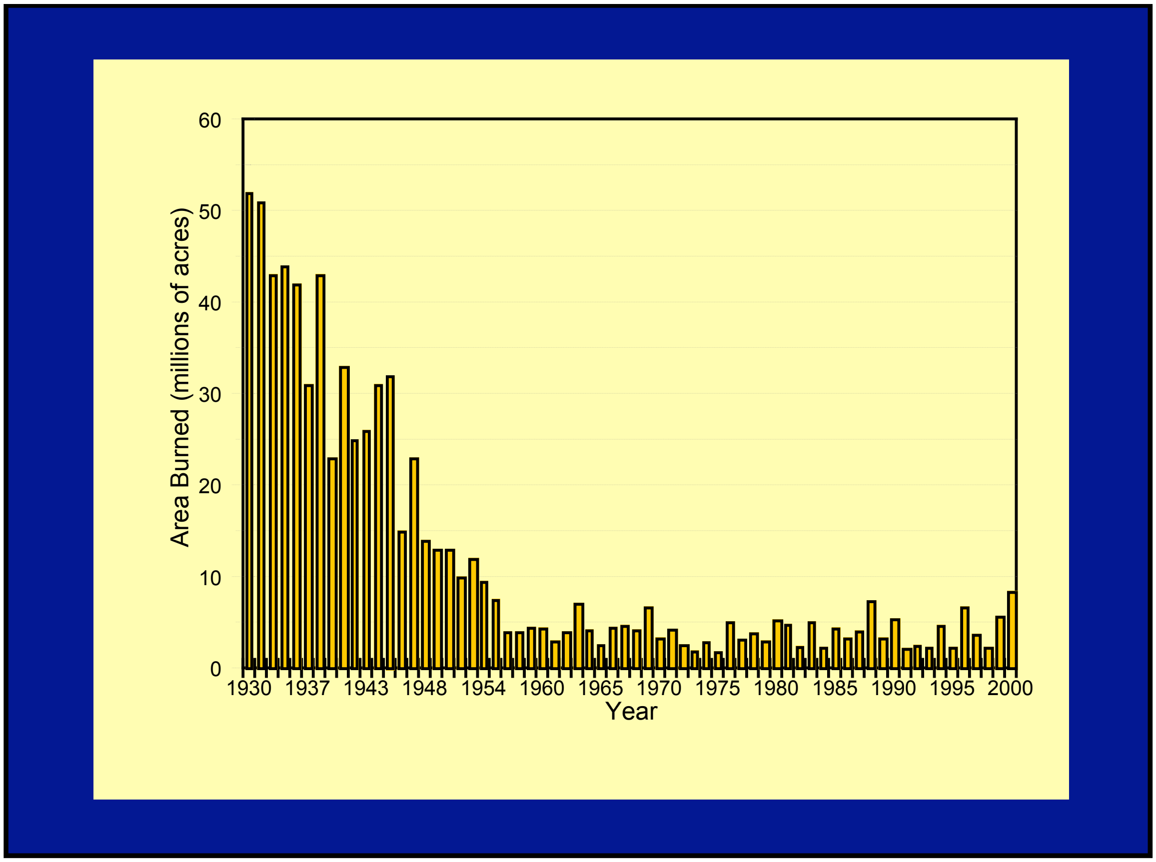

Figure 6. Land area burned by wildfires in the United States between 1930 and 2000 Source: Powell and others 1994 (updated).

WILDFIRE OCCURRENCE

Annual fire statistics available from NIFC and Forest Service reflect the general pattern of the fire model shown in Figure 5 (Figs. 6 and 7, Tables 4-5). Between 1930 and 1950, in excess of 10 million acres were burned by wildfires annually. Most of the area burned during this period was in the Southeastern United States (South RPA Region) and were primarily incendiary fires.

Since 1960, between 2 and 5 million acres were burned annually by wildfires. In recent years, the average area burned has increased, especially in the Rocky Mountain and Pacific RPA Regions. A peak fire year occurred in 1988 when 7.4 million acres burned. This was the year of the extensive fires in the Greater Yellowstone Basin. **The range of *recent* variation (since 1960), in terms of area burned, was exceeded in 2000 when wildfires burned more than 8.4 million acres**. This was the largest area burned in more than 40 years.

Table 4. Area burned by wildfires in the United States by RPA Region – 1938 to 1978 *
(Thousands of acres)

Year	RPA Region				
	North	South	Rocky Mountain	Pacific	U.S. Total
1938	1,234	31,898	33	650	33,816
1939	1,343	27,786	183	1,136	30,448
1940	No data available				25,848
1941	2,739	23,133	116	414	26,402
1942	2,485	28,531	147	574	31,737
1943	1,643	29,395	760	534	32,332
1944	1,407	14,059	290	903	16,659
1945	1,139	15,341	290	1,027	17,797
1946	1,950	18,204	223	1,271	21,648
1947	1,592	21,006	508	1,739	24,845
1948	1,620	14,447	233	256	16,566
1949	1,411	13,747	186	341	15,685
1950	1,099	13,675	223	2,581	17,578
1951	1,377	8,702	201	513	10,793
1952	3,963	9,951	84	249	14,247
1953	2,575	6,951	137	759	10,422
1954	1,334	7,217	102	1,234	9,887
1955	808	6,781	67	444	8,100
1956	943	5,290	172	621	7,026
1957	769	2,218	201	4,938	8,126
1958	557	2,159	265	611	3,592
1959	814	2,227	160	952	4,153
1960	911	2,738	339	488	4,476
1961	628	1,804	207	395	3,034
1962	819	2,935	125	198	4,077
1963	1,542	4,329	1,156	91	7,118
1964	1,087	1,431	1,363	314	4,195
1965	651	1,549	299	150	2,649
1966	644	1,972	995	961	4,572
1967	587	3,043	774	251	4,655
1968	257	977	355	1,248	2,837
1969	239	915	282	4,258	5,694
1970	162	973	285	840	2,260
1971	246	1,372	765	1,163	3,546
1972	112	590	492	1,088	2,282
1973	236	507	551	469	1,763
1974	142	1,024	705	842	2,713
1975	155	810	371	338	1,674
1976	516	1,170	619	330	2,635
1977	531	1,279	439	648	2,897
1978	108	1,106	304	236	1,754

Sources: NIFC 2000, Forest Service 1939–1978

Table 5. Area burned by wildfires in the United States – 1979 to 2000

Year	Area burned (acres)	Year	Area burned (acres)
1979	2,986,826	1990	5,452,874
1980	5,260,825	1991	2,237,714
1981	4,814,206	1992	2,457,665
1982	2,382,036	1993	2,310,420
1983	5,080,553	1994	4,724,014
1984	2,266,134	1995	2,315,730
1985	4,434,748	1996	6,701,390
1986	3,308,133	1997	3,672,616
1987	4,152,575	1998	2,329,709
1988	7,398,889	1999	5,661,976
1989	3,261,732	2000	8,422,237

Source: NIFC 2000

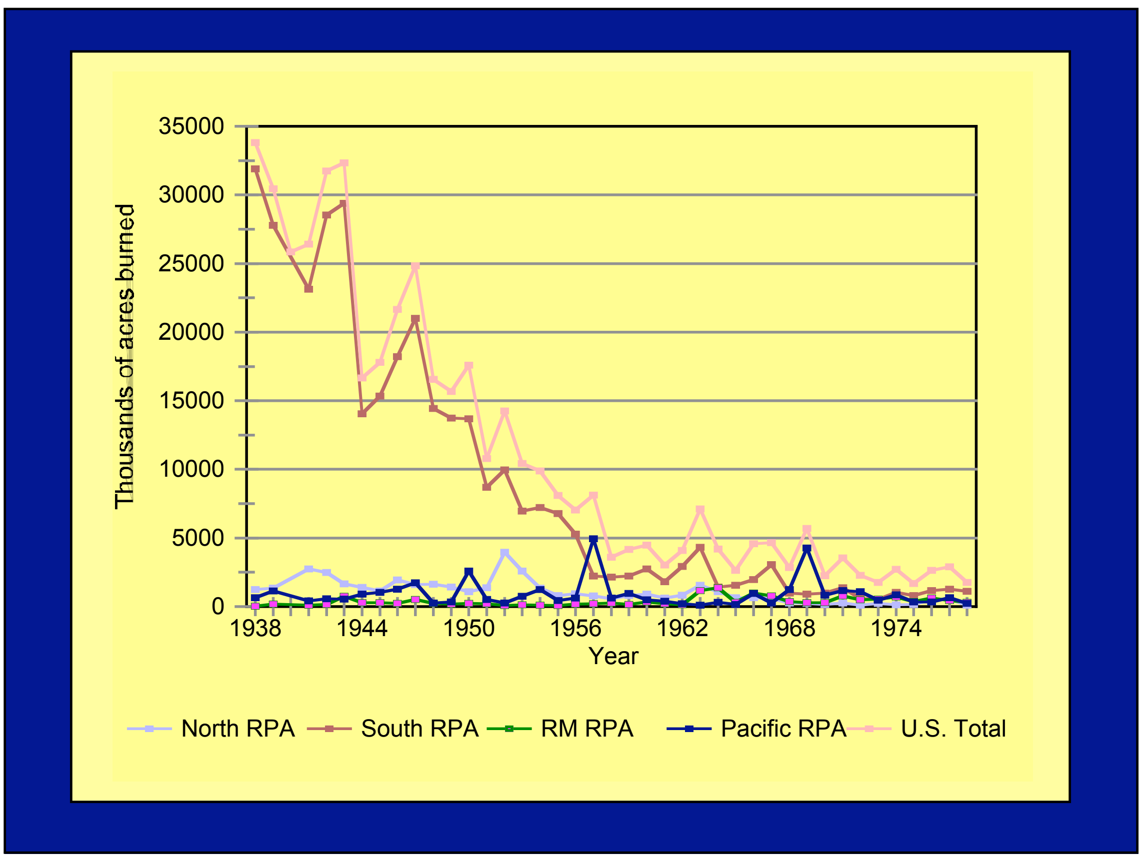

Figure 7. Area burned by wildfires by RPA region between 1938 and 1978

The 1997–98 ENSO, followed by an equally severe La Niña in 1998–1999 and a neutral period in 2000 that was drier than normal, set the stage for wildfires that were beyond the range of *recent* variation. For example, the fire season of 1998 was the most devastating in Florida's recent history. Approximately 50,000 acres burned in the forests of 18 northeastern Florida counties and resulted in economic impacts of at least $600 million (Butry and others 2001). High fire danger conditions occurred following the weakening of the 1999–2000 La Niña event, creating a set of conditions highly favorable for wildfires: hot dry weather, wind, low relative humidity, a source of ignition in the form of dry lightning storms, and the absence of seasonal monsoons in the Southwest. The season began in late February with two 40,000-acre fires in New Mexico. By midsummer 9 out of 11 geographic areas in the United States had active wildfires (NIFC 2000).

The fires of 2000 have also created large areas where conditions are favorable for a buildup of bark beetle populations in fire-damaged trees, especially Douglas-fir beetle and spruce beetle. These populations could move into unburned stands and cause additional tree mortality.

On a regional scale, metric data are available for area burned in 11 Western States (Arizona, California, Colorado, Idaho, Montana, Nevada, New Mexico, Oregon, Utah, Washington, and Wyoming) for an 84-year period between 1916 and 2000. These data show a number of years between 1916 and 1944 with an excess of 1 million acres burned per year. Beginning in the late 1940s, the area of forest burned annually by wildfire in these States decreased and was well below 1 million acres per year until 1979, when 1.36 million acres burned. The area of forest consumed by wildfire annually in the West has increased since 1979 with 17 years during which more than 1 million acres were burned. In 1988, 3.6 million acres burned and in 1996, 1998, and 2000, more than 4 million acres of western forests were burned (Figure 8). **Therefore, based on the approach used in this analysis, the area burned in 1996, 1998, and 2000 is considered to be beyond the range of *recent* variation.** The increased area and intensity of wildfire in the West (most wildfires that have occurred in recent years are high-intensity stand replacement fires) is believed to be the result of fire exclusion in many forests, resulting in changes in species composition, stocking levels, and levels of fuels. Moreover, the West has experienced warmer drier weather during the past two decades.

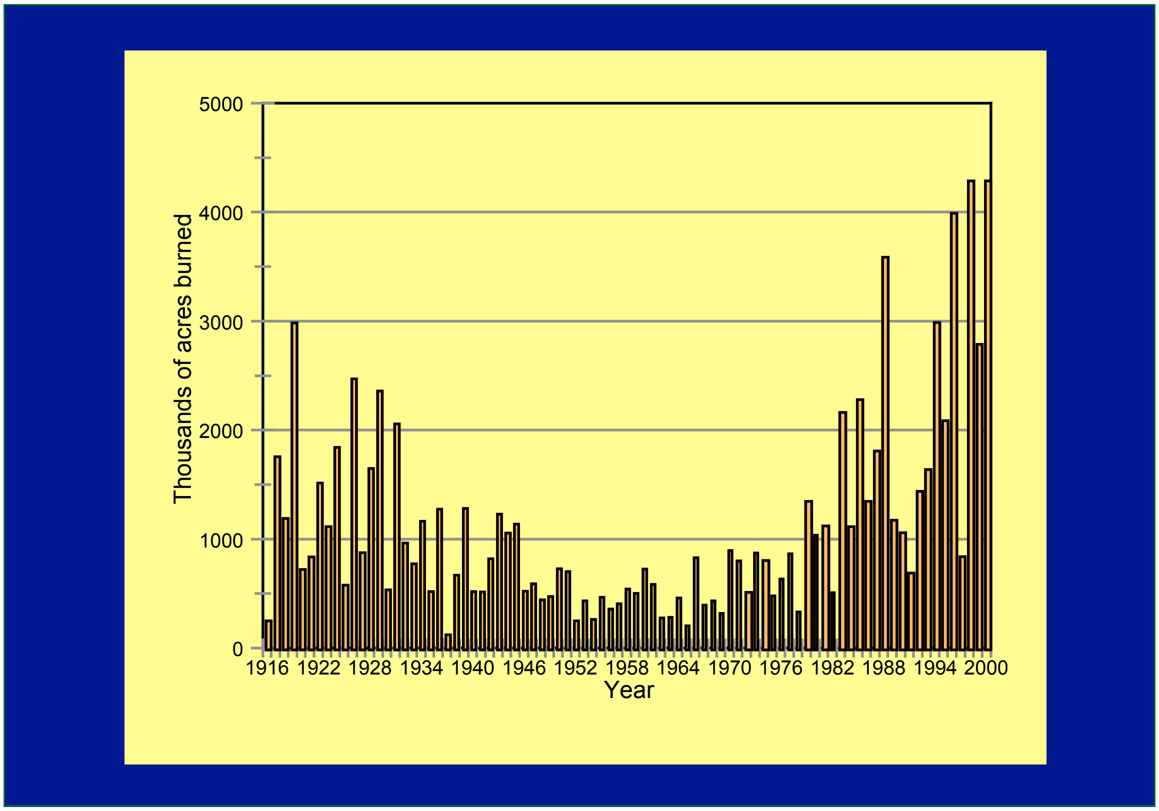

Figure 8. Area burned in 11 Western States between 1916 and 2000 (Source: J. Menakis, Fire Sciences Laboratory, Forest Service, Rocky Mountain Research Station, Missoula, MT)

INSECTS

NATIVE SPECIES

Native insects are major disturbance factors in U.S. forests. Many insects are dynamic, periodically reaching outbreak levels. In between outbreaks, there are periods of low population levels during which they may be difficult to find. Outbreaks may be of relatively short duration (e.g., 2 to 3 years) followed by a sudden population collapse or they can occur for periods of up to a decade. Factors that predispose forests to insect outbreaks include tree stress due to drought, poor growing conditions, overstocking and senescence.

Some insects are of local importance whereas others are of regional and national importance. Both anecdotal historic data and metric data exist for seven native species of regional or national importance, from which comparisons can be made regarding their status during the current analysis period (1996–2001). These include:

1. Southern pine beetle, *Dendroctonus frontalis*

2. Mountain pine beetle, *Dendroctonus ponderosae*

3. Douglas-fir beetle, *Dendroctonus pseudotsugae*

4. Spruce beetle, *Dendroctonus rufipennis*

5. Spruce budworm, *Choristoneura fumiferana*

6. Western spruce budworm, *Choristoneura occidentalis*

7. Douglas-fir tussock moth, *Orgyia pseudotsugata*

SOUTHERN PINE BEETLE

The southern pine beetle, *Dendroctonus frontalis*, is considered to be the most destructive insect pest of pines in the Southeastern United States (South RPA region), Mexico, and Central America. This insect attacks all species of pines within its range but in the Southeastern United States prefers loblolly, shortleaf, Virginia, pond, and pitch pines. It occurs primarily in the loblolly-shortleaf forest type groups. This insect kills trees by constructing breeding galleries in the cambium layer of host trees and introducing blue stain fungi (Thatcher and Barry 1982). Southern pine beetles can complete from three to seven generations per year in the Southeastern United States.

A comprehensive review of southern pine beetle, from the earliest historical records to 1996, is presented by Price and others (1997). The ability of the southern pine beetle to kill pines has been known since the latter part of the 18^{th} century. Accounts of Moravian settlers and others in the Carolinas dating back to 1750 describe the destruction of vast numbers of pines due to the "mischief" of what appeared to be bark beetle attacks.

During the late 1800's, A.D. Hopkins, regarded as the father of American forest entomology, reported on a major outbreak of this insect in West Virginia. In the early 1900s people in east Texas reported taking advantage of beetle infestations to help clear land for pasture (Payne 1979).

Factors favoring the development of southern pine beetle outbreaks are low tree vigor and/or stress, evidenced by a consistent association of this insect with reduced radial growth and certain stand disturbances. Reduced radial growth is associated with overstocked stands and the presence of certain soil conditions. Lightning strikes frequently serve as focal points for southern pine beetle infestations (Hicks 1979).

In any one area, southern pine beetle outbreaks tend to be short lived, generally of 2 to 3 years duration, interspersed with periods when the insect is hard to find. However, a series of maps showing the general location of southern pine beetle outbreaks from 1960 to 1996 indicate that outbreaks have been present somewhere in the Southeastern United States each of those 36 years (Price and others, 1997). Data on the status of southern pine beetle outbreaks from 1979 to the present reported in Forest Service Forest Insect and Disease Conditions Reports on a statewide basis also verify the occurrence of outbreaks somewhere in the region each year (Table 6). In 1986, more than 26 million acres of southern pine forests had southern pine beetle populations of at least one multiple tree spot per 1000 acres of host type (50.2 percent of the loblolly-shortleaf pine type groups). This 1986 event and another severe region-wide outbreak that occurred in 1995 when 21,675.9 million acres of pine forests were infested (41.2 percent of the loblolly-shortleaf pine type groups) are record high levels for southern pine beetle (since 1979) **and probably beyond the range of historic variation** (Table 7).

Levels of southern pine beetle activity were relatively low during 1996–2000 in comparison 1979–1995 (Table 7) although severe outbreaks occurred in Alabama, Louisiana, North Carolina, South Carolina, and Tennessee. In 2000, major outbreaks developed in the Southern Appalachian Mountains, including a major outbreak in southwestern Virginia and in southeastern Kentucky. The outbreak in Kentucky was so vigorous that trees other than yellow pines were attacked. Eastern white pine was frequently attacked and killed, and attacks occurred in Norway spruce and eastern hemlock. An outbreak in 33 counties of Florida represents a recent high for a State that has not experienced high levels of southern pine beetle activity in the past. **Both these outbreaks are considered to be outside the range of *recent* variation and probably *historic* variation.** Southern pine beetle activity also occurred in Arizona on more than 11,000 acres in the southeastern corner of the State. This is the largest outbreak ever recorded in Arizona. **Because of the magnitude of this outbreak and its location in the Southwest (southern pine beetle outbreaks in the United States typically occur in the Southeast), this outbreak is considered outside of the range of *recent* variation and probably *historic* variation.** (Figure 9, Forest Service 2001). These outbreaks continued into 2001.

Table 6. Area Infested by Southern Pine Beetle, *Dendroctonus frontalis* in the Southeastern United States, 1979–2000*

(Thousands of Acres)

State	Year										
	1979	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989
AL	5156.6	2227.8	0.0	1388.2	1880.1	No data available	3926.0	7.529.3	6034.0	4762.4	724.0
AR	0.0	0.0	0.0	830.0	2817.6		648.5	1.372.9	774.0	0.0	0.0
GA	4574.8	2498.5	22.0	720.5	774.8		1007.0	1.839.1	183.0	1.057.4	850.0
FL	0.0	0.0	0.0	0.0	0.0		0.0	0.0	0.0	0.0	0.0
KY	0.0	0.0	0.0	0.0	0.0		0.0	0.0	0.0	0.0	0.0
LA	0.0	0.0	0.0	123.6	248.4		3094.8	6431.8	376.0	17.0	17.0
MS	1324.2	2408.4	0.0	1106.3	452.6		270.2	2.383.2	1626.0	715.0	319.0
NC	386.3	1539.0	236.0	0.0	81.8		0.0	343.5	555.0	497.0	342.0
OK	0.0	0.0	0.0	2924.2	0.0		0.0	0.0	1.0	0.0	0.0
SC	3389.7	3367.0	606.4	0.0	3190.0		2066.1	2904.2	2.904	609.1	753.0
TN	134.4	84.0	0.0	0.0	0.0		0.0	0.0	440.0	278.1	427.0
TX	0.0	0.0	0.0	234.7	1220.4		4433.8	3409.8	475.0	0.0	1901.0
VA	0.0	0.0	0.0	0.0	740.3		0.0	175.1	428.0	0.0	0.0
Total	**14966.0**	**12124.7**	**864.4**	**7327.5**	**11406.0**		**15446.9**	**26388.9**	**13796.0**	**7936.1**	**5333.0**

State	Year										
	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000
AL	0.0	3.937	5815.7	2753.4	2951.4	6552.4	1177.9	4535.5	5241.3	5002.0	6936.1
AR	0.0	0.0	55.8	649.1	429.6	2112.9	1420.6	0.0	0.0	0.0	0.0
AZ	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	11.6
GA	0.0	346.5	871.0	587.3	315.4	1326.0	101.3	312.9	65.0	171.0	321.3
KY	0.0	0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	220.6
LA	0.0	1197.6	3112.4	2291.9	0.0	2908.8	165.3	110.0	228.0	0.0	0.0
MS	0.0	1278.4	406.1	331.5	689.6	2714.3	1150.9	892.1	73.0	0.0	210.6
NC	111.4	40.1	334.3	569.6	47.9	2755.6	747.1	702.3	234.0	252.0	437.9
OK	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
SC	2.320.6	2.413.6	469.2	366.4	332.8	2542.9	2496.6	843.0	944.0	8.7	1218.3
TN	0.0	0.0	45.9	173.0	148.6	0.0	41.2	30.3	35.0	685.0	1441.0
TX	1800.0	1495.9	2663.3	1106.8	238.3	0.0	0.0	649.6	0.0	0.0	0.0
VA	0	35.1	533.6	1584.6	0.0	27.0	0.0	0.0	0.0	0.0	268.0
Total	**4232.0**	**10744.2**	**14307.2**	**10413.6**	**5250.7**	**21675.9**	**7300.9**	**8476.8**	**6820.3**	**6158.7**	**12132.4**

Source: USDA Forest Insect and Disease Conditions Reports – 1979 to 2000

* Acres of outbreak are defined acres of host type having one or more multi-tree spots per 1,000 acres

Table 7. Status of Southern Pine Beetle Outbreaks in the Southeastern United States, 1996–2000 in Comparison With Recent Highs

Year	Area infested (thousands of acres)*	Percent of host type infested**
Current Analysis Period		
1996	7300.9	13.8
1997	8476.9	16.1
1998	6820.3	13.0
1999	6158.7	11.7
2000	12132.4	23.1
Historic Highs		
1986	26,388.9	50.2
1995	21,675.9	41.2

* Area with at least one multiple tree southern pine beetle spot per 1000 acres of host type.

** Based on an estimate of 52,530,000 acres of loblolly-shortleaf pine type (Source: http://fia.fs.fed.us/library/final_rpa_tables.pdf)

MOUNTAIN PINE BEETLE

Mountain pine beetle, *Dendroctonus ponderosae*, is a native bark beetle affecting ponderosa, lodgepole, and other pines. This insect is found throughout western pine forests from Canada to Mexico (Pacific Coast and Rocky Mountain RPA Regions) and is considered to be the most serious insect pest of western pines. Mountain pine beetle reaches epidemic levels in mature lodgepole pine forests and mature and overstocked forests of ponderosa pine. Dense ponderosa pine forests, often the result of fire exclusion, are especially susceptible to outbreaks of this insect.

An outbreak of mountain pine beetle from 1894 to 1908 in ponderosa pine forests in the Black Hills of South Dakota first called public attention to the extensive killing of trees by bark beetles in the West. Between 1 and 2 billion board feet of pine were killed during that outbreak. Between 1917 and 1926, about 300 million board feet of pine were killed on the Kaibab Plateau in northern Arizona. A series of outbreaks between 1925 and 1935 in Idaho and Montana killed more than 7 billion board feet of lodgepole pine and vast numbers of whitebark pines. This insect is continuously at epidemic levels somewhere in the West (Furniss and Carolin 1977).

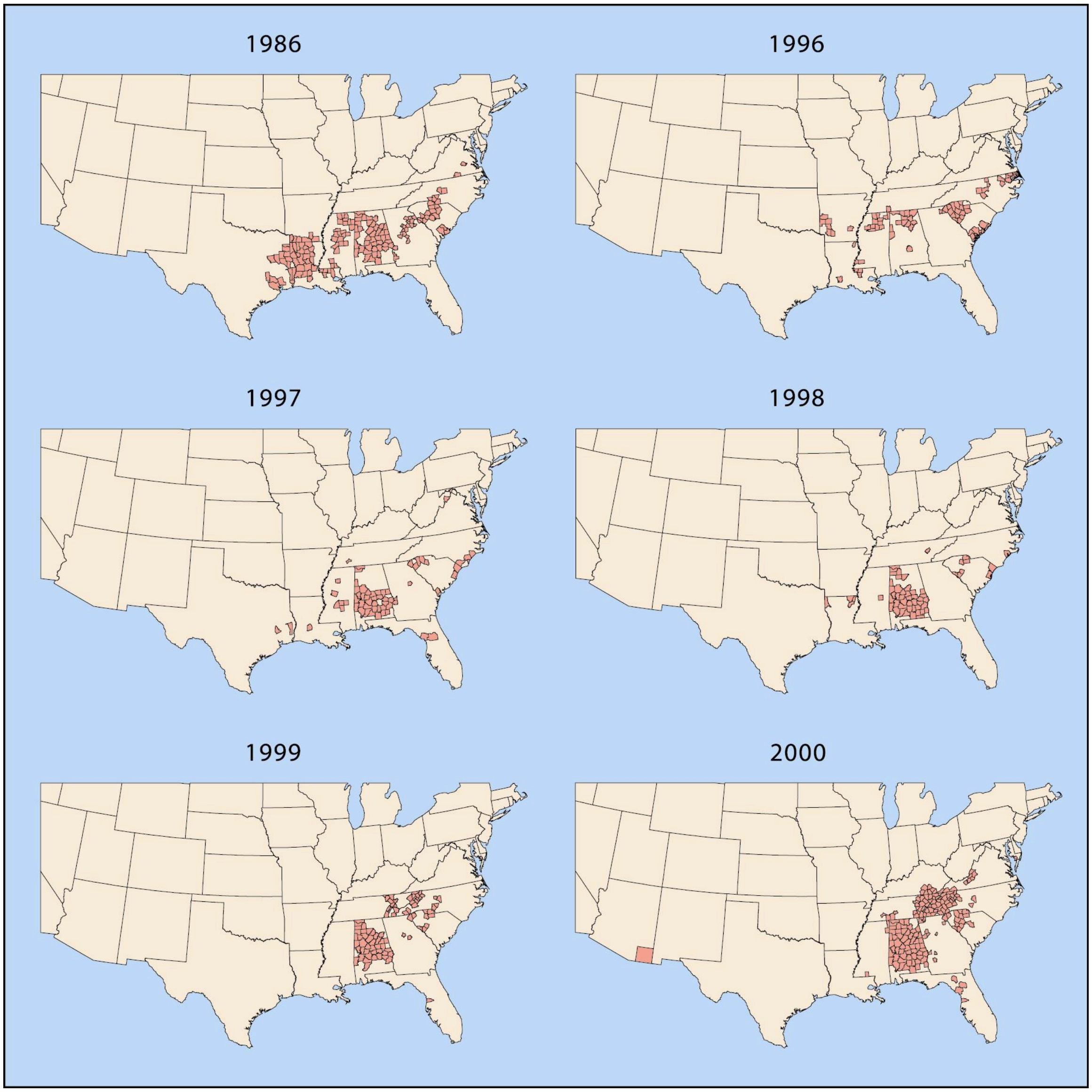

Figure 9. Location of southern pine beetle outbreaks in the Southeastern United States – 1996–2000 in comparison with 1986, the year of the highest recorded level of outbreaks (Sources: Forest Service 1987, 1997, 1998, 1999, 2000, 2001)

Mountain pine beetle outbreaks increased in intensity during the late 1960s with major outbreaks occurring in Colorado, Idaho, Montana, Oregon, Washington, and South Dakota. **Area affected by this insect reached a peak in 1981 with 4.7 million acres affected, a record high that may be beyond the range of *historic* variation (Table 8).** The increase in mountain pine beetle activity is believed to be the result of fire exclusion in western forests. Frequent low intensity fires in ponderosa pine forests maintained open, park-like stands of large trees. Fire exclusion resulted in increases in the numbers of trees per acre, making these forests susceptible to attack. In lodgepole pine forests, exclusion of stand replacement fires at 60–80 year intervals created older forests more susceptible to this insect. Following the 1981 peak (possibly a time when outbreaks *were* outside of the range of historic variation), mountain pine beetle activity declined as the area of suitable host type for this insect was decimated. During the analysis period (1996–2000), less than 1 percent of the susceptible host type was affected compared to 9.5 percent of the susceptible host type during 1981 (Table 9). Since these levels are relatively low compared to the decade of the 1980s, they are considered to be within the range of historic variation when compared to the 1979–1995 metric data. However, major outbreaks still occurred in Colorado and Idaho during the analysis period and this insect will continue to cause severe damage as long as suitable host type in the form of overmature or overstocked pine forests is available.

DOUGLAS-FIR BEETLE

Douglas-fir beetle, *Dendroctonus pseudotsugae*, is the most serious insect pest of Douglas-fir in the Western United States. This insect is capable of building up to epidemic levels in windthrow or trees weakened by drought, insect defoliation, or fire.

In past years, Douglas-fir beetle infestations were expressed by various Forest Service regions as either acres infested or number of trees killed. This inconsistency makes it impossible to develop a meaningful historical database. Some anecdotal records report massive losses to Douglas-fir forests caused by this insect. For example, four outbreaks in western Oregon and Washington between 1950 and 1969 killed 7.4 billion board feet of timber. A 1966 outbreak in California killed 800 million board feet of timber, and an outbreak in northern Idaho killed 109 million board feet of timber between 1970 and 1973 (Furniss and Orr 1978).

Since 1996 infestation levels have been summarized on a statewide basis in terms of acres infested, based on aerial survey data (Table 10). These data show an increasing trend in area of infestation from 1997 to 2000. Almost half of the infested area was in Idaho, where an outbreak developed in trees damaged by an ice storm during the winter of 1996. The proportion of the total area of Douglas-fir type groups infested by this insect varied between 0.1 and 0.8 percent (Table 11). While the levels of Douglas-fir beetle infestation were high in some areas during the analysis period, when compared to existing historical information, they are within the range of historic variation. However, since Douglas-fir beetle is known to infest fire damaged trees, already high populations of this insect in northern Idaho could increase significantly as a result of the extensive fires that occurred in the northern Rocky Mountains in 2000.

Table 8. Area Infested by Mountain Pine Beetle, *Dendroctonus Ponderosae,* in the Western United States, 1979–2000 (Thousands of Acres)

State	Year										
	1979	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989
AZ	97.9	25.2	14.9	3.8	0.3	No data available	0.1	0.1	0.0	0.6	0.9
CA	0.0	0.0	0.0	0.0	10.0		20.0	20.0	20.0	0.0	0.0
CO	383.3	99.0	107.5	183.0	237.0		260.0	159.5	2.5	13.0	12.0
ID	673.1	713.9	709.8	571.3	57.5		27.0	34.7	48.1	42.3	41.6
MT	1419.1	2205.6	2418.1	2142.1	1492.1		933.0	867.0	694.4	546.7	421.5
NV	0.0	0.89	2.3	0.0	0.5		0.0	0.0	0.0	0.0	0.0
NM	76.16	9.3	8.7	2.9	2.0		0.9	2.0	4.8	1.0	1.0
OR	1099.9	1002.9	591.5	702.2	1129.2		1400.0	1600.0	1400.0	1311.4	887.9
SD	300.0	300.0	380.0	5.5	11.0		7.0	4.6	2.3	2.6	2.4
UT	48.6	62.1	148.8	289.9	276.1		477.0	560.4	97.4	12.5	4.5
WA	124.7	84.0	123.8	116.7	146.6		100.0	157.0	158.0	220.3	231.4
WY	175.0	185.0	205.0	200.0	213.0		115.0	44.9	14.7	55.6	11.4
Total	**4397.9**	**4687.9**	**4710.5**	**4217.4**	**3575.2**	**3300.0**	**3340.0**	**3450.2**	**2442.1**	**2206.0**	**1614.6**

State	Year										
	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000
AZ	0.6	0.0	0.0	0.0	0.2	0.2	2.2	10.0	7.4	0.0	0.0
CA	0.0	0.0	0.0	121.0	115.0	58.9	25.1	15.2	26.8	9.7	30.4
CO	9.8	1.5	0.0	0.0	1.2	4.7	12.8	22.2	23.1	71.8	139.5
ID	15.2	22.5	22.4	43.7	7.8	13.9	33.4	54.0	81.6	84.3	122.3
MT	195.2	160.0	65.9	43.4	19.2	31.3	27.6	33.4	39.2	77.4	40.6
NV	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.4	0.8
NM	0.8	1.4	1.2	1.4	2.8	0.4	1.1	0.1	0.0	0.0	0.0
OR	245.1	249.6	303.0	345.6	161.1	234.4	112.6	82.3	65.5	46.2	43.6
SD	6.8	10.0	13.6	13.6	1.4	42.6	2.2	9.4	10.0	19.0	13.9
UT	2.0	1.3	4.1	10.0	18.7	20.9	24.6	20.9	4.5	3.7	2.2
WA	431.7	155.4	125.2	200.0	76.4	205.9	56.7	74.7	30.3	65.0	63.1
WY	28.3	15.4	106.0	2.8	1.6	2.3	1.7	6.7	2.5	6.2	9.5
Total	**935.0**	**617.0**	**641.4**	**781.8**	**405.4**	**575.5**	**300.0**	**328.0**	**290.9**	**384.7**	**465.9**

Table 9. Status of Mountain Pine Beetle Outbreaks in the Western United States – 1996–2000 in Comparison with Recent Historic High

Year	Area infested (thousands of acres)	Percent of Host type infested*
Current Analysis Period		
1996	300.0	0.6
1997	328.0	0.6
1998	290.0	0.6
1999	384.0	0.8
2000	465.9	0.9
Historic High		
1981	4,710.5	9.3

** Based on an estimate of 33,151,000 acres in ponderosa pine type groups and 17,515,000 acres in lodgepole pine type groups for a total susceptible host type of 50.6 million acres. Source: http://fia.fs.fed.us/library/final_rpa_tables.pdf

Table 10. Area of Aerially Detected Douglas-Fir Beetle Outbreaks, 1996–2000 (Thousands of Acres)

State	Year				
	1996	1997	1998	1999	2000
AZ	0.7	4.5	1.2	5.2	1.7
CA	0.0	0.0	T	0.1	T
CO	20.1	3.5	15.8	5.9	13.2
ID	71.2	8.6	65.3	170.4	150.8
MT	4.3	4.0	8.3	38.2	34.7
NV	0	0.0	0.0	0.2	T
NM	T	0.0	0.3	T	0.2
OR	3.3	2.5	16.7	56.5	57.4
UT	5.2	15.0	6.7	7.4	5.8
WA	6.5	5.9	16.9	59.8	78.5
WY	2.1	4.8	2.8	8.2	10.9
Total	**113.4**	**48.7**	**134.0**	**351.9**	**353.2**

T = Trace (> 50 acres)

Source: Aerial survey data base maintained by Forest Service, FHTET

Table 11. Proportion of Douglas-Fir Type Groups Infested by Douglas-Fir Beetle, 1996 to 2000

Year	Area Infested (thousands of acres)	Proportion of host type infested
1996	113.4	0.3
1997	48.7	0.1
1998	134.0	0.3
1999	351.9	0.8
2000	353.2	0.8

Based on a total area of 41,875,000 acres of Douglas-fir type groups Source: http://fia.fs.fed.us/library/final_rpa_tables.pdf

SPRUCE BEETLE

Spruce beetle, *Dendroctonus rufipennis*, is a major bark beetle pest of spruces throughout the United States. Outbreaks typically develop in windthrown trees and beetles emerging from the windthrow attack standing trees (Schmidt and Frye 1977).

The historical occurrence of spruce beetle outbreaks in the New York-New England area was reviewed by Weiss and others (1985). According to early records, spruce beetle was first recognized as a serious pest of spruce in the Northeastern United States in the early 1800s, when several major outbreaks killed millions of board feet or red spruce (Hopkins 1901). These outbreaks continued until the beginning of the 20th century but have since dwindled to smaller outbreaks covering several thousand acres, probably due to a reduction in the area of mature spruce forests.

Spruce beetle has also been a serious pest of spruces, especially Engelmann spruce, in the Rocky Mountain RPA region and in portions of Oregon and Washington (Pacific Coast RPA region. Historical records summarized by Schmid and Frye (1977) report an outbreak on the White River National Forest and the Grand Mesa in the mid 1870s. During this same period an outbreak killed 90 percent of the spruce on more than 13,000 acres in the White Mountains of the Lincoln National Forest, New Mexico. The "most damaging outbreak in recorded history" took place on the White River National Forest in between 1942 and 1948. The amount of tree mortality was so incredible that a precise estimate of the volume of dead spruce is unavailable although 3.8 billion board feet is commonly quoted (Furniss and Carolin 1977).

Spruce beetle has been a continuing problem in forests of white, Lutz and Sitka spruce in Alaska since the early 1970s. **The outbreak began to increase significantly in 1992 and peaked in 1996 at 1,130,000 acres, which is beyond the range of *recent* variation (Table 12). The outbreak in Alaska is the most widespread spruce beetle outbreak ever recorded and *may represent a departure from historic variation*.** This outbreak has been on a steady decline since 1996 with 86,000 acres infested in 2000. The decline is probably due to the decimation of the susceptible host type by this insect.

Outbreaks were generally low in the West during the period 1996–2000, probably because previous outbreaks have removed much of the susceptible host type. However, populations are building up in windthrow that resulted from the 1997 storm in Colorado and in portions of Wyoming. This outbreak has the potential to cause severe damage to mature Engelmann spruce forests in the affected area in future years. and some spruce beetle damage is occurring on the coast of Maine (Forest Service 1999, 2000). Based on anecdotal historical information available for this insect (Schmid and Frye 1977 Weiss and others 1985), these outbreaks are considered to be within the range of historic variation.

Table 12. Area Infested by Spruce Beetle, *Dendroctonus rufipennis* in Alaska—1979–2000

Year	Area Infested (thousands of acres)	Year	Area Infested (thousands of acres)
1979	333.8	1990	232.4
1980	No data available	1991	No data available
1981	227.5	1992	600.0
1982	477.0	1993	700.0
1983	328.0	1994	641.0
1984	416.0	1995	892.8
1985	256.0	1996	1130.0
1986	386.0	1997	544.3
1987	285.1	1998	316.8
1988	387.0	1999	253.3
1989	117.3	2000	86.0

Source: Forest Service, Forest Insect and Disease Conditions Reports – 1979 to 2000

SPRUCE BUDWORM

Spruce budworm, *Choristoneura fumiferana*, is a defoliator of balsam fir, *Abies balsamea*, and spruces, *Picea* spp. in the boreal and sub-boreal forests of Canada and the United States. Historically, this insect has been a more severe pest in Canada, which has vast areas of suitable host type. In the United States, spruce budworm has periodically been a pest of spruce-fir forests in the northeastern and north central United States (Northern RPA Region), usually as a part of much more extensive outbreaks in adjoining portions of Canada.

Tree ring analysis of old white spruce trees in Eastern Canada suggest that spruce budworm outbreaks have occurred in the boreal forests for at least 200 to 300 years. These studies suggest that outbreaks have occurred more frequently during the 20th century than previously. Regionally, 23 outbreaks took place during the 20th century compared to 9 in the preceding 100 years. Moreover, there is evidence that earlier infestations were restricted to specific regions but in the 20th century, they coalesced and increased in area (Blais 1985).

A major spruce budworm outbreak began in Maine during the mid 1970s and reached an historic high (probably exceeding the range of *historic* variation) of 7.75 million acres in 1978 (Figure 10). This represented 44.5 percent of the spruce-fir type in the Eastern United States. The outbreak resulted in extensive mortality of balsam fir throughout the infested area. Since 1979 populations have declined sharply with aerially visible defoliation occurring primarily in the North Central States. No defoliation has been reported from Maine since 1989 (Table 13).

During the period 1996–2000, the area defoliated by spruce budworm in the Eastern United States ranged between 28,500 and 222,500 acres, (all in Michigan and Minnesota). This represents between 0.2 and 1.2 percent of the 17.6 million acre spruce-fir type in the Eastern United States (Table 14)

An outbreak of spruce budworm began in Alaska in 1991, peaked in 1997 at 384,000 acres and began to decline. By 2000 there was no aerially visible defoliation in the outbreak area (Table 15). The occurrence of a *C. fumiferana* outbreak in Alaska is considered unusual. Previous reports of spruce budworm outbreaks in Alaska have been attributed to a closely related species, *C. orea.* A published distribution map of spruce and fir feeding species of *Choristoneura* indicates that the range of *C. fumiferana* does not extend into Alaska (Harvey 1985). However, other reports indicate that *C. fumiferana* has been collected in Alaska in conjunction with the *C. orea* outbreak. This outbreak and a concurrent outbreak of larch sawfly, *Pristiphora erichsonii,* were the most extensive defoliator outbreaks recorded in Alaska in more than 30 years. The sparse historical record of insect outbreaks in Alaska does not describe past occurrences of extensive defoliation. Therefore it is difficult to determine if these outbreaks are "beyond the range of historic variation."[4]

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[4] Personal communication, E. Holsten, Forest Service, R-10, Anchorage, AK and R. Werner (retired), Corvalis, OR, formerly Forest Service PNW Research Station, Fairbanks, AK

Table 13. Aerially Detected Spruce Budworm Defoliation in the Eastern United States, 1979–2000 (Thousands of Acres)

State	Year										
	1979	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989
ME	5900.0	5000.0	4000.0	3800.0	6000.0	5500.0	4800.0	600.0	250.0	65.0	4.8
MI	258.8	859.5	161.0	129.1	145.9	192.4	93.8	1.6	430.0	0.0	0.0
MN	150.0	103.0	110.0	126.7	127.00	361.6	307.3	440.0	0.0	200.0	140.0
NH	70.0	90.0	42.0	39.0	5.8	930.0	0.0	0.0	0.0	0.0	0.0
NY	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0
PA	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
VT	101.9	110.7	96.5	153.9	178.1	0.0	0.0	0.0	0.0	0.0	0.0
WI	141.3	439.0	84.0	0.0	20.9	22.050	15.0	0.0	0.0	0.0	0.0
Total	**6622.1**	**6602.2**	**4493.5**	**4248.7**	**6477.8**	**6076.9**	**5216.4**	**1041.6**	**680.0**	**265.0**	**144.8**

State	Year										
	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000
ME	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
MI	2.5	0.0	0.0	0.0	6.8	51.2	12.9	61.6	33.0	0.0	0.0
MN	198.0	108.0	126.0	116.0	770.5	505.0	207.6	276.2	256.4	70.0	28.5
NH	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
NY	0.0	0.0	0.0	0.0	0.1	0.4	0.0	0.0	0.0	0.0	0.0
PA	0.0	0.0	0.0	0.0	0.0	0.0	2.0	0.0	0.0	0.0	0.0
VT	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
WI	0.0	0.0	0.0	0.0	1.0	0.0	0.0	9.6	0.0	0.0	0.0
Total	**200.5**	**108.0**	**126.0**	**116.0**	**778.4**	**569.1**	**222.5**	**347.4**	**305.5**	**70.0**	**28.5**

Table 14. Status of Spruce Budworm Outbreaks in the Eastern United States – 1996–2000 in Comparison with Recent High

Year	Area infested (thousands of acres)	Percent of host type infested*
Current Analysis Period		
1996	222.5	2.8
1997	347.4	4.2
1998	305.5	3.9
1999	70.0	0.4
2000	28.5	0.2
Historical High		
1981	7,750,000	43.9

** Based on an estimate of 17,640,000 acres of spruce-fir e type groups in the Eastern United States. Source: http://fia.fs.fed.us/library/final_rpa_tables.pdf

Table 15. Aerially Detected Spruce Budworm Defoliation in Alaska, 1991–2000 (Thousands of Acres)*

State	Year									
	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000
AK	25.0	160.0	330.0	232.5	279.1	235.9	384.0	87.9	0.7	0.0

* Source – Forest Service, Forest Insect and Disease Conditions Reports, 1991–2000.

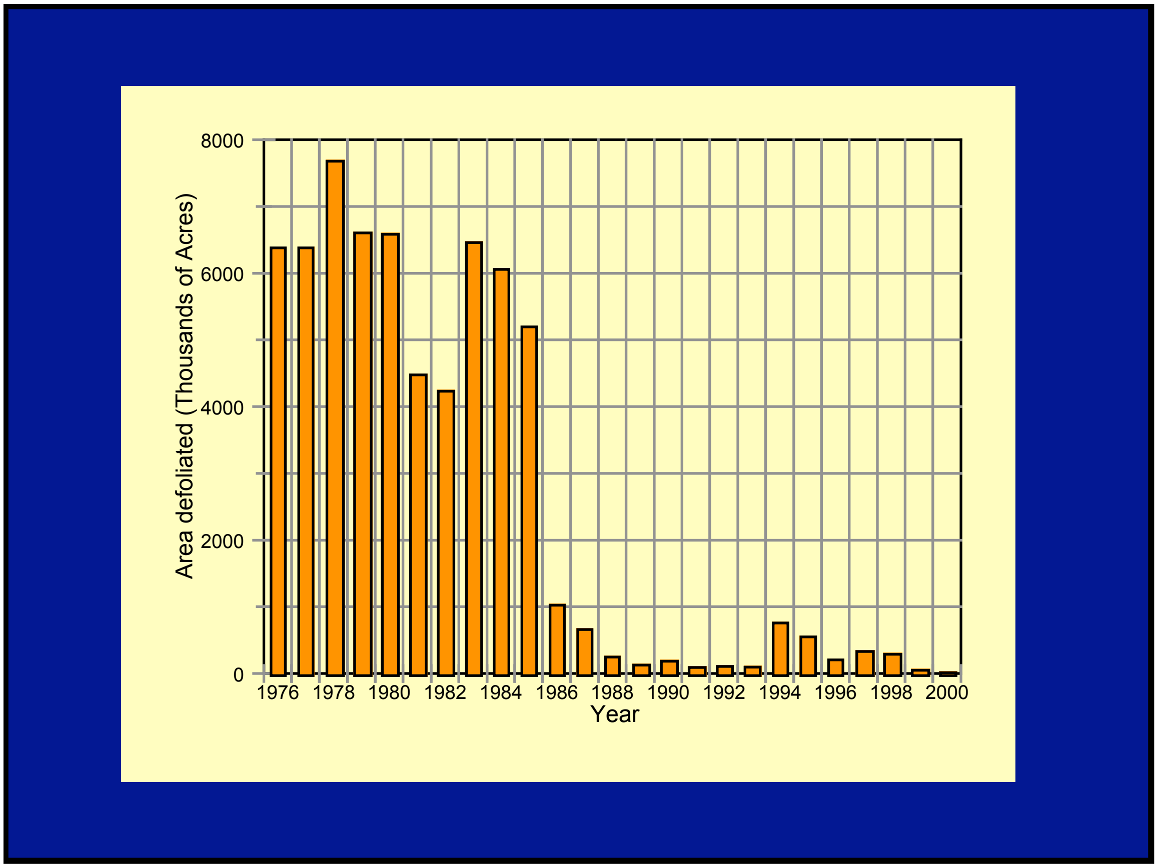

Figure 10. Area defoliated by spruce budworm in the Eastern United States, 1976–1999

Source: Forest Service 2000.

WESTERN SPRUCE BUDWORM

Western spruce budworm, *Choristoneura occidentalis* is native to Western North America from Arizona and New Mexico north to Idaho, Montana, Oregon, and Washington (Pacific Coast and Rocky Mountains RPA subregions). Host trees include Douglas-fir, true firs, and to a lesser degree, spruce and larch. This insect is the most serious defoliator of western conifer forests and has defoliated millions of areas causing growth loss, top kill, and mortality of host trees. Trees defoliated for several successive years are weakened and become subject to attack by bark beetles such as fir engraver, *Scolytus ventralis*, and Douglas-fir beetle*, Dendroctonus pseudotsugae*. This insect is generally in outbreak status somewhere in the West.

The first recorded outbreak of western spruce budworm was reported in 1909 on Vancouver Island, British Columbia, Canada. Two outbreaks occurred in Idaho in 1922 (Furniss and Carolin 1977). Most outbreaks that occurred between 1922 and 1946 were small and widely scattered. They subsided quickly and resulted in little or no damage. These infestations interested forest entomologists, but land managers had no means for suppression nor did they consider action necessary (Brookes and others 1987). Aerial survey data from the Pacific Northwest (Oregon and Washington) show two distinct spruce budworm outbreak cycles since 1947. These occurred from 1948 to 1963 and from 1971 to the present. **The second outbreak cycle was much more extensive and in 1986, some 6 million acres were defoliated in these two States, a recent high that**

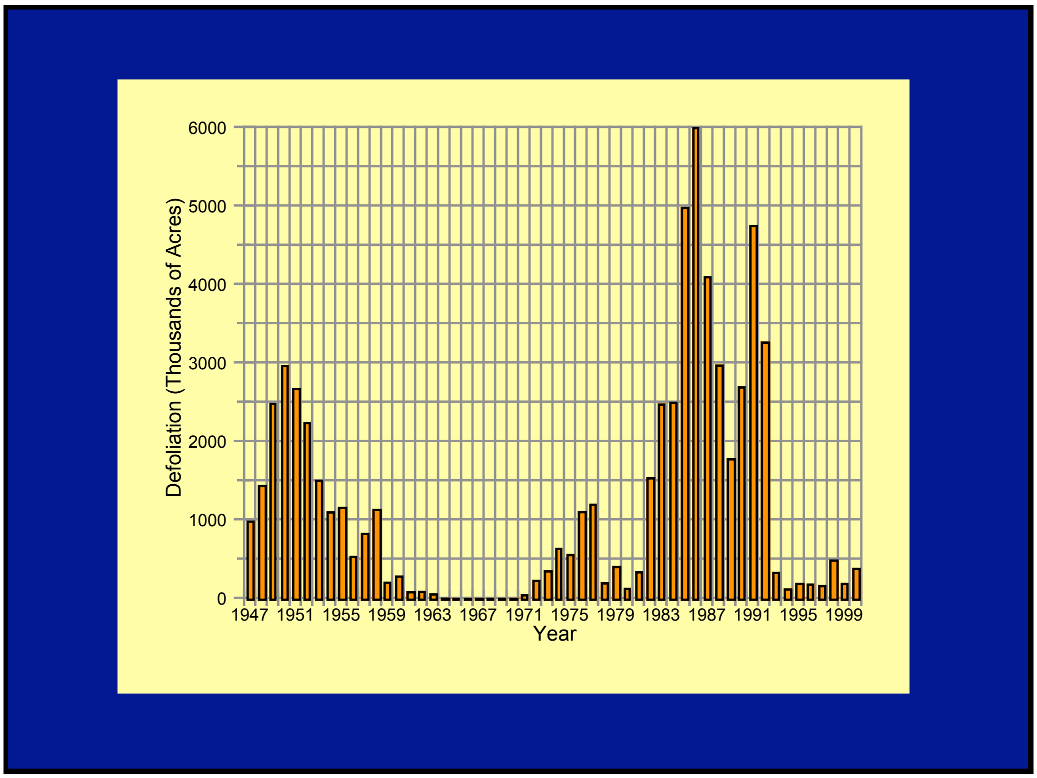

Figure 11. Western spruce budworm defoliation in the Pacific Northwest (Oregon and Washington), 1947–2000 (Sources: Dolph 1980, Forest Service 2000)

is probably beyond the range of *historic* variation (Figure 12). Increased spruce budworm activity in the Pacific Northwest and other parts of the West during this period may be the result of two factors: fire exclusion, which allowed more shade tolerant species such as Douglas-fir and true firs to become established in the understory and removal of overstory ponderosa pines via commercial timber sales. These factors resulted in a conversion of large areas of pine forests to species susceptible to western spruce budworm. The recent decline of western spruce outbreaks westwide is most likely the result of high rates or tree mortality due to defoliation and secondary bark beetle attacks, which reduced the amount of suitable host type.

Tree ring analyses in various parts of the Western United States show patterns of growth reduction suggestive of western spruce budworm outbreaks. For example, in the Southwest, studies by Swetman and Lynch (1989,1993) identified patterns suggestive of nine regional western spruce budworm outbreaks between 1690 and 1989, with a periodicity of 20 to 33 years and a duration within stands of about 11 years. These studies also suggest that while the periodicity of outbreaks has not changed during the 20th century when compared to the past, their spatial and temporal pattern has changed. The most recent outbreaks in northern New Mexico have been more synchronous, more extensive, and more severe than previous outbreaks. Dendrochronological studies conducted in eastern Oregon and the Northern Rockies show similar patterns. In the Blue Mountains of eastern Oregon, major changes in outbreak patterns have occurred since fire suppression and harvesting of nonhost species became widespread in the early 1900s. Outbreaks have become more frequent and severe since the early 1990s, though no change in outbreak duration was found. In the Rocky Mountains, outbreaks have been more severe and in the northern Rockies (Northern Idaho and Montana) they have lasted for longer periods (Anderson and others 1987, Swetnam and others 1995).

National data summarized since 1979 indicate that 1986 was the recorded recent high for western spruce budworm defoliation with more than 13 million acres defoliated that is probably outside of the range of *historic* variation (Table 16). The area of defoliation has declined steadily since 1986 (Table 16) and during the 1996–2000 period, defoliation was at low levels ranging between 332,900 and 843,000 acres. The largest areas of western spruce budworm defoliation during this period were in New Mexico and Washington.

Assuming that the combined area of Douglas-fir and fir-spruce type in the West is 111.6 million acres, the portion of area infested by western spruce budworm during 1996–2001 ranged between 0.29 and 0.76 percent. This is considered to be within the range of recent variation (Figure 12). By comparison, during 1986, the recent high year, 11.9 percent of these forest type groups were affected (Table 17). **However, dendrochronological studies do suggest that outbreaks throughout the Rocky Mountains and in Oregon are now more synchronous, extensive, and severe than those observed prior to 1850 and therefore are considered to be beyond the range of *historic* variation.**

Table 16. Aerially Detected Western Spruce Budworm, *Choristoneura occidentalis*, Defoliation in the Western United States, 1979–2000 (Thousands of Acres)

State	Year										
	1979	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989
AZ	87.1	66.5	120.0	31.5	19.9		102.6	86.5	15.5	5.8	0.7
CA	0.0	0.0	0.0	0.0	0.0		0.0	0.0	0.0	0.0	0.0
CO	930.0	1052.0	1400.0	1800.0	2600.0	No data available	1.567.0	1080.0	833.0	427.0	52.2
ID	1124.1	1244.2	1402.2	2262.6	2399.4		2.631.3	2916.9	898.2	61.0	26.6
MT	2185.1	848.3	894.7	2210.2	2545.3		2.675.0	2497.0	1802.0	2064.0	1191.3
NM	44.5	232.7	358.3	337.0	330.9		529.5	382.9	250.4	477.7	90.1
OR	28.6	2.3	312.6	1530.7	2439.2		4567.4	5600.0	3700.0	2740.4	1416.7
UT	0.0	6.0	5.1	51.4	78.5		87.6	95.6	37.7	0.0	0.0
WA	378.1	126.8	30.1	9.3	37.9		415.3	400.0	400.0	231.6	362.3
WY	235.0	339.6	971.2	445.3	586.2		220.5	164.5	16.3	55.8	0
Total	**5012.4**	**3978.4**	**5494.4**	**8677.9**	**11037.3**	**10633.8**	**12796.2**	**13223.4**	**7.953.1**	**6063.3**	**3139.6**

State	Year										
	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000
AZ	25.6	0.0	11.5	0.0	0.0	7.0	3.0	1.1	10.1	10.2	25.8
CA	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
CO	52.1	509.0	272.2	1.2	0.0	97.0	21.8	0.0	15.8	41.0	20.6
ID	48.0	61.5	89.8	0.9	0.0	0.0	0.0	0.0	0.0	3.6	4.4
MT	1492.4	1595.7	941.3	44.2	2.4	0.0	0.0	0.0	0.0	0.0	0.4
NM	310.5	218.6	9.4	66.4	369.2	183.8	123.9	197.1	310.5	282.6	165.0
OR	2344.3	3724.9	1937.7	87.7	37.4	14.9	1.0	0.0	0.0	0.0	0.9
UT	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	19.9	1.2	16.7
WA	351.0	1027.7	1329.5	243.8	85.4	175.1	183.2	165.9	486.8	189.7	383.7
WY	8.1	33.5	2.5	2.5	1.1	0.0	0.0	0.0	0.0	0.6	0.8
Total	**4632.0**	**7170.9**	**4593.9**	**446.7**	**495.5**	**477.8**	**332.9**	**364.1**	**843.1**	**528.9**	**618.3**

(Source Forest Service Forest Insect and Disease Conditions Reports 1985 – 2000)

Table 17. Status of Western Spruce Budworm Outbreaks in the Western United States – 1996–2000 in Comparison with Recent High

Year	Area infested (thousands of acres)	Percent of host type infested*
Current Analysis Period		
1996	332.9	0.29
1997	364.1	0.33
1998	843.1	0.76
1999	528.1	0.47
2000	618.3	0.55
Recent High Year		
1986	13,223.4	11.9

** Based on an estimate of 41,875,000 acres of Douglas-fir type and 69,686,000 acres of fir-spruce type for a total susceptible host type of 111,561,000 acres. Source: http://fia.fs.fed.us/library/final_rpa_tables.pdf . Note that Douglas-fir type on the western slope of the Cascades is rarely defoliated by western spruce budworm.

DOUGLAS-FIR TUSSOCK MOTH

Douglas-fir tussock moth, *Orgyia pseudotsugata,* is another defoliator of Douglas-fir and true fir throughout the Western United States. This insect is cyclic with outbreaks occurring at 7 to 10 year intervals. Outbreaks last 2 to 3 years during which severe damage can occur to host trees. Unlike budworm, which feeds on just the new growth of

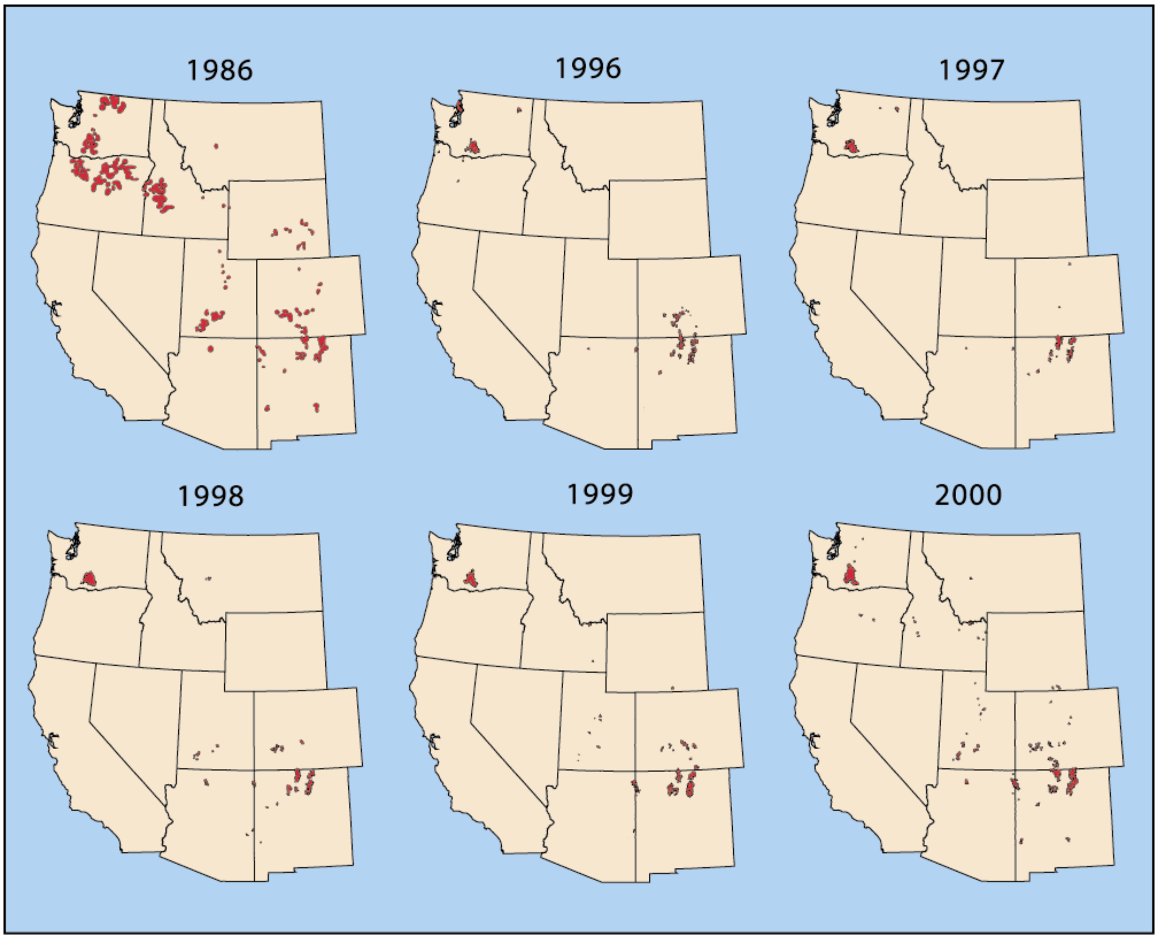

Figure 12. Maps of western spruce budworm defoliation 1996–2000 in comparison with 1986, the year of the historic high. (Source Forest Service Forest Insect and Disease Conditions Reports, 1986, 1996–2000)

host trees, Douglas-fir tussock can strip a tree of all of its foliage in a single season. Outbreaks collapse as dramatically as they erupt, usually due to a nucleopolyhedrosis virus (NPV).

Douglas-fir tussock moth outbreaks have been recorded in various parts of the West since 1925 (Figure 13). Data compiled since 1971 on area defoliated (Table 18) indicate that the historical high year for defoliation by this insect was 1973, when 788,300 acres (1.9 percent of 41,875,000 acres of Douglas-fir type groups) were defoliated in portions of Idaho, Oregon, and Washington. **This recorded high in 1973 probably exceeds the range of *historic* variation.** During the period 1996–2000, Douglas-fir tussock moth outbreaks developed in 1999 and 2000 in portions of California, Idaho, Oregon, and

Washington with 48,864 and 290,961 acres defoliated, representing 0.1 and 0.7 percent of the total area of Douglas-fir type groups, respectively. Based on available records, activity by this insect during the current period is considered to be within the range of *recent* variation.

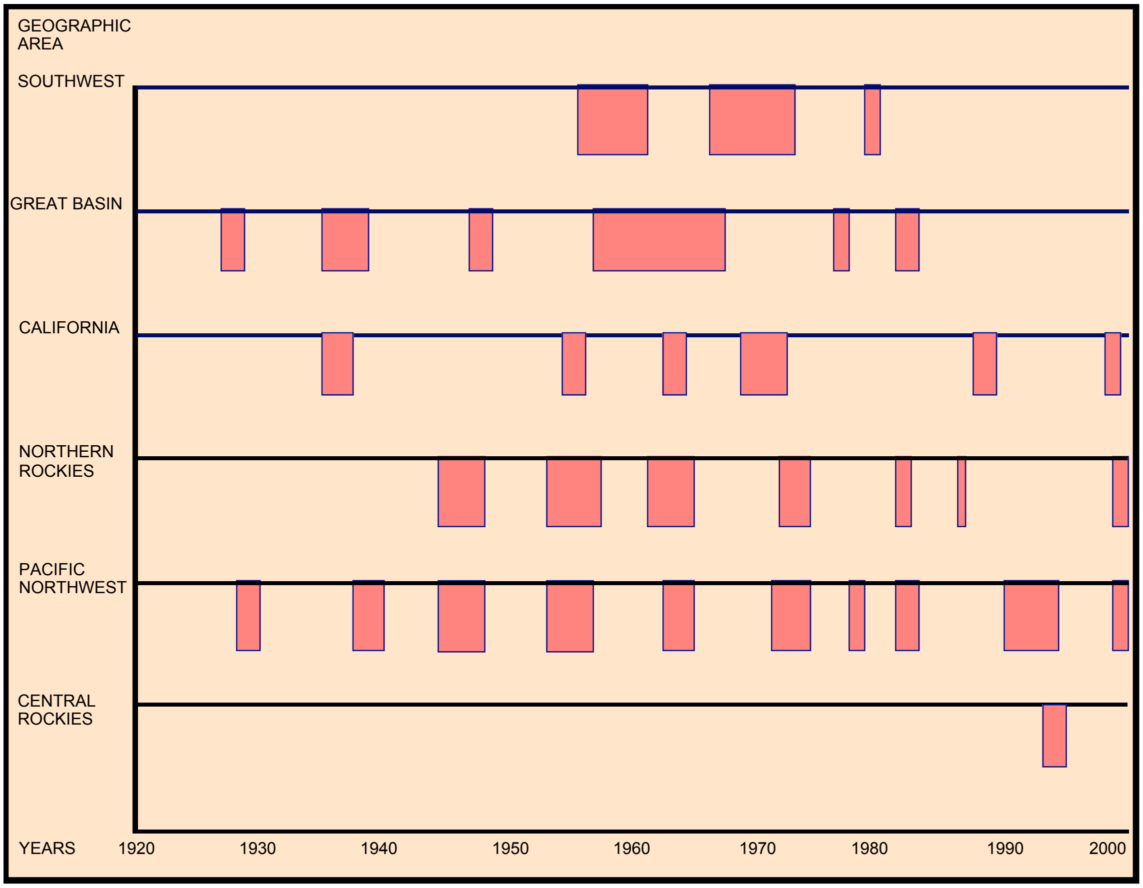

Figure 13. Occurrence of Douglas-fir tussock moth outbreaks from 1927 to 2000 (Redrawn and updated from Forest Service, Insect and Disease Conditions in the United States, 1979–83)

Table 18. Area of Aerially Detected Defoliation by Douglas-Fir Tussock Moth, *Orgyia pseudotsugata*, in the United States, 1972–2000 (Thousands of acres)

State	Year						
	1972	1973	1974	1975	1976	1977	1978
AZ	0.0	0.0	0.0	0.0	0.0	0.0	0.0
CA	X	0.2	0.0	0.0	0.0	0.0	X
CO	0.0	0.0	0.0	0.0	0.0	0.0	0.0
ID	0.0	115.6	116.1	0.0	0.0	0.0	0.0
MT	0.0	0.0	11.2	T	0.0	0.0	0.0
NM	0.0	0.0	0.0	0.0	X	2.5	7.0
OR	196.1*	672.5*	0.0	0.0	0.0	0.0	0.0
UT	0.0	0.0	0.0	0.0	0.0	0.0	0.0
WA	--	--	0.0	0.0	0.0	0.0	0.0
WY	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Total	**196.1**	**788.3**	**127.3**	**0.0**	**0.0**	**2.5**	**7.0**

State	Year										
	1979	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989
AZ	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
CA	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	5.0–10.0	105.0	105.0
CO	0.0	0.0	0.0	0.0	0.0	T	X	X	X	0.0	0.0
ID	0.0	0.0	0.2	4.0	14.2	T	0.0	3.4	0.0	0.0	0.0
MT	0.0	0.0	0.0	0.0	0.0	T	0.0	0.0	0.0	0.0	0.0
NM	1.6	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
OR	0.0	0.0	0.0	0.0	T	0.0	0.0	0.0	0.0	0.0	0.0
UT	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
WA	0.0	0.0	0.0	1.6	17.5	0.0	0.0	0.0	0.0	0.0	0.0
WY	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Total	**1.6**	**0.1**	**0.2**	**5.6**	**31.7**	**0.0**	**0.0**	**3.4**	**5.0–10.0**	**105.0**	**105.0**

State	Year										
	1990	1991	1992	1993	1994	1995	1996	1997	1998**	1999	2000
AZ	0.0		0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
CA	0.0		0.0	0.0	0.0	0.0	0.0	0.0	0.0	22.2	0.0
CO	0.0		0.0	0.3	6.1	1.5	0.0	0.0	0.0	0.0	0.0
ID	X		0.0	0.0	0.0	0.0	0.0	0.0	0.0	5.4	70.5
MT	0.0		0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	T
NM	0.0		0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
OR	X		7.5	46.2	26.5	2.9*	0.0	0.0	0.0	21.1	173.8
UT	X		0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.5
WA	0.0		0.0	0.0	0.0	--	0.0	0.0	T	0.1	46.1
WY	0.0		0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Total	**X**		**7.5**	**46.5**	**32.6**	**4.4**	**0.0**	**0.0**	**T**	**48.9**	**290.9**

Source: Forest Service, Forest Insect and Disease Conditions Reports – 1972–2000
* Includes Washington
** Data from aerial survey data base (FHTET)
T - Trace (< 50 acres)
X - Defoliation reported but no estimate of acres defoliated given

DISCUSSION

The available metric data for indigenous forest insect outbreaks indicates that six of the seven insects discussed in the preceding section reached record highs and are possible departures from their ranges of *historic* variation between 1973 and 1997 (Figure 14)**:**

1. **Southern pine beetle—1986 and 1995**
2. **Mountain pine beetle—1981**
3. **Spruce beetle (Alaska)—1996**
4. **Spruce budworm—1978**
5. **Spruce budworm (Alaska)—1997**
6. **Western spruce budworm—1986**
7. **Douglas-fir tussock moth—1973**

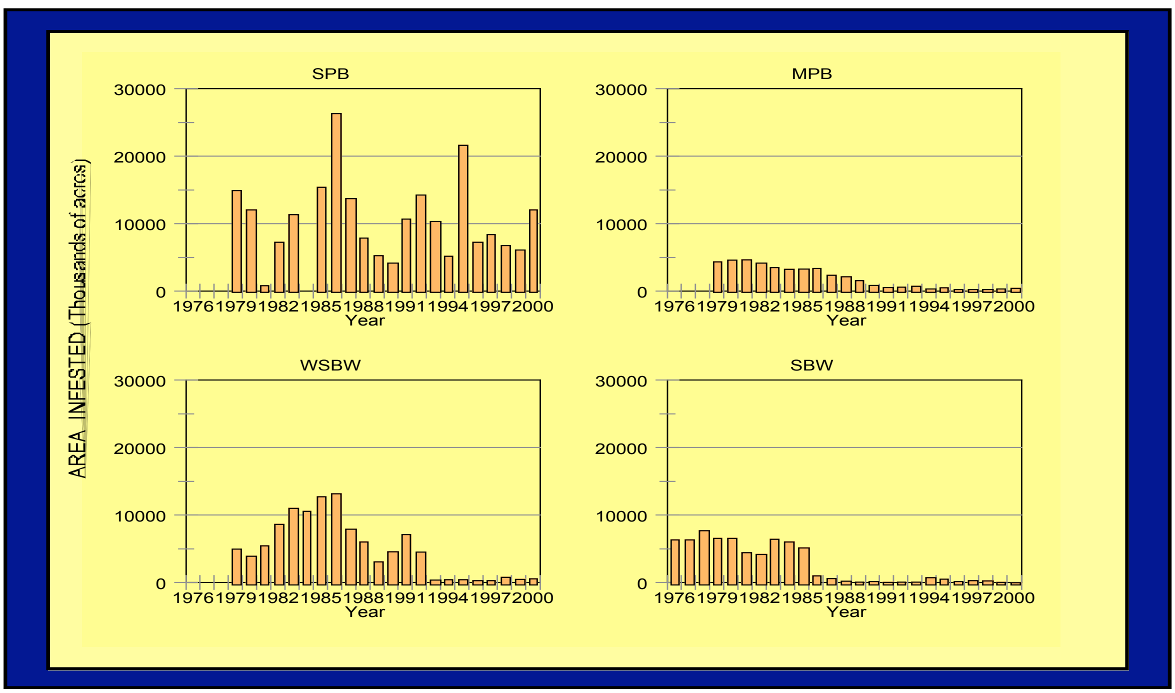

Figure 14. Area infested by four indigenous insects between 1979 and 2000 showing recent high levels of activity (SBP - southern pine beetle, MPB - mountain pine beetle, WSBW – western spruce budworm, Western United States, SBW – spruce budworm). Source – Forest Service, Forest Insect and Disease Conditions Reports, 1979–2000.

Two of these peaks (spruce beetle and spruce budworm in Alaska) took place during the 1996–2000 analysis period. In addition, southern pine beetle outbreaks occurred in several unusual locations during this period (SE Kentucky, Central Florida and SE Arizona) **and exceeded levels of *recent* variation**.

Most forest insect specialists and silviculturists agree that these massive outbreaks are the direct result of forest management practices carried out over the past 100 years. These include:

1. Forest fire management, which has reduced the natural fire interval in most forests resulting in changes in stocking density, age, and species composition.

2. Harvesting of old, large ponderosa pine forests in portions of the West, allowing Douglas-fir and true firs developing in the understory to replace the pines.

INTRODUCED SPECIES

A number of exotic forest insects have been introduced and established in the forests of the United States. Data recently compiled by Forest Service indicates that about 117 species of exotic insects have caused varying levels of damage to forest or urban trees. Most introductions have occurred in the South RPA region (Figure 15). While some have had relatively low socioeconomic impacts or have been successfully managed via classic biological control programs, others have become major pests (Table 19). Several recently introduced insects have spread rapidly once they became established in areas of suitable host type (Figure 16), however most forest insects introduced into the United States still have relatively localized distributions. Even gypsy moth, which has defoliated millions of acres of oak-hickory forests in the Eastern United States, is distributed over less than 10 percent of its potential range. Since most of these insects were introduced relatively recently and are still expanding their ranges, their present status, at least in terms of area infested, is at historic highs, with a potential for significant expansions of their ranges in the future.

GYPSY MOTH

Gypsy moth, *Lymantria dispar,* an insect native to Europe and Asia, was intentionally introduced into the United States from France in 1869 to begin a domestic silk industry. Some moths escaped, found the oak forests of southern New England to contain suitable host material, and have since spread south and west over large areas of broadleaf forest in the Eastern United States (North and South RPA regions). Spot infestations have also appeared in many parts of the United States This insect can feed on at least 500 species of woody plants but prefers oaks, *Quercus* spp. Damage by this insect consists of weakening and death of host trees, including episodes of oak decline and a gradual shift in species mixture of defoliated forests away from oaks (McManus 1980). Moreover, the presence of larvae in large numbers in urban areas is considered a nuisance.

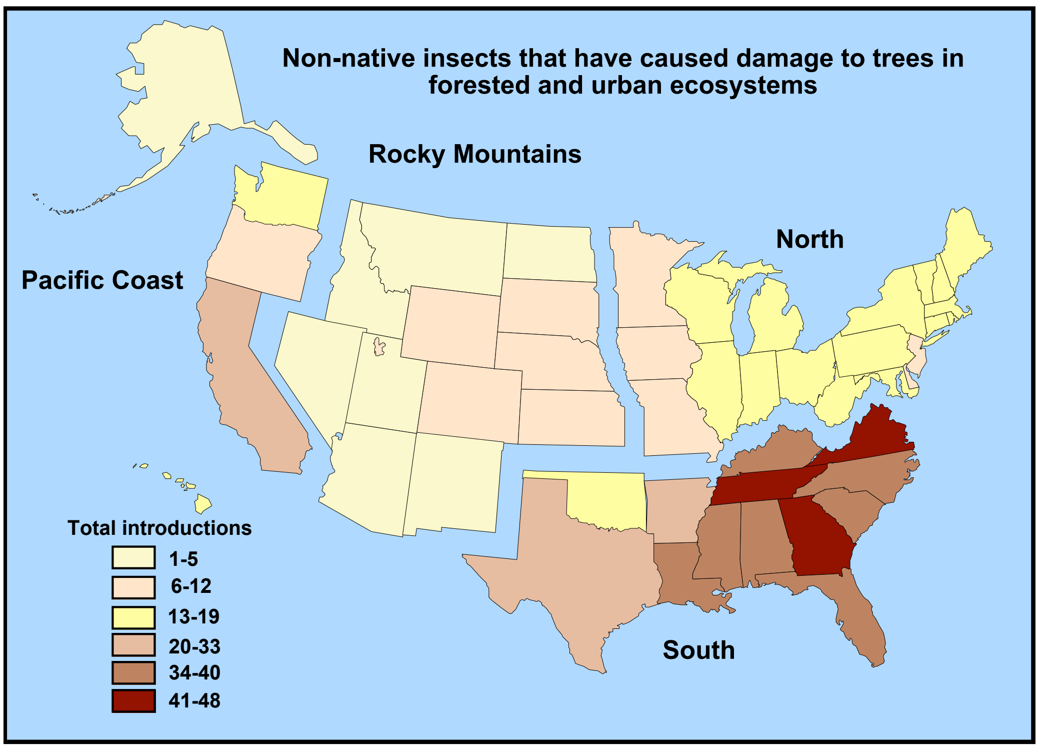

Figure 15. Distribution of species of nonnative insects that have caused damage to trees in forested or urban ecosystems in the United States by State and RPA Region (Source: R. Pywell, USDA Forest Health Technology Enterprise Team, Ft Collins, CO. "Non-native insects and pathogens that have caused damage to trees in forested and urban ecosystems [draft web pages]).

Of the susceptible host type, 12 million acres or 9.9 percent was defoliated, a recorded high (Figure 17 and Table 22). Gypsy moth populations declined in subsequent years, in part due to the successful introduction of a pathogenic fungus, *Entomophthora maimaiga*, into infested areas but again increased to more than 7 million acres in 1990. In addition, aerially visible defoliation has appeared in additional States: Virginia in 1984, West

Virginia in 1985, and Ohio in 1990. In Michigan, the total area of defoliation by gypsy moth increased from less than 100 acres in 1979 to a high of 712,200 acres in 1992. These data suggest that the gypsy moth is continuing its spread into new locations. Consequently the potential for the area of defoliation to exceed the recorded level established in 1981 sometime in the future is high.

Annual statistics on the total area defoliated by gypsy moth in the Eastern United States are available from 1924 to the present (Table 20) and on a statewide basis from 1979 to 2000 (Table 21). These data show relatively low levels of defoliation until 1953, when nearly 1.5 million acres were affected. The area of defoliation continued to increase gradually as the insect spread south into oak forests in Pennsylvania until 1981 when over 2.5 million areas were affected.

During the period of this analysis, gypsy moth defoliation remained at relatively low levels, ranging between 47,300 and 524,800 acres until 2000 when the level of defoliation increased to more than 1.6 million acres (Table 21). Also in 2000, aerially visible defoliation was detected for the first time in Wisconsin (Table 21). Assuming that the total area of oak-hickory forest in the United States is 130,250,000 acres, defoliation by gypsy moth ranged from 0.04 to 1.25 percent of the total area of susceptible host type during this period (Table 22 and Figure 16).

Two forms of gypsy moth are known, a European form with a female moth that is incapable of flight and an Asian form, which has a flying female moth. Introductions of the Asian form of gypsy moth into the United States occurred in 1991 and 1993 and were eradicated. No new introductions of Asian gypsy moth occurred during the reporting period.

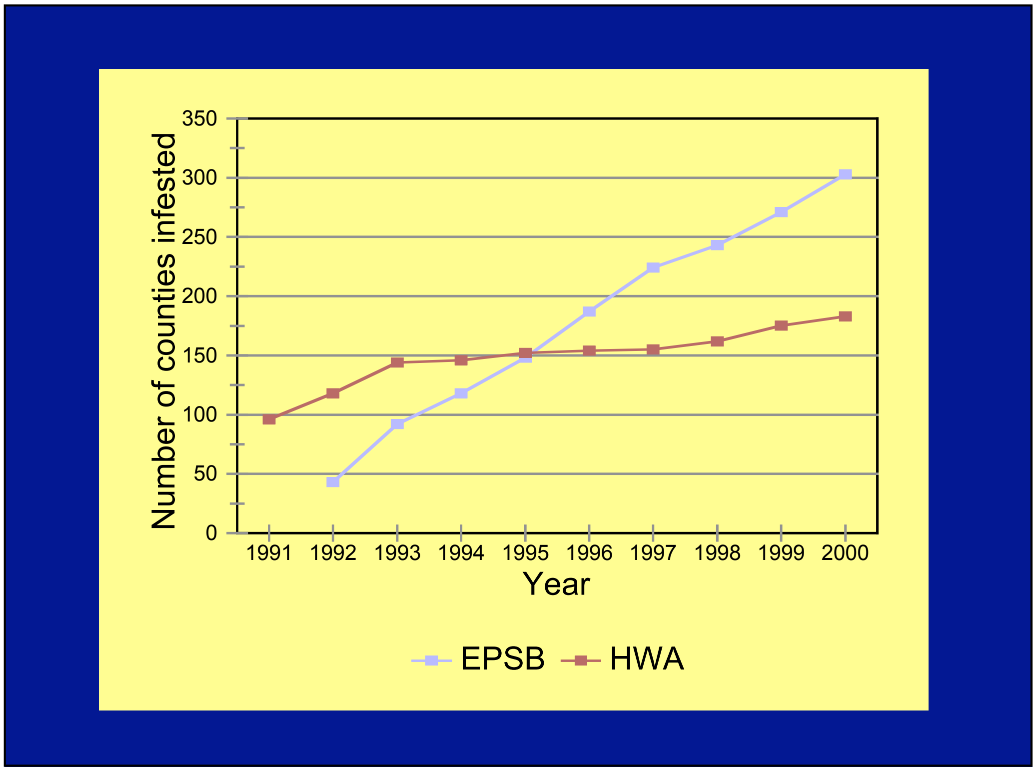

Figure 16. Change in numbers of counties infested by two recently introduced insects: European pine shoot beetle (EPSB) and hemlock woolly adlegid (HWA), 1991–2000.

Table 19. Partial List of Exotic Forest Insects Introduced and Established in North America

Species	Origin	Year of introduction	Site of introduction	Hosts affected	Type of damage
Gypsy moth, *Lymantria dispar* (European form)	France	1896	Boston, MA	Broadleaf trees	Defoliation
Larch sawfly, *Pristiphora erichsonii*	Eurasia	1880	Boston, MA	Larches	Defoliation
European pine sawfly, *Neopdiprion sertifer*	Europe	1925	Somerville, NY	Pines	Defoliation
Introduced pine sawfly, *Diprion similes*	Europe	1914	New Haven, CT	Pines	Defoliation
Balsam woolly adelgid, *Adelges piceae*	Europe	1908	Brunswick, ME	True firs	Gouting and tree mortality
European pine shoot moth, *Rhycionia bouliana*	Europe	1914	Long Island, NY	Pines	Shoot damage, deformity
Hemlock woolly adlegid, *Adelges tsugae*	Asia	1924, 1950	West Coast Richmond, VA	Hemlock	Tree mortality (eastern hemlock)
Larch casebearer, *Coloeophora laricella*	Europe	Prior to 1886	New England	Larches	Growth loss, tree mortality
Spruce aphid, *Elatobium abietinum*	Europe	Prior to 1953	West Coast	Spruces	Foliar damage, tree mortality
Gypsy moth, *Lymantria dispar* (Asian form)	Russia Europe/Asia	1991,1993	OR, WA, NC	Broadleaf trees	Defoliation
Common pine shoot beetle, *Tomicus piniperda*	Europe	1992	Ohio	Pines	Shoot damage, deformity
Asian longhorned beetle, *Anoplophora glabripennis*	Asia	1996	New York, Chicago	Broadleaf trees	Wood borer

Data sources: Drooz 1985, Furniss and Carolin 1977, Forest Service 1996,

Table 20. Area Defoliated by Gypsy Moth, *Lymantria dispar,* in the United States – 1924–2001

Year	Acres defoliated	Year	Acres defoliated	Year	Acres defoliated
1924	825	1950	5,368	1976	853,662
1925	48,560	1951	21,314	1977	1,598,662
1926	80,822	1952	293,052	1978	1,259,266
1927	140,920	1953	1,487,077	1979	643,609
1928	262,514	1954	491,448	1980	5,189,734
1929	551,133	1955	52,061	1981	12,886,535
1930	288,226	1956	43,158	1982	8,177,431
1931	204,721	1957	6,458	1983	2,386,838
1932	286,395	1958	125	1984	996,425
1933	397,730	1959	14,467	1985	1,728,331
1934	492,361	1960	48,722	1986	2,502,301
1935	540,769	1961	67,480	1987	1,373,099
1936	482,622	1962	308,312	1988	723,561
1937	608,760	1963	172,922	1989	2,917,255
1938	313,954	1964	254,983	1990	7,275,403
1939	492,640	1965	263,201	1991	3,952,234
1940	485,636	1966	51,865	1992	2,710,472
1941	468,021	1967	52,373	1993	1,676,674
1942	44,577	1968	80,123	1994	877,381
1943	34,845	1969	256,129	1995	1,418,537
1944	250,149	1970	972,833	1996	199,377
1945	821,497	1971	1,945,224	1997	49,180
1946	622,919	1972	1,369,130	1998	363,354
1947	7,422	1973	1,773,846	1999	470,537
1948	32,467	1974	750,905	2000	1,613,893
1949	78,673	1975	464,451	2001	

Source: Forest Service, GM Digest, Northeastern Area, Morgantown, WV

Table 21. Area of Aerially Detected Defoliation by Gypsy Moth, *Lymantria dispar*, in the United States, 1979–2000 (Thousands of Acres)

State	Year										
	1979	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989
CT	7.5	272.2	1482.2	803.8	153.2	0.5	89.5	237.2	65.4	1.6	1.8
DE	0.0	0.0	0.5	1.3	2.9	14.2	5.1	3.1	2.5	0.8	78.4
ME	23.2	221.2	655.8	574.5	16.3	1.8	6.7	11.6	0.6	0.1	35.0
MD	0.0	T	8.8	9.2	15.9	41.8	83.5	58.2	76.0	58.5	97.9
MA	266.3	907.1	2826.1	1383.3	148.1	185.5	414.1	343.1	28.7	0.0	0.9
MI	0.1	T	T	0.1	0.5	6.4	18.5	61.4	39.4	70.4	294.3
NH	5.9	183.9	1947.2	878.3	0.6	0.0	0.0	0.0	0.3	1.0	18.4
NJ	193.7	411.9	798.8	675.9	340.3	98.7	239.4	280.3	95.1	7.4	137.3
NY	162.3	2449.5	2303.9	825.7	290.8	33.7	129.8	175.4	55.2	5.7	421.1
OH	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
PA	8.6	440.5	2527.7	2351.3	1360.8	444.9	581.1	987.8	880.3	312.1	1506.8
RI	0.7	43.9	272.6	658.0	53.8	164.6	133.9	219.2	5.1	0.7	0.0
VT	15.4	75.1	48.9	9.9	0.0	0.0	0.0	0.0	0.0	0.7	237.3
VA	0.0	0.0	0.0	0.0	0.0	0.4	5.2	27.3	67.7	191.0	289.3
Washington, DC	0.0	0.0	0.0	0.0	0.0	0.0	2.3	0.0	T	0.0	0.0
WV	0.0	0.0	0.0	0.0	0.0	0.0	2.5	8.3	12.6	59.3	86.7
Total	**643.6**	**5005.4**	**12872.7**	**8171.2**	**2383.4**	**992.7**	**1709.3**	**2412.9**	**1328.9**	**709.3**	**2995.6**

State	Year										
	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000
CT	176.6	50.2	31.6	0.0	0.0	2.7	1.4	0.0	0.0	0.0	200.0
DE	3.8	13.46	4.9	26.7	60.7	65.5	0.5	0.0	0.0	0.0	0.0
ME	270.4	614.5	278.5	50.7	1.7	0	0.1	0.0	0.0	0.0	2.5
MD	133.1	75.2	38.7	68.9	93.2	93.9	11.2	0.7	0.5	1.2	23.2
MA	83.6	282.1	123.8	88.7	76.7	8.7	7.0	0.1	12.9	9.8	64.1
MI	358.3	626.7	712.2	399.3	97.3	85.9	5.0	36.9	310.7	176.6	106.3
NH	133.2	180.9	182.6	10.1	8.1	1.7	0.0	0.0	0.0	0.0	0.1
NJ	431.2	169.9	165.9	27.7	17.8	39.6	27.8	1.9	1.8	1.4	133.3
NY	354.2	175.9	60.0	2.0	0.5	0.2	16.3	2.2	9.4	6.0	27.5
OH	0.1	0.4	1.1	0.6	0.1	34.4	49.0	5.0	1.6	48.2	23.6
PA	4357.7	1230.1	641.4	318.1	18.0	132.5	6.7	0.0	31.6	281.6	843.0
RI	0.0	0.0	0.0	0.0	0.4	0.0	4.0	0.0	3.0	0.0	5.5
VT	63.0	3.6	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
VA	594.0	616.2	748.0	589.1	452.5	84.9	0.0	0.0	0.0	0.0	71.0
Washington DC	T	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
WV	388.7	122.9	67.5	202.5	53.4	103.0	70.7	0.5	0.8	0.0	323.1
WI	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1
Total	**7297.9**	**4152.1**	**3056.5**	**1784.4**	**880.4**	**653.0**	**199.7**	**47.3**	**363.3**	**524.8**	**1623.5**

T - Trace (< 50 acres)
Source: Forest Service, Forest Insect and Disease Conditions Reports – 1979–2000

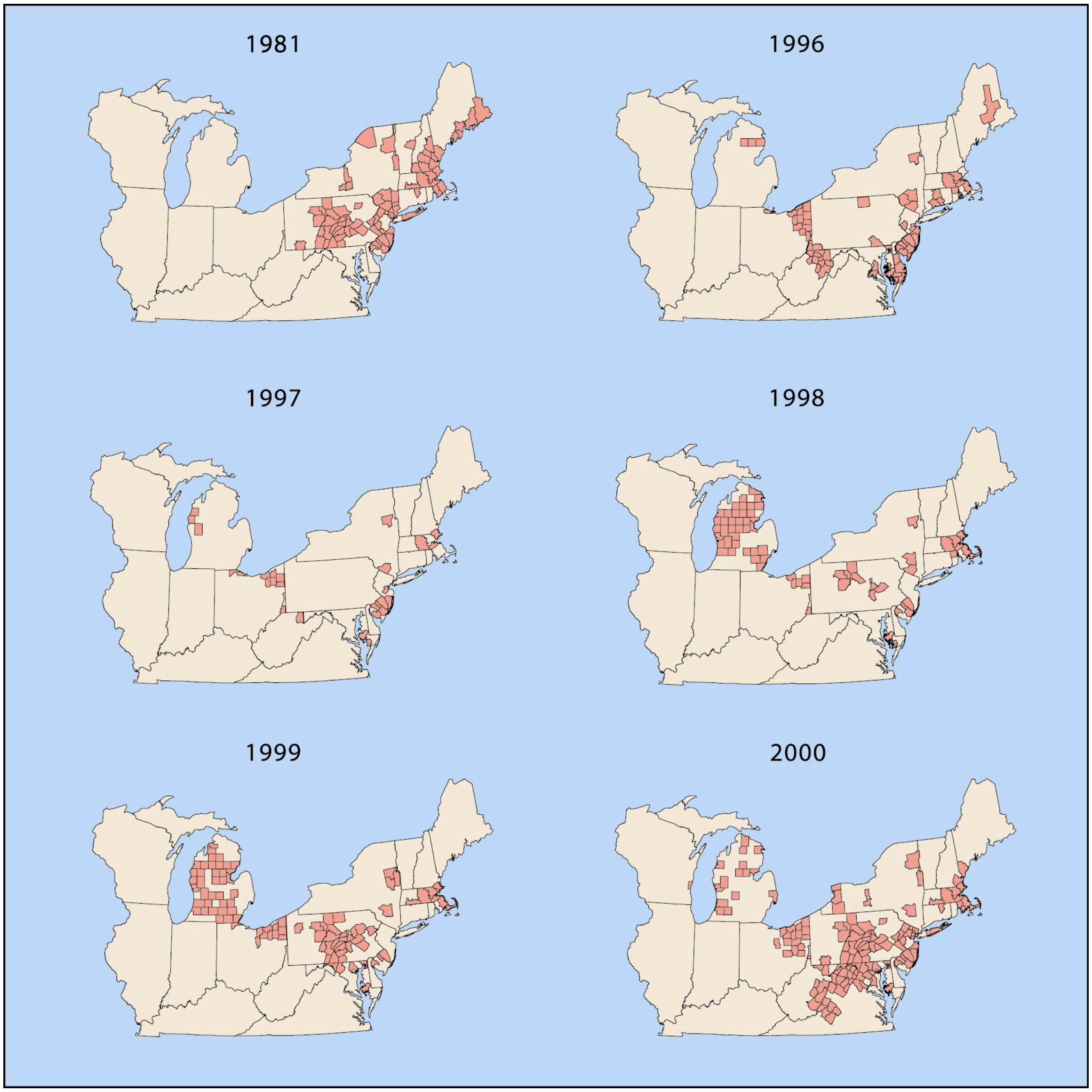

Figure 17. Area defoliated by gypsy moth in the Eastern United States – 1996–2000 in comparison with 1981, the year of historic high defoliation. (Source: Forest Service 1985, Insect and disease conditions – 1979–83 and Forest Service Forest Insect and Disease Conditions Reports 1996–2000)

Table 22. Aerially Visible Defoliation by Gypsy Moth in the Eastern United States – 1996–2000 in Comparison with Recent Highs

Year	Area defoliated (thousands of acres)	Percent of host type defoliated
Current Analysis Period		
1996	199.7	0.15
1997	47.3	0.04
1998	363.3	0.27
1999	524.8	0.42
2000	1623.5	1.25
Historic Highs		
1986	12872.7	9.9
1995	8171.2	6.2

** Based on an estimate of 130,250,000 acres of oak-hickory forest type groups. Source: http://fia.fs.fed.us/library/final_rpa_tables.pdf

HEMLOCK WOOLLY ADELGID

Hemlock woolly adelgid was first introduced into the Western United States and Canada, around 1924 and is now found in Oregon, Washington, and British Columbia where it infests western hemlock, *Tsuga heterophylla*, and causes little damage.

In 1950, hemlock woolly adelgid was discovered on the Atlantic Coast near Richmond, VA. Both eastern hemlock, *T. canadienses*, and Carolina hemlock, *T. caroliniensis*, are highly sensitive to the insect's feeding, and tree mortality can occur within 3 to 5 years after initial attack. Hemlocks are often concentrated in riparian zones, and the loss of these trees represents a loss of biodiversity as well as changes in the temperature of stream water, possibly affecting fish and other aquatic organisms.

Since the early 1990s hemlock woolly adelgid has spread to 11 States from North Carolina north to Massachusetts with an isolated infestation in New York. This insect has the potential to spread throughout the natural range of eastern and Carolina hemlocks (Forest Service 1999, 2000, Figure 18).

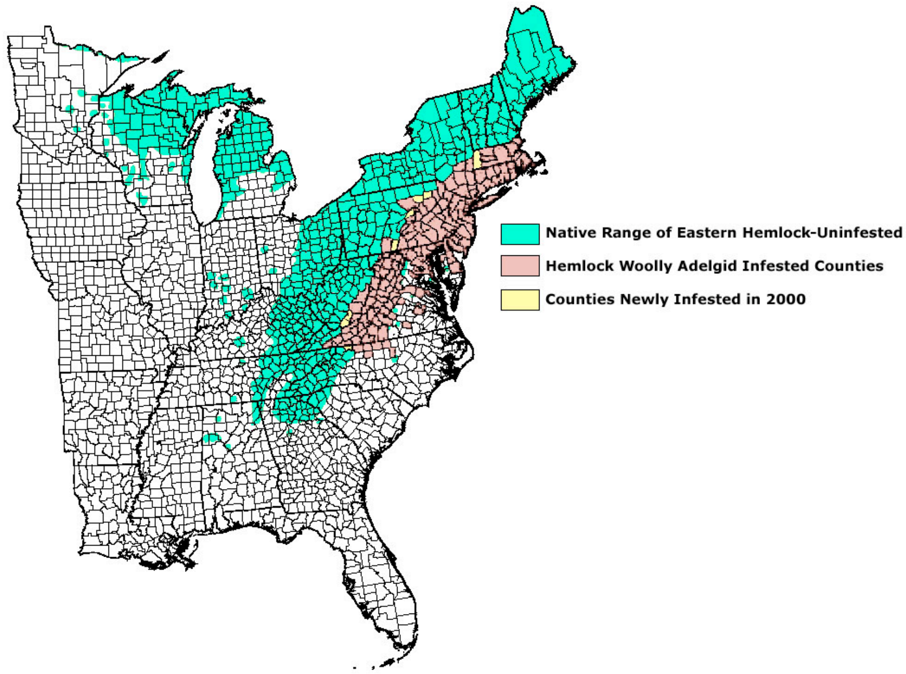

Figure 18. Distribution of hemlock woolly adelgid – 2000 in comparison with range of eastern hemlock. (Source: Forest Service, n.d. Hemlock Woolly Adelgid Web site. http://www.fed.fs.us/na/morgantown/fhp/hwa/hwasite.html)

EUROPEAN PINE SHOOT BEETLE

The European pine shoot beetle, *Tomicus piniperda*, is native to Eurasia where it attacks a variety of pines and other conifers. Damage is caused by adult maturation feeding in the shoots of host plants. The feeding causes shoot mortality, causing ornamental and Christmas trees to be unsightly.

Tomicus piniperda has often been intercepted at U.S. ports of entry. Between 1985 and 1998, it was intercepted 120 times, primarily in shipments of trade goods from France, the United Kingdom, Spain, and Italy (Haack and Cavey 1998, Stephen and Gregorie 2001). This insect was first discovered in North America in 1992 near Cleveland, OH. By the end of 1992, it was found in 43 counties in six States in the Great Lakes Region of the United States (Haack 1997, Haack and Kucera 1993). As of 1998, this insect had been found in 243 counties in 9 States in the United States (Illinois, Indiana, Maryland, Michigan, New York, Ohio, Pennsylvania, West Virginia and, for the first time, Wisconsin) (Forest Service 1999). In 1999 infestations continued to spread and were detected for the first time in New Hampshire and Vermont (Forest Service 2000, Figs. 19–20). This insect has the potential to spread more than much of the United States and Canada, causing damage primarily to ornamental trees and Christmas tree plantations.

ASIAN LONGHORNED BEETLE

The Asian longhorned beetle (ALB), *Anoplophora glabripennis*, is a serious pest of mature poplar plantations in portions of east central China. A common control method for this insect in China is to hire a large number of people to climb plantation trees as the adults are emerging, collect the beetles, and kill them by dropping them into a jar of kerosene (Author's observation).

Host trees attacked by this insect include various species of maple, *Acer* spp., poplar, *Populus* spp., willow, *Salix* spp., horse chestnut, *Aesculus hippocastanum*, black locust, *Robinia pseudoacacia*, elm, *Ulmus* spp., and birches, *Betula* spp. Another Chinese control method for this insect is to plant box elder, *Acer negundo*, and sugar maple, *Acer saccharum*, as trap trees to protect more valuable broadleaf trees (Forest Service and APHIS 1999, Haack and others 1997).

ALB was detected in the United States in 1996 in Brooklyn, NY. The discovery was made on August 19, 1996, by a concerned Brooklyn resident who notified the New York City Department of Parks and Recreation that all of the Norway maples, *Acer platanoides*, lining the street in front of his house were riddled with large holes. He also reported seeing large black and white beetles. The insects were subsequently identified as ALB. Additional surveys detected a second infestation in September 1996 in Amityville, NY, about 50 km east of the original discovery. The Brooklyn infestation is believed to have come from Asia in infested wooden packing materials entering the United States sometime in the 1980s or early 1990s. The Amityville infestation, on the other hand, is believed to have originated from the transport of infested tree sections from Brooklyn to Amityville for final disposal or sale as firewood (Haack and others 1997).

In 1998 three separate infestations were found in the Chicago metropolitan area, and additional infestations were detected in the New York City area. The first detection of ALB in the Chicago area was reported to APHIS PPQ by a local truck driver who had delivered a load of beetle infested wood. All infested areas are under quarantine and eradication efforts are underway (Forest Service 1999). ALB has also been found at 26 scattered warehouses and neighboring residential sites in 14 States across the United States (Excalibur Pallet Group 2000).

Additional spot infestations were found at O'Hare International Airport in 2000. Intensive surveys are underway, in both the infested areas and neighboring sites in Maryland, New Jersey, and Ohio, however these have thus far produced negative results (Forest Service 2001). This insect has the potential to spread throughout much of the United States and become a major pest of shade and ornamental trees and native broadleaf forests. Its present, relatively localized distribution represents an historic high and a threat to native broadleaf forests.

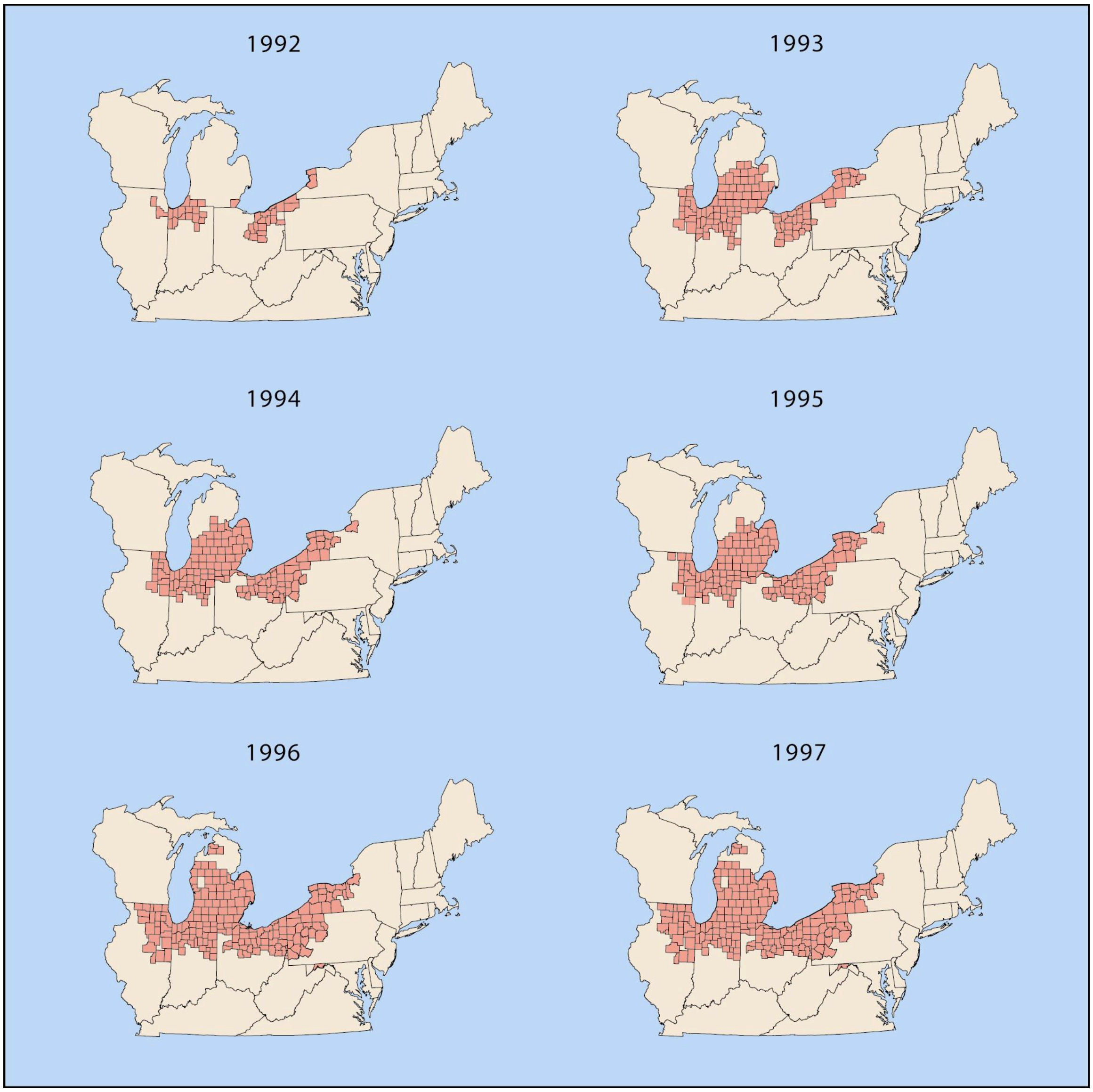

Figure 19. Known spread of the pine shoot beetle, *Tomicus piniperda*, in the United States from 1992 to 1997 (Source Forest Service Forest Insect and Disease Conditions in the United States, 1993– 2001)

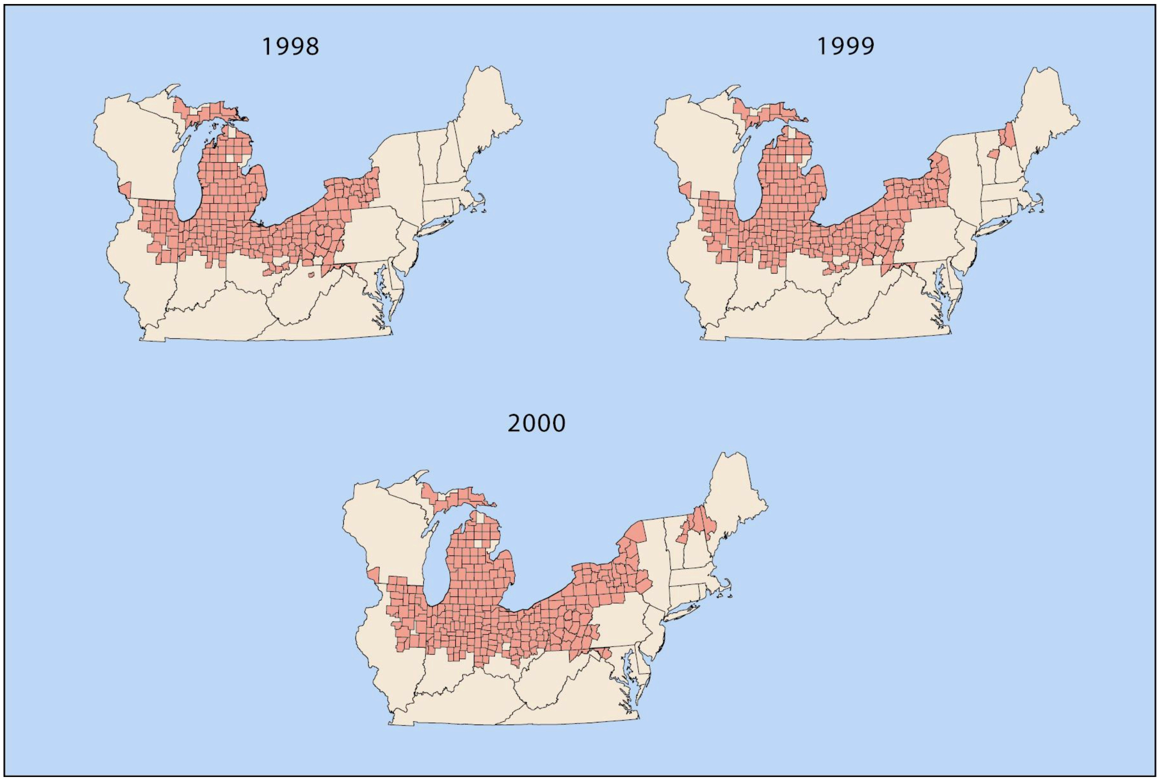

Figure 20. Known spread of the pine shoot beetle, *Tomicus piniperda*, in the United States from 1998 to 2000 (Source Forest Service Forest Insect and Disease Conditions in the United States, 1999–2001)

DISEASES

Agents and processes capable of causing disease in forests and trees include both biotic and abiotic factors. Biotic disease causing agents include fungi, viruses, and some parasitic plants. Air pollutants, chemicals, and climate are examples of abiotic factors that can cause disease.

NATIVE SPECIES

A large number of disease causing agents occur in the forests of the United States and play a vital role in their dynamics. Biotic agents capable of causing disease include fungi, viruses, phytoplasmas, and parasitic plants. Abiotic agents, such as certain chemicals, pollutants and climatic anomalies are also capable of causing disease. Among the more important disease pests of American forests are root diseases, rusts, wood decay fungi, and parasitic plants.

Unlike insects, most native disease causing agents are not dynamic and levels of activity do not change abruptly from year to year. Instead, they have a more or less even level of activity, and, unless management regimes are changed that favor or inhibit their development, they tend to cause about the same level of damage each year. Therefore, the range of historic variation for these agents is relatively narrow when compared to insects or wildfire.

DWARF MISTLETOES

Dwarf mistletoes, *Arceuthobium* spp., are parasitic plants that invade the branches of host trees. These parasites cause deformity (witches brooms), growth loss, and tree mortality.

The ecological and economic effects of dwarf mistletoes are reviewed by in a paper by Hawksworth and Shaw (1984). These parasites have many ecological effects. For example, dwarf mistletoes are involved in forest succession. Generally seral species, such as lodgepole pine, are severely parasitized whereas various species of spruce and fir are generally immune, or nearly so. The overall result of dwarf mistletoe parasitism is to hasten succession toward less infected climax species.

The fire history of forests is an important factor governing the distribution of dwarf mistletoes. In natural stands, fire has been a primary control agent. Fires severe enough to kill large areas of infected trees essentially eliminate the parasite from the stand except where an occasional infected tree may survive. Trees typically re-establish into burned-over areas much faster than the dwarf mistletoe. With a general increase in fire protection over the last 50 years or so, there has been a gradual increase in areas infested by these parasites. In a broader, long-term sense, fires also favor dwarf mistletoes in that, when less-susceptible climax forests are burned, they are often replaced by more susceptible seral species.

Dwarf mistletoes are considered to be the most serious disease agents in many western forests. In many areas they cause greater losses than forest fires, insects, and other pathogens combined. The proportion of dwarf mistletoe affected forests for several important tree species is summarized in Table 23. Dwarf mistletoes are estimated to affect at least ca. 21 million acres (8.5 million ha) of commercial forests in the United States and direct losses of about 418 million cubic feet per year were estimated in a 1982 report (Drummond 1982).

Considering the effects of reduced fire intervals in forests on the occurrence of dwarf mistletoe infections, it can be concluded that infestations have increased in magnitude and are currently exceeding the range of *historic* variation.

Table 23. Estimates of commercial forest types infected by dwarf mistletoes in the United States (Source Hawksworth and Shaw 1984)

Host and dwarf mistletoe species	Percent of area affected	Location
Lodgepole pine		
Arceuthobium americanum	50	Colorado, Wyoming
Arceuthobium americanum	42	California, Oregon, Washington
Arceuthobium americanum	35	Montana
Arceuthobium americanum	60	Southern Idaho, Utah
Western larch		
Arceuthobium laricis	47	Oregon, Washington
Ponderosa and Jeffrey pines		
Arceuthobium camplopodium	26	California, Oregon, Washington
Ponderosa pine		
Arceuthobium vaginatum	36	Arizona, New Mexico
Arceuthobium vaginatum	21	Southern Utah
Douglas-fir		
Arceuthobium douglasii	47	Arizona, New Mexico
Arceuthobium douglasii	58	Southern Idaho, Utah

FUSIFORM RUST[5]

Fusiform rust, caused by the fungus, *Cronartium quercumm f.sp. fusiforme*, is the most damaging disease of loblolly and slash pine in the Southeast (Southern RPA Regions). The disease disfigures and kills trees up to pole size and results in extensive stem breakage both in plantations and natural stands.

We do not know the "natural" level of fusiform rust, but according to the literature occurrence of this rust was relatively rare prior to 1900. "In spite of the lack of systematic data, there is no doubt that the incidence of rust is increasing," wrote Czabator, in his 1971 critical review of fusiform rust. Siggers and Lindgren (1947) reported, "Fifteen years ago trunk and branch cankers on southern pines were of interest mostly to classifiers of tree diseases. Fusiform rust…has become increasingly prevalent in the lower Gulf region during the past 35 years." Clearcut harvesting of old growth stands, through the 1920s and 1930s, led to the natural increase of pine in previously mixed forests. Scrub oak was also released. Fire suppression favored slash and loblolly pines to the detriment of longleaf pine. Because they were much faster growing, slash and loblolly were also the species of choice for replacement plantings. This resulted in slash pine plantations well north of their original range, and the spread of loblolly into areas further south than it originally occurred. The concurrent increase in the oak component also increased the availability of the alternate host, where sexual recombination occurs.

[5] Historical information in this section was provided by Kerry Britton, Forest Service, Southern Research Station, Athens, GA.

Unfortunately, the evidence of this early increase is piecemeal. Fusiform rust was not specifically assessed by FIA until the 4th inventory cycle, from 1968 to 1977. In 1929, Hayes and Wakely reported fusiform rust was of minor importance in plantations near Bogalusa, LA. In 1971, rust was a major problem there, despite the fact that Bogalusa lies a mere 100 miles from Livingston Parish, a famous center of origin for certain rust resistance genes. In 1938 and 1939, central Louisiana and northern Florida were areas of "light infection," but by 1940 Weber reported 64 percent of trees in a stand near Gainesville, FL, and more than 40 percent of several plantations near Foley, FL, were rust infected. In 1959, one 3,000-acre plantation near Alexandria, LA, was assessed as 70 percent infected with rust. In other parts of the South, slash pine plantations with severe mortality were common, and survivors often were 60–80 percent infected. Schmidt reported further increases in rust through the 1950s and 60s.

Forest nurseries, perhaps under the influence of increased inoculum, also witnessed the rise in rust incidence. It was not observed in nurseries until 1937. In 1938–39, incidence was 15–35 percent or less. Many plants were sent to the field with undetected infections, and the disease spread further across the south. By 1941, Bordeaux mixture was in frequent use in nurseries, but by 1949, losses of more than 65 percent were noted. In 1957, incidence was about 60 percent. By 1959, spray schedules had been refined, and incidence dropped to about 35 percent. Today fungicide sprays and seed treatments are routinely used in all pine nurseries to control rust.

Resistance screening began in the 1960s, using first generation phenotypic selections, which had often been selected prior to the rust epidemic. Second generation seed orchards were developed with rust resistant genes, and by the 1980s rust resistant seedlings were increasingly planted in the field. The best of these selections are now growing in third generation orchards. The availability of rust resistant seedlings has not been able to keep pace with demand, however. By 1994, one-half of planted slash pine had some rust resistance, with an expected gain of 40 percent. Only one-sixth of planted loblolly pine was resistant, and the expected gain was 30 percent.

Forest inventory data did not account for fusiform rust prior to cycle 4 (1968–77). Young pine plantations (less than 10 yrs old) assessed in cycles 4 and 5 (1978–86) were considered by Pye et al. to represent pine germplasm "before resistance gene deployment" for the purpose of their economic analysis of the cost/benefits of rust resistance research. They compared these data with rust incidence in cycle 6 (1986–1993), "post rust resistance deployment." On slash pine, rust incidence decreased only on high-quality sites. Probably this is due to the most resistant seedlings being planted on the best sites, to capitalize on the investment in resistant seedling cost. Rust incidence increased on low and medium quality sites, with 16 percent and 8 percent, respectively, of plantations assessed between 1986 and 1993 having more than 30 percent rust. Natural stands of slash pine had less rust than plantations, but increased in rust incidence over the same period, from 14 percent (cycle 4) to 35 percent (cycle 6) of the stands having more than 5 percent rust.

Loblolly pine showed decreased incidence in rust comparing "before" to "after" surveys, with an average of only 1 percent of plantations more than 30 percent infected in cycle 6. Rust incidence decreased on high- and medium-quality sites, and remained stable on low quality sites. Overall, about 23 percent of plantations were still at least 5 percent infected, however. Natural loblolly stands had more rust than plantations, and remained equally susceptible in cycle 6 as in cycle 4 (33 percent were more than 5 percent infected).

Acreage in plantation pines doubled between 1970 and 1993, and 75 percent of this increase occurred between FIA cycle 5 and cycle 6. Slash pine acreage decreased, primarily due to problems with fusiform rust, from 3.5 million acres to 2.6 million acres. Plantation pine increased from 2 million acres in 1952 to 32 million acres in 1999. Projections for 2040 are approximately 50 million acres (Wear et al. 2002). Rust incidence increased with site quality in cycles 4 and 5, but not in cycle 6, further evidencing the planting of resistant pine nursery stock on the higher quality sites.

Despite these seeming low numbers of plantation failures, wood product degradation due to rust stem infections led Pye et al. to conclude that rust resistance research had yielded a 6.3 percent (for pulp plantations) to 238 percent (for saw log timber) return on investment, as of 1994, and that these investments will continue to pay off as future plantings are installed and harvested. Future plantations are expected to be established on higher quality sites, and/or with higher inputs, possibly increasing their susceptibility to rust. Unless production of resistant seedlings is increased, we may expect fusiform rust incidence to increase, especially on slash pine. Identifying different rust resistance sources, and understanding their genetic structure could be important to staying ahead of the ability of the pathogen to adapt to resistance genes.

As of 1997, only four States had FIA fusiform rust incidence data from which a limited analysis of long-term changes in rust incidence could be made – Mississippi, North Carolina, South Carolina, and Virginia. These data show a level to slight reduction in fusiform rust incidence despite the fact that the total area of new pine plantations has increased slightly (Starkey and others 1997, Figure 21). Approximately 13.9 million acres of plantations are affected in 12 Southern States (Forest Service 2000, Table 24), however, the existing data are insufficient to establish historic highs in fusiform rust incidence.

These data indicate that fusiform rust is a pathogen that has been favored by forest management practices in the south, resulting in larger areas of susceptible host trees. **This pathogen has been outside of its range of *historic* variation since at least the 1970s and is expected to continue to occur at relatively high levels under present management regimes.**

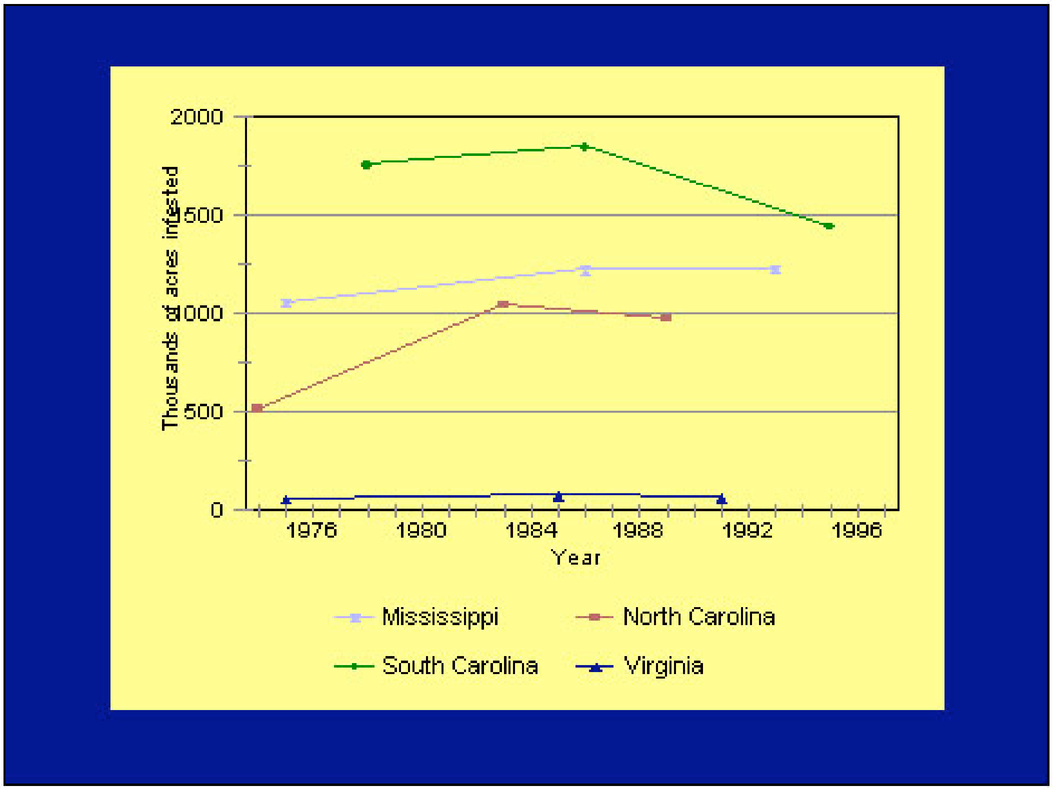

Figure 21. Acreage of loblolly and slash pine with ≥ 10 percent fusiform rust infection, estimated from FIA data for four States – 1974 to 1994 (Starkey and others 1997).

Table 24. Area Affected by Fusiform Rust in the Southeastern United States – 1999

State	Year of Survey	Area affected (thousands of acres)
Alabama	1990	1711.3
Arkansas	1995	285.4
Florida	1995	1468.4
Georgia	1989	4593.7
Louisiana	1991	1658.3
Mississippi	1994	1221.0
North Carolina	1990	968.9
Oklahoma	1993	33.9
South Carolina	1995	1437.2
Texas	1992	419.1
Virginia	1992	59.3
Total		13856.5

Source: Forest Service 2000

ROOT DISEASES

A number of fungi that cause root disease are found throughout the United States (Table 25). They cause a relatively inconspicuous level of mortality, however, their total impact is significant, especially in the way they influence management options on infected sites. Root diseases decrease timber production, increase management costs and root disease centers often serve as focal points for bark beetle infestations, especially during periods of dry weather.

West-wide assessments of root disease caused losses in the commercial conifer forests of the United States were made during the period 1979 to 1983. These indicated that root diseases were of concern on 16,805,300 acres with an annual mortality rate of 242,620,000 cubic feet (DeNitto 1985, Tables 26, 27). A second assessment, made in 1984, based on records from the Western United States, indicated that average annual loss on commercial forests of all ownerships was 237.4 million cubic feet or approximately 18 percent of the tree mortality in the Western United States (Smith 1984). Although it is generally believed that root disease losses will continue to increase, no data are available to substantiate this fact and no countrywide assessments have been made of the status of root diseases in recent years.

In portions of northern Idaho and western Montana, high levels of root disease have developed in areas where root disease susceptible trees (e.g., Douglas-fir and grand fir, have replaced western white pines following mortality caused by white pine blister rust (Byler and Hagle 2000). **These effects are considered to be beyond the range of *historic* variation.**

Table 25. Important Root Diseases in the United States

Disease	Causal fungus	Areas of management concern	Commercially important hosts
Annosus root disease	*Heterobasidium annosum*	South, West	All conifers, especially pines, hemlock and true firs
Armillaria root disease	*Armillaria* spp.	West	All woody species
Black stain root disease	*Ophiostoma wageneri*	West	Douglas-fir, ponderosa pine, piñon pine
Red-brown butt rot	*Phaeolus schweinitzii*	Idaho, western Montana, Wyoming	Douglas-fir
Laminated root rot	*Phellinus weirii*	Oregon, Washington, northern Idaho	Douglas-fir, White fir, Grand fir, Pacific silver fir, mountain hemlock, western red cedar
Littleleaf disease	*Phytophthora cinnamomi* (complex)	Southeast	Shortleaf pine, loblolly pine

Table 26. Acres on all Ownerships Where Root Diseases are a Management Concern and Average Annual Root Disease Related Mortality from 1979 to 1983 in the Western United States (DeNitto 1985)

Region/State	Area of management concern (acres)	Volume of mortality (1,000 cubic feet)
Pacific Coast States		
Alaska	24,000	-- [1]
California	8,132,500	19,398
Hawaii	700	--
Oregon	1,221,000	75,776
Washington	999,000	56,155
Rocky Mountain States		
Arizona	281,600	2,107
Colorado [2]	38,400	127
Idaho	1,929,000	41,210
Montana	1,400,000	40,000
Nevada	500	25
New Mexico	858,700	2,653
Utah	50,000	950
Wyoming	5,500	105
Total	14,940,900	238,506

1 Indicates no information available
2 Area and volume for subalpine fir in spruce fir type only

Table 27. Acres on all Ownerships where Root Diseases are a Management Concern and Average Annual Root Disease Related Mortality From 1979 to 1983 in the Eastern United States [1] (Denitto 1985)

State	Area of management concern (acres)	Volume of mortality (1,000 cubic feet)
Alabama	228,500	-- [2]
Florida	456,800	4113
Georgia	587,800	--
Kentucky	13,400	--
Mississippi	8,900	--
North Carolina	184,700	--
Ohio	1,500	--
South Carolina	204,100	--
Tennessee	40,700	--
Virginia	138,000	--
Total	1,864,400	4113

1 – indicates no information available. 2 – Acreage and volume data for the Eastern States, except Ohio, are for littleleaf and sand pine root diseases only.

OAK DECLINE

Oak decline is a "complex" disease caused by a number of predisposing, inciting, and contributing factors including drought, waterlogging, defoliation by insects or late spring frost, root disease, and ultimately attack by wood boring insects (Manion 1991, Wargo and others 1983). Symptoms are progressive and include a thinning of the foliage, crown dieback, and tree mortality. This condition has been recorded since 1900 and a number of episodes of oak decline have been reported throughout the oak-hickory and oak-pine forest type groups in the Eastern United States (North and South RPA regions).

Severe summer droughts occurred over portions of the Southeastern United States between 1998 and 2000 and caused widespread death of oaks and other trees, especially in portions of Virginia and Arkansas. Analysis of forest inventory data from 12 Southern States indicates that some 3.9 million acres of upland oak forest (oak-hickory forest types) or 9.9 percent of the susceptible host type in the South are affected by oak decline. Average annual mortality of oaks on affected sites was 45 percent higher than on unaffected areas (Forest Service 1999, 2000). **The level of oak decline in the Ozark and Ouachita mountains of Arkansas may be at an historic high with heavy oak mortality on 350,000 acres and an estimated potential timber loss on an additional 1 million acres. These effects are believed to be beyond the range of *historic* variation.** The affected oaks have been attacked by the red oak borer, *Enaphalodes rufulus,* an insect that is behaving in an exceptionally aggressive manner.

The extensive oak mortality in the Ozarks and other parts of the oak-hickory forest type groups may be the result of fire exclusion. Without periodic fires, the oak forests have become more heavily stocked. This, couple with episodes of drought or defoliating insects can trigger episodes of oak decline. Moreover, the shade created by dense oak forest favors regeneration by species such as red maple, ash, and elm. Therefore, the composition of oak hickory forests may be undergoing long-term changes to forests of less desirable species (Spencer 2001, Sutton 2001, Spencer and Sutton 2001).

OAK WILT

Oak wilt is a systematic, vascular wilt disease tree killing disease of oaks caused by the fungus, *Ceratocystis fagacearum*. It has been found in 21 States with considerable damage occurring in the Midwest. It was first recognized as an important disease in 1944 in Wisconsin. Surveys in Wisconsin showed that 11 percent of the annual growth increase in oak forests was offset by tree mortality caused by oak wilt (Rexrode and Brown 1983).

Oak wilt also occurs in central Texas, which according to one source, is outside its main range (Rexrode and Brown 1983) **and, therefore, for purposes of this analysis, is considered to be beyond the range of *historic* variation.** For many years, it killed trees in central Texas, principally live oak, *Quercus fusiformis*, and was referred to as "live oak decline." However in 1977, it was established that the oak wilt fungus was the cause of live oak decline (Lewis 1977). In central Texas, large areas of live oaks are connected through common root systems, resulting in large infection centers of up to 200 acres (Forest Service n.d.).

As of 2000, oak wilt was known to occur in 60 counties in central Texas where urban, suburban, and rural oaks are affected. Live oak is the premier shade tree species in the region and is highly valued for its beauty, shade and wildlife benefits. Beginning in 1983, The Texas Forest Service began a cooperative oak wilt suppression project. Since the project's inception, more than 2.4 million feet (greater than 450 miles) of barrier trenches have been installed around 2,065 infection centers in 34 counties to prevent the spread of this disease (Forest Service 2001).

According to Reisfeld (1995), the potential for an epidemic of oak wilt in Central Texas is believed to be the result of influences of European settlement. Fire suppression and introduction of livestock have contributed to the degradation of grasslands and an increase on woody plant species including vast clonal stands of Texas live oak susceptible to this pathogen. The low species diversity has significantly affected the vector-host relationship. Local residents also promote the spread of this disease by transporting infected firewood and creating infection courts.

The level of oak wilt damage is considered to have been outside of the range of *historic* variation for a number of years, including the period 1996–2000.

INTRODUCED SPECIES

Several diseases, accidentally introduced into U.S. forests, have caused high levels of damage and have irreversibly altered the character of some forests (Table 28). For example, chestnut blight, caused by the fungus *Cryphonectria parasitica* was introduced into the United States in 1904 and spread rapidly through the American chestnut, *Castanea dentata*, forests of the East, killing all native chestnuts. In some areas, one-third of the trees were chestnut, a tree highly valued for its excellent wood properties, resistance to decay and an edible nut prized by both humans and wildlife. This continues to prevent American chestnut from reaching sizes beyond those of a small shrub.

In 1906, white pine blister rust, caused by the fungus *Cronartium ribicola*, another native of Asia, appeared on eastern white pines. This disease also spread rapidly through eastern white pine forests and in 1921 was discovered in the West, where it continues to spread.

Dutch elm disease, caused by the fungus *Ophiostoma ulmi*, came to the United States via European elm logs infested by a bark beetle, *Scolytus multistriatis*, which is a vector of this disease. The introduction of Dutch elm disease virtually eliminated the American elm, a highly prized shade and ornamental tree, from cities and communities throughout the Eastern and Midwestern United States as well as natural forests, continues to infect trees and expand its range.

Recently compiled data by Forest Service indicates that about 46 species of pathogens that are either exotic or of unknown origin have caused varying levels of damage to forest or urban trees. Most introductions have occurred in the East, South, and Pacific RPA Regions (Figure 22).

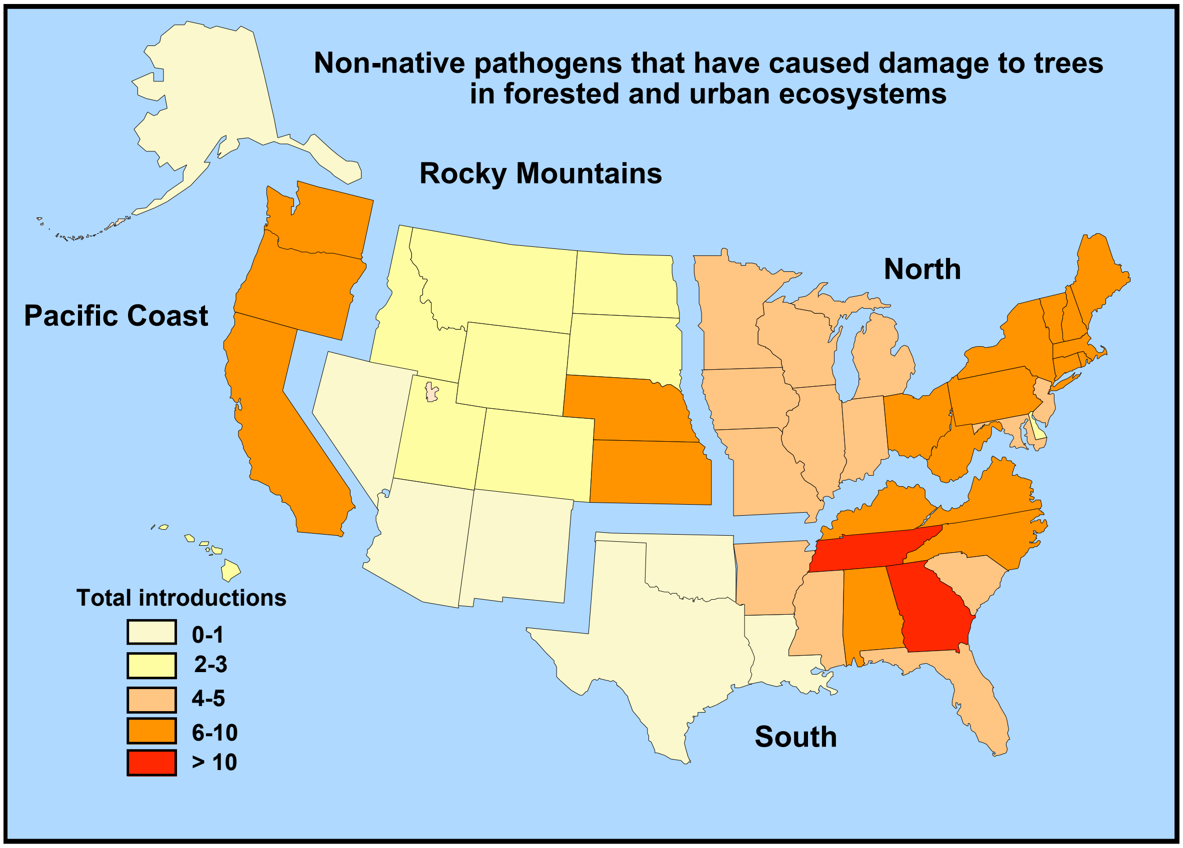

Figure 22. Distribution of pathogens of exotic or unknown origin that have caused damage to trees in forested and urban ecosystems in the United States by State and RPA Region (Source: R. Pywell, USDA Forest Health Technology Enterprise Team, Ft Collins, CO. "Non-native insects and pathogens that have caused damage to trees in forested and urban ecosystems [draft web pages]).

Since all exotic forest pathogens have been introduced relatively recently and are still expanding their ranges, they all must be considered outside the range of historic variation. Two examples are discussed in the following sections.

Table 28. Partial List of Exotic Forest Diseases and Diseases of Unknown Origin in North America

Disease	Origin	Year of discovery	Site of introduction	Hosts affected	Type of damage
Beech bark disease	Europe	1890	Nova Scotia, Canada	Beech	Decline and mortality
Chestnut blight	Asia	1904	New York	Chestnut	Mortality
White pine blister rust	Asia (via Europe)	1906 1921	New York British Colombia, Canada	5-needle pines	Mortality
Dutch elm disease	Asia (via Europe)	1930	Ohio	Elm	Mortality
Dogwood Anthracnose	Unknown	1976 1978	Northwest New York	Dogwood	Mortality
Port Orford Cedar root disease	Unknown	1952	Oregon	Port Orford Cedar, western yew	Mortality
Butternut Canker	Unknown	1970s		Butternut	Decline and mortality
Pitch Canker	Unknown	Unknown 1986	SE U.S. California	Pines Pines, other conifers	Deformity, tree mortality
Sudden oak death	Unknown	1995	California	Tanoak, oaks, other plants	Tree mortality

Data sources, Storer and others 1994, ODA n.d., Forest Service 1996, 2000.

WHITE PINE BLISTER RUST

White pine blister rust, caused by the fungus *Cronartium ribicola*, was first found in New York in 1906, arriving on white pine nursery stock from Germany. This disease has spread throughout the range of eastern white pine and has changed the way white pine is managed in many areas.

This disease was also introduced into Western North America on nursery stock imported from France in 1921 and has now spread throughout much of the West affecting all indigenous five-needled pines and causing significant tree mortality (Forest Service 1996).

A current concern about white pine blister rust is the high level of mortality it is currently causing in high elevation forests of whitebark pine, *Pinus albicaulis*, and limber pine, *P.*

flexilis. The extensive tree mortality may have significant effects on water and wildlife in these fragile ecosystems (Forest Service 2000).

This disease continues to spread to new areas in the West. In 1990, it was found affecting southwestern white pine, *Pinus strobiformis,* in New Mexico for the first time. By 1995, about one-half million acres were affected, resulting in mortality of commercially valuable five-needle pines, loss of five-needle pines in fragile alpine ecosystems, and loss of a source of food for several wildlife species (seeds of *Pinus albicaulis)* (Forest Service 1996). White pine blister rust was found for the first time in Colorado in 1998 (Forest Service 2000, Figure 23). This disease was introduced relatively recently and has continuously been beyond the range of historic variation.

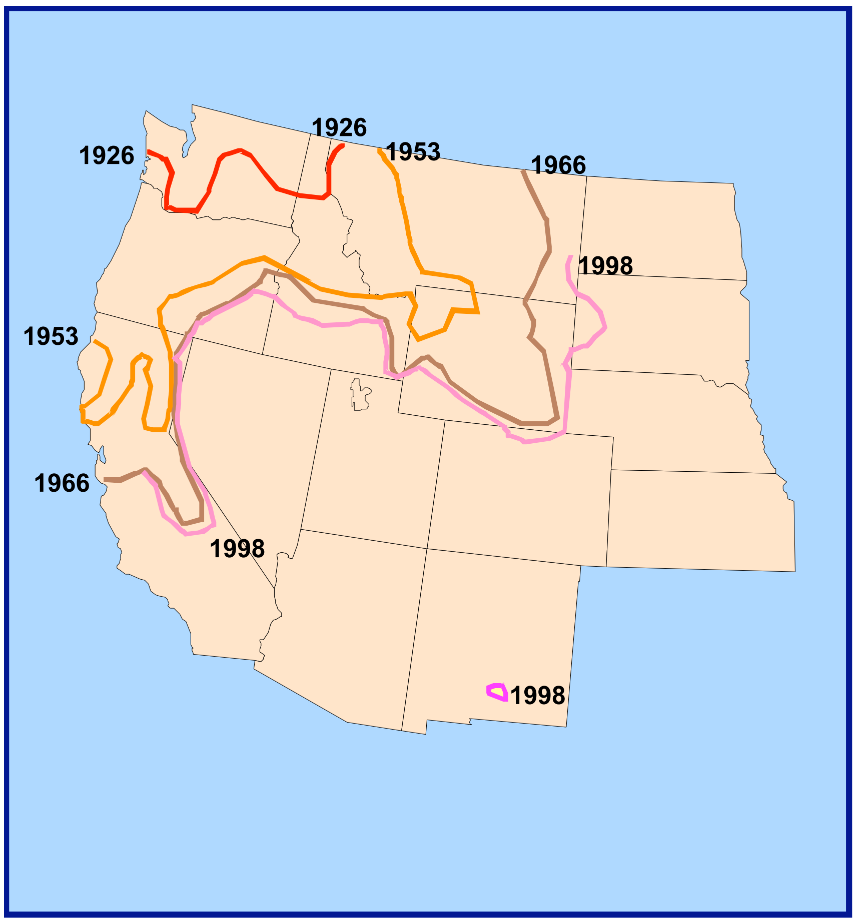

Figure 23. Spread of white pine blister rust in the Western United States from its original introduction to 1998.

BEECH BARK DISEASE

Beech bark disease is caused by a combination of an insect and a fungus. The beech scale, *Cryptococcus fagisuga*, attacks the bark of beech trees and causes a drying and cracking, which provides sites for the fungi *Nectria coccinea* and *N. galligena* to invade the tree. The scale, and probably the fungus, was apparently introduced into Nova Scotia, Canada, from Europe around 1890. The disease affects only beech trees. The European beech, *Fagus sylvatica*, is relatively resistant to the disease, but the American beech, *F. grandifolia*, is highly susceptible and many trees are ultimately severely deformed or killed. This has reduced the diversity of many northeastern forests. Since beechnuts are an important food for several wildlife species, the disease is also affecting wildlife populations.

Beech bark disease gradually spread across much of the Northeastern States as far south as northeastern Pennsylvania. More recently, the scale and the fungus have spread to two new locations. In 1981, a large area of infested beech forest was discovered in West Virginia, well ahead of the advancing front of the disease. Beech mortality was reported in adjoining counties of northern Virginia by the mid 1980s. In 1994, the disease was found affecting approximately 100 acres in three counties on the North Carolina-Tennessee border. This infestation is about 300 miles southwest of its previously known distribution (Figure 23).

During the period 1996–1999, no new infestations were found, but tree mortality continued to intensify in the South. In New York, 90 percent of the trees surveyed have some evidence of the disease (Forest Service 2000). In 2000, beech scale and beech bark disease was discovered in Michigan. To date affected beeches have been found in the northwestern Lower Peninsula and areas of the eastern Upper Peninsula (McCullough and others 2000, Forest Service 2001, Figure 24).

DISEASES OF UNKNOWN ORIGIN

Several disease agents have appeared in the forests of the United States whose origin is unknown (Table 28). They may have a yet to be determined natural range in some other part of the world or they may be native species that have recently evolved into pathogenic forms. Several of these have caused severe losses in forests in some regions of the United States and threaten the viability of certain tree species, thus the diversity of affected forest ecosystems. Most of these agents continued to expand their geographic ranges over the analysis period. **Consequently, even though their ranges may still be relatively localized, their distributions and effects to forests are considered to be beyond the range of *historic* variation.**

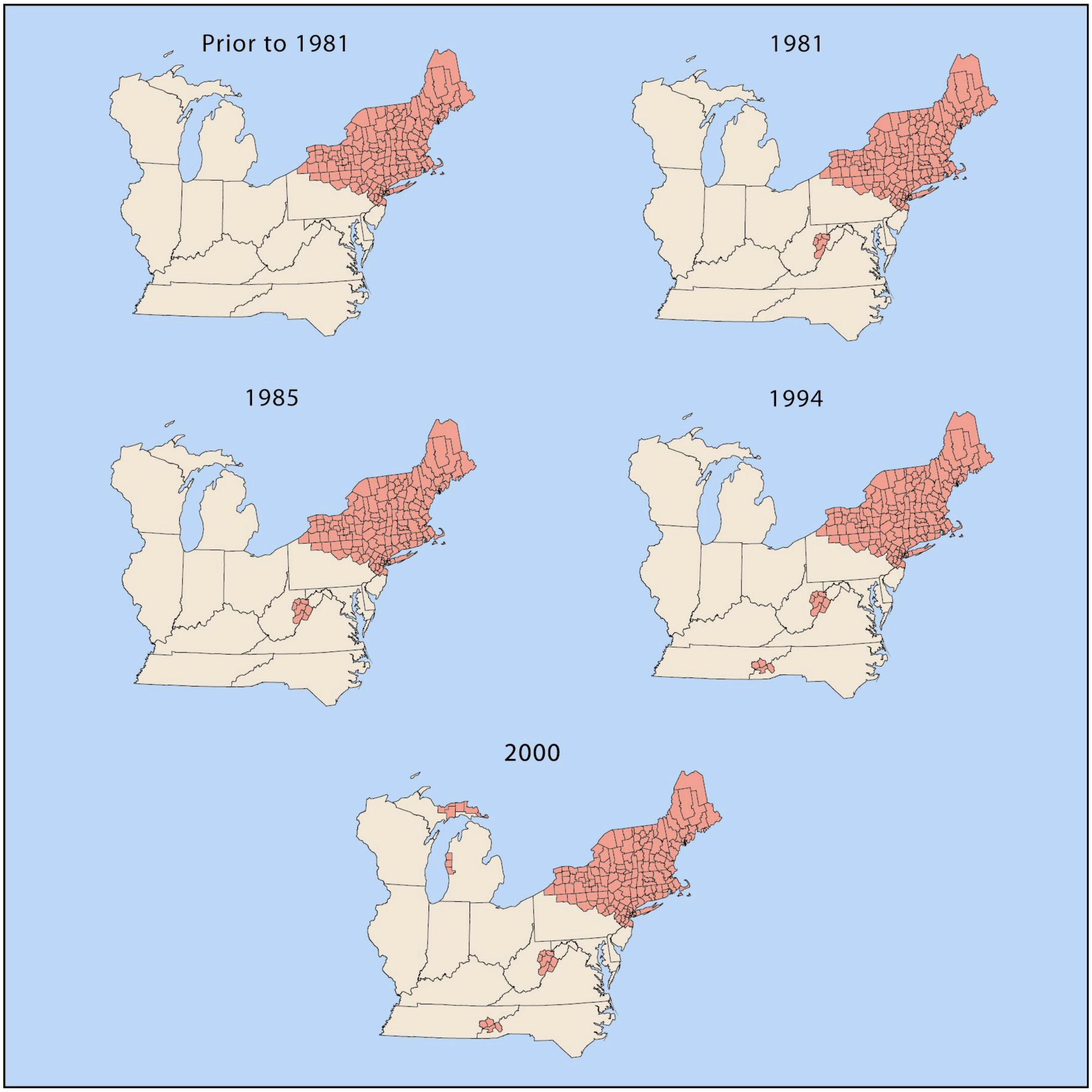

Figure 24. Distribution of beech bark disease in the Eastern United States prior to 1981, 1981, 1990, 1994 and 2000 (Sources: Forest Service, Forest Insect and Disease Conditions Reports, 1981, 1990, 1994, 2000)

DOGWOOD ANTHRACNOSE

Dogwood anthracnose, caused by the fungus *Disclusa destructive,* was first discovered in the Pacific Northwest on Pacific dogwood in 1976. Although the Pacific dogwood is more susceptible to this fungus than the eastern flowering dogwood, *Cornus florida,* the occurrence of drier summers in the West reduces the number of infection cycles.

In the Eastern United States, dogwood anthracnose was first found in southeastern New York in 1978. By 1994, this disease was found in 22 States from Maine to Georgia and west to Indiana and Missouri. The natural range of flowering dogwood extends from southern Maine to Florida and west to Michigan and eastern Texas (Forest Service 2000).

In the Southeastern United States, most of the flowering dogwood above the 3,000-foot elevation and in cool shaded areas below that elevation has been killed. The disease continues to intensify within infested counties. In 1999, 18 new infested counties were found in New York and in 2000 the disease was found in three new counties: one each in Kentucky, North Carolina and Virginia. In the Northeast, diseased dogwoods have been found in every county in Delaware, Maryland, and West Virginia (Figure 25). This fungus threatens the viability of a small but attractive flowering tree that was once a common component of many eastern forests.

BUTTERNUT CANKER

Symptoms of butternut canker, caused by the fungus *Sirococcus clavigignenti-juglandacearum*, have been known since the early part of the twentieth century. The causal fungus was not known until the late 1970s. Butternut, *Juglans cinearia*, is the only known host tree and ranges from Maine to Georgia and west to Minnesota and Arkansas. This disease is now known throughout the range of butternut and is a serious threat to the survival of the species – killing large trees, saplings, and regeneration (Figure 26). In North Carolina and Virginia, an estimated 77 percent of the butternut trees have been killed. Some trees have been discovered that exhibit some resistance to this disease. However, there are no control measures available (Forest Service 2000). **This disease is spreading continuously but slowly, and its current status represents a departure from *historic* variation.**

PORT-ORFORD CEDAR ROOT DISEASE

Port-Orford cedar root disease, caused by the fungus *Phytophthora lateralis*, was first discovered within the native range of this tree in 1952. The origin of this fungus is still unknown. Port-Orford cedar, *Chamaecyparis lawsoniana*, is a unique tree because of its high value and limited natural range, being found only in northwestern California and southwestern Oregon on 384,000 acres of National Forest System lands and 200,000 acres of other lands. This fungus attacks the tree's root system, and infected trees die rapidly. More recently, the same fungus has been found on Pacific Yew, *Taxus brevifolia*, a source of taxol, which is an important treatment for cancer. The fungus is easily spread into previously uninfected areas via swimming spores in surface or soil water and infected soil attached to logging equipment and other motor vehicles.

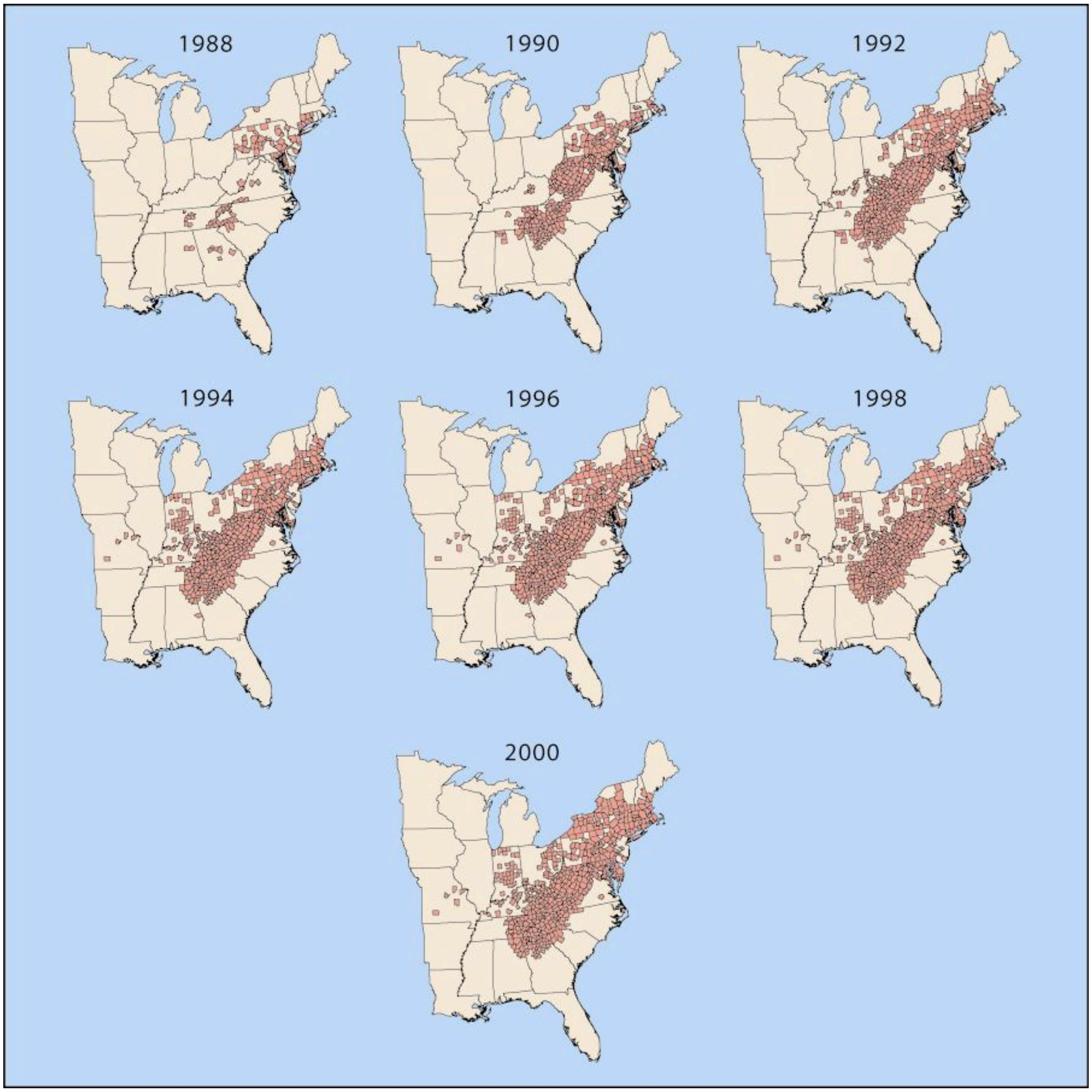

Figure 25. Changes in the known distribution of dogwood anthracnose in the Eastern United States between 1988 and 2000 (Sources: Forest Service, Forest Insect and Disease Conditions Reports 1988 – 2000)

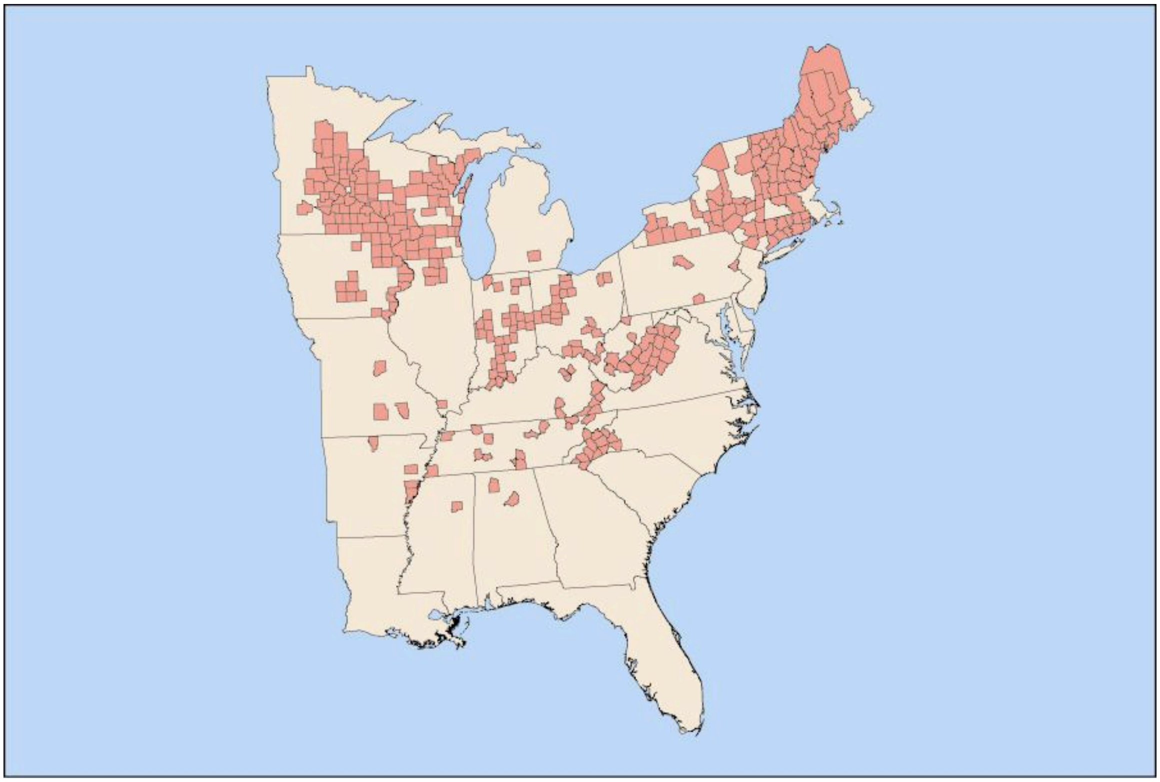

Figure 26. Known distribution of butternut canker in the Eastern United States as of 2000 (Source: Forest Service, 2001)

Management strategies have been developed to help slow the rate of spread of this disease. These include (Sammam n.d.):

1. Limiting timber harvesting operations to dry seasons.
2. Permanent or seasonal road closures.
3. Cleaning of vehicles.
4. Providing alternate drainage from roadways.
5. Selective removal of Port-Orford cedar along roads.

Approximately 10 percent of the area (about 60,000 acres) of the natural range of Port-Orford cedar is presently infected with this pathogen. Surveys in southern Oregon, conducted in 1998 and 1999, indicate that the disease was observed over an area of 1,600 acres in 1998 and 4,300 acres in 1999 (Forest Service 1999, 2000)

PITCH CANKER

Pitch canker, caused by the fungus *Fusarium circinatum*, was first discovered in the Southeastern United States where it periodically causes damage to all species of southern pines. However, it is believed to be an exotic of unknown origin.

Cankers on the boles of infected trees produce large amounts of resin. Trees with advanced infections have significant crown dieback resulting ultimately in tree mortality (Storer and others 1994).

In the United States, prior to 1986, pitch canker was only known from the Southeastern States. The disease was first recognized in California in 1986. It has also been found in various parts of the world, including Mexico, Japan, and South Africa. Within California, pitch canker is limited to coastal areas, mostly from San Diego to Mendocino counties. To date there are no confirmed records of pitch canker from the Sierra Nevada or other locations east of the central valley, or farther north than Mendocino County. All infested areas are presently on State or private lands. Worldwide, pitch canker is found in many countries. In addition to Haiti and the United States, it has been found in Japan and Mexico. In the 1990, it was reported to induce a root rot of containerized pine seedlings in South Africa and the mortality of pine seedlings in bare root nurseries in Spain (Dwinnel n.d.).

In California, pitch canker currently affects many species of pines including Monterrey pine, *Pinus radiata*, a tree that occurs in three relict stands along the California coast and is an important plantation species in the southern hemisphere (Australia, Chile, New Zealand and South Africa). It also infects Douglas-fir (Storer and others 1994). As of 1999, the total number of California counties infested with pitch canker was 19, a historic high for this pathogen (Forest Service 2000).

SUDDEN OAK DEATH

Sudden oak death (SOD) is a newly discovered disease, first detected near Mill Valley, CA, in 1995. The disease has rapidly spread throughout Marin, Monterey, Napa, Sonoma, San Mateo, Santa Clara, and Santa Cruz Counties, CA. Host trees are tanoak, *Lithocarpus densiflora*, several species of oaks, *Quercus* spp., and other plants. Symptoms include branch wilt and dieback followed by eventual tree death. The disease is caused by a newly described species of *Phytophthora, P. ramorum*. Although infected plants have also recently been discovered in Germany and the Netherlands, this disease is still considered to be of unknown origin (ODA n.d., Forest Service 2000). As of 2001, 10 California counties had confirmed cases of SOD. Moreover a small area of SOD was detected in Curry County in southern Oregon (Halstead 2001, University of California Coop Extension 2001) (Figure 27) indicating that this fungus has the potential to spread over long distances and is capable of threatening the biodiversity of the broadleaf forests of the Pacific coast. Recent seedling inoculation tests indicate that northern red oak, *Quercus rubra,* and pin oak, *Q. palustris*, trees native to the Eastern United States, are infected by this fungus. These data suggest that the oak-hickory forests of the Eastern United States could be susceptible to this fungus (SAF 2001).

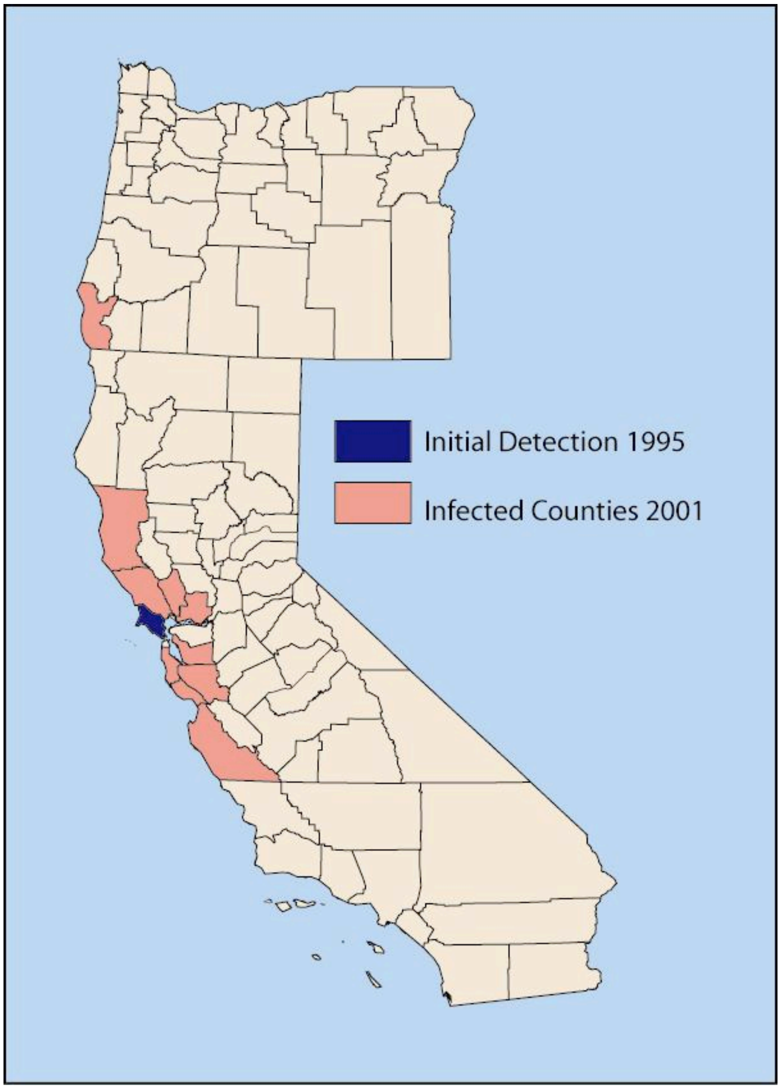

Figure 27. Known distribution of sudden oak death in California and Oregon as of 2001 (Source: University of California Coop. Extension 2001)

SHORT-TERM SPATIAL TREND ANALYSIS OF INSECT AND DISEASE CAUSED TREE MORTALITY AND DEFOLIATION

Indigenous insects and pathogens are a natural part of forest ecosystems and are essential to ecological balance in natural forests (Castello and others 1995). Their populations are influenced by climate, tree and stand vigor, biodiversity, human influences, and natural enemies. These agents influence forest succession, productivity, and stability through complex ecosystem interactions (Berryman 1986). They affect forest landscapes by causing tree mortality and/or reduced tree vigor and can occur at small scales (gap phase) or large scales (outbreaks) covering thousands of acres. Moreover, they can occur at any successional stage (Castello and others 1995).

Examining the trends of individual insects or diseases in terms of tree mortality and/or defoliation is useful in understanding the dynamics of the individual agents discussed in the preceding sections. In this section, the exposure of forests to insects and diseases is presented in terms of short-term spatial trends.

The Forest Service Forest Health Protection Program conducts annual aerial surveys to identify damage to forested areas countrywide. They record damage caused by a number of damaging agents such as insects, pathogens, and climatic events by aerial sketchmapping. Sketchmapping is a remote sensing technique used to observe and record forest damage from a small aircraft and manually recording the information on maps (McConnell and others 2000). This information is based on characteristics of overstory trees. Ground surveys are also used to assess insect and disease damage.

Using data collected in 1998 and 1999 from the aerial and ground surveys, pest agents in the nationally compiled database that cause either tree mortality or defoliation were analyzed. Short-term spatial trends in exposure of forests to mortality and defoliation were assessed on a county basis within each RPA region. Counties were used because this was the finest consistent spatial resolution of the database. Exposure was defined as the area, in acres, with mortality or defoliation causing agents present. The short-term spatial analysis was based on relative exposure (observed vs. expected exposures) on a county basis and was used to identify hot spots of activity during the time period.

Expected amounts of exposure were based on a Poisson model. The measure is referred to as relative exposure and is the ratio of observed to expected exposure. Relative exposure was calculated for mortality and defoliation agents and used to identify forested areas within RPA regions that had a higher incidence of mortality and/or defoliation as compared to the rest of the region. The calculated values ranged from zero to infinity. A value of less than 1 indicated low relative exposure and less than expected defoliation or tree mortality within a given RPA region. A value of greater than 1 indicated more than the expected exposure to defoliation or mortality causing agents in the RPA region. This measure is linear so that a relative exposure value of 2 indicates that an area has experienced twice the expected exposure, etc.

In the North RPA region, most forests had relative exposures of less than 1 to mortality causing agents. The South RPA region had a number of areas of double the expected exposure rates to mortality, largely due to outbreaks of the southern pine beetle, *Dendroctonus frontalis*. In the Rocky Mountain RPA region, hot spots of tree mortality occurred in the Black Hills (western South Dakota and eastern Wyoming), portions of Colorado, northern Idaho and western Montana, due to bark beetle outbreaks. The Pacific Coast RPA region had several areas of higher than expected mortality. These occurred in the eastern Cascades (Oregon and Washington), Blue Mountains (Oregon) and the Sierra Nevada (California) (Figure 28).

In the North RPA region, several areas had twice the expected exposure rate to defoliation causing agents for the 2-year time period. In the South RPA region, higher than expected defoliation occurred in southern Louisiana. In the Rocky Mountain RPA region, most of the defoliation occurred in Colorado, Utah, Arizona, and New Mexico. In the Pacific Coast RPA region, the Blue Mountains (Oregon), eastern Cascades (Oregon and Washington), and the Sierra Nevada (California) had large areas of forest with greater than expected exposure rates to defoliation (Figure 29).

This analysis identified several areas of greater than expected exposure to tree mortality and defoliation for each RPA region for the 1998–1999 time period. As more years of data become available, this analytical approach will help identify those areas that are continuously exposed to higher than expected levels of damage.

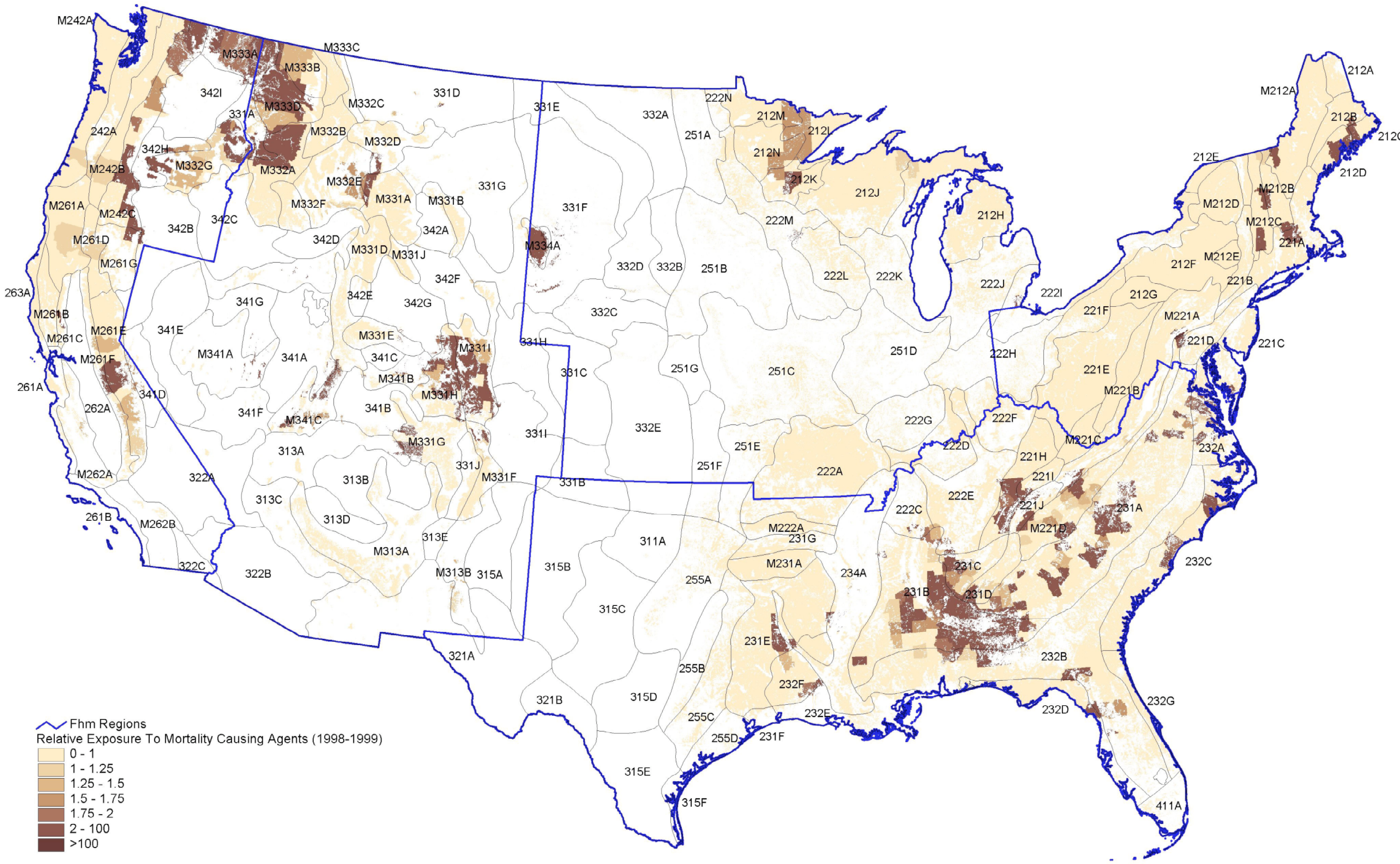

Figure 28. Relative exposure of forests to mortality causing agents by FHM region for the 1998–1999 time period

Figure 29. Relative exposure of forests to defoliation causing agents by FHM region for the 1998–1999 time period

WILDLIFE[6]

Wildlife populations, like those of insects and pathogens, are subject to periodic fluctuations. During the early days of European colonization of North America, there were many reports of abundant fish and game. For example, in 1803, as the Lewis and Clarke expedition traveled through the prairies adjacent to the Missouri River, they reported huge herds of deer, antelope, and elk (Ambrose 1996). Increased hunting pressure by Europeans, using more efficient weapons than those available to the indigenous tribes, coupled with habitat loss as a result of land clearing, resulted in drastic reductions in the numbers of many wildlife species. In the Western United States, for example, herd numbers of several species, including bison, Rocky Mountain elk and pronghorn antelope were reduced almost to extinction during the mid 1800s and early 1900s by professional hunters who supplied meat to mining and lumber camps.

Beginning in the early part of the 20th century, the American public became aware of the potential loss of key game species. As a result, revenues were raised through hunting licenses to establish State wildlife agencies whose role was to manage game populations in a sustainable manner. These programs were largely successful and many species threatened with extinction are present today in numbers sufficient to ensure their long-term survival.

In some cases, wildlife management programs have been “too successful” and large numbers of animals have caused habitat damage or have become a nuisance. In the Rocky Mountain RPA Region, for example, programs to establish viable Canada goose populations along the Colorado Front Range have been so successful that large numbers of geese, overwintering in urban areas, have become a nuisance. Moreover, the Rocky Mountain elk herd in Rocky Mountain National Park, CO, has reached such high numbers that they are a nuisance in surrounding communities such as Estes Park.

In portions of the Eastern United States, populations of white-tailed deer have reached numbers that are causing damage to plant species composition, community structure, and forest regeneration (Stromayer and Warren 1997). Two regions of the Northern United States (Northern RPA Region) have been studied intensively: the Allegheny Plateau and the Great Lakes.

Studies in the Allegheny Plateau indicate that browsing by white-tailed deer has profound effects on the establishment of forest regeneration, species composition, and density of broadleaf seedlings (Horesly and Marquis 1983, Marquis 1974, 1981). In the Great Lakes Region, studies on the effects of deer browsing suggest a replacement of conifers (e.g., hemlock, northern white cedar, and yew) by broadleaf species (Alverson and others 1988). Effects of deer browsing in other areas of the Northern RPA region include the decline of Atlantic White cedar, *Chamaecyparis thyiodes*, in the pine growing areas of New Jersey (Little and Somes 1965), and suppression of balsam fir, *Abies balsamea*, (Michael 1992).

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[6] Information on white-tailed deer presented in this section was provided by Connie Carpenter, Forest Service, Northeastern Area, Durham, NH.

These adverse impacts are related to an overpopulation of white-tailed deer. According to Jones and others (1993), the white-tailed deer has made a remarkable recovery throughout its range since being hunted to near extinction in the late 1800s and early 1900s. Protective game laws, lack of natural predators, and an abundance of early successional habitat, which produced abundant food were the factors responsible for population increase.

High deer populations were described in relation to three types of carrying capacities:

1. Cultural: The maximum number of deer that can co-exist compatibly with local human populations.

2. Biological: The number of deer that an ecosystem can support in good physical condition over an extended time period.

3. Biological Diversity: The maximum number of deer that can exist without negatively affecting floral and faunal diversity.

Plant species may be reduced or eliminated when deer numbers exceed biodiversity carrying capacity. Biodiversity carrying capacity is achieved at lower deer densities than either cultural or biological carrying capacities. White-tailed deer carrying capacities exceeding 20/mi^2 produce negative effects on the forest. The major effects are loss of tree regeneration, understory plants, biodiversity, and reductions in numbers of other wildlife species.

Studies in Pennsylvania indicate that deer can affect forest regeneration by reducing height and density and changing the species composition of seedlings and sprouts (Marquis and Brenneman 1981). Excessive browsing accounted for many regeneration failures and has virtually eliminated understory growth in many forests. Species most desirable for timber production in Pennsylvania include black cherry, *Prunus serotina*, sugar maple, *Acer saccharum*, white ash, *Fraxinus americana*. These are also the species favored by deer. Deer are also capable of removing advanced regeneration, which is the most important factor in ensuring satisfactory regeneration after a final harvest.

Understory plants are also affected when deer densities exceed 20/mi^2. Some species are eliminated and others are reduced in abundance and size. The usual result is fewer shrubs and wildflowers and more less-palatable species such as ferns, grasses, and sedges. While the more open, park-like appearance of the understory may be pleasing to some, species richness is reduced.

Excessive deer browsing also affects other wildlife species. Some of the impacts include reduced nesting sites for songbirds, changes in the composition of small mammals, reduced winter food for turkeys and reduced cover for black bear and ruffed grouse (Jones and others 1993). Species richness and abundance of intermediate canopy songbirds decreased along with nesting and foraging habitat (DeCalesta 1994).

The effects of white-tailed deer populations in the Northern RPA Region are considered to be beyond the range of *recent* variation.

INVASIVE PLANTS

Invasive plants are exotic or nonindigenous species that have been introduced into ecosystems in which they did not evolve and thus have no natural enemies to limit their reproduction and spread, thus displacing native vegetation (Federal Interagency Committee for the Management of Noxious Weeds 1998a).

Of the thousands of plant species introduced and established in the United States, about 1,400 are recognized as pests. Currently 94 species of exotic weeds are officially listed as Federal Noxious Weeds, and many more species are designated on State noxious weed lists. States with the largest number of listed noxious weeds are in the South and Pacific Coast RPA Regions (Figure 30). Experts estimate that invasive plants already infest more than 100 million acres and they continue to increase by 8 to 20 percent annually. Invasive plants are a direct threat to agricultural production and biodiversity. Croplands, rangelands, forests, parks, preserves, wilderness areas, wildlife refuges, and urban spaces are all adversely affected. The habitat of about two-thirds of all threatened and endangered species is threatened by invasive plants (Federal Interagency Committee for the Management of Noxious Weeds 1998).

On Federal lands in the Western United States, it is estimated that invasive plants or weeds occur on more than 17 million acres. Good estimates are not available for the Eastern United States. On National Forest System lands, an estimated 3.6 million acres are currently infested[7] and potentially increasing at the rate of 8–12 percent per year (Forest Service 1999).

Some methods of introduction and spread of invasive plants include:

* Contaminated agricultural seeds.
* Purposeful introduction of plants for agricultural or landscape purposes.
* Seeds attached to birds and mammals.
* Feed contaminated with weed seeds brought into wilderness areas to feed pack animals.
* Human transport of seeds attached to footwear and motor vehicles.
* Road construction in remote forest areas creates disturbed sites suitable for plant invasion.
* Wildfires resulting in bare soils suitable for occupancy by invasive plants.

[7] Personal communication, Rita Beard, Range Ecologist, Forest Service, Fort Collins, CO.

Despite integrated pest management programs designed to slow their spread and reduce ecological, economic, and social impacts, invasive plants are continuously expanding their ranges. Therefore all species are considered to be at historically high levels. The following sections describe the status of several major invasive plants that are adversely

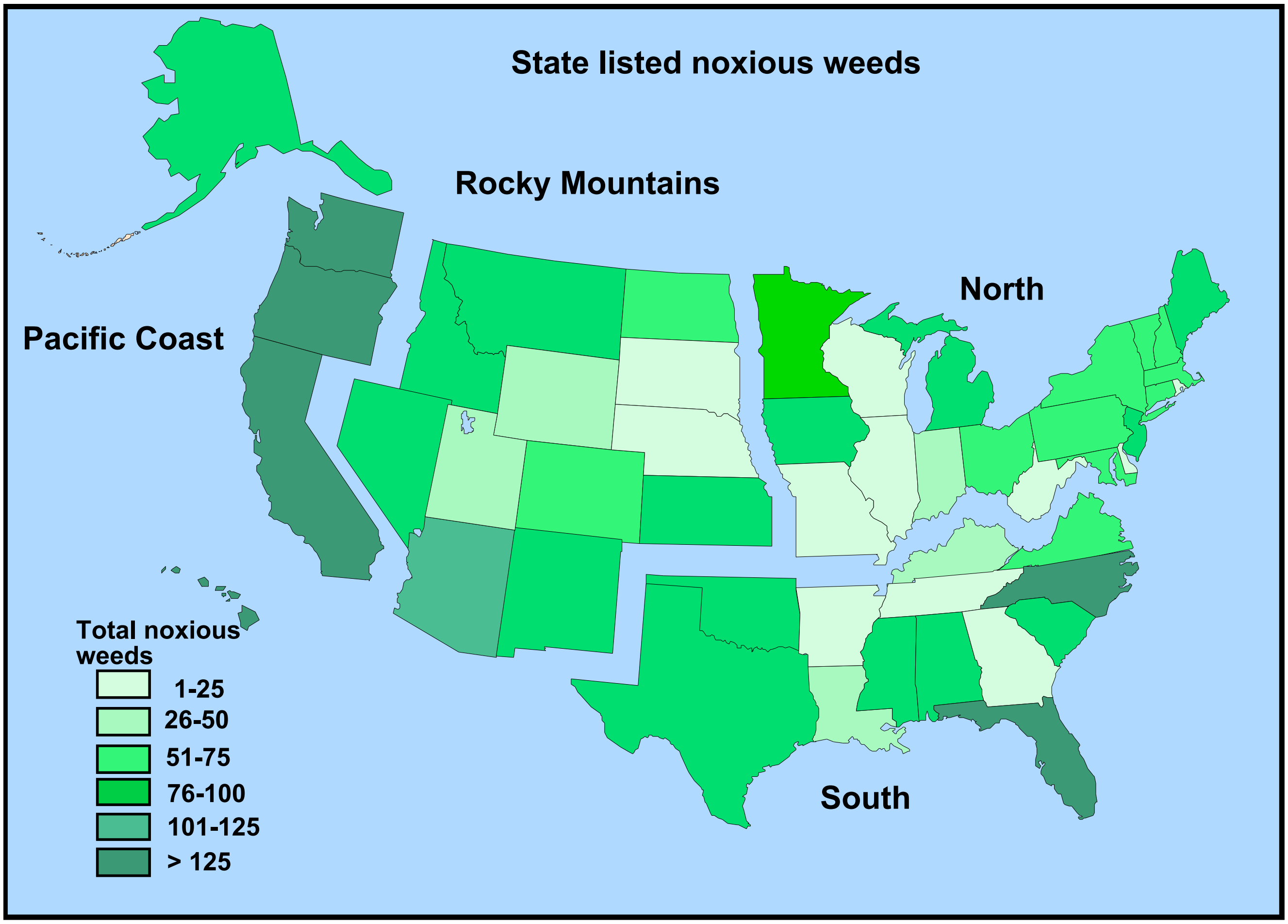

Figure 30. Numbers of State listed noxious weeds by State and RPA Region (Source: Invaders Database System, University of Montana: http:// invader.dbs.umt.edu/noxious_weeds/.

affecting forest and wildland ecosystems. All the effects of the invasive plants discussed in this analysis are considered to be outside of the range of *historic* variation.

INVASIVE PLANTS IN TROPICAL ECOSYSTEMS

Several species of invasive plants have invaded tropical ecosystems in Florida and Hawaii. While it may be inappropriate to include them in a document that addresses temperate and boreal ecosystems, they are having major ecological impacts in the areas they have invaded and are, therefore, briefly reviewed in this section.

MELALEUCA

Melaleuca, *Melaleuca quinquenervia*, is a major invasive tree problem in south Florida and also occurs in California. Also known as punk tree or paper bark tree, it is native to Australia where it is valued as an ornamental and as a source of honey. Melaleuca was purposely introduced into Florida as an ornamental in 1906 and was subsequently planted as an agricultural windbreak, soil stabilizer, and landscape ornamental around Miami. In 1936, melaleuca seeds were broadcast by airplane over south Florida in a private campaign to drain the Everglades (Westbrooks 1998).

Over the past 40 years, melaleuca has undergone an explosive invasion of wetlands in south Florida. In freshwater wetlands, melaleuca almost completely displaces native vegetation and degrades wildlife habitat. Its flowers and foliage produce volatile emanations that cause asthma-like symptoms or a burning rash coupled with headache and nausea in sensitive people. State officials estimate that melaleuca infests about 50,000 acres of native wetlands in south Florida and is expanding at a rate of 50 acres/day. A melaleuca control project is underway; however at current funding levels, the project is able to remove only 1 acre of melaleuca per day (Westbrooks 1998).

MICONIA

Miconia calvescens, commonly known as Miconia, is native to tropical forests in Central America. It begins life as a shrub but can grow to a height of 50 feet at maturity. This plant was introduced into the Hawaiian Islands as an ornamental in the 1960s. Miconia was discovered in the wild on east Maui in 1990, some 20 years after its introduction at a botanical garden in the community of Hana. It has been found at nine east Maui locations and on several other islands of the Hawaiian chain. When a miconia forest becomes established, all other plant life ends. It forms dense thickets that block sunlight from reaching the forest floor so that few understory plants are able to survive. This plant was recognized as an invasive plant in the 1980s and from 1991 to 93 some 20,000 trees were removed from private lands on Maui. In September 1993, another infestation of 250 acres was detected on Maui. Containment and control of this infestation began in 1994 (Westbrooks 1998).

INVASIVE PLANTS IN TEMPERATE ECOSYSTEMS

LEAFY SPURGE

Leafy spurge, *Euphorbia esula*, is a deep-rooted perennial herb native to Eurasia. It was brought into the United States as a seed contaminant around 1827 and is now found everywhere in the United States except the Southeastern States (Figure 31). Leafy spurge infests about 2.7 million acres, mostly in Southern Canada and the Northern Great Plains of the United States. Leafy spurge can successfully compete against native plants and often forms dense stands that crowd out most other vegetation. Infestations cause loss of plant diversity, loss of wildlife habitat, and reduction of land values (Westbrooks 1998). Leafy spurge also infests forest openings and displaces more desirable native plants.

Leafy spurge is primarily a problem on rangelands. Cattle refuse to graze in areas with 10–20 percent leafy spurge cover because the milky sap produced by the plant is a digestive irritant to cattle and causes lesions around the eyes and mouth. From a management standpoint, a range with 80 percent leafy spurge cover reduces the carrying capacity of the land to zero. In 1994, grazing capacity lost to leafy spurge in Montana, North and South Dakota, and Wyoming would have supported a herd of 90,000 cows, generating about $37.1 million in livestock sales (Westbrooks 1998).

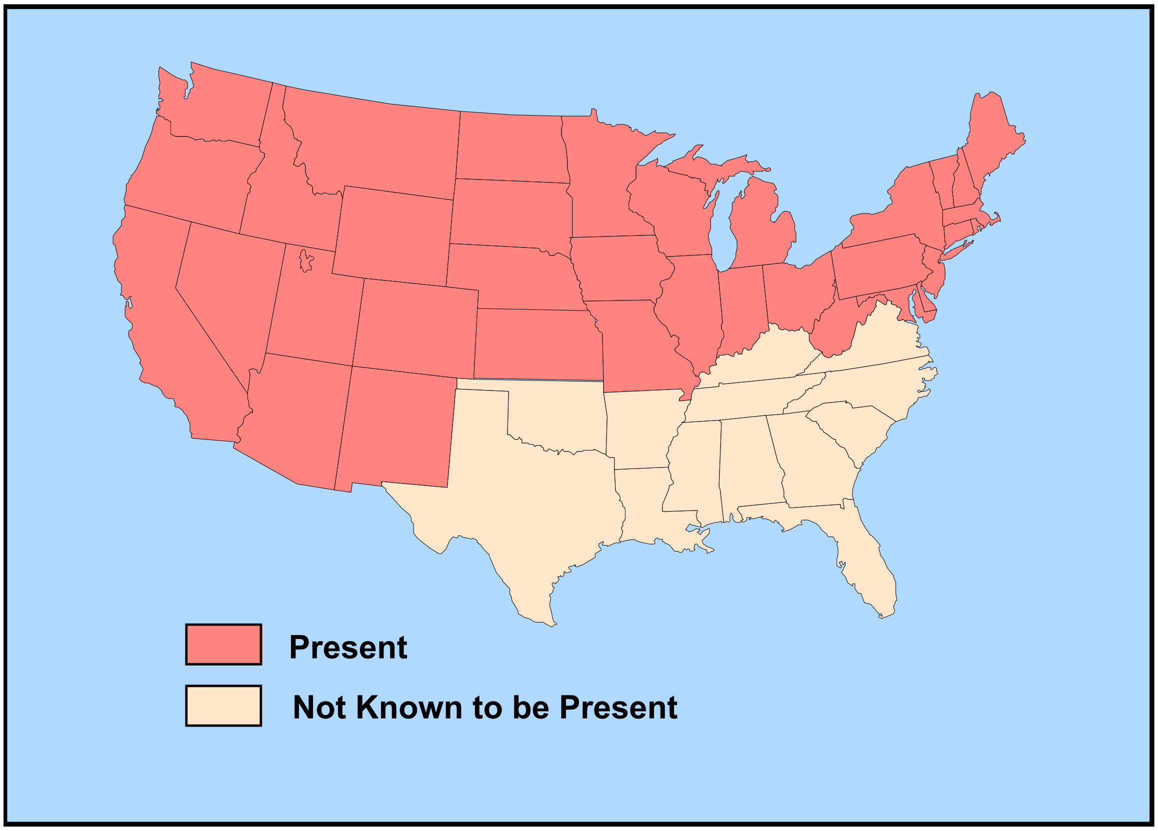

Figure 31. States with leafy spurge, *Euphorbia esula*, infestations in the United States

YELLOW STAR THISTLE

Yellow star thistle, *Centuarea solistitalis*, is an annual herb, native to dry habitats in Mediterranean Europe. This plant was first introduced into southeastern Washington as a contaminant in alfalfa seed. It is currently estimated that yellow star thistle infests 9.25 million acres of rangeland in the Western United States and is also present in a number of Eastern States (Figure 32, Westbrooks 1998). In the Hells Canyon National Recreation Area, on the border of Idaho, Oregon, and Washington, infestations of star thistle threaten the existence of a rare species of mariposa lily that grows only in Hells Canyon (Federal Interagency Committee for the Management of Noxious and Exotic Weeds 1998a).

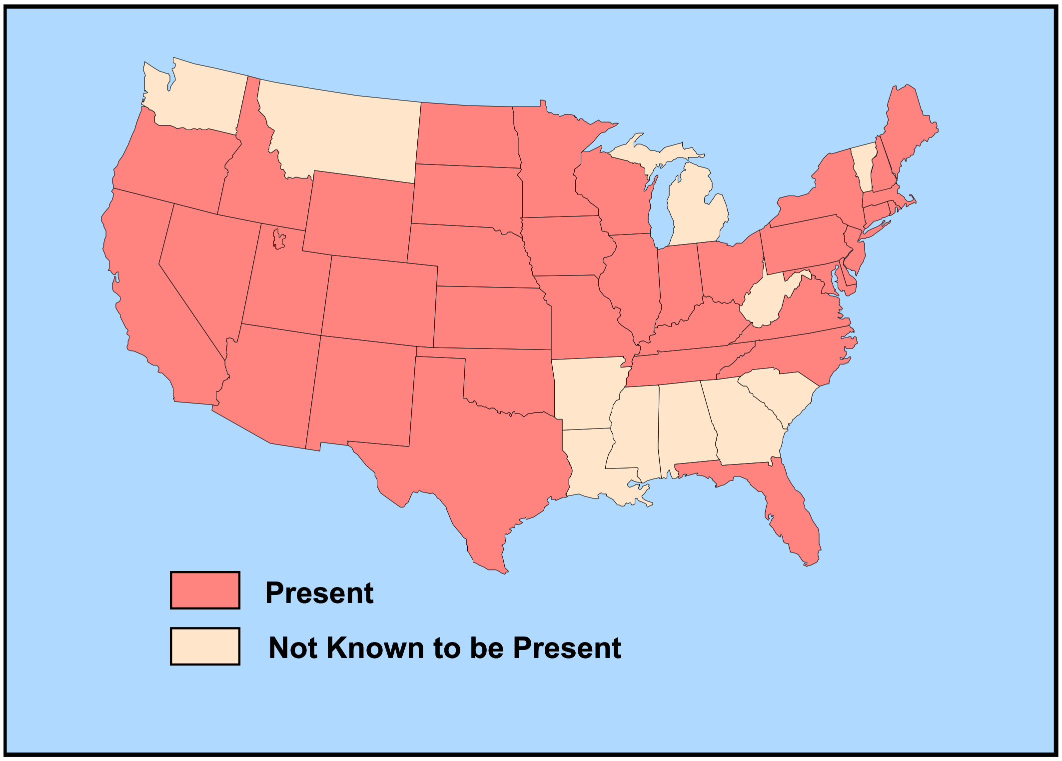

Figure 32. States with yellow star thistle, *Centuaurea solistitalis*, infestations in the United States

CHEATGRASS

Cheatgrass, *Bromus tectorum*, is a winter annual grass that was introduced from the Mediterranean Region in the late 19th century. By the 1940s, this grass had spread into every State in the United States and portions of Canada and Mexico. It is universally present on 100 million acres of rangelands in the Rocky Mountain and Pacific Coast RPA assessment subregions (Mitchell 2000).

Cheatgrass is highly competitive and has replaced most of the natural bunchgrasses in the sagebrush ecosystem of the Great Basin. Growing more densely than the naturally scattered bunchgrasses, it utilizes much of the open space between shrubs. During the dry summers of this region, cheatgrass rapidly dries out and becomes an abundant and pervasive source of fuel. The highly flammable cheat grass alters the frequency and intensity of fires on western rangelands. Instead of major fires occurring every 60 years, they now occur every 3 to 5 years (Westbrooks 1998).

By the 1950s, cheatgrass had also invaded piñon-juniper woodlands, and fires became more common in this forest type. Trees were slow to return, with little cover reestablished even after 60 years. Cheatgrass, however, was quick to reestablish. Therefore, some piñon-juniper woodlands have been replaced by annual grasslands (Billings 1994). FIA data from the Rocky Mountain Research Station indicate that the occurrence of cheatgrass in piñon-juniper forests is still relatively low, however. The westwide average of the FIA plots occurring in piñon-juniper woodlands having 5 percent or more cheatgrass cover is 1.7 percent (Table 29).

Table 29. Levels of Cheatgrass Infestation in Pi_on-Juniper Forests in the United States

State	Year of last FIA survey	Number of FIA plots In pinion-juniper forests	Number of plots with ≥ 5 percent cheatgrass cover	Percent of plots with ≥ 5 percent cheatgrass cover
AZ	1999	2065	21	1.0
CA	--	58	3	5.2
CO	1983	326	9	2.8
ID	1991	141	11	7.8
MT	1989	240	2	0.8
NV	1989	482	12	2.5
NM	2000	1919	12	0.6
SD	--	17	0	0.0
UT	1993	1483	42	2.8
WY	2001	269	8	3.0
Total		**7000**	**120**	**1.7**

Source: FIA, Rocky Mountain Research Station, Ogden, UT.

KUDZU

Kudzu, *Pueraria montana*, was introduced into the United States at the Centennial Exposition in Philadelphia in 1876 as part of a Japanese garden exhibit. The plant quickly attracted American gardeners. Later it was planted for forage. During the Great Depression of the 1930s, the Soil Conservation Service (SCS) promoted kudzu for erosion control and it was widely planted throughout the Southeastern United States for that purpose.

Kudzu is an aggressive vine that can grow as much as 1 foot/day in summer, climbing and completely covering trees, power poles, and sometimes homes. Under ideal conditions, kudzu vines can grow 60 ft/year. While kudzu is an effective means of erosion control, the vines can kill trees and damage forests by preventing trees from getting sunlight. Kudzu was declared a noxious weed in 1972 (University of Alabama 2001).

Kudzu has spread outside of the Southeastern United States and infestations have become established in a number of Northern States, including Connecticut, Massachusetts, New York, Pennsylvania, New Jersey, and Illinois. In 2000, kudzu was found on a patch of about one half acre in Clackamas County, OR. It was discovered near Vancouver, WA, in early 2002 (NAPPO 2002). The plant is now known to occur in 29 States (USDA, NRCS 2001, Figure 33), which represents a historical high in terms of its

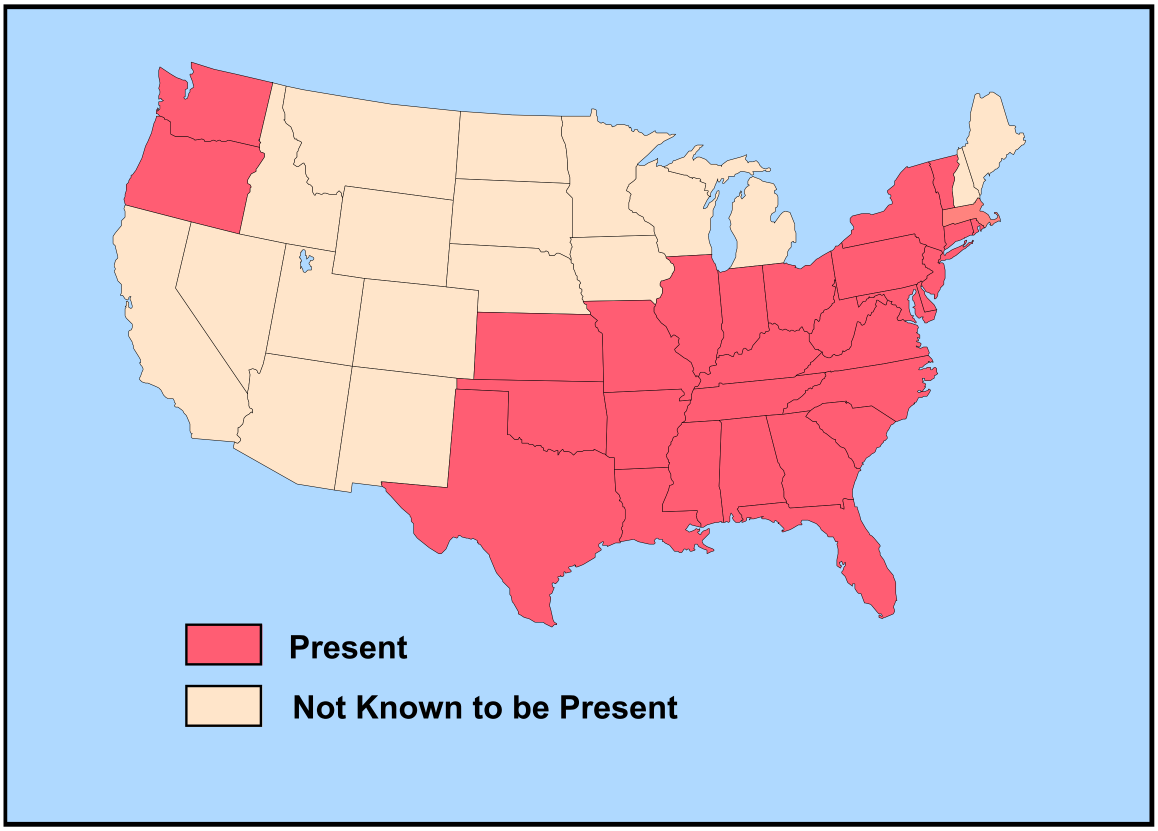

Figure 33. States with known infestations of kudzu (Source: USDA, NRCS 2001)

distribution. More than 7 million acres of forest in the Southeastern United States are estimated infested by Kudzu (Federal Interagency Committee for the Management of Noxious and Exotic Weeds 1998b).

GARLIC MUSTARD

Garlic mustard, *Alliaria petiolata*, is native to Europe and was first recorded in the United States about 1868, in Long Island, NY. This plant was probably introduced by settlers, who planted it for food and medicinal purposes. Garlic mustard poses a severe threat to native plants and animals in forest communities in much of the Eastern and Midwestern United States (Northeastern and North Central RPA subregions). Many native wildflowers that complete their life cycle in spring (e.g., spring beauty, wild ginger, bloodroot, Dutchman's breeches, hepatica, toothworts, and trilliums) occur in the same habitats preferred by garlic mustard. Once introduced into an area, garlic mustard outcompetes native plants by monopolizing light, moisture, nutrients, soil, and space. Wildlife species that depend on these native plants for their foliage, pollen, nectar, fruits, seeds, and roots are deprived of their food sources when garlic mustard replaces them. Garlic mustard poses a threat to one of our rare native insects, the West Virginia white butterfly, *Pieris virginiensis*, which feeds on toothworts, one group of plants displaced by garlic mustard (Rowe and Swearingen 2001).

Now present in 33 States, the range of garlic mustard extends from the Northeast south to Georgia and west to Colorado, Utah, and Oregon (USDA, NRCS 2001, Figure 34). Since

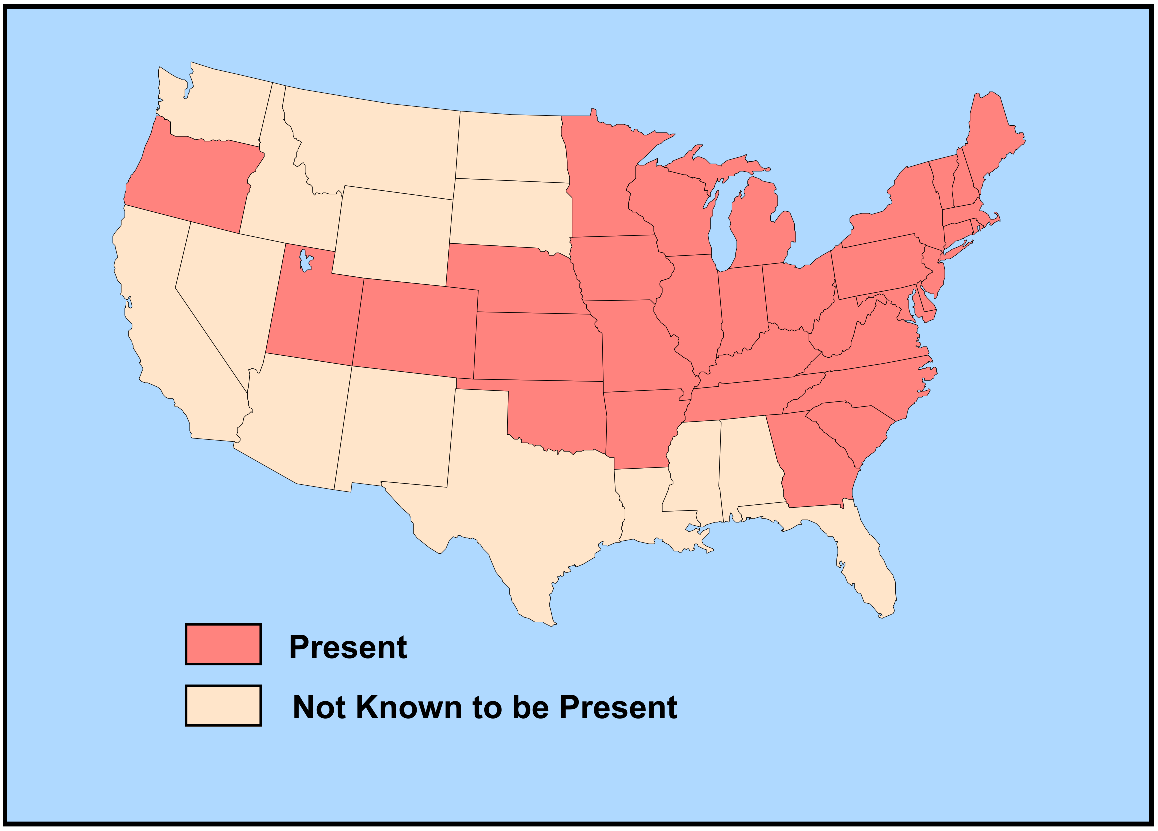

Figure 34. Known distribution of garlic mustard in the United States (Source: USDA, NRCS 2001).

this plant is spreading continuously, its present distribution represents a historic high, beyond the range of historic variation.

MULTIFLORA ROSE

Multiflora rose, *Rosa multiflora*, was introduced into the Eastern United States from Japan in 1866 as a rootstock for ornamental roses. During the 1930s, the USDA SCS promoted it for use in erosion control and as living fences to control livestock. State agencies found it to provide excellent cover for several species of game birds. Its tenacious and unstoppable growth habit was eventually recognized on pastures and unplowed land where it interfered with cattle grazing.

This plant is extremely prolific and can form impenetrable thickets that exclude native plant species. Multiflora rose readily invades open woodlands, forest edges, successional fields, savannas, and prairies that have been subjected to land disturbance (Bergmann and Swearingen 1999).

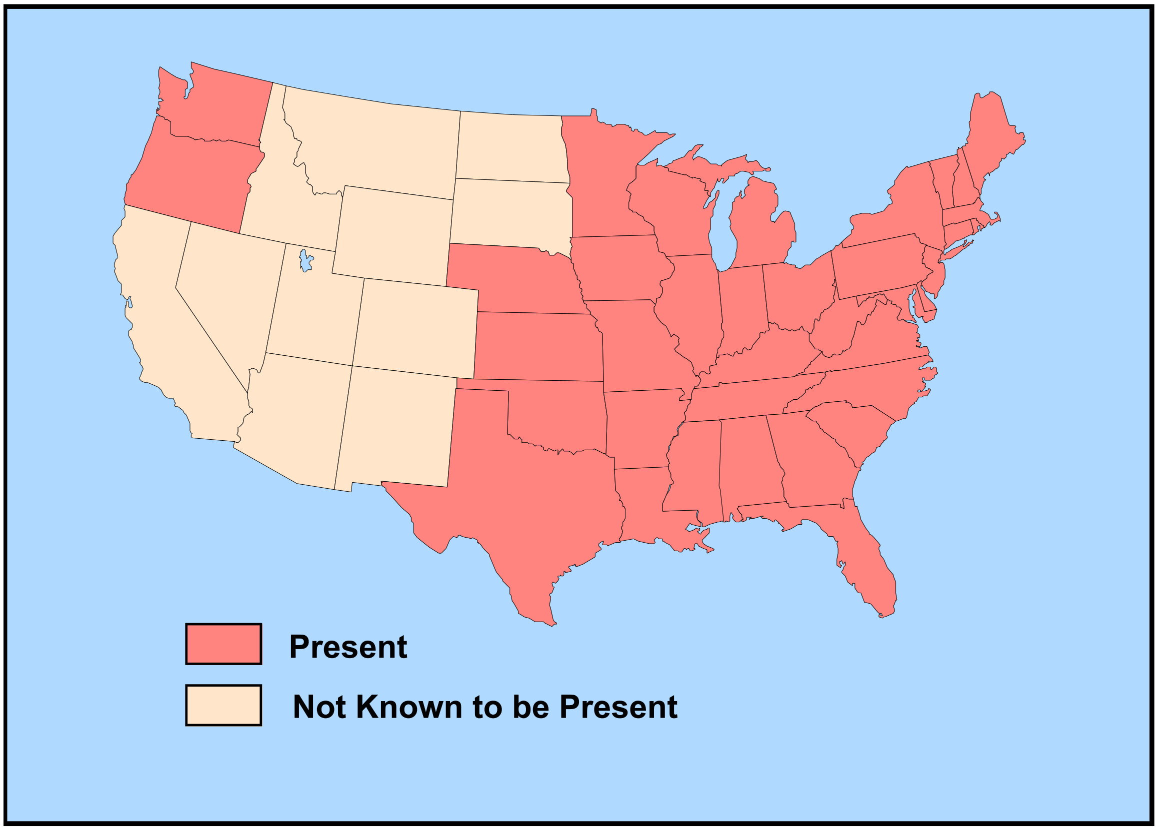

Figure 35. Known distribution of multiflora rose in the United States (Source: USDA, NRCS 2001).

Multiflora rose is now present in 37 States including all of the Eastern States and Washington and Oregon (USDA NRCS 2001, Figure 35). This distribution represents an historic high.

INVASIVE TREES

Several species of trees introduced into the temperate regions United States for a variety of purposes (e.g., ornamentals, potentially fast growing trees for lumber) have escaped cultivation and are considered in some areas to be invasive. Examples include:

NORWAY MAPLE, *Acer plataniodes*

Norway maple is a widely planted ornamental and street tree, native to Europe, that is often found to escape from cultivation in vacant lots and is now spreading into successional forests. It is listed as invasive in Maine, Maryland, Massachusetts, New York, Ohio, Pennsylvania, and Vermont (Shackleford and others 1998).

TREE OF HEAVEN, *Ailanthus altissima*

Tree of heaven is native to Eastern Asia and has been widely planted as an ornamental, especially in the Eastern United States. It readily escapes from cultivation and invades sites in alleys, meadows, dumpsites, on shores and riverbanks, and along railroad tracks. It is reported as invasive in Connecticut, Indiana, Iowa, Maryland, Massachusetts, Missouri, New York, Ohio, Pennsylvania, Vermont, and Wisconsin (Shackleford and others 1998).

RUSSIAN OLIVE, *Elaeagnus angustifolia*

Native to Eastern Asia, Russian olive has been widely planted in the Eastern and Midwestern United States. In the Great Plains it has been used widely as a shelterbelt or windbreak species. This tree readily escapes cultivation and invades fields and riverbanks.

AUTUMN OLIVE, *Elaeagnus umbellate*

Also native to Eastern Asia, autumn olive has been planted as an ornamental or for wildlife habitat. It readily escapes to roadsides, forests, fields, gravel pits, and other habitats. It is reported as invasive in 11 Eastern States and is listed as a noxious weed in 23 counties of West Virginia (Shackleford and others 1998).

EMPRESS OR PRINCESS TREE, *Pawlonia tomentosa*

This fast growing tree, native to China, was initially used as an ornamental tree. Timber plantations have also been established. It has escaped cultivation in the Eastern United States. In warm climates, it can grow in almost any habitat and is often seen in vacant city lots. It is of most concern in the Mid-Atlantic and Southeastern States (Shackleford and others 1998).

SALTCEDAR, *Tamarix spp* ***Saltcedar***

Native to the arid regions of Eurasia, Saltcedar was first introduced into the United States as an ornamental in the early 19th century and has become a robust invader of riparian rangeland ecosystems over the past 60 years. Saltcedar has replaced native riparian trees such as cottonwoods and willows. In some areas, water management practices and dam construction have stopped repeated scouring of riverbanks, thus further reducing the competitiveness of the native trees. Saltcedar has an extremely high transpiration rate and can lower water tables on its own, giving it a competitive advantage.

Saltcedars have successfully invaded nearly every drainage system in arid and semi-arid areas in the southwestern United States and occupy more than 1 million acres. Saltcedars now occupy most suitable habitats west of the Great Plains, north into Montana, and south into Northwestern Mexico (Westbrooks 1998).

Since all of these invasive trees are continuously expanding their ranges, their current distributions can be considered to be at historic high levels.

DISCUSSION AND CONCLUSIONS

A number of processes and agents affect the dynamics of temperate and boreal forest ecosystems in the United States. They include climate, fire, insects, diseases, and invasive plants. While many of the indigenous agents and processes are an integral part of the dynamics of these forest ecosystems, and essential to maintaining forest health, they may be as pests because they interfere with management objectives.

The dynamics and character of the forests of the United States have been significantly affected by human influences. These include:

- Clearing of land for agriculture, a portion of which has been abandoned and has regenerated to a second growth forest.

- Timber harvesting, establishment of forest plantations and other forest management activities.

- Protection of forests from wildfire, which reduces the natural fire cycle and results in changes in fuel conditions, species composition, and tree stocking levels.

- Accidental or purposeful introduction of exotic insects, diseases, and plants, some of which have caused ecological, economic and social impacts.

These influences have altered the behavior of some indigenous processes and agents. Moreover, a number of new agents have been introduced into U.S. forests.

A number of data gaps exist, which preclude a more complete assessment of the processes and agents affecting U.S. forests. For example, historical data on the area burned by wildfires for the South and North RPA Regions was not available at the time this writing. Suitable historic or modern data could not be found to represent the effects to forests from permanent flooding, salinization, or domestic animals. Moreover, despite the fact that national forest insect and disease conditions reports have been prepared since the early 1950s, consistent historic data on the status of forest insects and diseases is available for just a few important species from 1979 to the present. Distributional data on many invasive insects, pathogens, and plants is generally limited to States or counties within States that are infested by a given species. The lack of metric historical data for many agents and processes may make it impossible to *ever* establish a true baseline condition for the historic time period (1800–1850).

During the period of this analysis, several agents and processes exceeded the known range of *historic* and/or *recent* variation:

1. The period 1997–1998 was influenced by an El Niño of historically high proportions followed by 2 years of La Niña, which resulted in lower than normal precipitation over much of the United States and is considered to be outside the range of *recent* variation.

2. An ice storm, which occurred during January 1998 in the Northeast and affected 17.5 million acres (38 percent of the region's forests), is considered to be outside the range of *recent* variation and probably beyond the range of *historic* variation.

3. Nationally the 2000 fire season, with more than 8 million acres burned, is beyond the range of *recent* variation based on one data source for the area burned by wildfires in the United States between 1960 and 2000. In the West another data source for area burned indicates that the range of *recent* variation (since 1916) was exceeded in 1996, 1998, and 2000 when wildfires in the 11 Western States burned more than 4 million acres per year.

4. Several species of indigenous insects (southern pine beetle, mountain pinebeetle, spruce beetle, eastern and western spruce budworms, and Douglas-fir tussock moth) reached extremely high levels of activity between 1973 and 1996 and have declined in recent years, primarily due to decimation of susceptible host types. During the current analysis period the southern pine beetle reached outbreak proportions in several areas outside of its "normal or traditional" epidemic range. These outbreaks, which occurred in central Florida, southeastern Kentucky and southeastern Arizona, are considered to be outside the range of *recent* variation and *may also be a departure from historic variation*. The area affected by mountain pine beetle in the Western States reached a recorded high in 1981 and may be beyond the range of historic variation. In 1996, a long-standing outbreak of spruce beetle in Alaska reached a recent high that is beyond the range of *recent variation and may also be a departure from historic levels.* A major eastern spruce budworm outbreak in Maine reached a recorded high in 1978 that probably exceeded the range of *historic* variation. Similarly, in 1986 western spruce budworm reached a recorded high in the West, when more than 13 million acres were defoliated, that is probably beyond the range of *historic* variation. A recorded high for Douglas-fir tussock moth in 1973 probably exceeds the range of *historic* variation.

5. Dendrochronological studies suggest that recent western spruce budworm outbreaks throughout the Rocky Mountains and in Oregon are now more synchronous, extensive, and severe than those that occurred prior to 1850 and, therefore, are considered to be beyond the range of *historic* variation.

6. While epidemic populations of most indigenous forest insects were at lower levels during the current analysis period than during recent history represented by metric data (e.g., 1979–1995) they still caused serious regional and local damage to forests. Management actions to prevent or suppress some of these insects are underway (e.g., bark beetle species in the West). Moreover, the potential for increased insect activity is high, especially the hazard of bark beetle outbreaks in areas damaged by the fires of 2000.

7. Several species of indigenous pathogens are considered to be at levels that are causing effects beyond the range of *historic* variation. Areas infested by dwarf mistletoes, parasitic plants of western conifers, have probably increased due to fire exclusion. Fusiform rust, a disease of southern pines, has increased dramatically as a result of intensive forest management and the extensive use of plantation species susceptible to this disease. The effects of western root diseases, resulting from an increase in Douglas-fir and true firs caused by the loss of western white pine and fire exclusion, are considered to be beyond the range of *historic* variation. Oak wilt, a tree killing disease of oaks, is causing extensive losses in live oak woodlands in central Texas where it is considered to be "outside of its normal range." Also, a severe occurrence of oak decline and mortality in portions of Arkansas accompanied by an aggressive infestation of red oak borer is believed to be causing effects that are beyond the range of *historic* variation.

8. Populations of white-tailed deer in parts of the Northern RPA Region are at high levels and are causing damage to forest regeneration and understory plants. This damage is considered to be beyond the range of *recent* variation.

9. All exotic insects and diseases, diseases of unknown origin, and invasive plants considered in this analysis are believed to have been introduced or at least not established until after 1850 and therefore all their effects are beyond the range of *historic* variation. Some exotic species have had significant effects on U.S. forests, for example, chestnut blight, Dutch elm disease, white pine blister rust, and gypsy moth.

   a. The area defoliated by the exotic gypsy moth in the Eastern United States during the current analysis period (1996–2000) was well within the range of *recent* variation (1924–1995), but continued to spread South and West. In 2000, defoliation was detected for the first time in Wisconsin. Several recent exotic introductions (e.g., European pine shoot beetle and Asian longhorned beetle) are causing severe damage and have high potential for expanding their ranges.

   b. Several diseases of unknown origin are continuing to expand their geographic ranges and cause severe damage. Dogwood anthracnose has eliminated flowering dogwood from many eastern forests and a disease known as sudden oak death, first discovered in California in 1995, is affecting an increasing area of tan oak and oak forests.

c. More than 1,400 species of exotic invasive plants are known to occur in the United States. Many affect forest ecosystems by displacing trees or understory vegetation. The ranges of many of these plants are continuously expanding throughout the United States, despite pest management measures.

d. The largest number of exotic invasive species introductions has occurred in the coastal regions of the South and Pacific RPA regions (Figure 36).

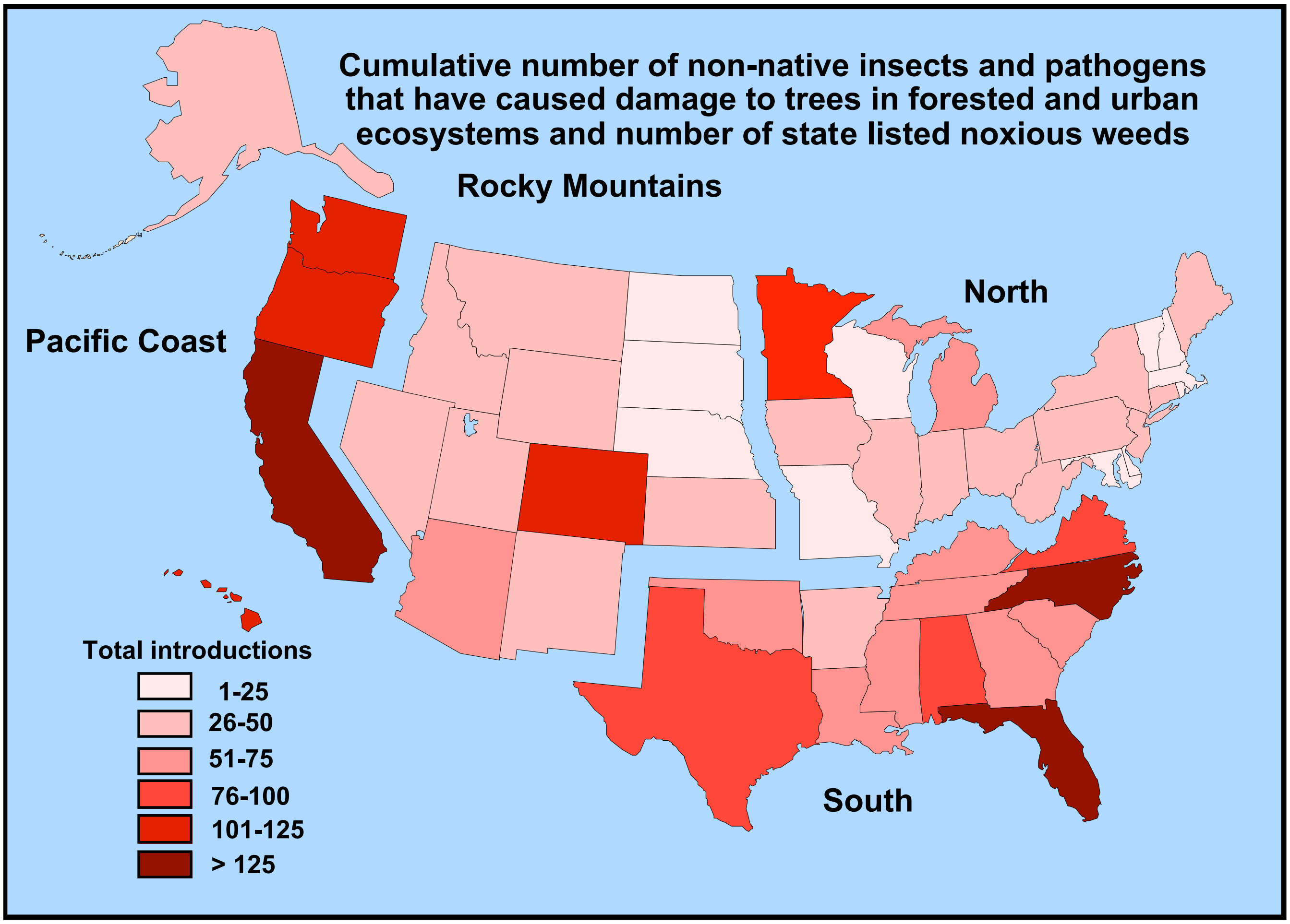

Figure 36. Cumulative number of nonnative insects and pathogens that have caused damage to trees in forested and urban ecosystems and number of State listed noxious weeds (Sources: R. Pywell, USDA Forest Health Technology Enterprise Team, Ft Collins, CO. Nonnative insects and pathogens that have caused damage to trees in forested and urban ecosystems [draft web pages]; Invaders Database System, University of Montana: http:// invader.dbs.umt.edu/noxious_weeds/)

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