Chapter 16: Insect and Disease Pests of Southern Forests

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Key Findings

- Nonnative pest species have increasing impacts in the South regardless of climate change, patterns of land ownership, or changes in the composition of vegetation.
- "New" nonnative invasive insects and diseases will have serious impacts on southern forests over the next 50 years. Some species such as emerald ash borer, laurel wilt and thousand cankers disease are expanding rapidly; they threaten the ecological viability of their hosts throughout large areas of the South.
- Given the trend in introductions of non-native insect pests and plant pathogens over the last 100 years, we can expect additional introductions of previously undocumented pests (insects, fungal pathogens, plant parasitic nematodes, etc) from foreign countries that will have serious consequences for some native forest plant species.
- When host material for a given insect or disease is projected to increase over the next 50 years as a result of climate change or management choice, we can expect more pest activity; for example, more pine acreage enables more southern pine beetle damage. Conversely, if host material decreases, the overall impact of pests utilizing that host material will likely decrease.

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- Very few indisputable projections can be made about the effects of climate change on native or naturalized pests. Although climate-change-induced host abundance is expected to increase the activity of some pests, others (such as gypsy moth) may become less active with warmer temperatures despite relatively similar levels of host availability.
- The scientific literature and the body of expert opinion are inconclusive in predicting the effects of climate change on many pests' activity levels, often even lacking historic trend data. However, based on anecdotal reports from professionals, and in the absence of other data, we generally assume that pest activity levels over the next 50 years will be similar to the past 50 years with respect to impact on preferred hosts.
- A significant source of uncertainty in projecting pest impacts is the adequacy of prevention and suppression methods: how effective are existing methods, compared with those that might be available in the future; how willing and able are land managers or landowners to adopt management/control methods; how much funding is available compared to the amount needed for implementation.
- Under the influence of climate warming host plants, pests and pest complexes are expected to migrate northward and to higher elevations. Because migration rates differ among the affected species, migrating plants are expected to form new associations, which will then affect the pests, their host populations, and the interactions among them. Unexpected pests very likely will become important, while some that are currently active will be less severe in their new habitats. As host plants "migrate" to the north an increase in the incidence of decline syndrome of plants in their previous range is expected.
- Although not expected to be a significant problem in the next 50 years, the migration of lower elevation plants to higher elevations could ultimately eliminate or at least severely restrict the host

ranges of current high elevation plant associations. Pests that act on a restricted host base, such as the balsam woolly adelgid and butternut canker, could become far more significant ecologically in areas of relict host populations.

- Climate change will lead to extra uncertainty in decisionmaking, especially in areas where the changes cause increased variability in local (fragmented) climate regimes that exceed historical variability of local weather patterns.

Introduction

An important part of the southern forested landscape is the array of insect and disease pests that significantly affect the management of forest resources on a relatively broad scale. The list of 21 key pests that were documented less than a decade ago (Ward and Mistretta 2002) has already expanded to 30.

The goal of this chapter is to project the behavior of insect and disease pests that we anticipate will affect forest resources over the next 50 years, based on changing climate, human activity and biologic factors. Our primary focus is on climate change and its sub-elements of temperature régime (dominated by temperature extremes), overall pattern of solar radiation, and rainfall pattern. All available climate change scenarios predict an environment in which we expect vegetation changes to occur (Iverson and others 1999). Concurrent with ecological changes will be a shift in the pests that function within an altered vegetative landscape under changed temperature, rainfall, and other climatic conditions. The impacts on pest activity, in turn, may influence the direction or scope of other changes in forest type and structure.

The focus of this chapter is the 30 species of pest insects or fungal pathogens that cause diseases projected to be of future concern, with emphasis on the following key issues:

- The historical and forecasted future spread of high threat insects and diseases
- Other pests invading southern forests and other high threat species poised to enter the region
- Expected consequences of the spread of high-threat pest species for forest productivity, ecosystem composition and biodiversity, threatened and endangered species and their habitats, watershed and soil health, carbon storage, and fire dynamics
- Potential severity of pest species threats relative to other threats and to future forest sustainability
- Forest species or populations that are likely to be lost or dramatically degraded by pests; the resulting changes in the composition of southern forests over the next 50 years; and the degree of certainty in these outcomes
- Adaptive strategies and methods for invasive pest management that could mitigate the effects of predicted future outbreaks

Methods

In response to the issues developed above, we present a brief extract of relevant information about the pests that are well established in southern forests (Ward and Mistretta 2002); we add more detailed descriptions of several new pests or pest complexes that have emerged in the past few years; we apply the results of past research on pests and pest management to expected changes in southern forests over the next 50 years; we identify management strategies for responding to pests in a changing environment; and, we identify research needed to improve our knowledge about pests with their hosts and their interactions with their changing environment, thereby enabling a more quantitative approach to forecasting in the future.

Data Sources

Information for this chapter is derived from two primary sources, selected items from the extensive body of published scientific literature, and the experience of the authors and their colleagues in State and Federal agencies, universities, and other public or private organizations that are engaged either in research or field-based pest management activities. Additional information about forest pests and their control is readily available from State and Federal forestry agencies or on the Internet (two good starting points are http://fhpr8.srs.fs.fed.us/ and http://www.na.fs.fed.us/spfo/pubs/fth_pub_pages/fidl.htm). Also, Appendix B contains additional resources ("References") not cited here but which provide valuable additional background for understanding the biology and ecology of the pests discussed.

Results

The scientific literature on climate change and other environmental considerations is summarized in Appendix D, which also provides the background information on our approach to pest modeling and future projection of impacts. Below we address the pests projected to influence the forests of the South over the next 50 years, their potential damage, potentially effective management strategies, and research needed to better understand and manage them (table 16-1).

Of the 30 forest pests in the South discussed below, 21 are well established and 9 are relative newcomers. Pests are roughly evenly divided between those affecting softwoods and those affecting hardwoods.

Insect Pests of Softwoods

Balsam woolly adelgid—Impacts of balsam woolly adelgid, *Adelges piceae*, were first documented in 1957 on Fraser fir in the Southern Appalachians. The five major areas of high-elevation, spruce-fir forest in North Carolina, Tennessee, and Virginia are highly valued for their scenic and recreation values, attracting several million visitors annually (Ward and Mistretta 2002). In addition, several species of

flora and fauna rely on mature spruce-fir habitat for survival, and many are found only in this environment. The balsam woolly adelgid has infested Fraser fir in all five areas. Damage caused by the adelgid has degraded scenery and recreation value and put this habitat of dependant tree species at great risk.

The spruce-fir forests of the Southern Appalachians are declining (Dull and others 1988, Nicholas and Zedaker 1990, Hollingsworth and Hain 1991). Balsam woolly adelgid has eliminated 95 percent of mature Fraser firs, and mortality continues at a steady rate. The residual population consists of trees generally younger than 40 years. Several laws enacted to maintain limited or threatened ecosystems and preserve spruce-fir forests direct the management of Fraser fir and provide decisionmaking guidance for resource managers. Insecticides are effective for control of this adelgid in Christmas tree plantations, but they are not feasible in forested settings.

Increased temperature and decreased precipitation will likely have the effect of both shrinking the range of spruce-fir forests now isolated on mountaintops and increasing adelgid activity and damage. If these trends continue unabated, natural populations of southern Fraser fir could disappear over the next 50 years. In addition, northern firs in the Lake States, New England, and Canada may become more susceptible to infestation as a result of milder winters and greater survival of the insect.

Hemlock woolly adelgid—Hemlock woolly adelgid, *Adelges tsugae*, an Asian native, was first identified in the early 1950s in Richmond, VA. Over the past 20 years, it has expanded rapidly into the southern range of eastern hemlock (*Tsuga canadensis*) (fig. 16-1). Hemlocks generally die within five years of initial infestation by this adelgid (McClure 1987), however some trees may live longer before succumbing.

Eastern hemlock is an important component of riparian ecosystems, providing streams with cooling shade and nutrient-rich litterfall, and wildlife with winter shelter. This tree may also be important as a feeding and nesting niche for neotropical migrant birds (Rhea and Watson 1994). The ecology of Carolina hemlock, *T. caroliniana*, is less understood. Although it generally occupies drier sites on ridges and rock outcrops, it is as likely as the eastern hemlock to serve as cover and nesting habitat for birds and small mammals.

Given the adelgid's current rate of spread, it could infest nearly the entire southern range of eastern hemlock and Carolina hemlock within the next 50 years. Some isolated areas within the infested range and some areas of hemlocks that are separated from the main range (in northwestern Alabama, for example) may escape infestation. In all likelihood, within the next 50 years hemlock woolly adelgid will kill most of the hemlocks that are alive today in the South. The loss of hemlock will be one of the major impacts caused by non-native invasive species to Southern forests in the next 50 years

A number of suppression tactics show some promise for preventing the loss of significant numbers of hemlocks over the next 50 years. Treatment of trees with imidacloprid effectively controls hemlock woolly adelgids for several years (Cowles and others 2006). Distribution of the insecticide into tree crowns is more effective with soil drench or injection than with stem injection (Dilling and others 2010). Dinotefuran is also being used with success. Current insecticide treatments are applied to individual trees and function primarily as a temporary protection measure for a relatively small number of trees. At this time, insecticide application over large areas is neither logistically feasible nor cost-effective. Several biological control agents (beetle predators) have been and are being released, and some are successfully establishing (Mausel and others 2010). More time is likely needed before conclusive impacts of biological control agents on the health of hemlock forests can be shown. Establishment of a

complex of natural enemies in a given area is desired to achieve long-term success. In June 2009, researchers and forest health professionals began evaluating the efficacy of *Lecanicillium muscarium*, an insect-killing fungus that is registered as a bio-pesticide in Europe (Grassano 2008).

Research and work is being done on hemlock host resistance and ex-situ conservation of hemlock seedlings and genetic diversity (Bentz and others 2002, Jetton and others 2008, In press, Montgomery and others 2009, Pooler and others 2002, Potter and others 2008, in press). These efforts may allow scientists and land managers to reintroduce adelgid-resistant hemlocks in the future.

Climate change is unlikely to reverse the spread of hemlock woolly adelgids. In the northern part of the hemlock woolly adelgid range, low minimum winter temperatures can significantly knock back populations and appear to limit spread. Therefore, we can assume that climate warming would likely promote a northward expansion of the adelgid (Paradis and others 2008). The southern range of hemlock is currently not benefitting from much cold winter knockback—a warming climate would presumably only exacerbate the situation.

Nantucket pine tip moth—The Nantucket pine tip moth, *Rhyacionia frustrana*, is one of the most common forest insects in the South (Berisford 1988). Although it is usually considered a southern pest, its range includes most of the eastern half of the United States.

Most commercial pine species are susceptible to attack by the Nantucket pine tip moth, but there are considerable differences in relative susceptibility. Among the southern pines, longleaf nursery seedlings and all ages of shortleaf, loblolly, and Virginia pines are highly susceptible, while slash and older longleaf pines are highly resistant.

Damage is normally transitory or negligible in forest stands but can be severe for seedlings and saplings younger than 5 years, resulting in deformities and loss of growth.

Based on the warmer and possibly drier climate that is expected over the next 50 years, the activity and damage levels of Nantucket pine tip moth are likely to increase in the South and extend to northern areas (Midwest, New England) where tip moth has not been much of a management concern. Activity may increase and continue into the winter months, as could the number of generations per year. Nantucket pine tip moths are primarily a problem in young loblolly monocultures. To the extent that land managers increase the planting of loblolly monocultures in the next 50 years, damage from the Nantucket pine tip moth is likely to increase.

A number of effective, chemical control options exist for this pest (Asaro and others 2003). If population levels are monitored in a timely and regular fashion, and are followed up by appropriate insecticide applications, tip moth damage can be minimized. Chemical control options are effective, especially the new systemic insecticides. However, they are often prohibitively expensive and will probably not be adopted under most commonly accepted climate scenarios unless tip moth population pressure becomes quite high.

Other bark beetles—Although the southern pine beetle is the most damaging insect in southern pine forests, it is only one of five pine bark-beetle species of concern for forest managers in the South. The others are the six-spined engraver, *Ips calligraphus*, the southern pine engraver, *Ips grandicollis*, the small southern pine engraver, *Ips avulsus* and the black turpentine beetle, *Dendructonus terebrans.* These beetles are usually considered secondary pests because they normally infest only stressed, weakened, damaged, or downed pines. They also colonize pines that have been attacked by southern pine beetles or another bark beetle species. Host species in the South include loblolly, shortleaf, Virginia, longleaf,

eastern white, pitch, slash (*P. elliotii*), and sand (*P. clausa*) pines. Both pure pine and oak-pine stands may be affected (Conner and Wilkinson 1983, Smith and Lee 1972, U.S. Department of Agriculture Forest Service 1985a).

Attacks by black turpentine beetles may continue for several months but infestation is not always fatal. Multiple attacks around the entire circumference of the tree are required to cause mortality (Smith and Lee 1972, Staeben and others 2010, U.S. Department of Agriculture Forest Service 1985a).

The small southern pine engraver and the six-spined engraver are the most aggressive and may kill small groups of trees. Losses may be extensive during periods of drought (Conner and Wilkinson 1983, U.S. Department of Agriculture Forest Service 1985a).

The secondary bark beetles play a vital role in shaping forest structure and may have a greater impact on regulating pine stands than southern pine beetles (Paine and others 1981; Thatcher 1960a). They attack weakened or severely stressed trees and those reaching senescence. Large infestations develop only occasionally, usually after widespread environmental stress, such as that caused by drought, storm damage, or wildfire. Their action serves to thin the pine forests, reducing competition, leaving the stronger trees, and decreasing the risk of SPB outbreaks.

The impact of these beetles depends largely on management activities (Coulson and others 1986). Engravers also breed in downed material, so it is difficult to substantially reduce populations, but prevention methods (such as lowering planting densities, thinning stands, and cutting and removing groups of infested trees) can reduce damage.

In unmanaged stands, they attack single trees or small groups of pines and reduce pine basal area. They provide openings for pine reproduction or for established hardwoods to grow. The effects are

often not noticeable except during periods of extended drought, after storm damage, or at the end of SPB epidemics.

Increased temperature and decreased precipitation would stress pines and could therefore increase the impacts of these bark beetles, but it is unlikely that they will become primary pests that kill large areas of trees. These four bark beetle species may move northward as winters become warmer.

Pine reproduction weevils—Pales weevil *(Hylobius pales)* and pitch-eating weevil *(Pachylobius picivorus*) are two of the most damaging insect pests of pine seedlings in the southeastern United States. In southern forests, they are found wherever pine occurs. Adult weevils of both species are attracted to newly harvested sites, where they breed in logging slash, stumps, and old root systems; they cause economic losses by feeding on the bark and often killing planted seedlings. If seedlings are planted on or adjacent to sites with fresh stumps or damaged trees, it is common to have 30 to 60 percent weevil-caused mortality among first-year seedlings, with instances of 90 percent or more mortality recorded (Thatcher 1960b). A third species, the eastern pine weevil (*Pissodes nemorensis*), is generally less common but is known to kill terminal and lateral branches and girdle the stems of small trees (Doggett and others 1977, Nord and others 1984). The reproduction weevils are almost never a problem in forest management unless seedlings have been planted on or adjacent to sites with fresh stumps or damaged trees. Foresters usually avoid this problem on cutover sites by delaying planting or by planting treated seedlings.

Pales and pitch-eating weevils prefer loblolly, shortleaf, pitch, and eastern white pines. They almost never attack longleaf and slash pines, but on rare occasions have been observed feeding on hardwoods. Although the eastern pine weevil prefers cedar, it also attacks most southern yellow

pines, such as loblolly, slash, and shortleaf. Pales and eastern pine weevils may serve as vectors for various pathogenic fungi.

The future outlook for the activity and damage levels of reproduction weevils is similar to the recent past. A warmer climate may allow these insects to extend their ranges north into Canada. Warmer southern winter months may allow them to increase and/or prolong activity and to produce more generations per year. Decreased precipitation may reduce their activity. The impacts of these pests are more dependent on stand management (and whether seedlings were treated with insecticides) than on climatic conditions. If pines are planted and then left unmanaged over the next 50 years, we can expect increased damage from pine reproduction weevils.

Sirex woodwasp—Sirex woodwasp, *Sirex noctilio*, is native to Europe, Asia and northern Africa and has been introduced to North America, South America, New Zealand, Australia, and South Africa. In Australia, South Africa and South America, it is considered an important pest, causing significant mortality (Oliveira and others 1998) in stands planted with North American pines, especially Monterey pine (*P. radiata*) and loblolly pine. Haugen and Hoebeke (2005) report that other known susceptible pines include slash, shortleaf, ponderosa (*P. ponderosa*), lodgepole (*P. contorta),* and jack (*P. banksiana*).

Females can produce up to 450 eggs and deposit them (mostly singly) below the surface of the bark close to the cambium. The female also deposits mucus and a basidiomycete symbiotic fungus, *Amylostereum areolatum*, which grows rapidly and excretes wood-digesting enzymes. When the larvae hatch they bore into the wood, but feed on wood already colonized by the fungus. The fungus and mucus act together to kill the tree and create an environment suitable for the development of the larvae.

Sirex woodwasp has not caused widespread mortality in the North American areas where it is established, nor have any populations been reported in the South. However within the next 50 years, it is very likely that natural or human-aided spread will introduce this pest to southern forests. Many of the South's most important pine species are susceptible to *Sirex* and many trees will succumb if attacks are as aggressive as they are in South America and Australia. Although this scenario could result in catastrophic ecological and economic losses, the complexity of southern forests (mixed stands, high biodiversity, many possible competitors, predators and parasitoids) contrasts with the monoculture pine plantations in other countries where the pest has been most damaging. Many studies are underway to assess the potential level of danger to southern forests. A national risk map for Sirex has been developed (see http://www.fs.fed.us/foresthealth/technology/invasives_sirexnoctilio_riskmaps.shtml) and risk maps specific to the South are in development.

If the Sirex woodwasp becomes established in the South and acts as a primary "tree killer", effective prevention and suppression techniques are available, including the current practice of thinning stands to increase growth and vigor and reduce susceptibility to bark beetles. . In other countries, Sirex woodwasp has been successfully managed using biological control agents. The key agent is a parasitic nematode, *Deladenus siricidicola*, which infests Sirex woodwasp larvae, and ultimately sterilize the adult females. Infested adult females lay infertile eggs that are filled with nematodes, which further spreads the nematode population. The nematodes can effectively regulate the woodwasp population below damaging levels. As Sirex woodwasp establishes in new areas, this nematode can be easily mass-reared in the laboratory and introduced by inoculating it into infested trees. Biological control employing these nematodes is being evaluated for use in U.S. forests. If effective, it should provide a good control option for southern landowners and land managers.

The effects of changes in temperature, carbon dioxide, and precipitation on Sirex woodwasp activity and aggressiveness are unknown. If pine acreage increases throughout the South or in certain areas of the South, susceptibility of these areas to attack will increase.

Southern pine beetle—Southern pine beetle (SPB)*, Dendroctonus frontalis,* is the most destructive insect pest of pine forests in the South (Thatcher and Conner 1985). Populations build rapidly during periodic outbreaks and kill large numbers of trees. For example, during the outbreak of 1999 to 2002, SPB killed more than a million acres of pines valued at greater than $1.5 billion. However, during periods of low activity, SPB populations may be so low that it is difficult to locate a single infested tree (Thatcher and Barry 1982, Thatcher and others 1980) or capture beetles in pheromone traps (Billings and Upton 2010.)

The SPB, which attacks all species of pines, prefers loblolly (*Pinus taeda)*, shortleaf (*P. echinata*), Virginia (*P. virginiana*), slash (*P. elliottii*) pond (*P. serotina)*, and pitch (*P. rigida*) pines but seldom attacks longleaf pine (*P. palustris*). SPB has been observed to successfully infest eastern white (*P. strobus*) and Table Mountain (*P. pungens*) pines. Mature trees in pure, dense stands have long been considered most susceptible to SPB attack, but in recent years unthinned pine plantations have increasingly supported SPB infestations (Cameron and Billings 1988). Attacks are rare for trees younger than 5 years or smaller than 2 inches in diameter at breast height (d.b.h.)

During outbreaks, SPB activity peaks in early summer in states on the Gulf of Mexico and in late summer and early autumn farther north.

In the last five decades, large acreages of pine plantations have been established in the South. Even-aged, single-species plantations become increasingly susceptible to SPB infestations as they age. Millions of acres of pine across the South are at high hazard for SPB attack as shown by regional and State maps

(Nowak [N.d.]). SPB hazard maps and information about their development can be viewed at: http://www.fs.fed.us/foresthealth/technology/nidrm_spb.shtml.

SPB impacts over the next 50 years are expected to be significant, especially if the pine acreage increases in the South, high-susceptibility species are planted in dense plantations, and the plantations are left unthinned. A warmer, drier climate is likely to increase SPB activity and impacts. Warmer temperatures will likely allow an increase in the number of SPB generations per year as well as the portion of the year that the beetles are active. The northern edges of the southern region and pine stands that are farther north than the historical SPB range (such as in the Lake States, New England, and Canada) are almost certain to experience SPB activity and impacts that are unprecedented or at least significantly greater than in the past.

There is some uncertainty and debate about the potential effects of a warmer climate on SPB (Tran and others 2007), and general predictions are difficult to make. An increase in temperature (particularly warmer winters) would allow more generations per year. Gan (2004) and Rivera Rojas and others (2010) predict outbreaks to become more frequent as climate changes, although lack of landscape-scale data on host abundance and distribution may have led Gan to overestimate future SPB activity. Very high summer temperatures may increase brood mortality, reduce spot growth rates, and hinder predation. Warmer winter temperatures may disrupt synchronization of the life cycles required for concentrated spring emergence that favors initiation of large, new infestations (Billings and Kibbe 1978).

The impact of outbreaks in the 1980s was magnified by an abundance of contiguous mature stands of sawtimber, many of which have been replaced with young plantations, at least on non-federal lands. If increased forest fragmentation, a younger age class distribution, and more thinning of plantations occur in the next 50 years, SPB impacts could be lowers in the future, despite increases in temperatures.

And although it is generally accepted that increased temperatures will increase SPB activity and damage, other factors (for example forest composition, forest management, direct suppression, etc.) may be more meaningful in determining future SPB activity and damage (Friedenberg and others 2008).

Similar to temperature's effect on SPB, the potential of moisture regime to increase or decrease SPB problems is open to conjecture and not fully understood. Some experts believe that drought is a major enhancer of SPB outbreaks, whereas others point to too much moisture as a primary facilitating factor.. If the frequency of precipitation extremes (years of extreme wetness or dyness) increases throughout the South over the next 50 years, it is probable that pines will become stressed and increased SPB activity and damage will result.

In addition to the effects that forest composition, temperature, and moisture will have on the SPB outlook, forest management will play a defining role. Planting the proper species for a given site, lower planting densities, and thinning of pine stands can increase stand vigor and resiliency and possibly reduce SPB damage. When outbreaks do occur, damage can be minimized by early detection and monitoring of spots, followed by prompt direct suppression of active spots (Billings 1980.)

Texas leafcutting ant—The Texas leafcutting ant, *Atta texana,* targets first- and second-year pine plantations in eastern Texas and west central Louisiana. In local areas where the ants are abundant, it is nearly impossible to establish pine plantations unless the ant colonies are eliminated. The annual loss of pine seedlings to Texas leafcutting ants is nearly 12,000 acres (Cherret 1986, Texas Forest Service 1982).

A warmer climate may lead to an increase and/or continuation of leafcutting ant activity during winter months. Decreased precipitation would likely have the opposite effect. Because this ant has a strong preference for well-drained, deep sandy soils (Moser 1984, Vilela 1986), climate-induced spread beyond its current distribution is unlikely. Although leafcutting ants are limited by average low

temperatures (warmer temperatures would lessen this limiting factor), their spread into new, northern areas is going to be limited due to the lack of preferred soils for the ant. There is a possibility that a warmer climate would allow northward movement into areas of Oklahoma and Arkansas that have deep, sandy soils. A new fipronil control product, PTM™ was registered in 2009, and an insecticidal bait is on the horizon. Regular and consistent application of these products has the potential to reduce the impacts of Texas leafcutting ants from historical levels.

Insect Pests of Hardwoods

Asian longhorned beetle—Asian longhorned beetle, *Anoplophora glabripennis*, was discovered attacking hardwood trees in the United States in the mid-1990s. Tunneling by beetle larvae girdles tree stems and branches. Repeated attacks lead to dieback of the tree crown and, eventually, death of the tree. The beetle probably travelled to the United States inside solid wood packing material from China. This pest beetle has been intercepted at ports and found in warehouses throughout the United States and is currently infesting trees in New York City, New Jersey, Worcester, (Mass.), and Toronto (Ontario, Canada). It was successfully eradicated from the Chicago area following a lengthy and aggressive campaign of detection and removal of infested trees (Antipin and Dilley 2004.)

This beetle is a serious pest in China, where it kills hardwood trees in roadside plantings, shelterbelts, and plantations. In the United States the beetle prefers maple species, including boxelder (*A. negundo*), Norway (*A. platanoides*), red (*A. rubrum*), silver (*A. saccharinum*), and sugar (*A. saccharum*) maples. Other preferred hosts are birches (*Betula* spp.), Ohio buckeye (*Aesculus glabra*), elms (*Ulmus* spp.), horsechestnut (*Aesculus hippocastanaeum*), and willows (*Salix* spp.). Occasional-to-rare hosts include ashes, European mountain ash (*Sorbus* sp.), London planetree (*Platanu*s sp.), mimosa (*Albizia julebrissin*), and poplars (*Populus* spp.). A complete list of host trees in the United States has not been compiled.

Asian longhorned beetles produce one generation per year. Adult beetles are usually present from July to October, but can be found later in the fall if temperatures are warm. Adults usually stay on the trees from which they emerged or disperse short distances to a new host to feed and reproduce. Each female usually produces 35 to 90 eggs (or more) during her lifetime. Eggs hatch in 10 to 15 days. The larvae feed under the bark in the living tissue of their host and then bore deep into the wood to pupate. Adults emerge by boring a tunnel and creating a large round exit hole in the tree (U.S. Department of Agriculture Forest Service and Animal and Plant Health Inspection Service 2008).

Currently, the only effective means to eliminate Asian longhorned beetle is to remove infested trees and destroy them by chipping or burning. To prevent further spread of the insect, quarantines are established to prevent transportation of infested trees and branches from the area. Early detection of infestations and rapid treatment response are crucial. Systemic insecticides can provide protection for individual trees or small numbers of trees, but individual tree treatment is not feasible in forested settings.

The future impact of Asian longhorned beetles on southern forests is unknown for several reasons. First, the pest may or may not spread into the South over the next 50 years. Significant eradication and containment efforts are being pursued in areas where trees are under attack. Although the beetle disperses slowly—it does not fly great distances and tends to remain in the same area until hosts are exhausted—it may be spread great distances in firewood or by movement of other infested material.

A wide variety of southern hardwood trees (especially maples) is at risk. It is unlikely, however, that vast areas of hardwoods would be killed within the next 50 years because the beetle takes several years to kill host trees and it is a slow disperser. If spot infestations are discovered early enough, the beetle can be eradicated before it becomes widely established. Successful eradication efforts require much time, funding, personnel, and strength of will.

Effects of southern climate on Asian longhorned beetle are completely unknown. Extreme heat in some parts of the South may inhibit activity and success. However, there is also the possibility that warmer temperatures would lead to quicker completion of the beetle's life cycle, which would mean larger populations and more damage to southern trees.

Baldcypress leafroller—Formerly named the fruittree leafroller, the baldcypress leafroller, *Archips goyerana*, periodically defoliates baldcypress in Louisiana and Mississippi. Kruse (2000) describes the baldcypress leafroller, and summarizes its biology and its effects on its host. This native insect causes growth reduction and dieback, but only causes mortality when multiple other stressors are at work.

The baldcypress leafroller was first recorded in 1983 in Louisiana, where it feeds almost exclusively on baldcypress. It annually defoliates an average of 35,000 acres in the oak-gum-cypress forest type. Although this insect is mainly a pest of flooded baldcypress, it can move into drier upland and urban settings during periods of heavy infestation.

Baldcypress trees of all sizes display canopy dieback and significant reductions in diameter growth resulting from repeated annual defoliation. Pole-sized to small sawtimber-sized trees growing on forest edges or in dense stands are most severely affected. In areas where chronic saltwater intrusion is a problem, trees die after as few as two consecutive years of defoliation.

Temperature and precipitation changes are unlikely to directly affect baldcypress leafroller's activity and impacts. However, higher sea levels resulting from warmer temperatures would further stress baldcypress trees because of increased saltwater intrusion, significantly increasing the likelihood that defoliation would damage and kill host trees. Human alterations to southern Louisiana's hydrology, greater saltwater intrusion, nutria feeding, defoliation by baldcypress leafroller, and other stressors are

all combining to threaten the baldcypress resource in southern Louisiana. Although unlikely to disappear in the next 50 years, this resource is expected to continue to be compromised.

Emerald ash borer—Emerald ash borer, *Agrilus planipennis*, is a devastating, wood-boring beetle native to Asia. It was first found infesting trees in North America in southeastern Michigan and adjacent areas of Ontario, Canada, in 2002 (Various 2010). Within the core infested area of Michigan, Indiana, and Ohio, more than 50 million ash trees are estimated to be dead, dying, or infested (Smith and others 2009). Elsewhere, the emerald ash borer already has killed tens of millions of ash trees, and continues to pose a serious threat to the ash resource of North America.

The emerald ash borer was first found in the United States in 2002, but it was likely introduced into the area around Detroit in the early 1990s (Kovacs and others 2009), probably in solid wood packing material from Asia. Soon after detection, five counties in Michigan were placed under quarantine. However, in the years before detection, infested material—such as nursery stock, unprocessed ash logs, firewood, and other ash commodities—was most likely moved to many areas around the United States. Inadvertent movement by humans continues into the present in spite of Federal and State quarantines restricting the export of potentially infested materials once the borer is detected in a county (U.S. Department of Agriculture Animal and Plant Health Inspection Service 2003, 2006). Surveys made in 2003 found infestations in 12 counties in Michigan and 3 counties in northern Ohio. By early 2011 infestations were located in an additional 13 states: Indiana, Illinois, Iowa, Maryland, Pennsylvania, Missouri, Virginia, West Virginia, Wisconsin, Kentucky, Minnesota, and New York (fig. 16-2). In Canada, infestations now occur in several areas of Ontario and Quebec. (U.S. Department of Agriculture Animal and Plant Health Inspection Service 2011)

Since its introduction, the emerald ash borer has had a significant negative impact on the ecology and economy of infested areas, with all 16 species of North American ash appearing to be susceptible. Ash trees are an important part of the rural and urban forests of the United States, valued at more than $282 billion (U.S. Department of Agriculture Animal and Plant Health Inspection Service 2003). Ash wood is used for a number of applications including tool handles, baseball bats, furniture, cabinetry and paper. As a result of wide-scale loss of American elms to Dutch elm disease in the 1970s and 1980s, ash has often become the tree of choice for landscaping in new residential areas and commercial plantings. Ashes are now among the most common tree species along streets, and in parks and yards. Annually, the nursery industry produces an estimated 2 million ash trees, valued at approximately $140 million. Ecologically, the 16 species of ash fill a number of niches, from riparian areas to upland forests.

In all likelihood the emerald ash borer will infest and kill many, if not most, of the ash trees in the South over the next 50 years. Generally, ash is not a dominant component of southern forests, but it is almost always common and in some areas (such as the bottomland hardwood forests of the Mississippi delta) ash makes up a considerable portion of hardwood harvests. Green ash is a small but significant component of most riparian forests in the South. The largest impact is likely to be in urban areas, where ash is a common street and yard tree in many communities.

In its native range in Asia, the emerald ash borer does not cause serious damage to ash trees. As a consequence, very little was known about its biology (life cycle, flight capabilities, host preferences, and natural enemies) and control. Also unknown were methods to detect the presence of the borer and the trees they had infested. One of the greatest challenges for managers is this limited ability to detect infestations early enough to effectively control them and prevent their spread.

There are a number of effective chemical control options available to protect individual trees from infestation (Herms and others 2009). Unfortunately, available time, funding, equipment, and expertise limit the number of trees that can be protected to urban/suburban settings and a very small number of high value trees in forested settings. With the emerald ash borer destroying every ash in its path, one practical option may be to delineate and protect small pockets of exceptional ash resource as "ash conservation areas."

Several larval and egg parasitoids are being investigated for use as biological control agents (U.S. Department of Agriculture Animal and Plant Health Inspection Service and others 2010). Although results are preliminary, it is reasonable to expect that biological control agents would mitigate populations but would not control or completely stop the spread and impacts of this insect invader.

The effects of changes in climate—such as increases in temperature, precipitation, and carbon dioxide— on emerald ash borer are uncertain. Warmer temperatures would likely result in more rapid life cycle completion resulting in increased population growth and impacts. However, the extreme heat of southern summers could actually inhibit activity and reduce the amount of ash mortality. The range of ash trees in the South is expected to shrink as the climate warms; between climate stress and the emerald ash borer infestations, the South is likely to lose millions of ash trees in the next 50 years.

Forest tent caterpillar—Forest tent caterpillar, *Malacosoma disstria*, occurs throughout most of the United States and Canada, where it defoliates a variety of hardwoods (Batzer and Morris 1978, Fitzgerald 1995, U.S. Department of Agriculture, Forest Service 1985b, Drooz 1985). In the South, it heavily defoliates water tupelo (*Nyssa aquatica*), sweetgum (*Liquidambar styraciflua*), blackgum (*N. sylvatica*), and various oak species (*Quercus* spp.). The most persistent and extreme outbreaks in the South occur on host trees in bottomlands, forested wetlands, and riparian areas. When populations

reach epidemic levels, the caterpillars often spread to urban and suburban areas where they defoliate shade trees and ornamental plants.

Outbreaks occur in several Southern states, where more than 500,000 acres can be defoliated in a single season; defoliation does not cause significant amounts of tree mortality and therefore control practices are rarely cost effective. However, significant loss of tree growth is often an outcome, and repeated, heavy defoliation of stands may cause significant dieback. If needed, control techniques are available and have proven effective but depend on the availability of both funding and technical expertise.

Tent caterpillar impacts occur mainly in the bottomland hardwood-cypress forest types (mapped as oak-gum-cypress and elm-ash-cottonwood), but they occasionally occur in upland northern hardwood forest types (mapped as maple-beech-birch, oak-hickory, and oak-pine).

Changes in temperature and precipitation are unlikely to increase defoliation by forest tent caterpillars. If climate change significantly stresses the forest types most vulnerable to tent caterpillar defoliation, the additive effect of multiple stressors could mean hastened or increased tree mortality.

Gypsy moth—Gypsy moth, *Lymantria dispar,* is native to Europe and Asia. In 1869, Leopold Trouvelot introduced the European strain of the gypsy moth. Since then, it has spread across the landscape of the eastern United States, defoliating vast acreages of forest (U.S. Department of Agriculture Animal and Plant Health Inspection Service 2010b). The insect was found in northeastern Virginia in the early 1980s. At its current rate of spread, specialists predict that a significant portion of the South will be infested in the next 50 years.

The impact of repeated gypsy moth defoliation on the health of oak forests is significant (Campbell and Sloan 1977). Repeated severe defoliation of oaks weakens trees to such an extent that they may be attacked and killed by secondary pest organisms, such as the two-lined chestnut borer (*Agrilus bilineatus*) and Armillaria root rot (caused by *Armillaria mellea*). Extended drought intensifies the death rate.

Gypsy moth caterpillars feed on a wide range of trees and shrubs (Liebhold and others 1995, Zhu 1994) but prefer oaks. Species are attacked preferentially without respect to forest type. Highly favored species include sweetgum, northern red oak (*Quercus rubra*), and American basswood (*Tilia americana*). Species of limited suitability include pines, maples (*Acer* spp.), ash (*Fraxinus* spp.), American beech (*Fagus grandifolia*), and cherry (*Prunus serotina*). Species that are not favored or are avoided include blackgum, yellow-poplar (*Liriodendron tulipifera*), black locust (*Robinia pseudoacacia*), baldcypress (*Taxodium distichum*), magnolia (*Magnolia grandiflora*), and tupelo (*Nyssa sylvatica*). As gypsy moth moves south and west, it will encounter lower concentrations of oak and cove hardwoods, and forest susceptibility will decrease in many but not all areas. However, with its wide host range it should still persist.

The most important disease agents affecting gypsy moths are the gypsy moth nucleopolyhedrosis virus (LdMNPV) and the gypsy moth fungus, *Entomophaga maimaiga* (Andreadis and Weseloh 1990, Hajek and others 1990).

The Slow the Spread Program decreases the gypsy moths' rate of spread from approximately 25 miles a year to 7 to 10 miles per year (Sharov and others 2002). If the program continues, we can expect the gypsy moth to move 350 to 500 miles farther into the South over the next 50 years, compared to total infestation within 25 to 30 years without the program.

Gypsy moths can also be artificially spread by human activities; continued vigilance to detect and eradicate the resulting small infestations help to prevent the moth's rapid spread into all areas of the South. In addition, methods exist to suppress areas of high populations in infested areas and to eradicate "satellite" infestations in advance of the moth's moving front; these methods include aerial applications of Bt (*Bacillus thuringiensis*) or dimilin (insecticides), or pheromone flakes (to disrupt mating).

Temperature changes alone are unlikely to have a dramatic effect on gypsy moth movement or impacts. The range of gypsy moth infestation is expected to expand regardless of changes in climate, and at a rate faster than can be attributed to any potential climate change-caused host range expansion. If warmer temperatures cause the oak-hickory forest type to displace boreal forests at higher elevations in the South, gypsy moth impacts will likely increase in these areas.

However, one hypothesis (John Ghent, personal communication) is that gypsy moth spread and damage will decrease as temperatures warm, thereby reducing the extent of southward spread. Gypsy moths need a cold snap to synchronize hatches (avoids different life stages from occurring at the same time) and thus improve mating efficiency. If this hypothesis is correct, as the moth moves farther south and as the temperatures warm, winters would not be cold enough or the necessary cold snap would come too late in the year to synchronize the spring hatch.

A drier climate would likely increase gypsy moth impacts because it would stress host trees and discourage build-up of the moth's fungal predator, which thrives during wetter springs.

Because the gypsy moth is still spreading into the South, barring unforeseen circumstances we can say with certainty that its impacts will increase over the next 50 years. How severe and widespread the impacts will be, however, is dependent on many factors including: the continuation of active programs

to slow the spread, suppress and eradicate gypsy moth; the amount and health of hardwood forests the moth encounters in the future; and potential unknown temperature and moisture effects on the moth, its hosts, and its natural enemies.

Hardwood borers—Insect borers are important pests of hardwood trees throughout the South. They tunnel in the bark, trunks, terminals, and roots, causing a variety of defects in wood, stem deformity, reduction of seed production, and tree decline.

Some of the major damaging borers in the South (Solomon 1995) are the carpenterworm (*Prionoxystus robiniae*), red oak borer (*Enaphalodes rufulus*), white oak borer (*Goes tigrinus*), redheaded ash borer (*Neoclytus acuminatus*), poplar borer (*Saperda calcarata*), oak timberworm (*Arrhenodes minutus*), Columbian timber beetle (*Corthylus columbianus*), and ambrosia beetle (*Xyleborus celsus*). Borers that are endemic to an area do not normally cause dieback and mortality, but in abnormally large numbers they contribute to tree decline and stand degradation. Excessive numbers of growth defects caused by borers affect between 25 and 88 percent of all hardwood logs (Ward and Mistretta 2002).

In the early 2000s, prolonged droughts compromised the vigor of oaks in northern Arkansas, leading to a massive red oak borer outbreak. Although they were not the primary cause of the oak mortality in that area, the borers soon became the most destructive agent in the decline complex. More than 340,000 acres of oak and mixed-oak-pine forest were severely impacted, with an estimated loss of 500 million board feet (more than $29 million) of oak.

Temperature change by itself is unlikely to have much effect on hardwood borer populations. As secondary insect pests, these borers are expected to have increased impact as populations of hardwood age and decline, especially during periods of drought stress. Hardwood borer activity and damage is

likely to increase throughout the South over the next 50 years if current predictions of future climate change prove accurate.

Soapberry borer—Soapberry borer, *Agrilus prionurus*, a native of Mexico, was first confirmed in eastern Travis County, Texas, in 2003. It infests and kills western soapberry (*Sapindus saponaria* var. *drummondii*), its only known host. Reports by landowners and arborists indicate that the insect had probably been infesting soapberry trees for several years prior to being identified. Infested trees were observed in Travis and McLennan counties as early as 1998. By January 2009, infestations had been reported in 18 Texas counties, including areas near Fort Worth, Dallas, Waco, College Station, Austin, Houston, and Corpus Christi. By December 2010, the number of counties had increased to 43 (Billings 2011). To date no infestations have been observed in adjacent States, although infestations in Roberts County in the Texas panhandle and Wichita County on the Texas-Oklahoma border suggest that the insect may already be in Oklahoma (fig. 16-3).

As soapberry borer populations expand rapidly in Texas, this wood-boring beetle is killing all soapberry trees larger than 2 inches d.b.h. Methods of prevention and control are being investigated. Among the most promising is injection of a systemic insecticide (emamectin benzoate, registered for the control of insects on conifers and hardwoods, including the prevention of emerald ash borer) into uninfested soapberry trees or those in early stages of attack. Test-injection trees are still being monitored, but early results look promising.

Regardless of climate change, it is likely that within 50 years the insect will threaten western soapberry populations throughout the tree's entire range, which extends from northern Mexico to Missouri, and west to Arizona.

Diseases of Softwoods

Annosum root disease—Annosum root disease (ARD), caused by the fungus *Heterobasidion annosum* (recently proposed to be renamed *H. irregulare* (Otrosina and Garboletto 2010), produces significant losses of conifers across the South. On sandy, well-drained sites, this disease causes growth loss and mortality. It is most often associated with thinning of loblolly, longleaf, shortleaf, slash, and white pine plantations. The fungus commonly infects fresh stumps and then grows through root grafts (roots that come into physical contact and then grow together, sharing water and nutrients) and infects residual trees on the site. Slash and loblolly pines are the most commonly planted species in the South and are both very susceptible to ARD (Robbins 1984, Stambaugh 1989).

A survey in the South documented: 44 to 60 percent occurrence of this root disease; and, 2 to 3 percent mortality in planted pine. Radial and height growth are significantly less for diseased pines (Applegate 1971, Froelich and others 1977, Morris 1970).

The primary risk factors associated with ARD are the amount of host type available, the soil type and condition, and the timing and degree of management activity. Risk decreases as clay content in the surface layer of soil increases, a condition that enables risk mapping (Ward and Mistretta 2002). In the South, risk of ARD is high or moderately high on an estimated 163.5 million acres, not all currently forested (Hoffard and others 1995).

The range of ARD already extends throughout southern forests and into the boreal forests of the North, making spread unlikely. Indeed, its range could decrease with efforts by many land management agencies to restore the less susceptible longleaf pine to its previous range while concurrently potential drought/temperature related dieback in the southernmost part of the loblolly/slash pine range further decrease its range. Increased temperatures, reduced rainfall, and increased host growth (from more carbon dioxide in the atmosphere) would all produce some increases in disease activity resulting from

increased host susceptibility, but would not significantly increase fungus virulence. It is improbable that climate warming/drying would affect pine susceptibility on well-drained, sandy sites and forested old farm fields since on these sites potentially affected pines are already highly susceptible to the disease.

Management for disease prevention using borax as a stump treatment in uninfected stands should continue to be effective. Depending on the rate of temperature increase, insolation (thermal treatment of the stumps by the sun) may be effective in preventing infection via stumps further north than the 35th parallel which is the currently accepted northern limit of its effectiveness.

Loss of area by host species favored by *H. annosum* should lead to a slight overall loss of the negative impact of this disease over the next 50 years.

Brown spot needle disease—Brown spot needle disease, caused by the fungus *Scirrhia acicola,* is considered the most damaging disease of longleaf pine. It primarily affects seedlings by delaying the onset of height growth and causing loss of potential wood production and mortality (if infection is severe). Brown spot is somewhat a disease of opportunity: the grasses that compete with longleaf seedlings also maintain a humid microclimate that contributes significantly both to infection of the seedling and to the general success of the disease.

This disease occurs from Virginia to Texas, primarily on the Coastal Plain. It is more severe in certain geographic areas (Ward and Mistretta 2002). Use of controlled fires to remove competing grasses and eliminate dampness is highly effective for controlling the disease and encouraging early growth of seedlings, provided steps are taken to avoid subsequent colonization by competing non-natives such as cogongrass.

At present, longleaf pine occupies only about 5 million acres of its former 60 million acre range. Recent restoration efforts have led to the production of healthier seedlings for planting and planting success has improved on sites where longleaf was once the dominant species (Cordell and others 1989, Kais 1989). Over the next 50 years, the emphasis on longleaf pine restoration should have a greater impact on this disease than climate warming. Longleaf pine is well adapted to summer temperatures in the South and it is unclear that increases even as high as 1 ^{o}C would have significant impact on the southern extent of the longleaf pine range. Higher temperatures might slightly favor increase in growth and longer summer heat spells might trigger early onset of height growth from the grass stage to the candle stage, ending the potential for brown-spot damage sooner. Reductions in rainfall, dew, and fog should favor the longleaf pine over the fungal pest. No shift in aggressiveness of infection or virulence of the pathogen is foreseen.

We anticipate a significant increase in the incidence of brown spot disease. This expectation is based more on increased out-planting of longleaf pine seedlings than on climate influences. Thus, although climate change is not expected to significantly change the disease profile (its virulence or host spectrum), human intervention to increase the quantity of host trees could result in increased incidence.

Fusiform rust—Fusiform rust, caused by the fungus *Cronartium fusiforme* f. sp. *fusiforme*, occurs primarily on slash and loblolly pines. It is considered the most destructive disease of southern pines, causing the production of cigar-shaped galls that are generally fatal if formed on the main stem of the host (Anderson and others 1980, Czabator 1971).

Extensive planting of susceptible slash and loblolly pines since the 1930s has resulted in an epidemic of fusiform rust, which now extends throughout its available host range in the South; infected trees being found throughout the southern pine region (Ward and Mistretta 2002). Losses are most serious on

Coastal Plain sites from Louisiana to southeastern South Carolina. Several variables including weather, amount of inoculum, abundance of oaks (the alternate host), and susceptibility of the individual pine species govern incidence of the disease. Effective strategies are available for managing fusiform rust impact in plantations and forests including avoidance of over-fertilizing seedlings in the nursery, silvicultural manipulation of young stands to favor healthy saplings, and favoring the deployment of genetically screened resistant seedlings in areas of historic high rust incidence.

Increase in disease range in this region under the influence of a warmer, drier climate change scenario is not a concern since the disease is already distributed host-range wide within the region. However, increased temperature and carbon dioxide in the atmosphere could cause the pathogen to become more virulent on its current host base. Although there is some disagreement on the effect of projected warmer, drier climate regimes on the geographic ranges for the pine hosts, it is anticipated that any losses of pine in coastal areas would be matched by gains in the Piedmont and in the lower reaches of the Appalachian Mountains.

Although research on rust fungi is inconclusive and primarily based on cereal grains and other field crops, results suggest that there would be greater incidence of fusiform rust simply as a function of healthier fungus and host trees (Chakraborty and others 1998). We also anticipate that loblolly pine at least will be planted in areas north of its current range; and that the rust, which infects juvenile tissue, will rapidly follow into these newly planted areas.

Over the next 50 years given the general availability of oak alternate hosts for the fungus and the only slight predicted migration of pine from coastal areas upward into the Appalachian Mountains, we expect that the pathogen will successfully fully colonize the extended range of its hosts. The potential effect of outplanting rust resistant seedlings in conjunction with potential geographic range and climate

shifts is uncertain at the present time. If the resistance is maintained in the face of changing conditions, a reduction of the impact of this disease would be expected to occur.

Littleleaf disease—Littleleaf disease is the most serious pest of shortleaf pines in the South. It is caused by a complex of factors including a nonnative fungus (P*hytophthora. cinnamomi*), low soil nitrogen, eroded soils, poor internal soil drainage, and a plow pan—a compacted layer of soil that has become less porous than the soil above or below, generally the result of tilling or other farming operations (Campbell and Copeland 1954). Often, native nematodes (microscopic roundworms) and native species of *Pythium* (also a fungus) are associated with the disease. Infected trees have reduced growth rates and commonly die within 12 years of symptom onset.

P. cinnamomi is distributed throughout (and well beyond) the range currently occupied by shortleaf and loblolly pine in the South. Shortleaf pine is the most seriously damaged softwood host, with loblolly pine affected to a lesser extent; American chestnut was its primary hardwood host. Littleleaf disease has also been reported on Virginia, pitch, slash, and longleaf pines. Affected pine stands are found on the Piedmont from Virginia to Mississippi. The disease has its greatest impact in Alabama, Georgia, and South Carolina (Ward and Mistretta 2002, fig. 17.10), with additional scattered pockets occurring in eastern Tennessee and southeastern Kentucky. Note that, although the fungus' range exceeds the range of its pine hosts, littleleaf disease is further restricted in within that larger range generally by site conditions.

The fungus has a mobile spore and needs water to spread from and infected host to uninfected potential hosts; however, the disease thrives under dry conditions that stress the host. Control strategies are available but most—such as sanitation thinning and salvaging dead materials—rely on treatment after infection when damage is imminent or already occurring.

Because of its specific site requirements, spread into uninfected southern forests is not expected. Further, rehabilitating sites by breaking up of the plow pans that favor this disease should result in better water relations and a reduction in infections. An increase in atmospheric carbon dioxide would result in increased growth of the host and greater disease expression in affected trees. Losses to this disease should continue at the same rate on affected sites. However, its range should contract if increased temperatures cause its hosts to migrate north, and its impact should decrease over time as sites are rehabilitated.

Loblolly pine decline—Reports of sparse, yellowing crowns, and low annual wood production in the pines of central-to-northern Alabama date back to the late 1960s (Brown and McDowell 1968, Brown and others 1969). Since the early 1990s, localized incidents of declining pines have been occurring throughout Alabama and into southwestern Georgia, with additional symptoms including root mortality and discoloration of many of the surviving rootlets (Hess and others 2003). Recent literature suggests the presence of fungi—including *Leptographium serpens, L. terebrantis, and L. lundbergii*–in the roots of affected trees (Eckhardt and others 2004b); but whether they are primary pathogens or simply taking advantage of already significantly weakened trees is still uncertain. A bark beetle, *Hylastes* sp., has been found in the root systems of many declining pines, and is suspected of vectoring the fungus from infected to uninfected trees (Eckhardt and others 2004a). Information is lacking on whether they select weakened trees to attack or are indiscriminate in their attacks (which would suggest that healthy trees may be able to overcome successful inoculation).

The symptoms of the decline primarily occur in loblolly pines older than 40 years, first becoming apparent in trees in the 40 to 50 year age class. Mortality can occur beginning as little as two to three years after first symptom expression. Little is known about the potential range and severity beyond that from field surveys in central northern Alabama (Hess and others 2005) and Fort Benning, Georgia

(Menard and others 2006). Nevertheless, there is strong speculation that both abiotic and biotic factors are involved in predisposing affected stands to decline. These factors include climate, wildfire, and human disturbances such as previous agriculture. Coincidently, many upland sites in northern and central Alabama were originally converted from subsistence farming to loblolly pine plantation because of loblolly's out-planting success rate and its rapid growth. One theory is that many of these sites are simply unable to sustain such rapid growth over the long-term.

Despite the uncertainties about the causes and progression of this disease complex, management strategies are in place that can be implemented with the expectation of improving resistance of future stands on affected sites. These strategies start with applying a risk rating model that uses digital elevation maps and mapped shape files for the sites in question combined with data on landform and root health of the trees in the stand. If the model predicts hazard to loblolly pine, the recommended alternative species is longleaf pine. For existing loblolly pine stands on high hazard sites, the recommendation is to thin them between ages 20 and 40 (Hess and others 2003). A previous recommendation, to allow a high- risk site to revert back to native hardwoods (Loomis 1976, Miller 1979), is still a viable (but seldom adopted) management option.

Tree decline is likely to increase in a warmer and drier climate, regardless of inputs from disease and insect vectors. This response to changing climate is a major factor in the northward movement projected for the southern pines. Increasing incidence of decline should eventually diminish as new adapted ecosystems form in the region, but this is not expected to occur within the next 50 years. 3

Diseases of Hardwoods

Beech bark disease—Beech bark disease is caused by a complex of two or more agents working in concert. The beech scale, *Cryptococcus fagisuga*, attacks the bark of American beech, creating infection

courts which are subsequently colonized by the fungus *Nectria coccinea* var. *faginata.* This fungus causes cankers that grow together and girdle host trees.

While the beech scale is now a common pest of the American beech, it is nonnative, having been introduced through the Canadian Province of Nova Scotia in the late 1800s. There is speculation that the fungus is also an introduced species. Discussion on that point is somewhat pointless since a native fungus, *N. galligena,* is also capable of inciting cankers and killing hosts after entering through scale-damaged bark. The scale is considered the pivotal introduction that allowed the invasive spread of this disease complex (Houston and O'Brien 1983, Southern Appalachian Man and the Biosphere 1996).

This disease complex, first identified in southern forests in the early 90s, continues to spread along a broad front and is expected to occupy the range of its host (Ward and Mistretta 2002). In the early phase of its cycle, more than half of the American beech trees 10 inches d.b.h. or larger are killed. Openings created by death or removal of the beech result in dense stands of root-sprouts, which produce stands dominated by beech but lacking any of its normal associates. In the second phase of the cycle, revegetated beech stands are attacked less severely, resulting in cankered survivors rather than in extensive mortality. Trees infected in this phase are rarely girdled, but they are generally severely deformed.

Since this disease complex affects only American beech, there is a direct relationship between the amount of beech in a stand and the intensity of the disease. Houston (1997) reports that stand age and density, tree size, and species composition affect disease severity, especially in forests affected for the first time.

Beech bark disease is enabled by an insect vector, so the projection of future condition is complicated beyond that of a simple pathogen or insect driven pest system. Vector mediation

corresponds to availability of spores and host susceptibility, and is expected to maintain synchronicity sufficient to cause a slight increase in infection. Temperature intolerance of the host should reduce the host's geographic range in the face of climate change. Increases in carbon dioxide should increase host growth allowing a slight increase in disease virulence.

Ultimately, however, the reduction in available host trees should result in an overall decrease of significance of beech bark disease in southern forests despite the probability that individual trees will experience a slight increase in disease severity.

Butternut canker—Butternut is being killed throughout its range in North America by a fungus, *Sirococcus clavigignenti-juglandacearum*, which causes multiple cankers on the main stem and branches of host trees. Butternut canker has been found in 55 counties in the South extending north from northern Alabama along the Appalachian Mountains into North Carolina, Tennessee, Virginia, and Kentucky, with scattered occurrences throughout Kentucky and Tennessee (Ward and Mistretta 2002). Butternut numbers have been dramatically reduced and the species is now a Listed as a species of Special Concern in Kentucky and as Threatened in Tennessee (U.S. Department of Agriculture Natural Resources Conservation Service 2011). In both states the species is listed as G4/S3. G4 indicates a plant which is "…apparently secure globally, though it may be quite rare in parts of its range…" while S3 indicates "…rare and uncommon in the state…" (U.S. Department of Agriculture Natural Resources Conservation Service 2008, 2009).

Detailed examination of cankers indicates that butternut canker has been present in the United States since the early 1960s. Its origin is unknown but its rapid spread throughout the butternut range, its highly aggressive nature on infected trees, the scarcity of resistant trees, the lack of genetic diversity in the fungus, and the age of the oldest cankers (40 years) support the theory that it is a recent

introduction. Data from forest inventories show a dramatic decrease in the number of live butternut trees in the United States (77 percent loss in North Carolina and Virginia).

Because butternut makes up less than 0.5 percent of the trees in the South, the overall current impact of its loss to the forested ecosystem in the South is considered by some to be minor. However, as butternut trees die, they are replaced by other already present species, contributing to a reduction of biodiversity.

Climate change would likely raise temperatures at the higher elevations of the Appalachians and the Cumberland Plateau. This coupled with drier conditions would significantly reduce the range of butternut at its southern edge. Although the higher temperatures and predicted increases in atmospheric carbon dioxide could increase the host trees' growth, drier conditions resulting from reduced precipitation would act against this increase. Overall we expect to see more cankering and mortality occurring on fewer butternut trees in the South.

Chestnut blight—Introduction of the chestnut blight fungus, *Cryphonectria parasitica*, from Asia, probably in the middle-to-late 1890s, led to a permanent change in forest ecosystems. The American chestnut (*Castanea dentata*) was essentially lost, not only as a valuable timber species but also as the most important producer of hard mast for wildlife. Oaks and other species filled the voids in forest stands left by the death of chestnut (Hepting 1974, Oak and others 1994). The fungus continues to survive on infected sprouts from old chestnut rootstock, various oaks, and some other hardwoods (Boyce 1961).

No control was found to stop the rapid devastation caused by this blight, and there is little chance that the pathogen will disappear or that the American chestnut will naturally recover its preeminent position in eastern forests. Researchers into hypovirulence have discovered a disease that weakens the

blight fungus, resulting in less damage to the infected tree (Anagnostakis 1978). Field-testing is underway on a genetically engineered virus that causes a hypovirulent reaction and has the potential to efficiently spread hypovirulence throughout the fungal population.

Attempts to cross American chestnuts with oriental varieties and then backcross to the American parent appear to offer a viable method of maintaining resistant chestnut in forests (Schlarbaum 1988). Selectively breeding chestnuts as described has produced chestnut hybrid clones that are undergoing field evaluation by the American Chestnut Foundation. If the seedlings overcome both the blight and another disease (caused by *Phytophthora cinnamomi*) that was devastating chestnuts at the time chestnut blight was introduced, a serious effort can be made to reintroduce chestnut into the American forests. It is too early yet to predict the outcome of this effort. However, even if the hybrids are resistant to the disease, large areas of forestland cannot be restored to chestnut in the next 50 years because the seedlings that would be needed for that effort are not expected to be available in large enough quantities Further, if climate change is considered, the impacts on chestnut deployed in the restoration effort would probably be similar to those predicted for oaks suffering from oak decline.

Dogwood anthracnose—Dogwood anthracnose is caused by an introduced fungus, *Discula destructiva.* It was first reported in the United States on flowering dogwood, *Cornus florida*, in 1978 and on western flowering dogwood, *C. nuttallii*, in 1979. For the past three decades, flowering dogwoods have been declining at a rate that threatens important cultural aspects of southern society. In some areas, they have been all but eliminated from the forest ecosystem above 3,000 feet (Ward and Mistretta 2002).

The eastern flowering dogwood is a small tree valued both as a sign of spring for rural communities and forest visitors, and as an important source of soft mast for over 100 different species

of wildlife that feed on its berries (Kasper 2000). It is typically an understory tree found growing with other hardwoods such as oak and hickory. Severe infection is restricted to fully shaded understory trees at higher elevations (above 3,000 feet) and to those on shaded sites with a northern exposure. The hazard of severe infection and mortality is greatest in shaded, moist, and cool areas.

The range of this disease stretches southward into South Carolina and Alabama and westward into central Tennessee and scattered western Kentucky counties (Ward and Mistretta 2002) with activity concentrated in the Appalachian Mountains. The southernmost limit of the dogwood anthracnose range relative to available host trees suggests that this disease is temperature limited in the South. Whether this limitation functions at the time of spore propagation or dissemination and host infection, or whether it acts directly to limit disease success is unclear.

Any projected increase in the incidence or virulence of dogwood anthracnose based on increased host and fungal growth resulting from higher carbon dioxide levels in the atmosphere should be eclipsed by the temperature increases and possible rainfall reductions projected to occur under climate change. Increased temperature and aridity encroaching at higher-than-current elevations in the Appalachian Mountains should diminish the importance of this disease in the region, especially if it has reached a temperature barrier farther south. A recolonization of some areas currently denuded of dogwood by this disease might be possible.

Dutch elm disease—The Dutch elm disease pathogen is vectored by one of two bark beetles and can be caused by either of two closely related species of fungi: *Ophiostoma ulmi* (formerly called *Ceratocystis ulmi*); and *Ophiostoma novo-ulmi*, which is more aggressive in causing disease (Brasier 1991). These fungi were first introduced to the United States on diseased elm logs from Europe prior to 1930. It is unknown when the more aggressive species became established; however it was possibly

present as early as the 1940s to 1950s, and most likely caused much of the devastating elm mortality through the 1970s. The less aggressive species is becoming increasingly rare in nature, and the aggressive species is thought to be the primary cause of current mortality. Although some local resurgence has been observed, there is no evidence that the pathogen has further changed. Localized resurgence is more likely the result of decreased monitoring and sanitation vigilance, a buildup in populations of the insect vectors, or high densities of susceptible host trees in the wild (French and others 1980, Haugen 2007, Hubb 1999).

Native species of North American elms vary in their susceptibility to Dutch elm disease. American elm (*Ulmus americana*) is generally highly susceptible To the disease while winged elm (*U. alata*), September elm (*U. serotina*), slippery elm (*U. rubra*), rock elm (*U. thomasii*), and cedar elm (*U. crassifolia*) range from susceptible to somewhat resistant. No native elms are immune, but some individuals or cultivars have a greater resistance or a higher tolerance to infection (and therefore may recover or at least survive). Many European and Asiatic elms are less susceptible than American elm (Haugen 2007).

In addition to genetic factors present in some cultivars and species, physical factors affect tree susceptibility. These factors include season of the year, climatic conditions (such as drought), and vitality of the tree. Water conducting elements are most susceptible to infection because they are produced in the spring, making susceptibility highest from first leafing to midsummer and lowest during drought conditions. Vigorously growing trees are generally more susceptible than slower growing trees (D'Arcy 2005).

Roots of the same or closely related tree species growing in close proximity often cross each other in the soil and eventually fuse (become grafted). The fungus can move from infected trees to adjacent

trees through these grafted roots. Infections that occur through root grafts can spread very rapidly throughout the tree, because the fungus is carried upward in the sap. Root graft spread is a significant cause of tree death in urban areas where elms are closely spaced (French and others 1980, Haugen 2007).

Current management options in urban, suburban, and other high value settings include sanitizing to reduce insect vectors, applying insecticides to kill insect vectors, disrupting root grafts; injecting trees with fungicide, eradicating the fungus from newly infected trees (pruning), and planting resistant or tolerant trees (French and others 1980, Haugen and Stennes 1999, KLS 2009, Newhouse and others 2007, Scheffer and others 2008).

Although the most effective action is prompt removal of stressed, dead, and dying elms this intensity of treatment is often not feasible (Haugen 2007).

Despite the presence of several elm species (American elm, winged elm, and slippery elm, at least) very little Dutch elm disease can be found in areas below northern North Carolina, Tennessee, and Arkansas. It appears that either the beetles or the fungi involved in transmitting/causing the disease are temperature limited. Barring significant changes in its pathogen/vector combination, increasing temperature and migration of the host slightly to the north is expected to diminish the disease's overall impact in the South.

Laurel wilt—Laurel wilt is an insect-vectored disease that is currently decimating the redbay (*Persea borbonia*) population of the southern Coastal Plain. This disease was first identified near Port Wentworth, Georgia, in 2003 and has subsequently spread north, south, and inland (west) from that location (fig. 16-4). It is caused by an introduced and only recently classified fungus, *Raffaelea lauricola*, (Harrington and others 2008) that is vectored from host to host by an ambrosia beetle (*Xyleborus*

glabratus, also an introduced species). The beetle carries the fungus in pouches located near its mandibles. When the beetle bores into the sapwood the fungus inoculates the xylem. Once inoculated, the host rapidly develops a vascular wilt; its leaves die generally downward from the top, and the wood beneath the bark becomes discolored from streaking (Fraedrich and others 2008). Infected hosts display rapid dieback (wilted leaves and discolored sapwood) and may or may not exhibit extrusion of frass (the fine powdery sawdust and excrement that insects pass as waste after digesting plant material) from the insect's entry holes.

Several additional hosts have been identified for this vectored disease including swampbay (*Persea palustris*), sassafras (*Sassafras albidum*), avocado (*Persea americana*), camphor (*Cinnamomum camphorate*), pondberry (*Lindera melissifolia*), and pondspice (*Litsea aestivalis*). Redbay, however, is the favored host for the ambrosia beetle and to the present the severest damage has been limited to redbay (Hanula and others 2008).

At the present time there is no effective control known for this disease for forest and woodland use. While preliminary results using propiconazole (a fungicide) show promise for preventing the disease in treated trees, the necessity of retreating them and the cost of treatment suggests that in the future use may be limited to the protection only of high value trees (Mayfield 2008). Research into chemical treatment, centered on control of the vector, is ongoing but has yet to identify a chemical effective for this purpose. Management recommendations emphasize early sanitation (removal) of killed material but with the strong concurrent recommendation that the dead materials not be moved offsite, or if moved offsite then not out of the known infested/infected area. Further, it is recommended that whenever possible material that has been cut down should be chipped or buried rather than left intact (Mayfield 2008).

Based on the current rate of spread (estimated to be about 20 miles per year), the known distribution of redbay, and regional climate projections, Koch & Smith (2008b) have extrapolated probable spread of this disease through 2040 (fig. 16-5). According to their projection, the disease complex will have reached its northern extent (host based) by 2020, and will reach the western extent of its host range in eastern Texas by 2040. The basis of their projections is the combination of redbay's natural range and climatic barriers that affect the vector and fungus, which will likely stall further progress of the disease in the South. Their caveat is that projections are limited to the redbay host.

Unanswered at this point in time is whether this fungus/vector complex could become established in other parts of the country on other lauraceous hosts (such as the California laurel) should fungus-carrying beetles be introduced into potential new host ranges. Further, potential for affecting the spread of and possibly controlling some of the loss through implementation of the Recovery Plan for Laurel Wilt on Redbay and Other Forest Species (Mayfield and others 2009) is as yet an unknown factor in the management of this disease.

Unfortunately, in 2009, laurel wilt was detected in the Sand Hill Crane National Wildlife Reserve in southern Mississippi—a location that was not predicted by Koch & Smith (2008b) for infection until about 2017—apparently through human introduction. Regardless whether this is a new introduction or movement from the east coast infected area, it has reduced by 8 years the disease's expected arrival in Texas.

Of concern is whether the disease might expand its host range under the influence of climate change or through a modification of the fungus/vector complex that would allow a new insect vector to become involved. If either occurs, there is strong potential for currently unpredicted involvement of

new hosts and unpredicted spread; newness of this complex in the South leads to extreme uncertainty when attempting to project future behavior.

Given the rapid and severe damage done to the infected hosts coupled with predicted shifts in coastal vegetation resulting from projected temperature increases and possibly decreasing precipitation, the potential of this disease to spread beyond its projected range is highly uncertain.

Oak decline—Because of the history of woods grazing, widespread wildfire, and exploitive logging for wood products, and the loss of American chestnut to chestnut blight, oaks probably represent a larger component of the southern forest ecosystem today than at any time in the past (Millers and others 1990).

Decline of oaks in upland hardwood and mixed oak-pine forests is a complex involving environmental stressors (often drought), root diseases, insect pests of opportunity such as the two-lined chestnut borer (*Agrilus. bilineatus*), introduced pests such as the Japanese beetle (*Popillia japonica*) and Asiatic oak weevil (*Cyrtepistomus castaneus*), and physiological maturity of the trees (Staley 1965, Wargo 1977, Wargo and others 1983). Bottomland oak forests are also subject to oak decline but at a lower incidence. Stress agents of bottomland hardwoods also include seasonal, sometimes prolonged flooding.

Decline progression is measured in decades rather than months or years. Introduction of the gypsy moth into northern parts of the region has worsened oak decline because oaks are preferred hosts, and spring defoliation contributes to the chain of events that increase susceptibility. Although decline development may take decades from inception to the appearance of visible symptoms, susceptible trees die within a few years after dieback exceeds a third of the crown volume. Not all affected trees reach this point. Species in the red oak group (particularly

black and scarlet oaks) are most susceptible. Hickories are the only non-oak species group commonly observed with symptoms in decline areas (Starkey and others 1989).

Forest workers have reported oak decline occurrences since the mid-1800s (Balch 1927, Beal 1926) and in every decade since the 1950s (Millers and others 1990). A severe drought in the 1950s may have led to the current cohort of trees being highly susceptible to oak decline (Dwyer and others 1995, Tainter and others 1990). Also, within about 60 years after the loss of American chestnuts in the Southern Appalachians, the oaks that replaced them began to decline and dieback, due in part to growth stress from sites better adapted to chestnuts. Significant oak decline episodes continue to occur in the region (primarily in Arkansas and Virginia) where predisposing conditions, inciting events, and contributing factors are coincident (Starkey and others 2000, Gysel 1957, Oak and others 1988).

With increased temperature and (possibly) less rainfall being predicted, oak decline is expected to increase, possibly significantly. Decline resulting from the stresses imposed should be contributory to elimination of oak in some drier areas, and it is uncertain what community of plants would replace the oak on these sites.

Oak wilt—Oak wilt is a vascular wilt disease of oaks that is found only in North America. The causal fungus, *Ceratocystis fagacearum*, was first identified in Wisconsin in 1942. Scientists believed the disease to be native to North America and to have been present long before its discovery (MacDonald 1995, Tainter and Baker 1996). Recently strong speculation has been voiced that the fungus is actually a non-native introduction, possibly from South America, where it occurs without causing disease (Juzwick and others 2008). Oak wilt occurs in 21 Central and Eastern

States (Rexrode and Brown 1983); 9 of the 13 Southern States are known to harbor the disease, but severe mortality is limited to a recent outbreak in central Texas (Ward and Mistretta 2002).

Oak wilt causes affected trees to wilt and usually to die. All species of oak are susceptible, but species in the red oak group—northern red (*Quercus rubra*), scarlet (*Q. coccinea*), and black (*Q. velutina*) oak—are most readily killed. Oaks in the white oak group—white (*Q. alba*), post (*Q. stellata*), and chestnut (*Q. prinus*) oaks—are infected but mortality occurs much less frequently and more slowly. Live oaks (*Q. virginiana*) die at a rate generally intermediate between red and white oaks.

Sap-feeding beetles can carry fungal spores to nearby healthy trees, the fungus can colonize neighboring uninfected trees by growing through root grafts, and human mediated transmission is also possible (moving infected firewood with intact bark allows fruiting of the fungus in areas currently not infected.)

It is unclear whether the north-to-south progress of the disease was halted by a temperature barrier that limits migration of the fungus. The existence of such a barrier could mean that the Texas outbreak is the result of a relatively recent adaptation of the fungus to a higher temperature regime or an adaptation to the hosts (live oak) attacked in that area. Regardless of what caused the recent surge in oak wilt activity in Texas, its rapid spread raises the practical question of whether the fungus can now spread throughout the uninfected areas from Louisiana to Georgia and Florida. We anticipate that this question may be answered within the next 10 to 20 years as the disease appears to be spreading (or being spread by humans) at a fairly rapid rate.

Increasing soil temperature might provide a further barrier to spread, if indeed temperature has been a barrier. Predicting the direct effects of temperature and atmospheric carbon dioxide

on this disease will require an understanding of the pathogen-host mechanisms at play: whether damage to the root system is sufficient to cause symptoms and death, or whether the fungus must grow from the root system (where most of the transmission is occurring) into and throughout the vascular system aboveground to cause the same effect.

Little can be said with any degree of certainty about possible insect transmission of this disease. Consistent but inefficient transmission by sap-feeding beetles (Nitidulids and Scolytids) is an accepted mode of spread. Shothole borers have also been suggested, but these, and other possible insects, are less accepted. Longer periods of activity of these insects, resulting from the lengthening of summers (already being observed), could greatly increase transmission. However, this increase could only occur if fruiting mats of the fungus (which, in Texas, is associated with cooler and moister fall, winter and spring conditions; not the anticipated conditions) were present during the time in which the insects are active. Unless increased temperature triggers more mat formation than has been historically reported in Central Texas (unlikely), it is not expected that additional insects would become significant carriers of the fungus to uninfected trees. Possible loss of some coastal forest to savanna should have only a slight impact: simply reducing the number of hosts lessens disease incidence.

Management of this disease has proven to be expensive and is generally reserved for high value (aesthetically desirable) trees. Given the apparent adaptation of the fungus to warmer temperatures and relatively dry conditions, and the limitations of control tactics available, there is a high probability of significant oak loss in previously unaffected areas along the Gulf of Mexico and in Georgia within 50 years. However, if the apparent adaptation to warmer and drier conditions proves inadequate for continued disease spread, we would expect an overall slight lessening of the impact of oak wilt in the South.

Sudden oak death—First reported in California in 1995 sudden oak death (SOD) is now a well-established pest with a fairly limited range in California and Oregon. However, despite this relatively limited current range, it is believed that if introduced into the eastern oak forest the consequences could be dire.

Literature relating to this disease is extensive, but has recently been reviewed (Kliejunas 2010) and much of what follows has been extracted from or cross checked with that review to limit the number of citations included here. This publication which includes a 58 page bibliography of relevant literature is available on the internet at http://www.fs.fed.us/psw/publications/documents/psw_gtr234/.

Sudden oak death is caused by *Phytophthora ramorum, a* fungus, which causes several nonspecific symptoms depending on the host and host part affected. Symptoms include stem or bole cankers, twig blight (diback), and leaf blight. Individual plant species can display more than one or only one symptom type (see http://rapra.csl.gov.uk/background/hosts.cfm for links to images of symptoms on a variety of hosts.).

Cankers appear in the phloem (tissues that carry sugars away from the leaves of a tree) which may be discolored a bright red, and spread until they reach the xylem (tissues that carry water and minerals up from the root; wood fiber.) Cankers are sunken, “bleed“ sap, and are generally restricted to the lower portion of the tree trunk. The amount of bleeding is variable even on a single tree and may be related to environmentally available water and the age of the canker. Decline symptoms (loss of leaves) and crown death first appear at the top of the tree and spread rapidly down through the crown often resulting in tree death (Garbbelotto and others 2001).

The list of hosts currently reported for this pest is extensive. As of 2010 the list includes 45 proven regulated hosts plus another 82 associated hosts regulated in the nursery trade (U.S. Department of

Agriculture, Animal and Plant Health Inspection Service 2010a). Hosts with stem or branch cankering include California tanoak (*Lithocarpus densiflora*), coast live oak (*Quercus agrifolia*), California black oak (*Quercus kelloggii*), Shreve's oak (*Quercus parvula* var. *shrevei*). In addition, field and greenhouse inoculation experiments (Rizzo and others 2002) confirm that the fungus can cause a variety of leaf and branch symptoms, but generally not stem cankering, on rhododendron and azalea (*Rhododendron* spp.), madrone (*Arbutus menziesii*), huckleberry (*Vacinium ovatum*), manzanita (*Arctostaphylos* sp.), California bay laurel (*Umbellularia californica*), buckeye (*Aesculus californica*), bigleaf maple (*Acer macrophyllum*), toyon (*Heteromeles arbutifolia*), California coffeeberry (*Rhamnus californica*), honeysuckle (*Lonicera hispidula*), and a long list of other plants.

Although few of these species occur in eastern forests, several of them can be found in significant numbers. Early results by Rizzo and others (2002) show that northern red oak and pin oak (*Q. palustirs*) are susceptible to infection. In California greenhouse tests, seedlings of both eastern oak species developed lesions almost twice as long as those formed on the oak seedlings from Pacific coastal areas and roughly equal to those formed on tanoak (considered the most susceptible species in California). These results suggest that, all conditions being equal, these species should be highly susceptible to sudden oak death.

Kliejunas (2003) rated the risk posed by this disease as very high, but cautions that the degree of uncertainty related to future disease risk is also high based on lack of knowledge about the host range. Noting the absence of control measures, his risk assessment predicts rapid spread by wind, water, and human transport of infected plants; and suggests the potential for severe economic and ecologic losses, reductions in biodiversity, and indirect impacts on sensitive or critical habitat for at-risk plant and animal communities.

Based on past history with invasive species, it is easy to project that it is not a matter of "if", but "when", sudden oak death will gain a foothold in eastern oak forests (see alternative hypothesis below as "Note:") If the disease reaches southern forests, the role that climate would play is far from certain. Also uncertain, lacking basic epidemiological research, is the potential effects on eastern species; these could range from insignificant to potentially catastrophic (rivaling the effects of chestnut blight).

Sudden oak death appears to have the potential to devastate the eastern oak population, even absent climate change considerations (Kliejunas 2010 Chapter 4). Increased temperatures and atmospheric carbon dioxide could be expected to increase growth of both the pathogen and its host, at least in the short term. That effect would be somewhat counteracted by reductions in precipitation and increased ozone in conjunction with the warmer temperatures. Nevertheless, once acclimated to the eastern forest, the disease would probably spread even faster than it has in California.

Using the distribution of known or likely hosts, climate conditions adequate for the survival and propagation of the pathogen, and probable pathways of Introduction of the disease outside of its current range Koch and Smith (2008a; Fig. 16-6) project a potential range for this disease. Very similar potential range is indicated by DEFRA, Fowler and others, and Margary and others. Kelly and others and Venette and Cohen propose somewhat different potential ranges but both include significant Southern forest areas (Kliejunas 2010, Chapter 4).

Climate-induced losses of native oaks at their southern margins (Iverson and others 1999) would reduce the potential incidence of disease, but only slightly, and would not slow the progress of the disease in other parts of its potential range. Sturrock et al. (2011) state that, based on CLIMEX projections, changing climate will decrease substantially the area in the eastern USA favorable or very favorable for *P. ramorum*.

During technical review of this paper, one reviewer (2011) noted that to the present this disease has only been found in the woods in a very narrow geographic range in coastal California and Oregon; generally extending no more than 50 miles inland. It has yet to be confirmed as being established in other than West Coast forests despite the pathogen having been identified from diseased nursery plants shipped from California to many northern, eastern and southern nurseries. In addition, the fungus has been found in the East in water in the nurseries and in a limited number of cases in waters in ditches or outflow conduits outside those nurseries where it has been identified. The inference from these statements opposes the previously suggested scenario of explosive colonization of a susceptible population by a non-native pest. The risk maps presented by Kliejunas (2010, Chapter 4) from a variety of sources and using a range of predictive models show clearly the narrow, basically coastal range (present and predicted) for this disease in California and Oregon but also show a significantly larger area at risk in the eastern and southern forests.

Based on this conflicting information the future of sudden oak is unclear at the present. However, what is clear is that if it is introduced into the East its invasive spread will override climate change concerns during the next 50 years.

Thousand cankers disease—Thousand cankers disease of is caused by a fungus *(Geosmithia morbida*) (Kolarik and Freeland In press) and vectored from infected to healthy trees by the walnut twig beetle, *Pityophthorus juglandis* (Seybold and others 2010). The beetle is native to Arizona, New Mexico, on Arizona and California (also probably Texas). First identified on Arizona walnut, *Juglans major*, the fungus is also associated with cankering and dieback of *J. californica* and *J. hindsii* (Kolarik and Freeland In press). The fungus infects and subsequently kills black walnut, *Juglans nigra*, a species that is highly valued for furniture, paneling, and walnuts.

Thousand cankers disease was recently discovered and confirmed in urban/suburban settings in 4 Tennessee counties (with suspect trees occurring in similar settings in an additional 10 counties); as yet no woodland or forest surveys have been conducted. Although the Tennessee infections were the first reported east of the Great Plains, they may have been occurring since the 1990s. The full extent of this infection is as yet to be determined.

Symptoms of the disease include a foliage wilt in which the leaves progress rapidly from green through yellow and then to brown. Wilting progresses from the top of the crown downward as branches die. In the West, the dieback and ultimate death of infected trees takes about three years. Symptoms at first (and certainly when observed at a distance) resemble those caused by drought. Closer inspection of dead branches reveals many beetle entry holes through the bark and many (often small) cankers just under the bark. As cankers increase in number and each grows bigger until the areas of dead tissue coalesce and girdle the branch. In the later stages of dieback the beetles may attack the bole of the tree accelerating its death (Seybold and others 2010).

Control measures for thousand cankers disease have been proposed but not yet evaluated. Because the current range of the fungal pest is generally hotter and drier than Tennessee's climate, the disease is highly unlikely to encounter temperature barriers that would limit its spread into southern forests. And predicted a warming climate is likely to have little effect; the pathogen and vector originated in a hot dry area of the Southwest but both have moved into the cooler, moister climate of central Tennessee. Finding no barriers to spread, thousand cankers disease could occupy the entire range of black walnut within 50 years, similar to the projected spread of laurel wilt.

Additional concerns

Although we can make relatively uncertain predictions about the future of known pests, predicting currently endemic organisms that may become pests or organisms from other locations that may invade the South is virtually impossible. Lovett and others (2006) predict that forest pests will be the primary source of changes in eastern forests but cautioned against speculation on specific changes or specific pest introductions.

An important consideration is human caused change in the forest land base driven by increasing and shifting populations and economic conditions. As shown in chapter 5, all of the Cornerstone Futures forecast that total forest acreage will decline over the next 50 years, only planted pine is expected to expand, the oak-hickory type is expected to remain relatively stable, but the three other forest types considered are expected to decline. Additionally, total forest biomass is expected to increase at first but then decline somewhat.

A generalized implication of these potential shifts is relatively straightforward. Because pest activity appears to be basically a linear response to availability, less biomass would indicate less (in absolute terms) loss of biomass to pests. However, planted softwoods would be expected to show an increase in absolute loss proportional to the increase in planted acreage.

The possible effects of fragmentation, parcelization, and urbanization on pest impacts and management are so complex (and largely unknown) that it is not prudent or feasible to attempt to identify specific interactions. Generally, parcelization (greater number of landowners on smaller units of land) may complicate pest prevention and/or suppression by making it more difficult to attain effective management on significant acreages due to the greater number of landowners involved. On the other hand, fragmentation and urbanization would interrupt or decrease the amount and continuity of host species, thereby potentially decreasing the spread and impacts of pests.

We expect continuing introduction (through international and domestic commerce and tourism) of nonnative insects and diseases which could become pests of forest trees, despite imposition of inspections and quarantines. Which organisms might be introduced, and then which of these might become pest species is the source of significant speculation, but is relatively unpredictable.

Discussion and Conclusions

Future Considerations for Pest-Host Relationships

Planned adaptation (Spittlehouse and Stewart 2003) should reduce vulnerability for commercial tree species at selected sites. However, many forest species will have to adapt autonomously and society will have to adjust to the result (Winnett 1998). Forest pest distribution changes caused by climate change are likely closely tied to shifts in host distribution (Sturrock 2007).

Some ecosystems are expected to be new: new communities of tree and plant species with different suites of insects and pathogens. If forests do remain on a particular site, similar functional types of insects and pathogens are likely to remain, although they may be include different species than at present (Beukema and others 2007). Pathogens expanding their ranges and contacting 'new' hosts and vectors may mean that new pathosystems probably will emerge. Interactions between pathogens may change (Sturrock 2007).

Climate change may amplify the impact and aggressiveness of pathogens or alter the balance between pathogens and their natural enemies; it may also change the status of weak/opportunistic pathogens such that they are able to infect and damage stressed tree hosts (Sturrock 2007).

Tree decline is likely to increase in a warmer and drier climate, regardless of inputs from diseases and insects. The effect of warmer and drier climate is to stress trees used to a cooler and moister

regime. This stress alone should cause an increase in the incidence of declining trees, but compounded by the presence of opportunistic insects and pathogens, there is a strong possibility that this increase in declining trees could be significant. Increasing incidence of decline should eventually diminish as new adapted ecosystems form in the region, but this is not expected to occur within the next 50 years.

Almost every study and review of climate change effects on forests has a common caveat—the complexity the ecosystems and pest systems, about which relatively little is known (Sturrock 2007). The difficulty in predicting the future of plant disease is highlighted by Woods and others (2005), who report on an endemic needle blight fungus (*Mycosphaerella pini*) that previously had only minimal impact on native forest trees in British Columbia. However, recently, in apparent response to a local increase in summer precipitation, this disease has been causing extensive mortality of lodgepole pines. While admitting that establishing causality of the increased virulence of this endemic pathogen is fraught with risk of misinterpretation of the evidence, they indicate the link to precipitation (while dismissing warmer temperatures) appears to be far greater than "circumstantial". No prior indication of this shift to virulence appears in the literature—the event was unprecedented, unpredicted, and possibly unpredictable. In partial confirmation, Sturrock (2007) notes that wetter springs in some regions may result in increased foliage diseases without venturing to predict subsequent possible host/pest scenarios.

Endemic root rot fungi (*Inonotus schweinitzeii*, *I. tomentosus*, or *Ganoderma* spp.), which currently cause limited damage, or insects such as engraver beetles or species of wood borers could become important management concerns or could fade into obscurity from a management standpoint. The fungi that cause littleleaf disease, sudden oak death (Brasier and Scott 1994), and other infections are predicted to increase their activity in temperate zones in the Northern and Southern Hemispheres as they migrate away from the tropics. Under changing climatic conditions these fungi are expected to

cause more damage to existing urban and forest tree hosts in the South and to expand the number of species they can infect. Expected to be especially prevalent and damaging are those, like the littleleaf disease fungus, that can grow in temperatures higher than 28° C (Broadmeadow 2005).

Increased drought stress on hosts may mean increased mortality from root pathogens. Pathogenic *Armillaria* spp. fungi may be assisted by the impairment of host tolerance caused by climate change-induced stress: this may enable less pathogenic fungi to become more successful on stressed trees (Sturrock 2007). Incidence of oak and beech decline, highly complex disorders, is likely to increase if the predicted frequency and severity of summer drought stress prove accurate (Broadmeadow 2005).

A changing climate with increased temperatures, increased evapotranspiration, and extreme weather events would increase the frequency and severity of stress factors, which may lead to more frequent forest declines (Sturrock 2007). Pathogen evolution could be accelerated by mutation resulting from increased sunlight or increased reproduction rates (shorter life cycles under higher temperatures) that could lead to host resistance being overcome more rapidly (Coakley and Scherm 1996).

Based on these occurrences and trends, the following basic patterns have emerged on which we have built our projections of future impacts of pests:

- The current emphasis on longleaf pine restoration, coupled with increasing temperature and decreasing rainfall should result in a measurable shift in the population distribution of southern yellow pine types, both spatially and numerically.
- Boreal forest species such as hemlock, Frasier fir, eastern hemlock, American, beech, and others are expected to have their ranges reduced in the South due to the combined effects of increased temperature and decreased available water.

- Pests associated with southern host species are expected to migrate with their hosts with few exceptions. The exceptions are those pests that already occur throughout the South and extend into the northern part of the United States.
- Although long-term projections are suggest an increasing coastal savannah replacing forest types in many coastal and coastal plains locations, the progress of this change within the next 50 years is not expected to be severely impacting.
- Most root rotting diseases are expected to respond aggressively to the combination of warmer soil temperature and reduced precipitation. This combination of heat and drought is expected to result in an increase in dieback and decline among many tree species, often providing further stress that could act as a precursor to successful invasion/colonization by root rotting fungi. Newly stressed trees also may become the focus of insect attack.
- Trees suffering long-term stress may prove to be more resistant to secondary pest attack because of lower physiological activity and reduced availability of resources needed by pest organisms.
- Tree diseases which affect primarily stem and branch tissue are subject directly to the potential effects of warmer temperatures and a drier environment. At first, warmer temperatures and increased carbon dioxide in the atmosphere are expected to have a stimulatory effect on both host and pathogen. However, the anticipated lower availability of water should generally function more against the host plant than the fungi infecting it, favoring an increase in disease. This assumes that the temperature increase does not exceed the thermal death point of the fungus or its spores.
- Foliage attacking fungi are subject to significant pressure from light and the microclimate in the host's leaves. Although significant loss of spore viability is common on the upper surface of leaves, any change in the amount of sunlight will normally alter the survival rate; more sunlight results in lower spore survival and less successful infection and vice versa. The microclimate of the underside

of leaves is also critical to the success of foliar pathogens. Lower atmospheric moisture resulting from less rainfall, fog, and dew, (with a secondary effect of reduced secretion of liquids) is expected to reduce the effectiveness of colonization by leaf-infecting fungi.

- Longer and warmer summertime temperatures are expected to increase pathogen and insect activity. Insect populations may show simple increases in number due to the availability of additional host material on which to browse, or may be able to produce an additional generation each year.

Managing Pests Under Changing Conditions

Many land-management decisions made today are based on the assumption that the climate will remain relatively stable throughout a forest's life—an assumption that may have worked well in the past but is being challenged by climate change. Even without a clear view of the future climate and forest, it is possible to develop adaptive strategies now. Adaptation in forest management requires a planned response well in advance of the impacts of climate change (Spittlehouse and Stewart 2003). This is especially important when the rotation periods are long (Lemmen and Warren 2004).

Changes in climate, especially if they lead to greater variability among and within regions, tend to add extra uncertainty to decisionmaking (Garrett and others 2006). Burton and others (2002) appear to contest the conclusion of Spittlehouse and Stewart (2003) cited above with their conclusion that development of adaptation measures for some time in the future, under an uncertain climate, in an unknown socioeconomic context is bound to be highly speculative. Not so; reconciling the apparent contradiction here is the necessity that best professional judgment rather than proven science be brought to bear on planning for an uncertain, but generally predicted future.

Adaptive strategies include resilience options and response options. Mitigation options include options to sequester carbon and reduce overall greenhouse gas emissions (Millar and others 2007). Coping strategies for one disturbance type are often appropriate management responses to other disturbance types. Before disturbance occurs forests can be managed to reduce vulnerability or to enhance recovery. Trees can be planted that are less susceptible to disturbance. Species that promote disturbance can be removed (Dale and others 2001). Millar and others (2007) propose the following generalized strategies:

- **Improve resistance in hosts:** From high-value plantations near to harvest to high-priority endangered species with limited available habitat, maintaining the status quo for a short time may be the only or the best option. Resistance practices seek to improve forest defenses against direct and indirect effects of rapid environmental changes by reducing the undesirable or extreme effects of fires, insects, and diseases. Because they may require intensive intervention, these options are best applied only in the short-term.
- **Promote resilience to change:** Resilient forests are those that not only accommodate gradual changes related to climate but also tend to return toward a prior condition after disturbance, either naturally or with management assistance. Promoting resilience is the most commonly suggested adaptive option discussed in a climate change context. This process may also become intensive as changes in climate accumulate over time.
- **Enable forests to respond to change:** These adaptation options intentionally accommodate change rather than resisting it. Treatments implemented would mimic, assist, or enable ongoing natural adaptive processes such as species dispersal and migration, population mortality and colonization, community composition and dominance within communities, and disturbance regimes. Some potential practices include: (1) Increase redundancy and buffers, manage for asynchrony, realign

significantly disrupted conditions, and use establishment phase to reset succession; (2) Establish "neo-native" forests, experiment with refugia, and promote connected landscapes; (3) Develop indicators as a prerequisite for any kind of decisionmaking and surveillance networks to assess spatial and temporal evolution of diseases and improve epidemiological models; (4) Take an anticipatory and preventive approach based on risk analysis when addressing disease management in forest ecosystems (even more so than for crops), avoid total reliance on one or two control strategies (as Hain [2006] recommended when discussing the unsatisfactory results of balsam woolly adelgid control efforts), and anticipate surprises and threshold effects

- Disease management options could be altered (Coakley and others 1999) or imposed. For example, although it is known that movement of firewood, nursery stock, and even family trailers and boats is responsible for the transport of many species, there is no cohesive strategy for addressing this problem (Moser and others 2009). Other actions proposed for managing insects and diseases include:
- Avoid dissemination of pests into climatically favorable zones where they could find naïve host populations by practicing strict hygiene measures, based on the most probable dissemination pathways of organisms (in seeds, wood, and plants).
- Reduce vulnerability to future disturbance by managing tree density, species composition, forest structure, and location and timing of activities (Dale and others 2001).
- Increase light, water, and nutrient availability to the uninfected/uninfested trees and decrease susceptibility to pest attack by practicing precommercial thinning, sanitation removal, or selective removal of suppressed, damaged, or poor quality individuals (Gottshalk 1995; Wargo and Harrington 1991; Papadopol 2000).

- Underplant with other species or genotypes in forests where the current composition is unacceptable as a source of regeneration (Spittlehouse and Stewart 2003).
- Shorten rotations to reduce the period of stand vulnerability to insect or disease attack, and replant to speed the establishment of better-adapted forest types (Gottshalk 1995; Parker and others 2000).
- Use pesticides in situations where silvicultural or other means of pest management are ineffective (Parker and others 2000); however, because morphological or physiological changes in the host resulting from increased carbon dioxide uptake could affect uptake, translocation, and metabolism of systemic fungicides (Coakley and others 1999), incorporate integrated pest management practices.
- Expand and improve existing monitoring efforts to include an expected increase in the number of new, introduced plant diseases (Sturrock 2007).
- Assist in the migration of forests, by introducing carefully selected tree species (including using biotechnology techniques in some situations) in regions beyond their current ranges, being mindful of the potential for unforeseen consequences.

With respect to nonnative invasive species management, Moser and others (2009) recommend five priorities: (1) promoting education and awareness, (2) expanding early detection and active management and intensifying enforcement of quarantines, (3) building the capacity to increase understanding of and treatments for NNIS control, (4) strengthening the basic forest health curriculum, and, (5) encouraging cross agency collaboration and investment.

Although the process of planning and acting to prepare for a future most probably affected by climate change is fraught with uncertainty, not planning and acting will likely result in greater economic and social disruption. Success can only be achieved if those in environmentally sensitive management roles are well informed and exercise their best judgment.

The single consistent theme throughout the literature on pest impacts and climate change is that minimizing ecological change (and disruption) requires maximum possible biodiversity, either through a system of protected refugia or by direct adaptive management for specific characteristics.

Differing perceptions of risk and adaptation may lead to increased tension among various groups. Conflicting priorities and mandates could also lead to future problems (Lemmen and Warren 2004). In these situations, care must be taken to adopt a decisionmaking process that identifies and evaluates all issues and employs the best ecological science.

Knowledge and Information Gaps

As should be clear from the above discussion of current knowledge and from our projections of the future activity of known pests, huge uncertainty dominates the subject of pest management and climate change, with significant gaps existing in baseline knowledge making any generalized quantitative modeling of future conditions impossible. Although some specific pest behaviors have been projected, most of them are qualitative. Lacking generalized and often specific baseline data leaves modeling (quantitative projection) a desired tool whose time has yet to come. Currently unavailable data that would contribute to a generalized projection of potential future pest activity in forests (Beukema and others 2007; Chakraborty and Dalton 2003; Hain, 2006; Lemmen and Warren 2004; Logan and others 2003; Mamlstrom and Raffa 2000; Rogers and others 1994; Scherm 2004; and, Seem 2004) include:

- Information on host biology and response to pests: the role of changing secondary metabolites (primarily phenols or phenol like) under changing environmental conditions; the functional components of respiration (construction, maintenance, and ion uptake) as well as carbon costs due to root exudation; the role of water in tree health; the genotypic variability and plasticity of hosts; water balance threshold as it affects direct mortality of host plants, the effects of climate change on

host defensive mechanisms (physiological, morphological, or other); the impact of climate change on biodiversity and the role of biodiversity in ecosystem functions and pest management/prevention; and, projections of host migration and availability under the influence of climate change.

- Information on forest pests: current distributions and ranges of pests; influence of mycorrhizae on plant health under climate change; direct and indirect effects of carbon dioxide, ozone, and UV-B on roots and root-surface microfloras under natural conditions; knowledge of insects and pathogens from outside the area such as Mexican bark beetles and various Asian insects; mechanisms by which changes in carbon dioxide and precipitation alter pest survival, growth, susceptibility and interactions
- Information to add clarity and specificity on pest/host interactions: dispersal structure and distance and interconnectedness of temperature, phenology and pest population growth rate; phenological relationships among trees and pests; role of climate on insects and pathogens in relation to available water; baseline data on pests of natural populations that identify the separate of multiple climate variables and problems they cause (including forecasts of epiphytotics or epizootics, and evaluations the role of evolution); pest/predator interactions and responses, relationships among climate, pests, and their parasites; minimum and maximum temperature preferences of pests and pest/host interactions and response to temperature extremes; protocol for identifying the “drivers” that transform new insects and diseases into pests; disturbance regimes and their interactive impacts; and, synergies among fire, insects, and pathogens.
- Models and modeling protocols needed: models that incorporate local meteorological data; improved spatially-explicit climate predictions at finer scales (average daily patterns and projected variations from the average); effects of down-scaling or up-scaling data from various models and

appropriate linking tools for increasing the accuracy of these predictive processes to be more accurate predictors; functional group rather than single-species models; and, predictive models that incorporate data on disturbances and disturbance impacts.

- Management information needed: a new protocol for addressing the research needs of invasive forest pests that involves all stakeholders in a coordinated partnership; and management action plans developed in the face of no-analog vegetation systems and climate change.

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Tables

Table 16-1—Important insect and disease pests of southern forests

Pest	Pest's scientific name	Type of pests / abiotic factors	Origin	Forest type or species affected
Annosum root disease	*Heterobasidion annosum*	Fungus	Native	Pines in the loblolly-shortleaf and longleaf–slash forest types
Asian longhorned beetle	*Anoplophora glabripennis*	Insect	China	Most hardwoods
Baldcypress leafroller	*Archips goyerana*	Insect	Native	Baldcypress in oak-gum-cypress forest type
Balsam woolly adelgid	*Adelges piceae*	Insect	Europe	Fraser fir in the spruce-fir forest type
Bark beetles (other than southern pine beetle)	*Ips avulsus, I. calligraphus, I. grandicolli, & Dendructonus terebrans*	Insect	Native	Pine in the loblolly-shortleaf and longleaf–slash e forest types
Beech bark disease	*Nectria coccinea var. faginata, N. galligena* (fungi); 2 (at least) insect vectors	Complex of insects & fungi	Unknown	American beech in the oak-hickory forest type
Brown spot needle disease	*Scirrhia acicola*	Fungus	Native	Longleaf pine in the longleaf–slash forest type
Butternut canker	*Sirococcus clavigignenti-juglandacearam*	Fungus	Unknown	Butternut in the oak-hickory forest type
Chestnut blight	*Cryphonectria parasitica*	Fungus	Asia	American chestnut, chinquapins, several species of oak in the oak-hickory forest type
Dogwood anthracnose	*Discula destructiva*	Fungus	Unknown	Dogwood in the oak-hickory forest type
Dutch elm disease	*Ophiostoma ulmi (formerly called Ceratocystis ulmi)& Ophiostoma novo-ulmi* (fungi); two bark	Complex of fungi and insects	Europe	All elm species

	beetles			
Emerald ash borer	*Agrilus planipennis*	Insect	Asia	All ash species
Forest tent caterpillar	*Malacosoma disstria*	Insect	Native	Hardwoods in the oak-gum-cypress forest type
Fusiform rust	*Cronartium fusiforme f. sp. fusiforme*	Fungus	Native	Loblolly and slash pines in the loblolly-shortleaf and longleaf–slash types
Gypsy moth	*Lymantria dispar*	Insect	Europe and Asia	Hardwoods (all types)
Hardwood borers	Various	Insect	Native	All species of hardwoods
Hemlock woolly adelgid	*Adelges tsugae*	Insect	Asia	Hemlocks
Laurel wilt	*Raffiella lauricola* (fungus), *Xyleborus glabratus* (insect)	Complex of an insect and fungus	Unknown	Redbay
Littleleaf disease	*Phytophthora cinnamomi, Pythium sp.*	Tree decline complex; fungi and site factors	Southeast Asia (likely)	Shortleaf and loblolly pines in the loblolly-shortleaf forest type
Loblolly pine decline	As a minimum: various fungi *(Lophodermium spp.)* and insects *(Hylastes spp.)*	Tree decline complex; insect and fungi	Unknown	Pines
Nantucket pine tip moth	*Rhyacionia frustrana*	Insect	Native	Pines
Oak decline	*Armillaria* sp., and other secondary fungi	Tree decline complex; site conditions and fungi	Mixed	Oaks
Oak wilt	*Ceratocystis fagacearum*	Fungus	Native	Oaks in the oak-hickory forest type
Pine reproduction weevils	*Hylobius pales, Pachylobius picivorus*	Insect	Native	Pines
Sirex woodwasp	*Sirex noctilio* (insect), *Amylostereum areolatum* (fungus)	Complex of an insect and fungus	Europe, Asia, northern Africa	Pines
Soapberry borer	*Agrilus prionurus*	Insect	Mexico	Western soapberry

Southern pine beetle	*Dendroctonus frontalis*	Insect	Native	Pines
Sudden oak death	*Phytophthora ramorum*	Fungus	Unknown	Oaks
Texas leafcutting ant	*Atta texana*	Insect	Central and South America	Pine (reproduction)
Thousand cankers disease	*Geosmithia sp.* (fungus), *Pityophthorus juglandis* (insect)	Complex of an insect and fungus	Unknown	Black walnut

Figures

Figure 16-1—County-level distribution of established hemlock woolly adelgid populations, as reported by State forest health officials in 2009. Populations are not distributed evenly within infested counties (Adapted from U.S. Department of Agriculture, Forest Service 2010). Note: This map is undergoing rapid change due to the ongoing expansion of the range of this insect.

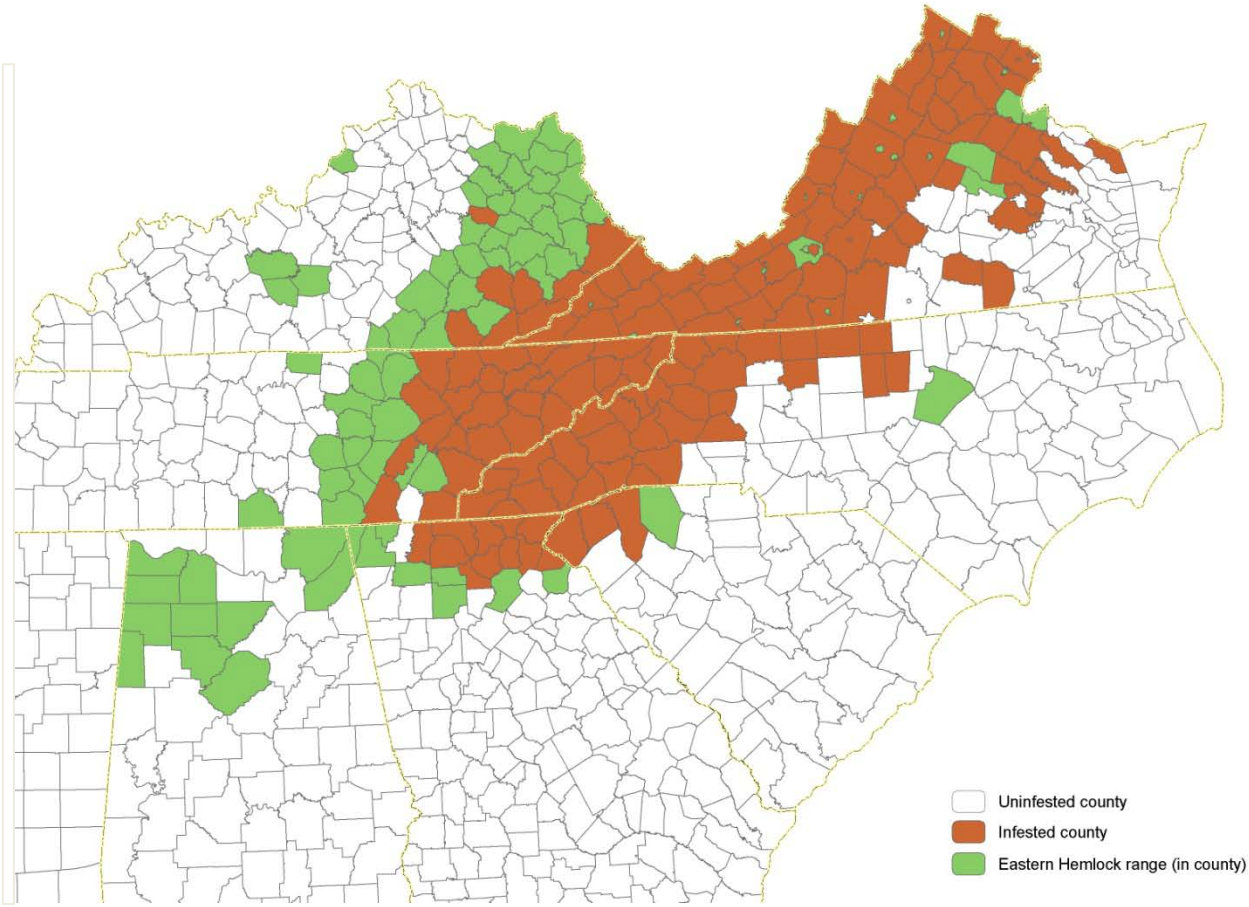


Figure 16-2—Emerald ash borer locations in the northern United States and southern Canada as reported February 1, 2011 (adapted from U.S. Department of Agriculture Animal and Plant Health Inspection Service 2011). Note: This map is undergoing rapid change due to the ongoing expansion of the range of this disease. The most current map can be found at: http://www.emeraldashborer.info/files/MultiState_EABpos.pdf.

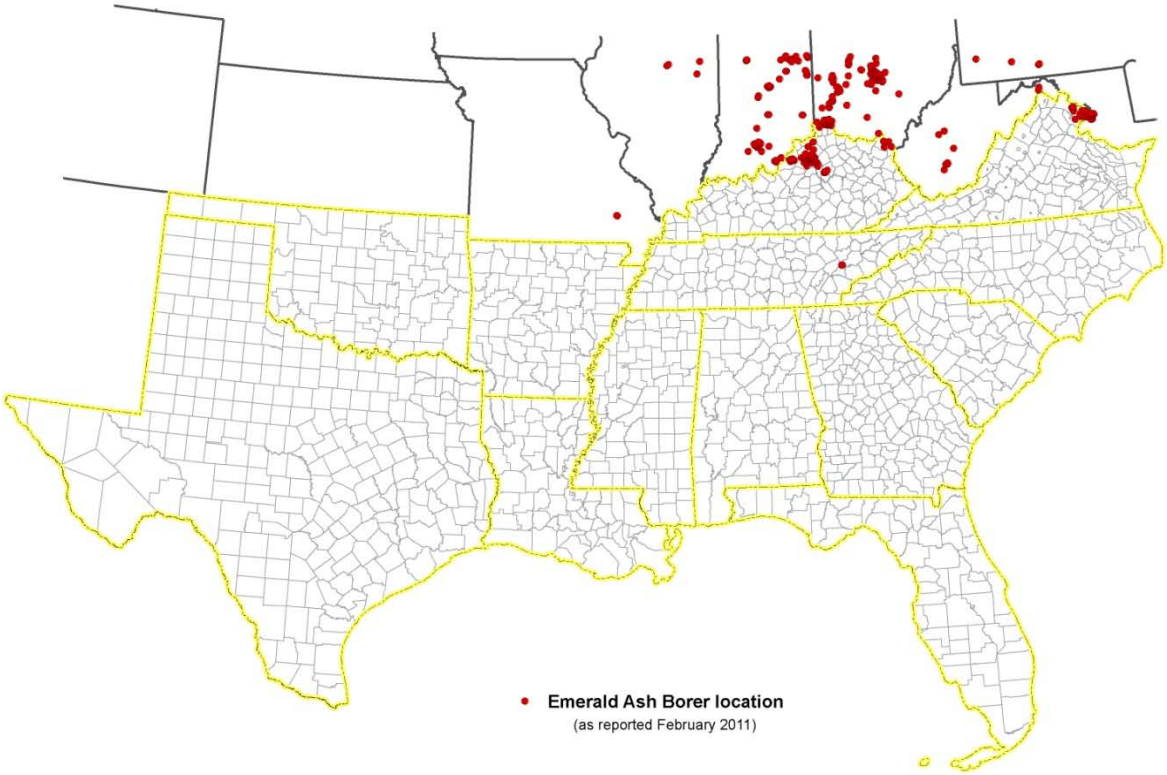


Figure 16-3—Counties where soapberry borer has been found in Texas through December 2010; courtesy of the Texas Forest Service, Texas A&M University System (adapted from Billings 2011). Note: This map is undergoing rapid change due to the ongoing expansion of the range of this insect pest.

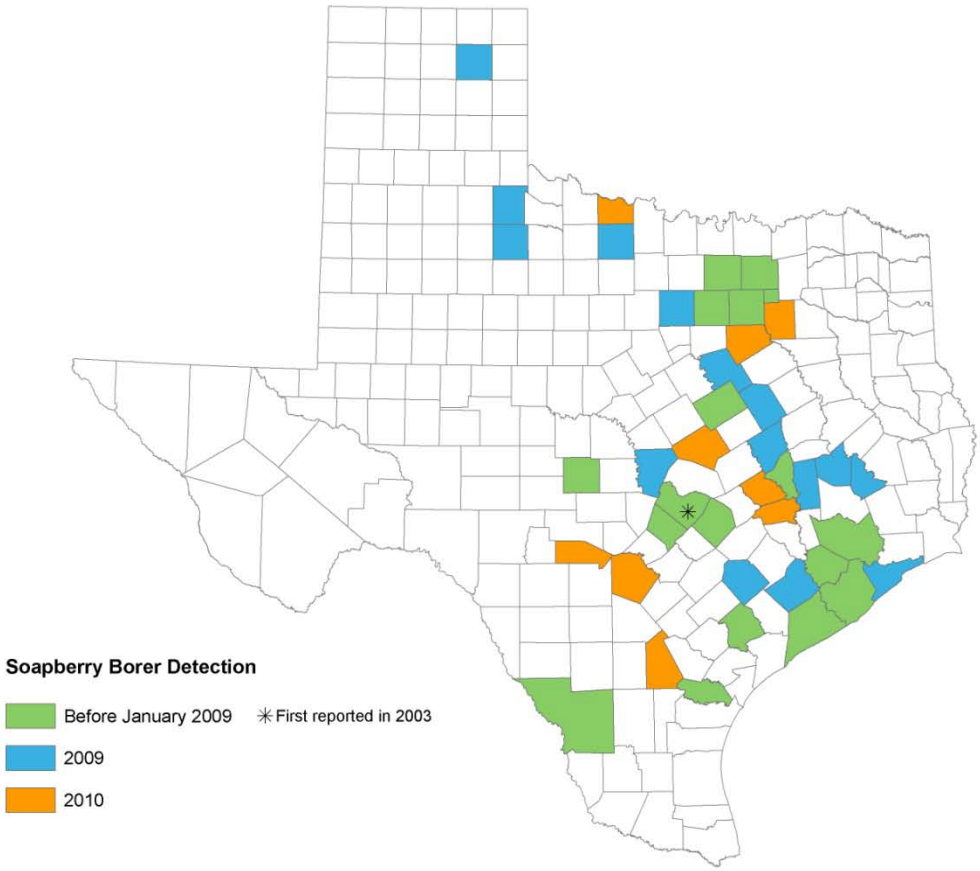


Figure16-4—Distribution of counties with laurel wilt disease by year of initial detection and as confirmed through laboratory analysis of host samples collected in the counties affected; updated March 3, 2011 (adapted from Reid and others 2011). Note: This map is undergoing rapid change due to the ongoing expansion of the range of this disease. The most current map can be found at: http://www.fs.fed.us/r8/foresthealth/laurelwilt/dist.map.shtml.

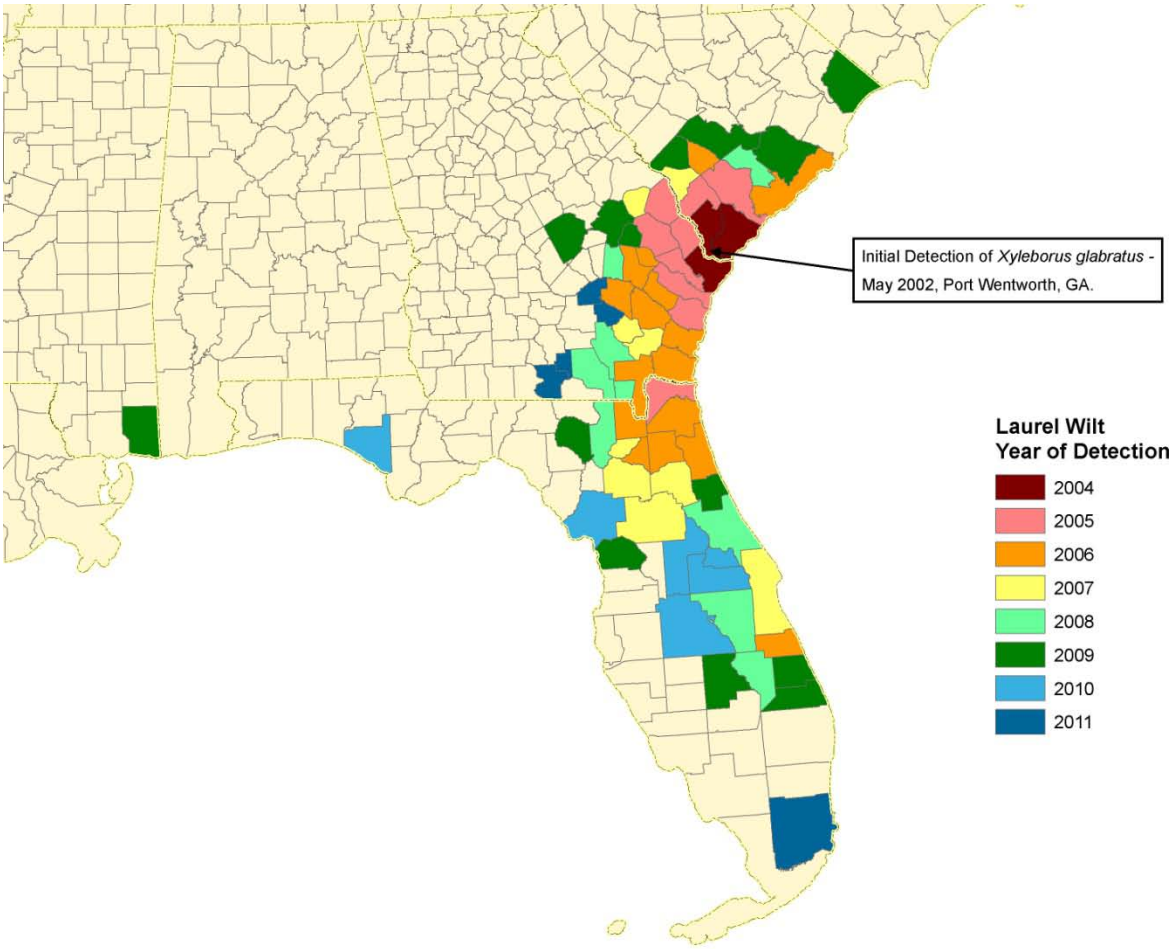


Figure 16-5—Probable spread of laurel wilt disease from 2006 to 2040, based on the current rate of spread and known distribution of the redbay host (Adapted from Koch and Smith 2008b).

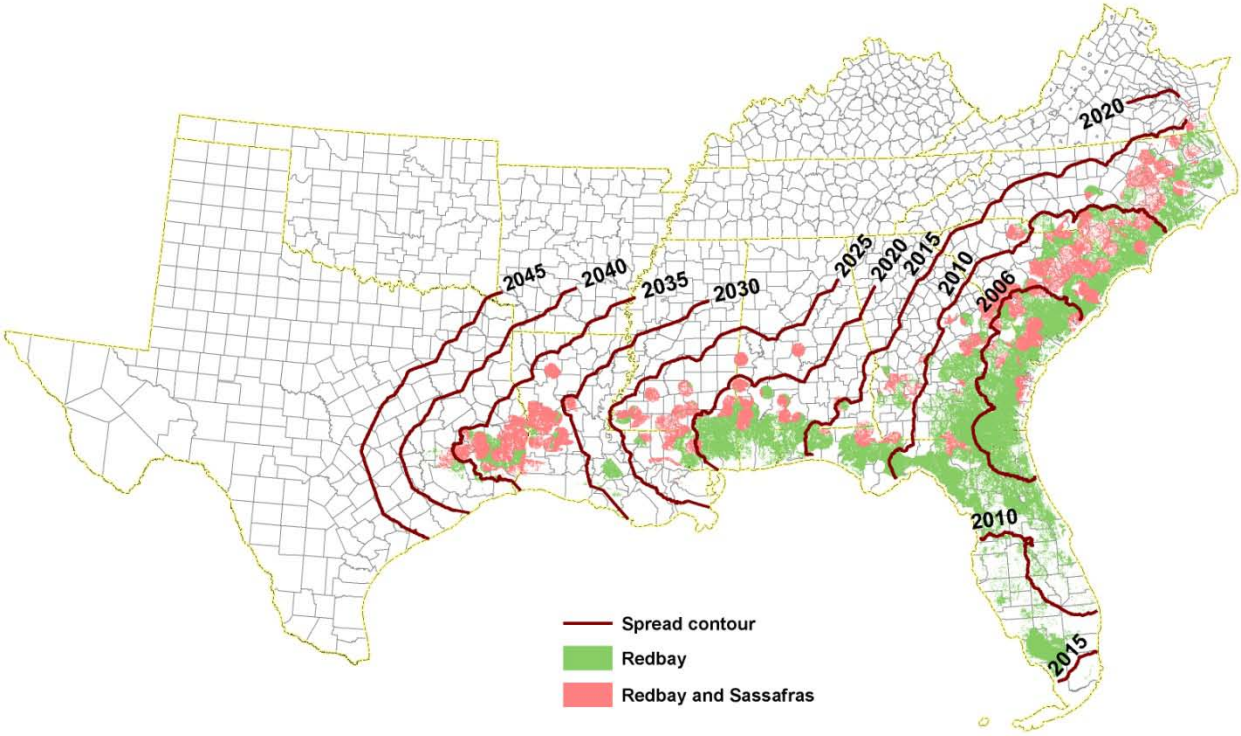
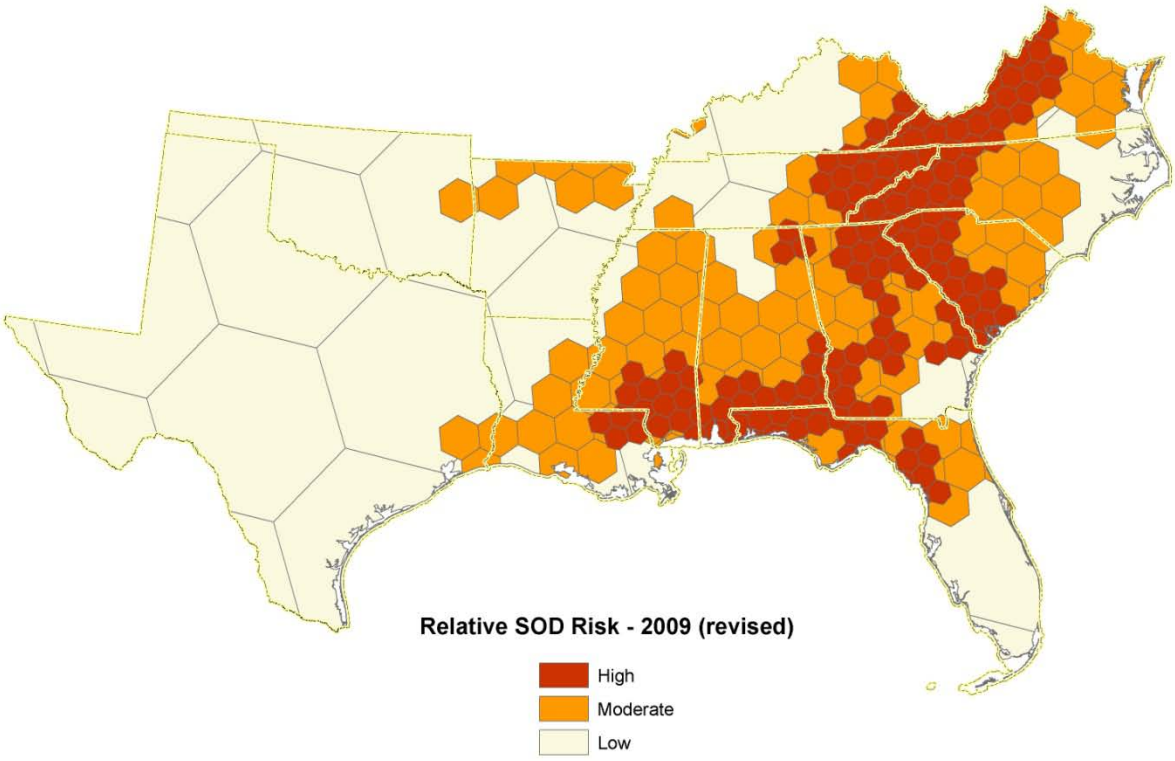


Figure 16-6—Potential range for sudden oak death in the contiguous United States based on the distribution of known or likely hosts, climate conditions adequate for the survival and propagation of the pathogen, and probable pathways of Introduction of the disease outside of its current range (revised from Koch and Smith 2008a by Koch [pers. Comm.])



Appendix A

Introduction

The *Southern Forest Resource Assessment* (SoFRA) (Wear and Greis 2002) presented a compilation of shorter reports dealing with the state of Southern forests as well as with various factors potentially affecting those forest resources. *Chapter 17 – Impacts of Pests on Forest Health* (Ward and Mistretta 2002) presented an overview of forest insect and disease pests which currently significantly affect on a relatively broad scale the management of forest resources in the South. Divided first as insect or disease pest, and then separated by having effect on conifer or hardwood, they were finally separated as native or non-native pests; 21 pests were documented in the 8 resulting categories. Since 2005 the listing of pests (either active or probable significant management concerns) has changed and 30 pests are discussed specifically in the current work.

In the current effort we are being challenged to work from where we are (slightly updated from Ward and Mistretta 2002), and project the behavior of insect and disease pests which we anticipate will affect the forest resource during the next 50 years, based on changing anthropogenic and biotic factors not considered in detail in the SoFRA. Climate change (and its subelements such as temperature régime [primarily temperature extremes], overall pattern of insolation, and rainfall pattern) is a primary focus of this work. Based on climate change scenarios we anticipate vegetation changes to occur (Iverson and others 1999). Concurrent with the edaphic and ecological changes just mentioned a shift in the pests affecting the changed vegetation under changed temperature, rainfall, etc. conditions is expected to occur. Pest activity, in its turn, may influence the direction or scope of the other changes in forest type and structure.

While initially we were to consider only non-native invasive pests (NNI pests) in the context of 'the future forest', it became apparent that the scope was too narrowly focused if we were attempting to project a reasonably dynamic future forest system. So, we have incorporated the pests from the previous effort (Ward and Mistretta 2002) and added several new ones, as mentioned above. For those which were discussed previously in the Southern Forest Resource Assessment data relevant to the current discussion has been excerpted, either directly or with necessary updating, from that document (Ward and Mistretta 2002). This data is presented in the body of the chapter.

During the scoping phase of this project several specific key issues were identified and are detailed in the chapter proper.

Methods

In order to bring some organization to the responses to these questions, our approach in this paper will be to present information:

- in this Appendix as a generalized summary of the relevant literature related to:

  Climate change

  Vegetation change (species and geographic range changes)

  Pest activity scenario classification as reflected in the current literature

- In the Chapter proper:

  Very brief extracts of relevant information for the pests described in the SoFRA (Ward and Mistretta 2002) alphabetically, separated by pest as 'Insect'or 'Disease', and any relevant new data (beyond the SoFRA itself)

  Full descriptions of each of the several new pests or pest complexes which have emerged in the past few years

Qualitative implications of the literature review with respect to pests and pest management in the Southern U. S. over the next 50 years (initial 10 year increments were found to be too finely drawn for discussion due to the overall gaps in the quantitative database discussed later); discussion will include key issues identified in the paper

Management strategies for responding to the pests in a changing environment

Research needed to allow refinement of our knowledge base relating to pests, their hosts, and their interactions as well as the interactions with their changing environment, thereby permitting the application of a more quantitative approach to forecasting in the future

Data Sources

Information for this appendix was derived from published science literature, which is cited as appropriate. Only a limited selection of articles is cited in the chapter, often only one of many on a given subject. These cited articles form only a small part of the very extensive literature about southern forests, their ecology, and the pests and edaphic conditions which affect them. Additional information about forest pests and their control is readily available from State and Federal forestry agencies or on the Internet (two good starting points are http://fhpr8.srs.fs.fed.us/ and http://www.na.fs.fed.us/spfo/pubs/fth_pub_pages/fidl.htm).

A variety of additional citations not referenced in either Chapter 16 or this appendix but which provide valuable additional background for understanding the biology and ecology of the pests discussed are listed in the “References” section at the end of this appendix.

Results

It is now an accepted premise among a large body of scientists that climate change is an ongoing phenomenon in the global environment. However over the last century a globally experienced warming trend with overall reduced rainfall has occurred and is predicted to continue to occur into the immediate future (Kliejunas and others 2009; Malcolm and Pitelka 2000; McNulty and Aber 2001; Intergovernmental Panel on Climate Change 2007; National Assessment Synthesis Team 2000).The primary factors of climate consistently noted as driving observed ecological effects are temperature and available water. In addition atmospheric gasses (CO_2 and air pollutants such as nitrogen oxides [NO_x] and sulfur dioxide [SO_2]) in excess of 'normal' ranges are often identified as drivers of the change being observed. It is generally conceded that these changes will directly impact crops and their pests, as well as their interactions (Runion 2003). Climate change is generally linked, at least in part, to human activity (Sturrock 2007; Malcolm and Pitelka 2000; Winnett 1998).

Reporting the results of a workshop attempting to understand the potential interactions between forests, insects, diseases and climate change, Beukema and others (2007) state: "Participants agreed that things will change. Most vegetation communities will not simply migrate from one location to another. Instead, many communities will be completely new, with new combinations of trees, understory plants, insects, and pathogens. At the same time it is important to bear in mind that we are not going to completely lose all forests and all vegetation. New plant communities will organize themselves and will replace plants that are unable to adapt to new climates. New communities could include current tree species, other tree species (*e.g.*, hardwoods or strongly dispersing species from warmer areas) or could become dominated by grass and shrub species."

At the present time major effort is being expended to create and use models to project potential scenarios for both the edaphic condition changes resulting from climate changes and subsequent ecological changes resulting from and possibly then influencing those factors. Modeling can contribute

to our projections of future conditions "…but requires sound knowledge of the causal factors determining spatial distribution, survival, reproduction, dispersal, and infliction of damage" (Goudriaan and Zadocks 1995). As recently as 2011 (Sturrock et al.) we find in the literature the unfortunate statement that "rating systems for most pathogens either have not been developed or do not account for climate change."

Several different broad-scale model types (Global circulation models [GCMs], process-based models, empirical models, etc.) are used depending on the specific questions being analyzed and the available relevant data. The application of GCMs is limited; the finest scale used for global climate simulation is far too coarse for meaningful ecological applications (Logan and others 2003). At a smaller scale models such as gap models, biogeography models, and biogeochemistry models are being used to attempt to refine probable broad-scale model projections to more local conditions (Winnett 1998).

Model projections of future conditions which will affect forest composition and productivity vary over a wide range of plausible scenarios (Logan and others 2003; Scherm 2000; National Assessment Synthesis Team 2000).

In his paper Scherm (2000) refers to a 1994 article by Millstein which points out that uncertainties in model input can compromise the credibility of the output because of error perpetuation or propagation. They are not alone in expressing this concern, and to it is added the concern that data selection can also significantly influence model output. The use of a crisp data set versus a fuzzy number set will have additional major impact on outputs (Coakley and others 1999).

Scherm and Coakley (2003) have identified three continuing problems with the application of models for predicting climate change effects: model inputs have a high degree of uncertainty; non-linear relationships and thresholds in the relationship between climatic variables and epidemiological

responses make it difficult to collect sufficient data for clear predictive understanding; and, the potential for adaptation by plants and pathogens is often ignored in modeling. (Note: This statement written in a discussion of plant disease is accurate when extended to include insects pests of plants.)

In Chapter 2 of this Assessment several maps and graphs of projections made for the six Forest Futures "Cornerstone Futures" are presented.

Forecasts of total growing stock for all scenarios end up above current growing stock levels after 50 years, but only one of the projections (Future F) is on an upward trajectory at the end of that time interval. All other scenarios increase at first but then decline. All futures indicate an increase in softwood growing stock, and all but one (Future F) show a decline to below current levels of hardwood stocking by the end of the 50-year projection window. All futures show an overall decline in total forest area; range of this loss across the cornerstone projections (visual approximation from the graphs) is between 5 million and 20 million acres of forest lost during the next 50 years. All projections show increased population density at the end of the 50 year window of projection, with some three of the subregions showing significant population density increases and two showing moderate increase in population density .Population shifts are reflected in two maps with some areas gaining significant numbers of people and others losing them.

In Chapter 2 of this analysis two figures are used to show projected impacts on temperature and rainfall from 2000 – 2060. These figures are briefly discussed immediately below.

Physical impacts of climate change

Increase in average temperature is consistently shown in results from a variety of models as being of concern. The Intergovernmental Panel on Climate Change (Intergovernmental Panel on Climate Change 2007) stated that data showing warming of the climate system is unequivocal, as is evident from

observations of increases in global air and ocean temperatures, widespread melting of snow and ice, and rising global average sea level. They then quantify the linear trend in temperature increase to be 0.74 [0.56-0.92]^{o} C for the century 1906-2005, and further point out that this is higher than the increase found for 1901-2000 (0.6 [0.4-0.8]^{o} C) published in their previous report. They also point out that land areas have warmed faster than oceans, and that the temperature increase appears to be greater in the northern latitudes.

In gross terms, climate change is predicted to lead to increasing temperature. Mean global surface air temperatures are predicted to increase in the range 1.4^{o} – 5.8^{o} C by the end of the current century. Both night-day and winter-summer mean temperature ranges are likely to reduce as minimum temperatures increase more than maximum ones; and continental and high-latitude areas will tend to warm more than coastal and lower-latitude ones (Burdon and others 2006; Harvell and others 2002). The magnitude of these changes is expected to vary both temporally and spatially (McNulty and Boggs 2010).

Chapter 2 includes four maps of projected temperature change in the South over the next 50 years, one map for each of the Cornerstone Futures A through D. All projections basically show a minimum 0.5^{o} C increase in temperature with the maximum projected increase being greater than 3° C in parts of the South. Cornerstone future A (and E which has the same temperature regime) shows the greatest overall increase in projected temperature, in a pattern far more extreme than is reflected in the other scenarios. With respect to the other Cornerstone Futures projected temperature increase overall declines in the sequence Future D (= F with respect to temperature regime), Future B, and with the least impact being projected in Future C. (Note that only several very small, scattered areas in Virginia, two spots along the Mississippi River [Louisiana-Mississippi border], and Lake Okeechobee, Florida consistently show 0-0.5^{o} C change).

Water is reported to be of great significance, second only to temperature, when projecting potential effects of climate change. Overabundance of water, lack of it, and seasonality of availability, all have significant impact on forest processes which govern overall health of the individuals present. Projections of overall effects to rainfall pattern vary greatly. Generalizations found in the literature include the following:

> In the South intense precipitation events have increased over the past 100 years (National Assessment Synthesis Team 2000).
>
> Sea-level rise has already had significant impacts on coastal areas, and these impacts will likely increase (National Assessment Synthesis Team 2000).

However, Malcolm and Pitelka (2000) summarized the effects of water as follows: future regional-scale precipitation changes remain particularly difficult to predict, and changes in the frequency and severity of extreme weather events such as storms are uncertain (Wigley 1999). Overall these changes will appear as a shift of climatic zones towards the poles as warmer temperatures reach further north in the United States.

This last observation introduces a critical concern when discussing climate change. Edaphic factors do not function in isolation, they interact. This is a fact easily forgotten when reading the literature, much of which discusses single factor effects at any of a variety of scales. However, edaphic factors influence each other, interacting in a significant manner as noted in the previous quote where precipitation *under the influence of warmer temperatures* [emphasis added] is expected to contribute to the poleward shift of climatic zones.

One figure in Chapter 2 shows projections of change in rainfall expected over the next fifty years. As for temperature, Cornerstone Future A is extreme in its projection; all areas of the South are shown

losing precipitation compared with the present (loss from 0-50 mm [0-1.52 inches] to > 150 mm [4.57 inches] lower than at present). Cornerstone Futures B through D show a mixed pattern of projected rainfall with increases and decreases ranging from more than 150 mm additional rainfall to a loss of greater than 150 mm rainfall. Of these three, Cornerstone C shows an overall pattern with the most area in the South losing rainfall, and Cornerstone B shows the least area losing rainfall from the present while simultaneously has the largest area gaining rainfall, Cornerstone D falls between B and C.

Carbon dioxide as a byproduct of the industrial revolution is routinely cited as a primary cause of global warming. The present consensus among informed scientists is that the increase in CO_2 and shifting percentage of other trace gasses (ozone, chlorofluorocarbons, nitrogen oxides, sulfur oxides and methane) will combine to bring about continuing global warming (Coakley 1988). While this is generally agreed to be an accurate projection of future condition, the spatial relationships involved, predicting where this effect will be significant vs. being not so significant, is extremely uncertain.

Solar radiation is the source of energy for most terrestrial processes, and anything that alters the amount of radiation reaching the earth's surface may alter climate. Fluctuations in solar output, volcanic eruptions, and other natural perturbations influence solar input to the earth's energy engine. In addition changes in land use and industry that alter the amount of radiatively active trace gasses, plus other factors, can affect climate (Coakley 1988). The quality of light and the duration of photoperiod have been shown to affect plants in a variety of ways. Yet except to note that greenhouse gasses can affect the quality of light, little is said in the literature about possible future shifts in light quality. Photoperiod is seldom discussed as changing for a given area. Effects of photoperiod only appear to be noted as significant in the context of the migration of plant populations under other influences, which will be discussed later in this paper.

In the early 1960s Hepting (1963) stated that wind, not temperature or rainfall, was the primary climatic factor within the context of changing climate in Great Britain. A more recent publication by Lemmen and Warren (2004), also discussing climate change in Great Britain, suggests that a warmer climate may be more conducive to extreme wind events and that these may in turn have consequences for other forest disturbances. One recent work on microclimatological effects (Yarwood 1959) suggests that wind, considered as it directly or indirectly influences plants, has significant impact on them and pests which affect them. Unfortunately, with the exception of discussions in the context of storm events, wind is little discussed in the literature, and no projection has been found of future wind events in the South.

Soil chemical properties do not appear to be directly affected by climate change, nor do they affect the change except as a complex of secondary effects. However, it is generally recognized that as air temperature increases so does soil temperature. Soil warming in conjunction with drought is noted as a major concern as predisposing to root and rootlet death in- or absent-the-presence-of root rotting fungi. Localized and often short term shifts in the albedo are predicted to occur due to the failure of vegetative cover resulting from soil warming. These predictions are not spatially explicit either as to size or location.

Rate of soil mineralization, acidification, nitrification, and carbon sequestration are all processes which are clearly influenced by climate change but generally these effects are more greatly influenced by (and subsequently influence) the local biota.

A variety of projections have been made for compounded edaphic factors; four are briefly noted below:

increased frequency of extreme weather events (Scherm 2003)

increased frequency and intensity of drought occurring under warmer temperatures (Breshears and others 2005)

more frequent winter waterlogging resulting from increased winter rainfall (Broadmeadow 2005)

increased duration of sunshine resulting from changes in temperature and humidity which in turn lead to reduced summer cloud cover (Broadmeadow 2005)

Physiological and ecological impacts of climate change on host plants (the southern pine forest)

Climate is the single most important factor determining the distribution of major vegetation types and individual species (Malcolm and Pitelka 2000).

Extrapolating the potential effects of climate change from its physical effects to the potential biological/ecological effects which they engender is often problematic. The simple description of the forests' future is that as the climate warms southern forests will migrate northward and upward (as higher elevation sites becoming available) displacing a portion of the temperate mixed hardwood forest. The temperate mixed hardwood forest in its turn migrates, displacing part of the northern boreal forest. While this presents an easy to understand generalization it masks an extremely complex reality.

Forests are not expected to migrate as an integral whole. The migration response to climate change, while driven by a set of individual physical parameters, will respond directly at the species and individual plant level, not at the association, ecosystem or any other higher ecological level of organization. Different species (and different individuals even within a species) will react in potentially very different ways to the various stimuli affected by climate change. The responses of ecosystems can be predicted

only in part by understanding the behavior of their convergent properties. One must consider the unique characteristics and responses of the individual species as well (Malcolm and Pitelka 2000).

On the positive side, increasingly sophisticated computer models have been developed that incorporate more fundamental ecological mechanisms than previously. Their primary limitation is that even these newer models cannot predict with accuracy what happens as the climate is changing (Malcolm and Pitelka 2000).

Despite the concerns expressed above, we have some clear reports of the effects of a changing climate which are already being observed. In response to an average 1° C increase in mean temperature, plant growth is reported to have increased and growing season has lengthened. Budbreak of trembling aspen in Alberta, Canada is reported to be 26 days earlier than 100 years ago, and white spruce budbreak in Ontario is reported as 'earlier' (Lemmen and Warren 2004). Ground based monitoring efforts in Europe documented an increase in growing season length of 11 days over a 34 year period (Malcolm and Pitelka 2000). Because temperature can affect ecosystems in many different ways, and because there are multiple pathways for feedback and interaction, evaluating or predicting the effects of temperature increases is not simple. Not surprisingly, the results to date are mixed (Malcolm and Pitelka 2000).

At the plant level at multiple scales, plant organs may increase in size as a response to elevated levels of CO_2. Increased leaf area, leaf thickness, greater number of leaves, higher leaf area per plant, and stems and branches with greater diameter have all been observed under increased CO_2. Enhanced photosynthesis, increased water use efficiency, and reduced damage from O_3 are also reported as responses to increased CO_2 (Garrett and others 2006).

Host range projections were made using Iverson and others (1999), and the interactive web site derived from this publication. Both the general type map and specific species maps were produced, and formed the basis for our initial projections of potential host ranges for potential forest insect and disease problems in the future. For this exercise we considered the current range of the host or forest type based on FIA data and on a model projection, and compared this to the average of six projections (3 models x a high and a low change scenario for each) generated by the web model.

In addition, the second periodic National Insect and Disease Risk Map (NIDRM) completed in 2006 presents a strategic assessment of the risk of tree mortality due to major insects and diseases. This is the definitive source at the present time for projected insect- and disease-caused mortality for the years 2006-2021 (shorter term than the 50 year window of this report). The NIDRM is compiled using nearly 190 separate models in a GIS-based framework. It assigns risk to individual one kilometer pixels based on forest type, host species basal area, and numerous other traditional factors associated with different host species and damage agents. (At this time climate change is not specifically factored into the models but the map provides an excellent short-term projection of pest activity from which to extrapolate.)

A composite map (fig. B-1) displays a summary of risk from all damage agents. The risk shown is the expectation that 25% or more of the standing live volume of trees 1" and larger in diameter will be lost over the next 15 years. The national composite map can be found at: http://www.fs.fed.us/foresthealth/technology/nidrm.shtml.

Region 8 Forest Health Protection has developed a portal web page that provides access to separate large-scale risk maps for each Southern state, color-coded by the degree of risk in each pixel. These maps, labeled by damaging agent (e.g., gypsy moth, southern pine beetle, etc.), show the risk associated with the most serious individual pest problems for each state. The portal page can be accessed at:

http://www.fs.fed.us/r8/forest health/programs/riskmap/maps/statemaps.shtml. Further information about these risk maps is available at http://www.fs.fed.us/foresthealth/technology/nidrm.shtml.

Plant disease and insect pest activity under the influence of climate change

Note that a variety of statements within the following section are written specifically about pathogens, insects or specific host species, based on the content of the initial work being cited. Most, if not all, apply to the broader spectrum of pest species or host species and should be interpreted in that context.

Yang and Scherm (1997) have shown climate change to be a driving force in the long-term dynamics of plant disease; changes can result in the emergence of new threats from minor diseases to range expansion of diseases into areas where they were not previously a concern (Lonsdale and Gibbs 1994). We are, however, working in an area where the number of reported clear-cut examples of climate change causing a shift in pest/disease patterns is very limited (Goudriaan and Zadocks 1995). While the number of recent attributions of pathosystem shifts resulting from climate change is increasing in the literature, field research is plagued by the very nature of "climate change" which is a longer term process than simple shifts in weather, historically the more commonly studied change (Coakley 1988).

Host-pest interactions will be affected by climate change in similar ways as other plants and animals. In its most simplistic form, species of pest will migrate, generally following the migration of their preferred hosts. While following their hosts into new locations all of the same edaphic and ecological elements affecting their hosts in the new environment will impact the pests themselves. Temperature, available water, quality and duration of light, air quality, soil condition, and other edaphic and biological factors will affect the pest's physiological/ecological responses. In addition the condition and possibly

altered physiology of the host in its new environment are additional factors influencing the new host-pest interaction (Moore and Allard 2008; Tubby and Webber 2010).

Pathogen evolution is another factor which presents complications when predicting the migration of pathogens into new areas. Pathogen evolution rates are determined by the number of generations of pathogen reproduction per time interval, along with the heritability of traits related to fitness under the new climate scenario (Garrett and others 2006).

A few recent publications have focused on the need to consider microclimate factors as being immediately relevant when describing pest-host interactions. This is a little studied area due to the complexity inherent in isolating micro-effects in a macro-scale ecosystem.

Gradual warming would probably lead to a general northerly shift in seasonal climatic regimes which in turn would sometimes adversely and sometimes favorably affect the range of oak (Brasier and Scott 1994). New disease complexes may arise and some diseases may cease to be economically important if warming causes a poleward shift of agroclimatic zones and host plants migrate into new regions. Pathogens would follow the migrating hosts and may infect remnant vegetation of natural plant communities not previously exposed (Coakley and others 1999).

The geographic range of fungal pathogens are to some extent determined by the temperature ranges over which they can grow (Lonsdale and Gibbs 1994). The growth and development of many fungal pathogens within the host may often be favored by climate warming, however, conditions when fungi arrive at the host surface are often critical for the establishment of the pathogen (Brasier and Scott 1994). They further note, the effects of temperature on the development and population dynamics of many potential oak pathogens have been little researched owing to the difficulties involved. Despite this statement he concludes that as warming increases in Europe, *Phytophthora cinnamomi,* a serious root

rot pathogen, will extend its northward range, survive winters better in root systems, show increased spread within the host, have greater infection frequency of new hosts, and cause markedly more rapid host decline and mortality.

This latter point is mirrored by Chakraborty and others (1998) who point out that changes in temperature will alter host plant physiology and thus host resistance to pests. Broadmeadow (2005) adds that increased temperatures will result in higher evapotranspiration. And, Burdon and others (2006) reiterate that when we turn to the impact of the more unpredictable aspects of global climate change on the pathogens themselves, it is likely that over time we will see significant changes in host-pathogen interactions, which are likely in both directions (increase and decreased activity).

Increased soil temperature has been shown to have negative effects on plant roots. Redmond (1955) reported that birch rootlets (55 year old yellow birch stand) with a normal background mortality rate of about 6%, suffered 19% root mortality when average soil temperature increased 1° C, and 60% root mortality if the temperature average increased 2° C. They also reported a change in microbial population and a change in the development of mycorrhizae.

Higher air temperatures commonly enhance the general activity, population size and potential for dispersal of insect pests. Higher temperatures could lead to greater overwintering population size, increased length of flight season, and length of daily flight periods (Brasier and Scott 1994). Climate change, and particularly global warming, will have a dramatic impact on pest insect species. As “cold blooded” organisms, they have a life history that hinges on temperature; the thermal habitat largely sets the boundaries of their geographic distribution (Logan and others 2003).

Extended periods of warm weather can favor the development of insect pests both directly and indirectly. Warm temperatures can accelerate the development of insect populations by reducing the

time needed for life-cycle completion. Indirect effects can be mediated through the host plant or by decoupling relationships with natural enemies (Mamlstrom and Raffa 2000). However, in some cases, warmer temperatures could actually inhibit insect activity or disrupt the build-up of populations. Although warmer winters would increase over-winter survival of some insect pests, reduced snow cover could increase the winter mortality of others (Burdon and others 2006). Enemies of insect pests will also be affected by climate change, but these effects are unknown and require more research. If warmer temperatures positively affect predators and parasitoids, natural enemies will exhibit greater control of pest species. Conversely, if warmer temperatures disrupt or decrease predator/parasitoid populations, pest populations will grow more quickly and will persist at higher levels for longer periods.

Pathogens will follow their migrating hosts, and because of their rapid response to small environmental changes may provide good early warning of impending climate change. The damage threshold from a disease may also change in a new geographical location (Chakraborty and others 1998).

Gilmour (1960) points to two opposite water related conditions which cause significant impact on trees. Drought conditions have been shown to be the cause of various disorders with or without any associated fungus pathogen. And, saturated soil has been found to cause disorders in many plants. Thus, both extremes in water availability have been shown to negatively affect trees. Saturated soils, while being somewhat deficient in oxygen, appear also to have altered chemistry from their drier condition. Garrett and others (2006) state that even without the added impetus of climate change the interaction of precipitation and disease is of primary importance for predicting disease severity.

Broadmeadow (2005) points out that increased winter rainfall will lead to more frequent winter waterlogging [of soil] and in some cases to fine root death extending into the soil surface horizons. This

in turn exacerbates the effects of subsequent summer drought. Black and others (2010) point out that Swiss needle cast disease is associated with spring and summer needle wetness, as well as wintertime temperatures.

Since most plant parasitic fungi are believed to require free water for spore germination, microclimate of leaf surfaces is an important consideration. The important sources of free water for foliage pathogens are rain, fog, condensed water, and guttation water. For foliar pathogens which require free water there is little germination at relative humidities below 95% (Yarwood 1959); further stating that fungi that infect the foliage can be categorized by their requirements for water in the phyllosphere during the infection stage, but centers his further discussion on rust fungi (specifically the urediospore stage) and does not present a broad categorization of this effect.

Lemmen and Warren (2004) emphasize a very important consideration when they state that forest characteristics and age-class structure also affect how forests respond to changes in moisture. They note that mature forests (with well established root systems) are less sensitive to changes in moisture than are younger forests and post-disturbance stands – at least in the short run. They add that different species have different drought tolerance which also must be considered. And, Lonsdale and Gibbs (1994) remind us that climate change with its associated change in frequency of summer droughts would alter the stability of associations between tree species and various members of their endophytic mycofloras – resulting in pathogenesis where, minimally, coexistence, and in some cases mutualism, had been the pre-change norm.

Hanson and others (2001) summarize the varying projected potential effects of water on forests in the following way: the impact of potential changes in drought or precipitation regimes will not only depend on the predicted scenario of change, but on the type of forest ecosystem and the climate

conditions to which it is currently adapted. He concludes his discussion by summarizing Loehle's (1996) conclusions. In that paper six reasons are given supporting the hypothesis that forests under the influence of climate change (including drought) would not exhibit catastrophic dieback. These six reasons support Loehle's argument that forests are somewhat buffered against climate change; will be replaced by faster growing trees; but over extended periods of time and not catastrophically, as has often been predicted.

Generally speaking, any precipitation regime that stresses host trees (whether it is too little or too much moisture) will make them more susceptible to insect attack.

While Yarwood (1959) cites wind as being a serious modifier of water relations, he actually suggests that wind commonly prevents the formation of dew, and causes raindrops or dew to evaporate more rapidly than they would in still air. Broadmeadow (2005) notes that an increase in the number of storms may make woodlands more vulnerable to wind damage.

Fungi preferentially grow when the sky is cloudy and are therefore active mainly on shaded parts of the plant or in non-irradiated angles of the ecosystem. Pathogenic fungi are additionally protected while growing partly or completely within the host's tissue (Manning and von Tiedemann 1995).

The great significance of light especially in the near UV band on fungal sporulation has been recognized since the first studies were performed on this phenomenon in the 1940s. Humphrey (1941) reports that exposure to light stimulated sporulation in 62 of 75 species of fungi tested; most required light for the initiation of sporulation. Sporulation was inhibited in none of the 417 fungal strains tested when exposed to light. However, enhanced UV-B radiation may increase, decrease or leave unaffected the severity of biotic diseases. A serious comparison of this contradictory information is not possible since for the underlying studies the ranges of light qualities, light intensities and light exposures were

too large and too variable as were the experimental designs and time courses applied (Manning and von Tiedemann 1995).

If some parts of the pathogen life cycle are photoperiod sensitive, populations might need to undergo extensive adaptation to make use of extended seasons in temperate areas (Garrett and others 2006). UV-B has positive and negative effects on fungal development; its effect on diseases is mainly through altered physiology and morphology (Chakraborty and others 1998).

As noted above, increased CO_2 in the atmosphere is generally cited as being a primary factor in driving physiological changes in plant populations. Runion (2003) working with a pasture legume and *Coletotricum gloeosporioides* (a fungus) at 2 times ambient CO_2 concentration, reports an increase of virulence of the pathogen against resistant cultivars of the legume (no change with respect to susceptible cultivars) and a significant increase in fecundity which was more pronounced in the aggressive fungal cultivars being tested. Chakraborty and Datta (2003) focuses this discussion by stating that of particular concern is whether this increased pathogen fecundity at elevated CO_2 levels could rapidly erode the usefulness of disease resistance. Altering the predisposition of the host to disease may be the predominant effect of rising CO_2 (Manning and von Tiedemann 1995).

Charkrabotory and others (1998) report an increase of disease severity in response to increased CO_2 for 6 of 10 biotrophic fungi and 9 of 15 necrotrophic fungi based on a survey of literature reports of effects. They conclude that predicting effects for unstudied pathosystems will be quite challenging and that combining the effects of elevated CO_2 on plants with the effects on disease will make projecting effects even more challenging.

Burdon and others (2006) suggest that the effect of CO_2 may be an increase in the efficiency of carbon fixation and a resultant increase in growth and improvement in the carbon status of the plant.

This increase will lead to morphological change generally expressed as enhanced growth, and the combined changes in nutrition and morphology, in their turn, can affect the suitability of the plant as host material for a variety of pathogens. This having been said the authors go on to say that there is limited reported research on the process they described. They end the discussion by moving from plant level to community level with the following: "…the predictability of the impact of these factors as on whole communities is even more uncertain with both indirect and direct effects of varying magnitude being likely."

Mirroring this concern is the statement that while numerous studies have investigated the impacts of elevated CO_2 on forest growth and health, the results are neither clear nor conclusive (Lemmen and Warren 2004).

Manning and Keane (1988) make several statements relating to air pollution and pest behavior including: bacterial diseases are generally inhibited by SO_2 which inhibits lesion size and often increases latent periods; fungal diseases have been reported to be either enhanced, inhibited or not affected at all by air pollutants; little is known about the effects of pollution on root diseases; and, virus affected plants are usually less affected by air pollutants than are virus-free plants. They also cite others for several other bits of information, including: inoculated stumps of O_3 stressed pines were more readily invaded by *Heterobasidion annosum* (James and others 1980a); O_3 stressed white pine in the Blue Ridge Mountains of Virginia were more subject to root disease caused by *Verticicladiella procera* (Skelly and others 1983); loblolly pine seedlings with ectomycorrhiae (*Pisolithus tinctorius*) were not adversely affected by O_3 , SO_2 or a combination of both (Mahoney and others 1985); O_3 caused significant decreases in ectomycorrhizae of white birch and white pine seedlings (Keane and Manning 1987). His final conclusion is that "in a theoretical sense, air pollution can increase, decrease or not affect the course of development of a disease epidemic".

Carbon dioxide concentration in soil is expected to be far less impacting to pathogens than is atmospheric CO_2. Soil microflora is routinely exposed to levels 10 to 20 times higher than atmospheric CO_2 levels (Coakley and others 1999; Manning and von Tiedemann 1995). Colonization of and persistence of mycorrhizae appears to be dependent, in part, on soil nutrient status (primarily N) and CO_2 concentration in soil although there does not appear to be a consistent pattern to observed responses. Not much further can be said here since the influence of mycorrhizae on plant disease is still not well understood.

Ozone does not penetrate the soil surface, affecting roots only indirectly via its effect on photosynthesis. Damage caused by several tree root disease pathogens has been reported to be enhanced when the host plant was stressed by O_3 (James and others 1980b, 1982; Fenn and others 1989; Skelly and others 1983).

O'Neill (1994) presents a detailed review of the potential effects of elevated levels of CO_2 on the rhizosphere. Introducing the subject she observes that ecosystems are largely constrained by the rates at which soil processes occur. Despite a fair volume of literature being reviewed and summarized the ultimate conclusion for all of the different issues addressed is that we need more data to begin the process of generalized modeling of effects on the rhizosphere.

Effects of soil saturation have already been briefly discussed under "Available water" above. Both the amount of water and timing of flooding affect the degree of negative impact on cover plants.

Soil characteristics, nutrient availability, and disturbance regimes may prove to be more important than temperature in controlling future ecosystem dynamics (Lemmen and Warren 2004). Climate and vegetation interact to determine the characteristic soils of a region, and different climatic zones are

characterized by different soil types (except where the presence of unusual rock such as serpentine results in unique soils) (Malcolm and Pitelka 2000).

Little is known about how environmental effects on tree physiology influence the inducible responses that are relevant to pathogens (ie. signal recognition, generation of phytoalexins and reactive oxygen species, hypersensitive responses, callus growth, and systemic acquired resistance) (Ayres and Lombardero 2000).

Carbon dioxide is a primary input to growth and development of all plant life, providing both a fertilization effect and an increase in the efficiency with which plants use water. The fertilization effect may be affected by the availability of water and other nutrients. It may also diminish after an initial period of adjustment by the plant. Increased CO_2 levels may also trigger changes in the chemical composition of vegetation such as affecting the C:N ratio in leaves (Winnett 1998). Positive response to CO_2 appears to occur under a wide range of nutrient availability (Rogers and others 1994). In addition Bazzaz and others (1992) stress the point that differential response of species to elevated levels of CO_2 suggests that there will be shifts in the competitive relationships among plants. Partial closure of the guard cells forming stomates has been proposed as the mechanism by which plants slow transpiration (Jones and Mansfield 1970), which in turn may be one mechanism of adaptive resistance to elevated CO_2 levels.

Ozone effects on plant diseases are host mediated. The principal effects of UV-B on plant diseases would be via alteration in host plants (Manning and von Tiedemann 1995).

Carbon dioxide in soils can be used as an additional C-source by some fungi, being incorporated into organic acids and eventually entering the Krebs cycle as an additional energy supply (Manning and von

Tiedemann 1995). This increase tends to increase root growth more than aboveground growth (Rogers and others 1994).

Host-pathogen relationships, defense against physical stressors, and the capacity to overcome resource shortages could be impacted by rises in CO_2 (Rogers and others 1994).

During winter dormancy, direct effects of climate on the host are generally less important than those involving the pathogen (Lonsdale and Gibbs 1994).

Increased summer temperatures and droughtiness could be expected to help shift the distributions of fungi northwards within the range of potential hosts, or at least to increase the geographic range over which they behave a pathogens (Lonsdale and Gibbs 1994).

Fungi appear to be largely tolerant of ozone levels experienced at present in most parts of the world. However, there is a strong negative correlation between rainfall or relative air humidity and photochemical ozone generation in the atmosphere. On wet days that are appropriate for vegetative growth of fungi on plant surfaces, ozone levels are usually low. Consequently, a coincidence of biologically harmful ozone concentrations in nature and germinating spores or actively growing mycelium seems to be largely excluded (Manning and von Tiedemann 1995).

Expected more vigorous growth due to elevated CO_2 levels in the atmosphere will almost certainly aggravate problems with pathogens. This having been said, it is likely that this effect will be offset by growth reductions caused by O_3 and UV-B (Manning and von Tiedemann 1995). CO_2 may greatly alter ecosystem structure and function (Bazzaz and Fajer 1992). Natural, unmanaged forest ecosystems may be seriously impacted (by CO_2 acting in combination with drought) but alterations in species composition will have little or no effect for intensively managed, monoculture tree farms. Overall the

interaction of CO_2 and temperature is not well understood; there continues to be some conflict in the experimental data (Rogers and others 1994).

Generally, an increase in the availability of all major nutrients (N, P, Ca, Mg, K, S) can be expected as a result of the increased circulation of nutrients in the soil-nutrition system. This will be caused by increasing water fluxes through soil and higher organic matter decomposition rates at increasing temperature. Also, nutrient circulation will increase due to higher growth rates of forest species at increasing atmospheric CO_2 concentration and warmer temperature (Nilson and others 1999).

Stressed trees are more susceptible to insect pests and diseases (Broadmeadow 2005). While firm projections of future pest activity cannot be made, expert judgment of forest pathologists and entomologists allows some assessment to be made. However, considerable caution should be exercised in extrapolating analysis to a future climate. For some insects and diseases, likely trends cannot be predicted even on the basis of expert judgment (Broadmeadow 2005).

Climate change will directly influence infection, reproduction, dispersal, and survival between seasons, and other critical stages in the life cycle of a pathogen (Coakley and Scherm 1996). Observed outcomes include modifications in host resistance, altered stages and rates of pathogen development, and changes in the physiology of host-pathogen interactions (Scherm 2003).

It is important to note that species will respond individually to climate change and that ecosystems will not necessarily shift as cohesive units. The most vulnerable species are expected to be those with narrow temperature tolerances, slow growth characteristics and limiting dispersal mechanisms such as heavy seeds (Lemmen and Warren 2004). How well plant and animal species adapt to or move with changes in their potential habitat is strongly influenced by their dispersal abilities and the disturbances to these environments. Introduced and invasive species that disperse rapidly are likely to find

opportunities in newly forming communities (Joyce and others 2001). However, if climate change causes a gradual shift of cropping regions, pathogens will follow their hosts (Goudriaan and Zadocks 1995) into less changed new communities.

The pattern of disturbance imposed on a landscape by a particular biotic agent is determined both by the landscape's structure and condition and by the nature of the biotic disturbance agent and its environmental responsiveness (Mamlstrom and Raffa 2000). Factors such as changes in land use or increase in resistant strains of pathogens may underlie range expansions (Harvell and others 2002).

Dale and others (2001) point out that many disturbances are cascading. Insect infestations and disease promote forest fire by creating fuels and the fires then promote future infestations and infections by compromising tree defenses. Invasive nonnative species are sometimes able to modify existing disturbances or introduce entirely new ones. Under climate change these compounded interactions may be unprecedented and unpredictable. They are likely to appear slowly and be difficult to detect because of tree longevity.

Climate change is a form of disturbance to natural ecosystems and thus could provide new opportunities for invasive species favored by disturbance to flourish and displace native species. An important feature of many invasive species is that they are effective at dispersing and have high reproductive rates (Malcolm and Pitelka 2000). Changes in phenological synchronicity of hosts and pests, as well as their relative abundance and physiological condition, may affect the frequency and consequences of outbreaks by indigenous species (Malcolm and others 2006).

As climate shifts, climatically-sensitive species will eventually die out, and only a subset of the potential pool of incoming plants may actually migrate sufficiently quickly to keep up with the shifting climate. Thus, plant communities could become progressively composed of the more climatically-

tolerant and fast-moving species, especially if warming is rapid. This change in plant communities, especially tree communities, is of considerable concern. The warm temperature mixed-evergreen forest type of the southeastern United States expands at the expense of other forest types. However, in some scenarios, parts of the Southeast become drier and grasslands or savannahs replace the current forest (Malcolm and Pitelka 2000).

The forest area impacted by insects and pathogens in the US is approximately 45 times that of fire, with an economic impact that is almost 5 times as great (Dale and others 2001). We believe that pests and diseases are likely to be the primary cause of species change in eastern forests in the next few decades. Forecasting the trajectory of those changes is nearly impossible… we cannot predict with any certainty which pests or pathogens will be established (Lovett and others 2006). Given the complexities of climate change, and biotic responses to it, prediction of the future impact of climate change on emerging infectious diseases of plants is difficult except on a broad scale. Climate change can lead to the emergence of pre-existing pathogens as major disease agents or can provide the climatic conditions required for introduced pathogens to emerge (Anderson and others 2004). Because climate change will allow plants and pathogens to survive outside their historic ranges, Harvell and others (2002) have projected an increase in the number of invasive pathogens.

Describing similar effects for pest insect scenarios Logan and others (2003) state that to date, the majority of results assessing individual pest species' response to climate change indicate intensification in all aspects of outbreak behavior, and this certainly characterizes our [modeling] work with the mountain pine beetle, gypsy moth, spruce beetle, and spruce budworm

Walther and others (2002) link changes in a variety of known phenological expressions occurring during the spring in a variety of organisms in Europe (including bird breeding, migrant bird arrival, the

appearance of butterflies, earlier choruses and spawning of amphibians, and earlier shoot growth and flowering of plants) to climate change, specifically suggesting a lengthening of growing season of 12 +/- 4 days as causal of these observations. Anderson and others (2004), citing grey leaf blight of corn, state that the ranges of several important crop insects, weeds and plant diseases have already expanded northward. They also note that fall phonological expressions (leaf color change and leaf fall) are not as clearcut in their response to the extension of growing season.

Plants have historically responded to climate change by migration and adaptation. Fragmentation and rate of seedling establishment may hinder some plant populations from successful migration to higher latitudes. Persistence of these populations may depend heavily on adaptive evolution. Unfortunately, predicted rates of evolutionary response are much slower than the predicted rate of climate change. Historical climate changes were generally much slower (by one or more orders of magnitude) than those predicted for the future (Etterson and Shaw 2001). This observation leads to concern that historical response patterns to climate change may not prove to be effective as predictors of future change.

And, a critical consideration not yet emphasized enough - climate change cannot be viewed in isolation, the effects of climate change on ecosystems must be considered in the context of a range of human-caused impacts on ecosystems, such as air pollution, water pollution, habitat destruction and fragmentation, and invasive species. Invasive species thrive and have their most serious effects in ecosystems already disturbed by human activities (Malcolm and Pitelka 2000).

A large number of unpredictable, unforeseen pest problems may arise as a result of changing temperature and/or precipitation regimes. Previously minor and/or infrequent pests may become significant causes of tree mortality. Some major pests may decline in importance. In addition to new

non-native invasive pests arriving from overseas, the ranges of insects and diseases native to North America may expand or contract dramatically. Because the above changes are largely unpredictable yet bound to occur, it will be important for land managers and scientists in forestry-related disciplines to practice early detection and monitoring of "new" problems and to follow up these observations with research and creative, adaptive management strategies.

The following discussion and analysis is excerpted with only very minor changes from Régnière and Bentz (2008) and provides an example for consideration of a pest present and destructive in the western United States which and its potential impact in the East and South under the influence of climate change.

> The mountain pine beetle (MPB) (*Dendroctonus ponderosae*) is a native insect pest of pine forests in western North America. While it has a broad geographical distribution, it has been historically confined to the western side of the continent; in the U.S. by the distribution of its pine hosts and in the northern half of British Columbia, Canada by the geoclimatic barrier of the Rocky Mountains. Since the early to mid-1990s, an outbreak of MPB has reached unprecedented levels in terms of acreages and numbers of pine trees attacked. Lodgepole pine is being killed throughout its range, most notably in Colorado and British Columbia. The MPB is also causing very high mortality among whitebark and limber pines at high elevations. Historical records from the past 100 years suggest these ecosystems have had pulses of MPB-caused mortality but not at levels currently being observed. Since 2006, MPB has extended its range into the Peace River area of north-central Alberta.

Climate change may well be involved in this pests' recent northeastward and upward range expansion. There is ample and mounting evidence of similar latitudinal and altitudinal shifts in insect distributions throughout the world, many convincingly linked to climate change. The main concern at this time is the likelihood that this insect will continue spreading east into the pines of Canada's boreal forest, eventually reaching the eastern provinces and threatening the pines growing on the Atlantic side of the continent, and then spread all the way into the Southern U.S. Because of its recent incursion at the gates of the Canadian boreal forest, MPB is being viewed as a potential invading species in eastern pine ecosystems.

There are three well-understood links between climate and MPB that form the basis for our belief that changing climate (temperature and precipitation) has had (and will have) a role to play in the recent outbreaks and range expansion of this insect. A well-synchronized adult emergence pattern is a prerequisite for successful mass attack of healthy pine trees by MPB. Such highly synchronized emergence is most likely to occur where (and when) the insect has a strictly univoltine (one generation per year) life cycle. For more than 20 years, process-based models describing MPB responses to temperature have been under development and, in addition to the synchrony issue, a hemivoltine life cycle (one generation every 2 years) leads to lower population performance mainly because it implies that the MPB is exposed to two winters. We know that cold winter temperature is the major cause of mortality in MPB ecology. A third weather factor is drought which affects the ability of pine trees to defend themselves against MPB attack. We thus have three model components available to study the impact of weather on MPB populations: a phenology model which predicts life

stage-specific developmental timing; a cold tolerance model which predicts probability of MPB larval mortality due to cold temperature; and, a drought-stress model which predicts fluctuations of tree susceptibility. All three models have been implemented within BioSim to make landscape-scale predictions of MPB performance under climate change scenarios.

The phenology model is very good at predicting the portions of the continent where the insect has a high likelihood of being univoltine. This model predicts the northward and upward shift of MPB. Under a conservative climate change scenario, it also predicts that by the end of the 21st century, the area at risk of univoltine MPB will shift considerably north, to a point that the insect may be misadapted over much of its current distributional range.

The cold tolerance model suggests that winter survival is very low and will remain so in the foreseeable future throughout the boreal pine forests of the Canadian central provinces from Alberta to Ontario. While drought stress is, and is predicted to be, more common in that same area, there is not a very large change in this risk factor predicted in the near future. Thus, with our current understanding of the insect's physiology and host plant interactions, the risk of seeing the MPB roll across the northern forests of Canada into the eastern pine forests seems rather low. This, of course, is contingent upon the insect not adapting (evolving) and changing its thermal responses, and upon the distribution of pines not changing appreciably over the time range under consideration.

Pathosystems

The subtle changes in climate attributed to climate change can affect plant-disease development. These changes are not easily determined, and, consequently, the ability to forecast how disease changes under altered growth conditions is not simple (Seem 2004).

Less stable relationships tend to occur in the simpler ecosystems that initially exist in man-made plantations, often involving new combinations of host and pathogen species that have been artificially transported beyond their natural geographic ranges. In such cases it can be envisaged that climate change would encourage major changes in disease incidence and severity (Lonsdale and Gibbs 1994).

Tree-disease problems cannot be fully understood without a thorough appreciation of the part played by environmental factors, particularly climate, as a precursor to fungal attack. The manifestation of many diseases often merely indicates unfavorable site factors, the presence of the pathogenic fungus being the result of an unhealthy condition rather than the primary cause of the tree's debility (Gilmour 1960).

During unusual climatic events or biologically induced stress periods the competitive dominant may be the most vulnerable. Its large size has stretched the limits of coordination of uptake, transport, storage, and photosynthetic systems. (Manion and Lachance 1992)

The timing of the stress event is also very important. Early season stress is frequently overcome while mid- to late-season stressors are not so, due often simply to a lack of sufficient time remaining in the growing season (Lundquist and Hamelin 2005).

Exotic insect pests and pathogens pose the most serious current threat to the forests of eastern North America. The litany of pest and pathogen introductions is long; chestnut blight,… Dutch elm

disease,… gypsy moth,… beech bark disease,… balsam woolly adelgid,… the hemlock woolly adelgid,… dogwood anthracnose (Lovett and others 2006).

According to Lovett and others (2006) ecologists need adequate information in only six categories in order to make rough predictions of the type and magnitude of potential ecosystem impacts of exotic pests; it includes knowledge of the pest and the host. Pest information is needed concerning: 1) the pest's mode of action, 2) its host specificity (host species, host age class, etc.), and 3) its virulence. With respect to the host they suggest that we need to know: 4) its ecological importance (position or bio-production values in the system), 5) its uniqueness, and 6) its phytosociology (pure vs. mixed stands, effectiveness of regeneration, etc.)

When climatic change has a significant and direct effect on plants, changes in composition may occur, which, given differential responses across plant species, may lead to relative changes in community composition, or when coupled with range extensions or contractions of individual species will lead to increased or decreased diversity of whole plant communities. Pathogens of one host species may thus be brought into more intimate contact with previously unencountered, naïve hosts (although the likelihood of spatial movements necessary for this to occur is perhaps low in the immediate future); may benefit from increasing overlap of obligate alternate host distributions; or may suffer significant reductions in population size as a consequence of the generation of allopatric distributions or incomplete congruence in the distribution of obligate alternate hosts (Burdon and others 2006).

Boland and others (2004) state that the literature provides some background on the potential impact of climate change on plant diseases. Unfortunately they continue: "…much of this literature focuses on diseases of agricultural crops." Despite this they tabulate data for climate change effects with respect to forest pathosystems under the headings: primary inoculum or disease establishment; rate of

disease progress; potential duration of epidemic; reasons for effects; and, net effect of the disease. A quick review of this article shows that of the 143 plant diseases that are listed, only 18 are forest tree diseases. They go on to say that predicting the effects on the pathogens is relatively intuitive to plant pathologists, but extending that knowledge to the effect on pathosystems or disease per se requires greater knowledge of how these factors affect the host's physiology and thus the host-pathogen interaction. And then they cite a specific need for further knowledge of the effects of elevated CO_2, UV radiation, and ground level O_3, as well as for the effects of environmental changes on insect vectors of pathogens.

An interesting sidebar to pathosystems activity is reflected in the capacity of fungi to perform their cleanup function (woody and leaf litter decomposition) under the influence of climate change. Yin (1999) states that the decay rate of forest woody debris is a key missing link in our quantitative understanding of carbon dynamics and the global carbon budget in the forest. He goes on to say that in the context of global climate change, a 2° C warming in air temperature in January and July would accelerate stem woody debris decay (in density loss); accelerated decay would decrease in the presence of increased precipitation (and vice-versa); but, the magnitude of increase would be smaller when adjusted for the detrimental effect of elevated CO_2 as part of climate change.

For many fungal diseases involving vectors, the effects of climate change and weather on the development of outbreaks or epidemics have not been studied in detail. In regions where a pathogen already occurs, weather conditions may favor outbreaks of its vectors in certain years, suggesting that climate change could influence long term prevalence of the disease (Lonsdale and Gibbs 1994). The introduction of new vector species and changes in vector overwintering and oversummering (several cited in Garrett and others 2006) and other effects of change on insects may have important effects on pathogen survival, movement and reproduction (Garrett and others 2006) .Pathogens that rely on biotic

vectors for dispersal may see significant shifts in their distribution and or intensity if environmental changes affect the behavior and or viability of their vector (Burdon and others 2006).

However, in some cases, warmer temperatures could actually inhibit insect activity or disrupt the build-up of populations. Enemies of insect pests will also be affected by climate change, but these effects are unknown and require more research. If warmer temperatures positively affect predators and parasitoids, natural enemies will exhibit greater control of pest species. Conversely, if warmer temperatures disrupt or decrease predator/parasitoid populations, pest populations will grow more quickly and will persist at higher levels for longer periods.

Futuring

In Spittlehouse and Stewart (2003) we are given a four step framework for mitigating the effects of climate change and adapting to it. The first step is defining the issue. The second involves the assessment of the vulnerability to change (sensitivity, adaptive capacity) of the forest, forest communities, and society. This allows the development of adaptive actions to be taken now (step 3) and those required for the future (step 4) as change occurs. Steps 3 and 4 must be flexible enough to incorporate new knowledge of future climate and forest vulnerability.

Planned adaptation will reduce vulnerability for commercial tree species at selected sites. However, many forest species will have to adapt autonomously and society will have to adjust to the result (Winnett 1998). Pathogen [and pest insect] distribution changes caused by climate change are likely closely tied to shifts in host distribution (Sturrock 2007).

Some ecosystems will be new: new communities of tree and plant species with different suites of insects and pathogens. If forests do remain on a particular site, similar functional types of insects and pathogens will likely remain, although they may be different species (Beukema and others 2007).

Pathogens expanding their ranges and contacting 'new' hosts and vectors may mean that new pathosystems will emerge. Interactions between pathogens may change (Sturrock 2007).

Climate change may amplify the impact and aggressiveness of pathogens; it may also change the status of weak/opportunistic pathogens such that they are able to infect and damage stressed tree hosts (Sturrock 2007).

Climate change may alter the balance between pathogens and their natural enemies (Sturrock 2007).

Almost every study and review of climate change effects on forests has a common caveat – that the "complexity of forest ecosystem relations and the paucity of predictive scientific data means that it is difficult to predict the effects of climate change on many forest factors, including host-pest interactions" (Sturrock 2007).

The difficulty of predicting the future of plant disease is high-lighted by a recent paper which details a scenario not specifically applicable to the South, but one which is indicative of a real pitfall embedded in predicting future pest trends. An endemic needlecast fungus which historically had not caused significant problems in British Columbia is reported to have become a virulent pathogen under the influence of climate change. Woods and others (2005) report that Dothistroma needle blight previously has had only minimal impact on native forest trees in British Columbia. However, recently, in apparent response to a local increase in summer precipitation, this disease is causing extensive mortality of lodgepole pines. While admitting that establishing causality of the increased virulence of this endemic pathogen is fraught with risk of misinterpretation of the evidence, they indicate the link to precipitation (while dismissing warmer temperatures) appears to be far greater than "circumstantial". No prior indication of this shift to virulence appears in the literature – the event was unpredicted, and possibly

unpredictable. Sturrock (2007) notes that wetter springs in some regions may result in increased foliage diseases without venturing to predict possible host/pest scenarios.

P. cinnamomi is predicted to increase its activity in temperate zones in the Northern and Southern hemispheres (i.e., to have considerable northward expansion and a decrease in the tropics). Under changing climatic conditions the fungus is expected to cause more damage to existing urban and forest tree hosts and to expand the number of species it can infect. Also predicted is a similar trend for many invasive tree-infecting *Phytophthora* spp. (eg. alder *Phytophthora* and *P. ramorum*) (Brasier and Scott 1994). Phytophthora root diseases are likely to become more prevalent and damaging, especially those which have higher growth optima (28-30° C) such as *P. cinnamomi* (Broadmeadow 2005).

Increased drought stress of hosts in many regions may mean increased mortality from root pathogens. Highly pathogenic *Armillaria* spp. may be assisted by the impairment of host tolerance caused by climate change -induced stress: this may result in less pathogenic species also becoming more successful on stressed trees (Sturrock 2007). Incidence of oak and beech decline, highly complex disorders, is likely to increase because of the predicted frequency and severity of summer drought stress (Broadmeadow 2005).

In a changing climate with increased temperatures, evapotranspiration, and extreme weather events, there will be an increase in the frequency and severity of stress factors, which may lead to more frequent forest declines (Sturrock 2007).

Pathogen evolution could be accelerated by climate changes due to increased mutation (e.g. from enhanced UV-B) or reproduction rates (shorter life cycles under higher temperatures) of the pathogen relative to the host. This could lead to host resistance being overcome more rapidly (Coakley and Scherm 1996).

Based on the above discussion some basic patterns emerge on which our projections of pest impact in the future will be based. The obvious caveat to anything said in this context is that every single paper encountered which projected pest activity into the future emphasized, often in very blunt language, that any projection of this type whether incorporating modeling or not is subject to huge uncertainty given the large number of currently unquantified variables acting in and on pest/host systems.

The current emphasis on longleaf pine restoration, coupled with increasing temperature and decreasing rainfall will result in a measurable shift in the population distribution of southern yellow pine types, both spatially and numerically.

Boreal forest species such as hemlock, Frasier fir, eastern hemlock, American, beech, and others will have their ranges reduced in the South due to the combined effects of increased temperature and decreased available water.

Pests associated with southern host species are expected to migrate with their hosts with few exceptions. However, note that many of the pests being discussed currently occur throughout the South and already extend into the northern part of the country. With respect to predictions, these pests will thus not appear to migrate in the South since they already extend throughout the region.

While long term projections are that an increasing coastal savannah will replace forest types in many coastal and coastal plains locations, the progress of this change within the next fifty years is not expected to be severely impacting.

Most root rotting diseases are expected to respond aggressively to the combination of warmer soil temperature and lower available water due to reduced precipitation. This combination of heat and drought is expected to result in an increase in dieback and decline among many tree species, often providing further stress which could act

as a precursor to successful invasion/colonization by root rotting fungi. Newly stressed trees also may become the focus of insect attack.

Trees suffering long-term stress may prove to be more resistant to secondary pest attack due to the fact that they have lowered physiological activity and available resources needed by pest organisms.

Tree diseases which affect primarily stem and branch tissue are subject directly to the potential effects of warmer temperatures and a drier environment. At first warmer temperatures and increased CO_2 in the atmosphere are expected to have a stimulatory effect on both host and pathogen. However, the anticipated lower availability of water will generally function more against the host plant than the fungi infecting it, favoring an increase in disease. This assumes that the temperature increase does not exceed the thermal death point of the fungus or its spores.

Fungi attacking the foliage are subject to significant pressure from light and the microclimate in the phylloshpere. While significant loss of spore viability is common on the upper surface of leaves, any change in the amount of UV-B radiation impacting the leaf surfaces (generally this means the upper leaf surface) will alter the survival rate of these reproductive propagules; more UV-B results in lower spore survival and less successful infection, and vice versa. Overall, it is believed that the microclimate of the atmosphere of the underside of leaves is critical to the success of foliar pathogens. Lower atmospheric moisture in the phyllosphere's microclimate as a result of lower rainfall, fog, dew, and as a secondary effect, guttation, is expected to reduce the effectiveness of colonization by leaf-infecting fungi.

Increasing length of warmer summertime temperatures is expected to result in additional pathogen and insect activity. Insect populations may show simple

increases in number due to the availability of additional host material on which to browse, or may actually have sufficient added warm weather to allow propagation of an additional generation per year.

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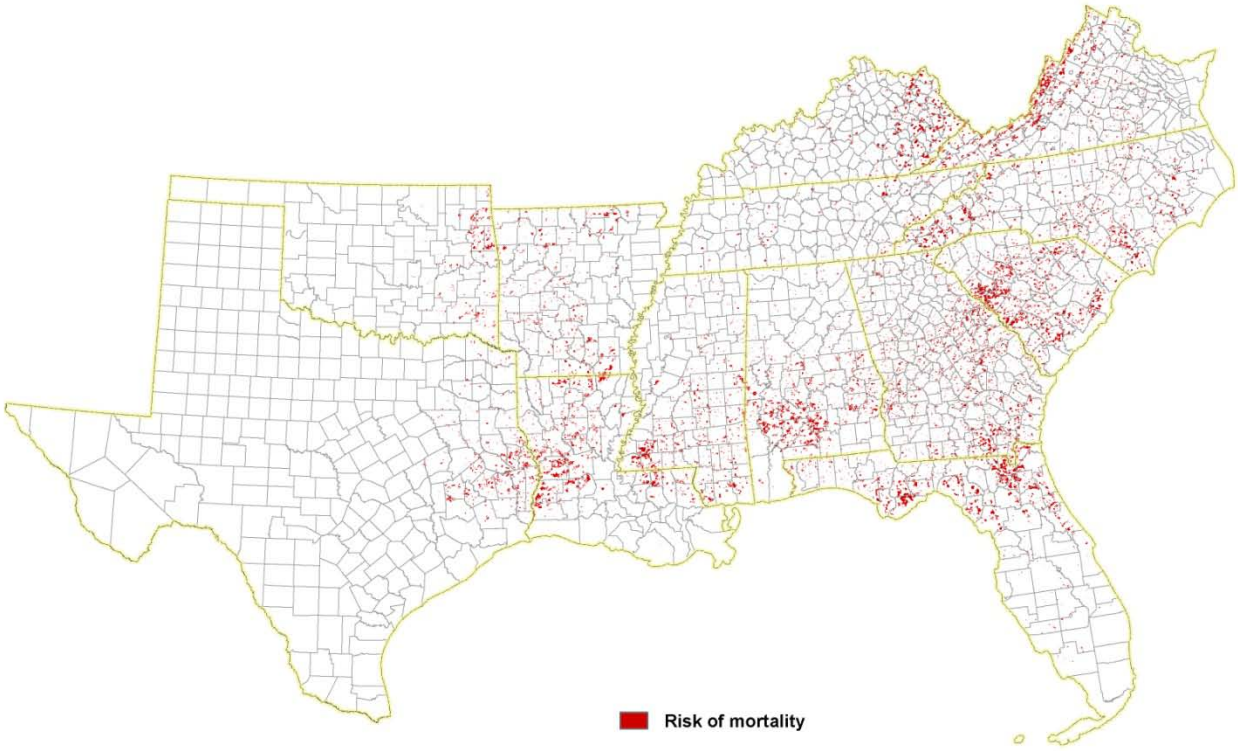
Figure D-1: Composite 2006 regional insect and disease risk map. Adapted from the National insect and disease risk map: National 2006. The composite national map is available at: http://www.fs.fed.us/foresthealth/technology/images/RiskMap_agents_hillshade_ma.jpg.