
6 Climate-Induced Changes in Vulnerability to Biological Threats in the Southern United States

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The South is characterized by hot summers, mild winter temperatures, and high humidity compared to the other regions of the continental United States. During the past three decades, the annual average temperature has risen in the South, particularly during the winter months (Karl et al. 2009), and the occurrence of temperatures below freezing has declined by 4–7 days per year in most of the region. Some areas have experienced >20 fewer days below freezing, and the average autumn precipitation in other areas has increased approximately 30% since 1901 (Karl et al. 2009). Although many areas have reported increases in heavy rainfall events (Karl and Knight 1998; Keim 1997), others have experienced moderate-to-severe increases in drought over the last three decades.

Globally, growing atmospheric greenhouse gas concentrations emitted from human activities could cause annual temperatures to increase between 1.8°C and 4.0°C by the end of the twenty-first century (Intergovernmental Panel on Climate Change 2007). Projected future climatic changes are expected to affect the condition, composition, distribution, and productivity of southern forests by the end of this century (www.srs.fs.usda.gov/futures/date accessed: August 13, 2012). At high latitudes and elevations in North America, the rise in temperatures could exceed global averages (Intergovernmental Panel on Climate Change 2007). Changes in environmental factors including temperature, precipitation, and associated factors could affect the occurrence and impacts of forest diseases, native and nonnative pest insects, and nonnative plant species in several ways, some of which are difficult to predict. Understanding the potential impacts of climate change on these biological threats is therefore of great importance to the future of southern forests. However, many uncertainties and gaps still exist in our current knowledge about interactions among these organisms and southern forests.

FOREST DISEASE AND CLIMATE CHANGE

Relatively few studies address the direct effects of climate change on forest diseases. As drought and other weather-driven abiotic stressors increase or are amplified in a changing climate, some scientists have predicted increases in the frequency and severity of forest diseases (e.g., Sturrock et al. 2011). Nevertheless, the examples and scenarios they provided have a significant degree of uncertainty. Although other scientists have reached similar conclusions for organisms such as the fungus (*Heterobasidion irregulare*) that causes annosus root disease (Kliejunas et al. 2009; Orosina and Garbelotto 2010), these studies are also couched in terms of significant uncertainty and scant empirical evidence.

One major limitation in such prediction is the biotrophic (host dependent) or at least hemibiotrophic nature (ability to continue survival on dead tissue) of the causal organisms behind most forest diseases. As such, host distributions would be the starting point for discussion of how climate change could influence forest diseases. Although well-reasoned and comprehensive predictions of host distributions under varying climate regimes exist, the complex and co-evolved host–pathogen interactions must also be considered, as these interactions add a considerable degree of uncertainty to forecasting efforts. Fungal pathogens are highly adaptable and their reproductive systems are geared for coping with highly variable climatic regimes, host genetics, and host physiology. A case in point is the annosus root disease pathogen that causes significant mortality of southern conifers. Gonthier et al. (2007) discovered that this species is causing extensive mortality of Corsican pine (*Pinus nigra* var. *maritime*) in the central western coast of Italy, where the climate is warm but drier than in the Southern United States; however, the fungus is apparently spreading and causing higher mortality as it invades newly forested areas, possibly supplanting or hybridizing the native variety. Evidence from genetic analyses of its origins and spread over geological time scales also suggests a high degree of adaptability and mobility. In fact, changing climates may have played a key role in the evolution and speciation of this fungus, affecting host specialization (Linzer et al. 2008; Orosina et al. 1993). The question remains, therefore, as to the degree to which this pathogen will respond to changes over the relatively short time scale of 50 years.

Climate can affect host genetics. Mitton and Duran (2004) found genetic variation in populations of two needle pinyon pine (*Pinus edulis*) that they associated with summer rainfall patterns—the selection for certain alleles correlated with sites having more rainfall during summer. Similar mechanisms may exist in host species from the Southern United States where a warmer, drier climate could cause the selection of certain genotypes in Southern pine hosts. Any such changes in hosts could, of course, affect interactions with pathogens.

Many soil processes that directly affect forest tree productivity are impacted by drought. Long-term drought events can result in increased host stress and lead to an increase in the severity and mortality caused by root disease. Conversely, tree mortality associated with drought could reduce stand density, thus reducing the opportunity for root contact, contagion, and demand for nitrogen. Nitrogen losses during a return to normal precipitation could affect soil nutrient status and further complicate potential host–pathogen interactions for a time (McNulty and Boggs 2010).

Increased air temperature can decrease the incidence of stem rust pathogens, which have exacting requirements for infection and spore survival on plant surfaces. Another complicating factor, especially with fusiform rust (*Cronartium quercuum* f. sp. *fusiforme*), is the requirement of an alternate host, principally water oak (*Quercus nigra*). The differing requirements for infection of oaks would also complicate any model that attempts to superimpose temperature and moisture regime layers on incidence and severity of this disease.

Increasing atmospheric carbon dioxide can also affect forest diseases by influencing the physiology of their hosts. For root disease, increasing atmospheric carbon dioxide can lead to increased root growth, and by extension, increased probability of infection by pathogens such as annosus root disease (Kliejunas et al. 2009). Although no empirical evidence supports this notion, elevated carbon dioxide concentrations could increase the amount of susceptible pine tissue available for infection by fusiform rust or by pitch canker (*Fusarium circinatum*), increasing the likelihood that disease incidence will increase. For example, increased atmospheric carbon dioxide concentrations may increase growth or amount of juvenile tissue available to infect. Juvenile tissue favors these pathogens. However, some experimental data suggest the opposite; that is, that these pathogens are seemingly unaffected by elevated carbon dioxide (Runion et al. 2010) and that changes in host defensive compounds could result in no increase in disease incidence.

These are only examples of the results of climate change—forest pathogen interactions are only a partial illustration of possible ramifications and complexities. In general, data are extremely rare or nonexistent for most forest diseases.

FOREST INSECTS AND CLIMATE CHANGE

Changes in climate have already contributed to various insect outbreaks over the past decades (Karl et al. 2009) and more outbreaks are expected in the future (Bale et al. 2002; Klapwijk et al. 2012) partly because of the direct link between insect population growth and seasonal temperature (Danks 1987). A warming climate would likely hasten proliferation of potentially devastating insect species such as bark beetles, defoliators, sap-suckers, borers, and weevils. Any future increase in outbreaks and severity of insect pests could increase host-tree mortality, heighten forest disturbance, and reduce tree vitality (Pimentel et al. 2004; Stadler et al. 2005; Volney and Fleming 2000). Recent years have seen increases in outbreaks of both native species, such as southern pine beetles (*Dendroctonus frontalis*) and pine root collar weevils (*Hylobius radicis*), and nonnative species, such as emerald ash borers (*Agrilus planipennis*), Sirex woodwasps (*Sirex noctilio*), gypsy moths (*Lymantria dispar*), hemlock woolly adelgids (*Adelges tsugae*), and redbay ambrosia beetles (*Xyleborus glabratus*).

Temperature plays a decisive role in the survival, growth, distribution, and diversity of many organisms. Increasing temperatures could expand the northern ranges of many nonnative insect species in the United States. Insect developmental thresholds, the rate of multiple life stages (generations), adult longevity, prolonged adult emergence, and insect flight are all affected directly by

temperature (Bentz et al. 2010). However, the spread of an individual species is also influenced by its adaptive capability and biological properties, by the diversity of potential host species, and by the prevailing ecosystem disturbance (Mack et al. 2000). Shifts in temperature and precipitation have already shown some correlations with outbreak of bark beetles, which are highly sensitive to changes in temperature and water stress (Berg et al. 2006; Powell and Logan 2005).

From 1960 to 2004, minimum temperatures have increased by approximately 3.3°C in the South. Over the same period, outbreaks of southern pine beetles have extended northward, matching physiological model predictions of cold tolerance for the species (Trân et al. 2007). Similarly, warming trends have been associated with shifts in the duration of generations for spruce beetles (*Dendroctonus rufipennis*) in the Western United States (Hansen et al. 2001; Werner et al. 2006) and of mountain pine beetles (*Dendroctonus ponderosae*) in high-elevation forests (Bentz and Schen-Langenheim 2007). The expansion in tree host range could lead to the dispersal of many insect species and expansion of communities that are associated with them (symbionts). Climate change may promote rapid adaptation of nonnative species to seasonal temperature changes (Balanyá et al. 2006; Bradshaw and Holzapfel 2006) and expansion of their ranges into the new niches created by increased temperatures (Battistia et al. 2006; Nealis and Peter 2009).

Warmer winter seasons could increase survival rates of insects throughout the winter, allow more generations per season, and possibly expand their ranges northward (Karl et al. 2009). Insects generally grow under medium thermal conditions ranging from 5°C to 37°C, with optimal growth occurring at 20–30°C and thermal death usually occurring at 46°C (Dwinell 2001). Bentz et al. (2010) identified direct correlations between increased minimum temperatures and reduced cold-induced beetle mortality, while Ungerer et al. (1999) described the development rate of southern pine beetles as a function of temperature, where air temperatures of –16°C result in almost 100% mortality. Hence, the range of the southern pine beetle is more likely dependent on winter temperatures. Therefore, warmer winters would likely benefit the southern pine beetle to expand its range northward.

In addition to changing developmental rates, the metabolic rates of insect species almost double with a 10°C increase in temperature (Clarke and Fraser 2004; Gillooly et al. 2001), a clear indication that insect physiology is highly sensitive to changes in temperature. Therefore, a warmer climate would likely accelerate insect feeding, development, and movements with possible effects on the rate of fecundity, survival, generation time, and dispersal (Bale et al. 2002). In addition to increasing insect metabolism during the growing season, warmer temperatures can also reduce insect mortality from extreme cold in winter (Ayres and Lombardero 2000; Bale et al. 2002), resulting in thriving insect populations through the spring and into summer.

Climate change could also indirectly affect insect populations through impacts on natural enemies (Burnett 1949), important insect symbionts (Lombardero et al. 2003; Six and Bentz 2007), host physiology, and host range distributions (Bentz et al. 2010). Bark beetles and redbay ambrosia beetles typically rely on a complex relationship with fungal symbionts to survive and complete their life cycles. However, these symbionts could have different thermal ranges for optimal growth (Rice et al. 2008). Therefore, a significant shift in thermal range could influence the fungal symbionts that are vectored by dispersing beetles (Six and Bentz 2007), and the physiological responses of hosts to environmental changes could also lead to shifts in the insects that feed on them (Seybold et al. 1995). Shugart (2003) noted that an approximately 2°C increase in temperature has led to significant changes in the composition and formation of new forest communities, most of which no longer exist today, over several thousand years.

As warming trends progress, insect distributions are extending northward (Parmesan 2006; Ungerer et al. 1999). However, some species could be limited to a specific area because of unfavorable climatic conditions rather than unavailability of tree hosts (Salinas-Moreno et al. 2004). Future warmer winter temperatures could remove existing range barriers, creating conditions that

are conducive for nonnative insects to spread into places where hosts are currently abundant. The resulting influx could result in competition with native species.

Although the complexity of insect species' responses to climate change results in uncertainties in predicting the extent and nature of impacts, predicted changes in temperature from various general circulation models serve as an important basis for estimating biological responses to changing conditions (Millar et al. 2007). The general circulation models and quantitative models based on documented biological parameters of individual species should provide some insight into how southern forest ecosystems would likely respond to climate change. Warming would only translate to insect population growth or range expansions if the insect species can adapt to changing forest ecosystems (Bentz et al. 2010; Bradshaw and Holzapfel 2006). Overall, the potential threat of future insect attacks (native and nonnative species), will depend on how well they response to shifts in environmental factors such as temperature or precipitation ranges, and on their interactions with host plant species and other associated organisms within the forest ecosystem.

NONNATIVE PLANT SPECIES AND CLIMATE CHANGE

In general, nonnative plant species will likely face similar effects from climate change as their native counterparts (Parmesan 2006), perhaps with varying level of impacts. Global climate change, along with other natural and human-caused factors, could lead to a major conservation threat, which could change species geographic range, community composition, and structure and function of ecosystems (Mack et al. 2000). Similar to native plants, many introduced species, especially those that are highly invasive, could shift their distributional ranges accordingly, albeit with a time lag in response. For example, under warming conditions, including those seen in the last few decades, invasives of all kinds are moving higher in latitude and elevation (Guo 2010), threatening those native species that have small populations and small distribution ranges (e.g., the species on mountaintops). In a recent study of 825 birds, mammals, and plants introduced between continents and from eastern Asia to North America, Guo et al. (2012b) found strong correlations between the latitudinal distributions in their native lands and the lands they invade; nevertheless, relatively more species occur at latitudes higher than those in their native lands, and fewer occur at latitudes lower than those in their native lands.

In addition, climate change will likely influence the establishment of new species and the effectiveness of control strategies. Nonnative plants that are fast growing and responsive to resources would be favored by environmental changes that increase resource availability. Many endangered species already have limited habitat on mountaintops and fragmented landscapes and no viable escape routes. A warming climate, upward movement of nonnatives, and increasing urbanization and fragmentation would further jeopardize their further existence.

Current forest management practices have given scant consideration to the possible consequences associated with climate change, even though most—if not all—simulations and models have projected significant shifts in species ranges after introduction of nonnative plants. The resulting alterations in ecosystems would greatly affect how we manage our forests and the future management policy (Guo et al. 2012b).

This chapter describes the interrelationships among insects, diseases, and nonnative plants, and assesses the vulnerability of the southern forest ecosystem within the context of future climate conditions. Specifically, we discuss the impacts of future climate change on selected insects, diseases, and plants species; along with projections and distributions of host species or the local conditions that affect host species at the subregional level and critical areas within subregions. Our goals are to benefit stakeholders including land managers and private landowners and to assist the Forest Service in future strategic planning of resource allocations to mitigate the effects of climate change on southern forests.

METHODS

FUTURE CLIMATE

One climate projection for 2080 (Karl et al. 2009) indicates that average temperatures in the South will rise by 2.5°C under lower emissions projections to 5.0°C under higher emissions projections, with an increase of approximately 5.8°C in summer. The states that border the Gulf of Mexico are expected to have less rainfall in winter and spring, compared to states farther north. The frequency, duration, and intensity of droughts will most likely increase with higher temperatures, more evaporation from soils, and more water loss from plants. The number of days per year with peak temperatures >32.2°C is expected to rise significantly, especially under the higher emissions projection. The largest temperature increase can be expected in summer, when increases in the number of very hot days could mean more and more-intense droughts and wildfires. Increases in temperatures on the very hot days are projected to outpace increases in average temperature (Karl et al. 2009).

To develop estimates of carbon dioxide that humans could produce over the next 100 years, the Intergovernmental Panel on Climate Change (2007) developed a set of greenhouse-gas emissions storylines, based on economic, environmental, global, and regional considerations.

In this chapter, we evaluate the impacts of future climate change by 2020 and 2060 on selected invasive insects, diseases, and nonnative plants (Table 6.1) by examining an ensemble of four general circulation models (CGCM3, GFDL, CCSM3, and HadCM3, described in Chapter 2), anticipated to be moderate and warm, in combination with three of the Intergovernmental Panel on Climate Change storylines (A1B, A2, and B1). A storyline is a narrative description of a scenario (or a family of scenarios), highlighting the main scenario characteristics and dynamics, and the relationships between key driving forces (Nakićenović et al. 2000): A1B representing moderate population growth and high-energy use; A2 representing continuously increasing population, regionally focused economic growth, and slower introduction of alternative fuels technology; and the B1 representing moderate population growth and an emphasis on global approaches to economic and environmental stability.

INSECT INFESTATIONS AND DISEASES

We examined representative species from five classes of insects (Table 6.1): borer (emerald ash borer and Sirex woodwasp), bark beetle (southern pine beetle), defoliator (gypsy moth), sapsucker (hemlock woolly adelgid), and weevil (pine root collar weevil).

The three representative diseases of major significance to the southern forests selected were fusiform rust, annosus root disease, and laurel wilt caused by a fungus (*Raffaelea lauricola*) carried by redbay ambrosia beetles. All plant diseases and insect species selected have unique biological and environmental requirements (Table 6.1) and distribution patterns, which depend on host species distributions across the United States (Figure 6.1). Temperature and precipitation are the two main factors represented in the models presented in this chapter. Optimum temperature range is critical for individual representative species to establish and thrive in a given environment. Hence, in this study, maps of suitable habitat were produced by using (1) individual species temperature range (i.e., lower and upper limits required by the organism at any given time through the season), (2) the corresponding host winter temperature range (upper and lower limits), and (3) the host water availability requirements (i.e., annual minimum precipitation required for growth).

Availability of water is perhaps the most important environmental factor affecting the geographic distribution and structure of vegetation (Woodward 1987). The northern range of distribution for many species is usually limited by minimum annual average temperature; minimum temperature also influences the growing degree-days (the average of the daily maximum and minimum temperatures minus a base temperature of 10°C) that are necessary for physiological processes, such as spring budburst, flowering, pollen production, and seed production (Karlsson 2002; Linkosalo

TABLE 6.1
Temperature and Precipitation Requirements for Several U.S. Invasive Insects and Diseases and Their Hosts

Invasive Species	Temperature (°C)			Preferred Host		
	Extreme	Range	Summer Extreme	Species	Precipitation Range (mm) ^a	January Temperature Range (°C) ^a
Insects						
Redbay ambrosia beetle (<i>Xyleborus glabratus</i>)	<−27 ^b	10.0–25.0	—	Redbay (<i>Persea borbonia</i>)	1020–1630	3–20
Emerald ash borer (<i>Agrilus planipennis</i>)	<−30 ^c	13.4–25.0 ^{d,e,f}	>32 ^g	Green ash (<i>Fraxinus pennsylvanica</i>)	380–1520	−18 to 13
Gypsy moth (<i>Lymantria dispar</i>)	<−20 ^h	3.0–28.0 ^{h,i}	>41 ^{g,h}	Northern red oak (<i>Quercus rubra</i>)	760–2030	4–16
Hemlock woolly adelgid (<i>Adelges tsugae</i>)	<−30 ⁱ	3.9–25.0 ^k	—	Eastern hemlock (<i>Tsuga canadensis</i>)	740–1270	−12 to 6
Pine root collar weevil (<i>Hylobius radialis</i>)	—	8.5–28.0 ^l	>40	Red pine (<i>Pinus resinosa</i>)	510–1010	−18 to −4
Sirex woodwasp (<i>Sirex noctilio</i>)	—	6.8–25.0 ^m	>35	Loblolly pine (<i>Pinus taeda</i>)	1020–1520	4–16
Southern pine beetle (<i>Dendroctonus frontalis</i>)	<−16	6.7–36.7 ⁿ	>37	Loblolly pine (<i>Pinus taeda</i>)	1020–1520	4–16
Diseases						
Annosum root rot (<i>Heterobasidion irregulare</i>)	—	5.0–32.0 ^o	>35	Loblolly pine (<i>Pinus taeda</i>)	1020–1520	4–16
Fusiform rust (<i>Cronartium fusiforme</i>)	—	16.0–25.0	—	Slash pine (<i>Pinus elliottii</i>)	1020–1520	−18 to 17
Laurel wilt (<i>Raffaelea lauricola</i>)	—	10.0–30.0 ^p	>34	Redbay (<i>Persea borbonia</i>)	1020–1630	3–20

Note: — means no information available.

^a Burns and Honkala, 1990 (Source: www.na.fs.fed.us/spfo/pubs/silvics_manual/table_of_contents.htm, date accessed: August 13, 2012.)

^b www.forestthreats.org/publications/fhm/frank-koch/Spatio-temporal_analysis_of_Xyleborus_glabratus_invasion_in_eastern_US_forests.pdf, date accessed: August 13, 2012.

^c www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5191794.pdf, date accessed: August 13, 2012.

^d www.insectscience.org/10.128/i1536-2442-10-128.pdf, date accessed: August 13, 2012.

^e nrs.fs.fed.us/pubs/gtr/gtr-nrs-p-51papers/54keena-p-51.pdf, date accessed: August 13, 2012.

^f esameetings.allenpress.com/2009/Paper17566.html, date accessed: August 13, 2012.

^g pestthreats.umd.edu/content/documents/EABBulletin.pdf, date accessed: August 13, 2012.

^h www.nappfast.org/caps_pests/maps/Documentation/Lymantriamathuradocumentation.pdf, date accessed: August 13, 2012.

continued

TABLE 6.1 (continued)**Temperature and Precipitation Requirements for Several U.S. Invasive Insects and Diseases and Their Hosts**

- ⁱ www.ipm.ucdavis.edu/PHENOLOGY/ma-gypsy_moth.html, date accessed: August 13, 2012.
- ^j www.biocap.ca/rif/report/Hunt_S.pdf, date accessed: August 13, 2012.
- ^k www.insectscience.org/10.62/i1536-2442-10-62.pdf, date accessed: August 13, 2012.
- ^l www.na.fs.fed.us/pubs/misc/prc_weevil/prc_weevil.pdf, date accessed: August 13, 2012.
- ^m www.aphis.usda.gov/plant_health/plant_pest_info/sirex/downloads/sirex-pra.pdf.
- ⁿ repository.tamu.edu/bitstream/handle/1969.1/ETD-TAMU-1360/GRAY-DISSERTATION.pdf?sequence=1, date accessed: August 13, 2012.
- ^o www.springerlink.com/content/06640483645677p1/fulltext.pdf, date accessed: August 13, 2012.
- ^p www.public.iastate.edu/~tcharrin/399harrington8-53.pdf, date accessed: August 13, 2012.
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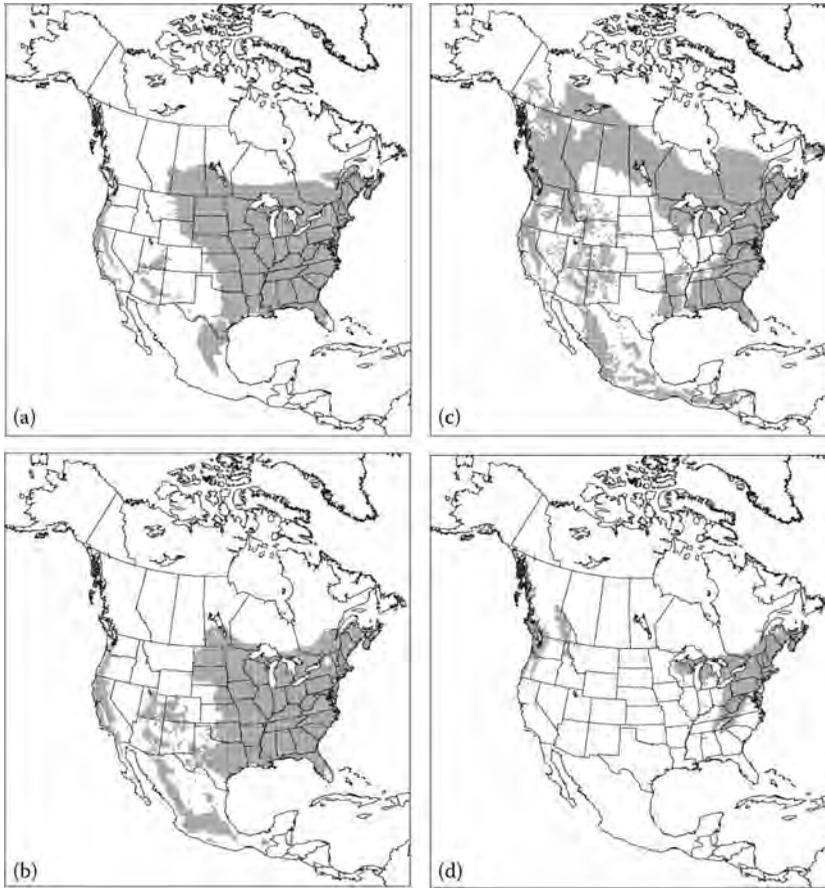


FIGURE 6.1 Distributions of conifer and hardwood hosts for several important invasive insects and diseases of the United States: (a) ash, (b) oak, (c) pine, and (d) hemlock. (Adapted from pubs.usgs.gov/pp/p1650-a/, date accessed: August 13, 2012.)

et al. 2000). In this chapter, insect/disease, host species, and climate scenarios were represented in models examined to obtain long-term projections (i.e., 2020 and 2060). We used annual minimum precipitation and the winter (i.e., January) temperature to produce a climate envelop, that is, a map of likely suitable habitats for individual host plants under different climate scenarios (A1B, A2, and B1 storylines for 2020 and 2060). First, we developed a map of potential suitable habitats for each disease/insect species using the temperature requirements for the insect/disease development in January and July (Table 6.1), which represents suitable habitat in the winter and in the summer, and also indicates the north and south limits for the species. Second, we produced a map of suitable habitat for the corresponding host plant species using the temperature range required by host plant species in January (upper and lower limits), and the annual minimum precipitation required for development of host plant (Thompson et al. 1999) (pubs.usgs.gov/pp/p1650-a/). The climate envelopes for representative hosts were comparable to distributions of major conifer and hardwood hosts (Thompson et al. 1999). Third, we overlaid host and insect/disease maps in DIVA-GIS version 7.5.0 (<http://www.diva-gis.org>) to produce a suitable habitat map for each representative species (Table 6.1).

To highlight the need for daily weather data in future research and the importance of accumulated degree-days, we used a simple regression analysis to establish relationships between accumulated degree-days from the dynamic downscaling daily temperature data (Mearns et al. 2009)

and the monthly average temperature ensemble general circulation model data (A2 storyline). The resulting regression equation was applied to the monthly average temperature from the ensemble data to calculate the degree-days accumulation from March to August, which represents the period between spring and summer. The degree-days data were used to produce maps of suitable habitat under the A1B, A2, and B1 storylines for 2020 and 2060.

NONNATIVE PLANTS

To study the potential impact of climate change on nonnative invasive plant species, we analyzed five representative species of different growth forms (Table 6.2): garlic mustard (*Alliaria petiolata*) for forbs, cogongrass (*Imperata cylindrical*) for grasses and canes, Chinese privet (*Ligustrum sinense*) for shrubs, kudzu (*Pueraria montana* var. *lobata*) for vines, and tallowtree (*Triadica sebifera*) for trees. To reconstruct the spatial distribution of each species, we used a spatial distribution database developed by the Eastern Forest Environmental Threat Assessment Center of the Forest Service (www.forestthreats.org, date accessed: August 13, 2012).

Because many species have been introduced to the United States over a relatively short period of time and are thus still spreading, their distribution boundaries (such as their northern and southern latitudinal limits) are difficult to define. For this reason, we used their native ranges as references to determine their warmest and coldest climatic limits (Guo 2006). We projected future distributions at the county level for the year of 2020 and 2060 (10-year and 50-year projections).

The geographic distribution of a species is the product of complex interactions among environmental adaptations, ecological and evolutionary processes, and alterations by humans (Guo et al. 2012a). Climatic variables are the first-order constraints of species distribution, as evidenced in many range-equilibrium studies (Geber and Eckhart 2005; Griffith and Watson 2006). In this

TABLE 6.2
Characteristics and Climatic Limits for Five Nonnative Plant Species of the Southern United States

Species	Growth Form	Native Region(s)	Years Since Introduction (Estimated)	Southern Limit Average Temperature (°C)		Northern Limit Average Temperature (°C)		Annual Precipitation (mm)
				January	July	January	July	
Garlic mustard (<i>Alliaria petiolata</i>)	Forb	Europe	200	9	28	-17	3.9	≥500
Cogongrass (<i>Imperata cylindrical</i>)	Grass	Asia	100	15	35	-14	21	1000–2500
Chinese privet (<i>Ligustrum sinense</i>)	Shrub	Asia	150	13.3	28.5	-10	24.7	700–1600
Kudzu (<i>Pueraria montana</i> var. <i>lobata</i>)	Vine	Eastern, southern Asia	100	>14	30	-14	16	>1000
Chinese tallowtree (<i>Triadica sebifera</i>)	Tree	Eastern Asia	300	10	30	-2	27.5	700–2800

analysis, we employed a bioclimate modeling approach to study the potential habitats, and a set of temperature and precipitation variables to represent the limits in native ranges (Table 6.2). The underlying assumption of bioclimate-based invasive species distribution modeling is that a species can only survive in environments with conditions that match those of its native range (Peterson 2003; Peterson and Vieglais 2001). This general consistency in environmental requirements after a species is introduced into a new area allowed us to ascertain the maximum extent of invasion and identify the geographic areas with native resources at risk. Bioclimate-based models are useful and, to some extent, effective tools in predicting the impacts of climate change on the distribution of species, at least as a first approximation (Pearson and Dawson 2003).

OUTLOOK FOR FOREST PESTS

It is important to note that temperature and precipitation were the only weather factors represented in the models used for producing the 2020 and 2060 projections in this chapter. However, references were made to other ecological factors that might be important, but not represented in the models (e.g., wet springs) in discussing on each representative species.

ANNOSUS ROOT DISEASE

Annosus root disease is a deadly disease of forest trees that is often associated with thinning susceptible hosts such as intensively planted loblolly, longleaf, shortleaf, slash, and white pines. The disease has resulted in significant losses of conifers across the South, where slash and loblolly pines are common susceptible hosts (Robbins 1984; Stambaugh 1989). Infection is particularly common in freshly cut stump surfaces and is often transmitted through root grafts that function as conduits to nearby susceptible hosts.

Impacts of climate change—Spores are disseminated throughout the year, making annosus root disease highly dependent on favorable environmental conditions for successful germination and establishment. The fungus does not do as well in cooler, wetter climates; the optimum temperature is approximately 24°C. Ambient temperatures of 45°C can render the spores inactive, and mycelia in wood can die from exposure to 40°C for an hour (Duerr and Mistretta in press). The range of annosus root disease extends throughout most southern forests with few areas remaining for further spread. Changes in environmental factors including increased temperatures, reduced rainfall, and increased host growth could result in increased activity attributable to increased host susceptibility. A drier/warmer climate could also increase pine susceptibility on well-drained sandy sites and on reforested farms where pines are already highly susceptible to the disease (Duerr and Mistretta in press), causing reductions in the distribution range of the host species and ultimately limiting the overall negative impact of this disease by blocking infestations in new locations.

By 2020 and 2060, suitable July habitat for annosus root disease is expected to cover most of the South, with the exception of southern Florida, some parts of Oklahoma, and a large part of western Texas. Under the A1B storyline, the northern limit could shrink by 2060. The pattern of suitable July habitat will likely be similar for the three storylines, with slightly larger January habitable areas under A2 storyline (Figure 6.2). A significant retreat of the range's southern edge is unlikely.

Although the disease is found throughout much of the United States, it is more common and severe in the South. Therefore, a warmer climate and a shift in host ranges could mean that the disease will become more common in latitudes farther north.

Management—As temperatures increase, thermal treatment of the stumps by sunlight could become even more effective in preventing infection. Other efforts, such as restoration of the less susceptible longleaf pine to its previous range, could further decrease the range of annosus root disease. Using borax as a stump treatment in uninfected stands is expected to continue to be an effective tool for managing the disease (Duerr and Mistretta in press).

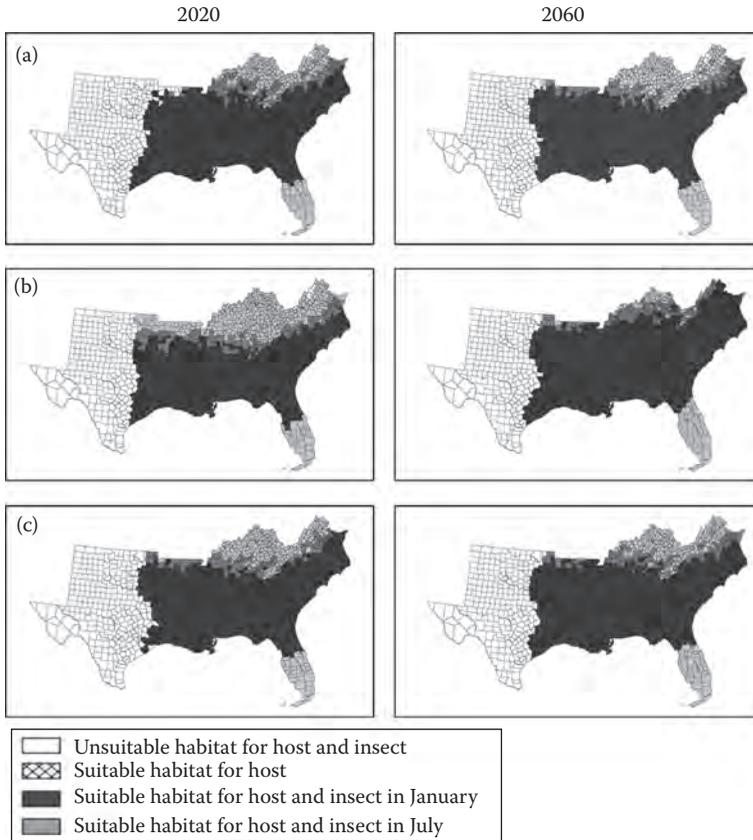


FIGURE 6.2 Suitable habitat for annosus root disease in January and July (2020 and 2060) in the Southern United States, projected using an ensemble of selected general circulation models (CGCM3, GFDL, CCSM3, and HadCM3) and assuming three Intergovernmental Panel on Climate Change (2007) emissions storylines; (a) A1B, (b) A2, and (c) B1 (see text and Chapter 2 for storyline descriptions).

FUSIFORM RUST

Fusiform rust is an important destructive disease of southern pines that primarily occurs on susceptible species such as loblolly and slash pines, where it causes the formation of fusiform-shaped galls that are fatal to host trees if they occur on the main stem (Anderson et al. 1980; Czabator 1971; Duerr and Mistretta in press). Among the most important factors affecting the incidence of fusiform rust in forest stands are weather, fire, soil cultivation, tree species, genetic strain, stand age, rate of height growth, and density of stocking (Goggans 1957). In the 1930s, extensive planting of susceptible slash and loblolly pines later led to the spread of fusiform rust throughout the host range, with infected trees found in all southern pine ecosystems (Duerr and Mistretta in press; Ward and Mistretta 2002). Heavy losses were observed in Coastal Plain sites from Louisiana to southeastern South Carolina (Duerr and Mistretta in press). Phelps and Czabator (1978) reported that the highest incidences of fusiform rust were in the loblolly and slash pine stands of Georgia, Alabama, South Carolina, and Mississippi. In Louisiana, incidence was low for loblolly and high for slash, whereas Florida had a higher incidence for loblolly and a lower incidence for slash pine. Incidence in Arkansas, Virginia, Texas, and North Carolina was generally low.

Impacts of climate change—Temperature (15–27°C) and humidity (97–100%) range requirements for infection (Phelps and Czabator 1978) and the late winter and early spring weather affect the extent and severity of infection during a given year (Goggans 1957). In the next 50 years, the

range of fusiform rust is expected to expand concomitantly as the ranges of its pine and (and alternatively, oak) hosts expand with migration from coastal areas upward into the Appalachian Mountains. Already, the disease is widely distributed within its geographical host range in the South; therefore, a warmer/drier climate could be of less concern than the possibility that a virulent strain will emerge on current hosts. Duerr and Mistretta (in press) and Chakraborty et al. (1998) predict that any losses of pine hosts in coastal areas (with warmer, drier climate regimes) would be offset by gains in the Piedmont and in the lower reaches of the Appalachian Mountains. Any northward shift in planting loblolly pine prompted by warmer climate regimes would likely be matched by a rapid movement of fusiform rust into the new planting areas.

By 2020 and 2060, the potential suitable habitat for fusiform rust will likely extend throughout most of the region with a few unsuitable habitats below the south limit in southern Florida, western Oklahoma, and parts of central and western Texas. The suitable July habitat under the A1B, A2, and B1 storylines in 2020 is expected to shrink by 2060 (Figure 6.3).

Management—The amount of fungal inoculum available, the susceptibility of individual pine species, the abundance of the alternate susceptible hosts (oaks), and favorable climatic conditions are critical for the spread of fusiform rust. Longleaf pines are moderately resistant and shortleaf pines are highly resistant to the fungus (Anonymous 1989). Although several uncertainties still exist, planting of resistant seedlings in anticipation of shifts in host geographic range and changes in

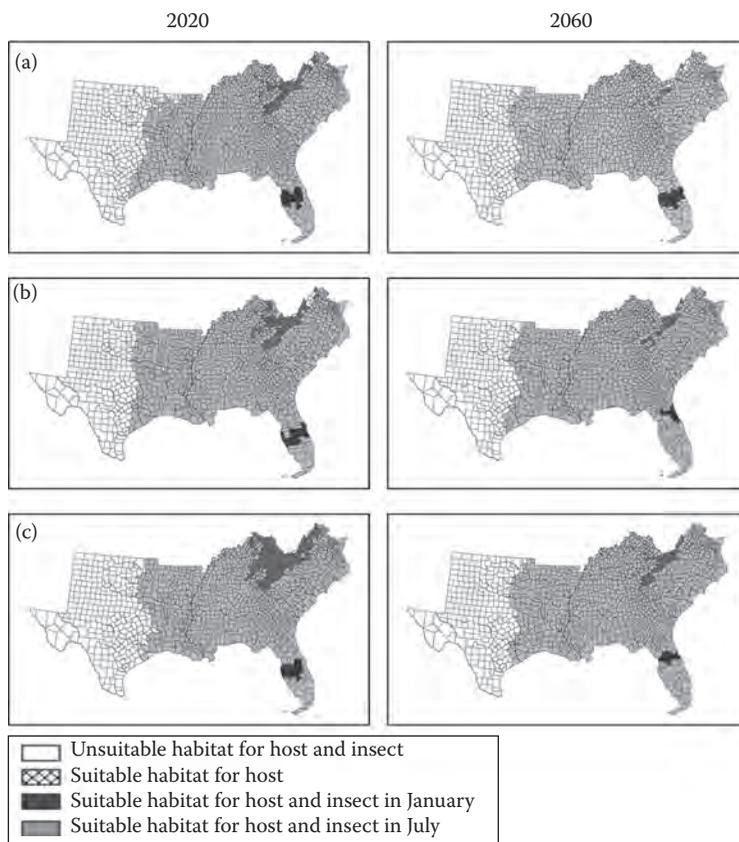


FIGURE 6.3 Suitable habitat for fusiform rust in January and July (2020 and 2060) in the Southern United States, projected using an ensemble of selected general circulation models (CGCM3, GFDL, CCSM3, and HadCM3) and assuming three Intergovernmental Panel on Climate Change (2007) emissions storylines; (a) A1B, (b) A2, and (c) B1 (see text and Chapter 2 for storyline descriptions).

climate could be a useful management strategy in reducing the impact of the disease. Other strategies for plantations and forests include avoiding over-fertilization of seedlings in nurseries, using effective systemic fungicides to protect seedlings in nurseries, applying silvicultural manipulations in young stands to favor healthy saplings, and favoring the deployment of genetically screened resistant seedlings (Duerr and Mistretta in press).

LAUREL WILT/REDBAY AMBROSIA BEETLE COMPLEX

The redbay ambrosia beetle was first reported in the United States in 2002 at the Port Wentworth in Georgia (Rabaglia 2005) and was later found to be consistently associated with unusual mortality of redbay (*Persea borbonia*) and sassafras (*Sassafras albidum*) in coastal areas of Georgia and South Carolina (Fraedrich et al. 2008). Most ambrosia beetles attack stressed, dead, or dying woody plants but the redbay species is unusual in that it attacks healthy trees. Initial attacks occur on healthy trees followed by reproduction on dead tissue. The beetle feeds on plants in the *Lauraceae*. It is thought to be attracted to host plant volatiles, which presumably trigger attacks (Harrington and Fraedrich 2010; Hanula et al. 2008). A single attack by a redbay ambrosia beetle is sufficient to transmit the fungal pathogen that causes laurel wilt, which quickly causes mortality (Fraedrich et al. 2008; Harrington et al. 2008).

Redbay, the favored host of the redbay ambrosia beetle (Hanula and Sullivan 2008), grows in a climate ranging from warm–temperate along the Atlantic Coast to semitropical in southern Florida and the coast of Texas (Brendemuehl 1990). According to the Global Invasive Database provided by the International Union for Conservation of Nature Invasive Species Group (2007a) (<http://www.issg.org/database/species/ecology.asp?si=1536&fr=1&sts=sss&lang=EN>, date accessed: June 26, 2013), other hosts of the redbay ambrosia beetle include swamp bay (*Persea palustris*), avocado (*Persea americana*), camphortree (*Cinnamomum camphora*), pondberry (*Lindera melissifolia*), and pondspice (*Litsea aestivalis*).

The laurel wilt fungus, a food source of the redbay ambrosia beetle, causes vascular damage in redbay, sassafras, and other southern laurel species (Fraedrich et al. 2008; Harrington et al. 2008). At present, the redbay ambrosia beetle is the only known vector. The disease is widespread in six southern states and continues to spread throughout the ranges of redbay and sassafras.

Impacts of climate change on the redbay ambrosia beetle—The redbay ambrosia beetle has a much narrower native geographic range than other ambrosia beetle species and is limited to subtropical or warm temperate climates that are characteristic of eastern Asia (Koch and Smith 2008; Schiefer and Bright 2004). Although flight activity is typically highest in the summer (Hanula et al. 2008), data suggest that adults can also fly during winter and summer months (Mayfield and Thomas 2006). Unlike any other parts of the United States, the southern Coastal Plain has sufficient rainfall across the months of the growing season to provide the amount of moisture needed to sustain the beetle's fungal symbiont. Whether temperature is more of a constraint than moisture remains uncertain. The beetle-symbiont interaction could be constrained to the southern Coastal Plain, and therefore unlikely to spread into other areas east of the Mississippi River (Koch and Smith 2008; Rabaglia et al. 2006). Duerr and Mistretta (in press) indicate uncertainty as to whether both the beetle and fungus will be able to establish on other hosts as their ranges shift under the influence of climate change, making the likelihood of laurel wilt spreading beyond its projected range highly uncertain.

The beetle and the laurel wilt are already as far south as Homestead, Florida. The three emission storylines (A1B, A2, and B1) evaluated in this study indicate a likely decline in potential suitable habitats for the redbay ambrosia beetle by 2060. The B1 storyline will likely have larger areas of suitable habitat in July 2020 compared to the other storylines (Figure 6.4); by 2060, suitable habitat in southern Florida and southeastern Texas is likely to shift slightly southward in January for all storylines.

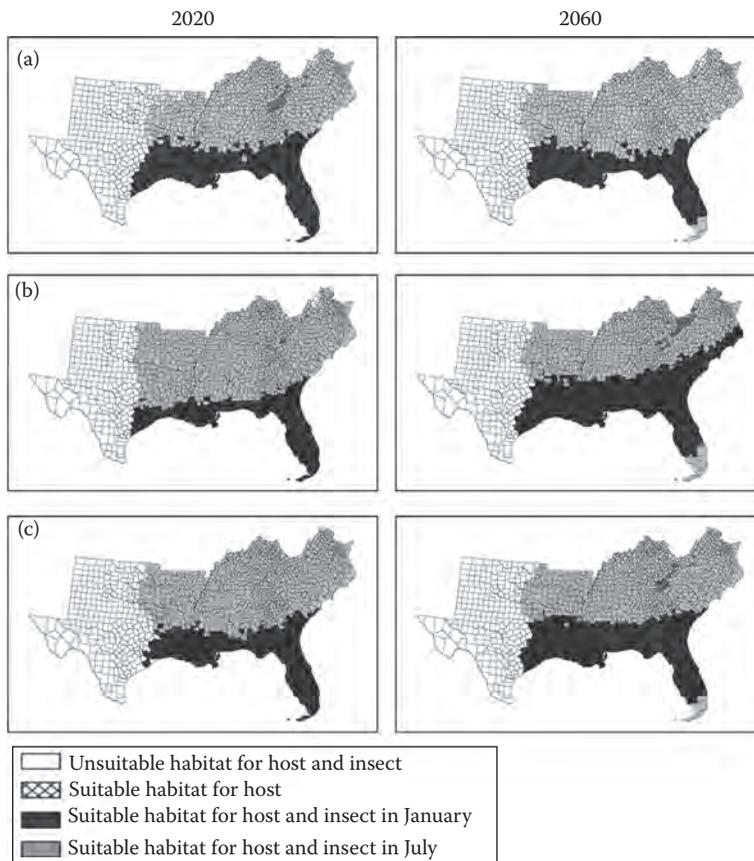


FIGURE 6.4 Suitable habitat for the redbay ambrosia beetle in January and July (2020 and 2060) in the Southern United States, projected using an ensemble of selected general circulation models (CGCM3, GFDL, CCSM3, and HadCM3) and assuming three Intergovernmental Panel on Climate Change (2007) emissions storylines; (a) A1B, (b) A2, and (c) B1 (see text and Chapter 2 for storyline descriptions).

Impacts of climate change on laurel wilt—Sustaining laurel wilt requires the presence of the redbay ambrosia beetle and viable host tissue, both of which need adequate water, although the fungus is not one that grows better under more moist or wet conditions. Koch and Smith (2008) projected that laurel wilt will reach its northern host range by 2020 and its western host range (eastern Texas) by 2040. This projection was only based on the natural range of redbay and on the climatic barriers that could limit the progress of the redbay ambrosia beetle and its fungal symbiont. In 2020, suitable July habitat is expected to be fairly similar for all three storylines, with few differences in the northern limit (Figure 6.5). Suitable July habitat is projected to shrink markedly by 2060. The projections indicate a northward shift in suitable January habitat in southern Florida and the lower Coastal Plain, where only few southern-most counties could become unsuitable in 2060. However, conditions will remain suitable for the hosts, redbay in the south and sassafras to the north.

Management—Because redbay ambrosia beetles can fly, laurel wilt can spread over short distances and perhaps longer distances when the beetles are carried by prevailing winds. The pattern of disease progression reported in Georgia and elsewhere strongly suggests that humans are aiding in its long distance spread. In one instance, an infection center was found adjacent to a campground, which most certainly occurred because a camper had brought in firewood that had died from laurel wilt and was infested with the redbay ambrosia beetle (Cameron et al. 2008). Other than the use of integrated management approach, no single control measure is effective in protecting

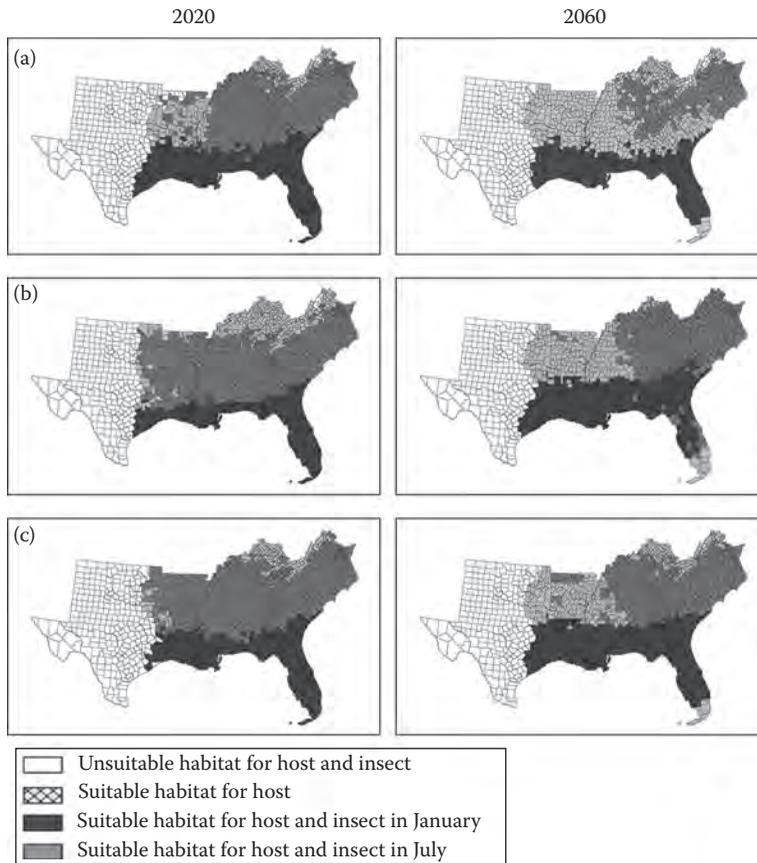


FIGURE 6.5 Suitable habitat for laurel wilt in January and July (2020 and 2060) in the Southern United States, projected using an ensemble of selected general circulation models (CGCM3, GFDL, CCSM3, and HadCM3) and assuming three Intergovernmental Panel on Climate Change (2007) emissions storylines; (a) A1B, (b) A2, and (c) B1 (see text and Chapter 2 for storyline descriptions).

forest and woodland trees against laurel wilt. A demethylation-inhibiting fungicide showed some promising results for preventing the disease in treated trees. However, the cost of treatments will likely limit its use in the future, except for high value trees (Mayfield and Thomas 2006). Therefore, adequate and effective dissemination of vital information and an increase in local awareness of the disease are critical in dealing with laurel wilt.

Local park-recreational activities could have played an important role in the spread of the redbay ambrosia beetle and host-tree mortality in Georgia. Therefore, effective phytosanitary practices to manage infested plant materials are critical, including monitoring, early detection, and effective disposal (chipping or burying) of infected materials (Mayfield and Thomas 2006).

SOUTHERN PINE BEETLE

The southern pine beetle is a destructive insect pest of southern pine forests (Meeker et al. 2004; Thatcher and Conner 1985) where it has killed >1 million acres of pines, valued at >\$1.5 billion, during outbreaks from 1999 to 2002 (Duerr and Mistretta in press). Southern pine beetle populations can increase rapidly, killing large numbers of trees, during periodic outbreaks; however, it could also go through periods of low activity when finding a single infested tree or capturing beetles in pheromone traps becomes rare (Billings and Upton 2010; Thatcher et al. 1980; Thatcher and

Barry 1982). Southern pine beetles attack all species of pines, including loblolly, shortleaf, slash, pitch, Virginia (*Pinus virginiana*), and pond (*Pinus serotina*), although impacts on longleaf pine (*Pinus palustris*) are generally lower. Successful infestations have been observed in eastern white pines and Table Mountain pines (*Pinus pungens*), and an outbreak was recently observed in New Jersey pitch pines. Attacks on trees <5 years or <2 inches in diameter are very rare, but plantations with uniform-aged single species are susceptible as they age (Duerr and Mistretta in press). Mature trees in pure, dense stands or unthinned pine plantations are most susceptible to attack (Cameron and Billings 1988); therefore, large acreages of pine plantations established in the South over the past five decades could continue to be particularly vulnerable.

Impacts of climate change—Trân et al. (2007) reported that southern pine beetle populations decline when winter minimum temperatures drop below -14°C ; and that developmental processes increase with temperatures ranging from approximately zero development at 5 to 10°C , and begin to decline when temperatures exceed 27 – 32°C (Wagner et al. 1981, 1984a). Extremely hot summer temperatures can be deadly to southern pine beetles (Wagner et al. 1984b) and could affect their interactions with associated forest communities (Hofstetter et al. 2006, 2007; Lombardero et al. 2003). Every stage of the southern pine beetle life cycle can be found in the South throughout all seasons (Thatcher and Pickard 1966, 1967), with a generation completed in about 50 days at 25°C , 100 days at 17°C , and 200 days at 12.5°C (Friedenberg et al. 2007; Wagner et al. 1984a). In the next 50 years, warmer winters could support more generations of southern pine beetles per year (Duerr and Mistretta in press, Gan 2004), but could also disrupt the timing of adult emergence with the onset of new infestations in spring (Billings and Kibbe 1978). Extremely hot summers could increase mortality, hinder flight, reduce physiological activities, and disrupt population growth. Pine stands in the northern edges of the region and even farther north than the historical southern pine beetle range will most likely experience significantly greater southern pine beetle activity and impacts than in the past (Duerr and Mistretta in press).

Under the three storylines, southern pine beetles would likely maintain a considerable suitable habitat from January to July based on 2020 and 2060 projections. Much of the South would provide suitable habitat, the exception being some southern Florida counties where environmental conditions could be unsuitable for either the host (loblolly pine) or the beetle. The pattern of suitable habitat in July will likely be similar for the three storylines, with a slight northern shift in under A2 by 2060 (Figure 6.6).

Management—Adequate management efforts, types of forest composition, direct suppression of new invasions, and other actions will play a significant role in determining southern pine beetle activity and future damages to hosts (Friedenberg et al. 2008). If, in the next 50 years, forest fragmentation increases, age distributions shift to predominantly younger classes, and thinning of plantations increases, the impacts of southern pine beetles could be lower, despite a warmer climate. Conversely, if the frequency of extreme precipitation events (drought and flooding) increases, pines could become stressed and vulnerable to increased southern pine beetle activity, resulting in significant damage (Duerr and Mistretta in press). Therefore, management options such as planting the appropriate species for a given site, lowering planting densities, and thinning pine stands (which increases stand vigor and resiliency) will likely reduce the risks of southern pine beetle outbreaks. Early detection and monitoring for infestation spots, followed by prompt direct suppression of active spots (Billings 1980), could help minimize damage when outbreaks do occur.

GYPSEY MOTH

Gypsy moth, a native of Europe and Asia, was accidentally introduced into the United States in 1869 (Lechowicz and Mauffelte 1986). Repeated defoliation by gypsy moth has significant effects on the health of oak forests (Campbell and Sloan 1977). Gypsy moth caterpillars feed on a wide range of trees and shrubs, but prefer oaks (Liebhold et al. 1995). Severe defoliation could leave oak trees vulnerable to secondary attacks by pests or pathogens such as the twolined chestnut borer (*Agrilus bilineatus*) or Armillaria root disease caused by a fungus (*Armillaria mellea*). Extended drought could also predispose oak trees to gypsy moth attack and increase oak tree mortality (Duerr

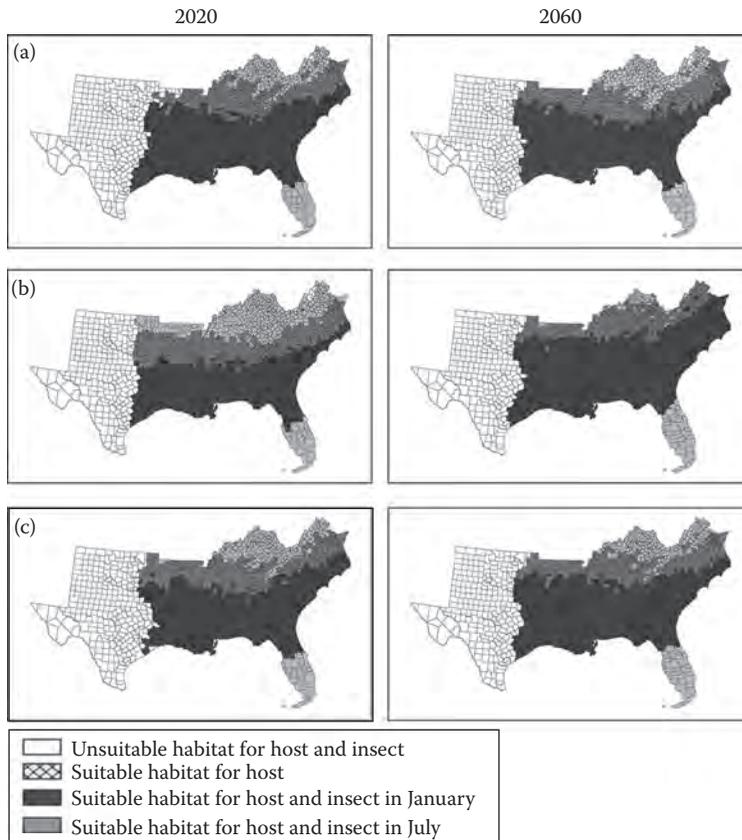


FIGURE 6.6 Suitable habitat for the southern pine beetle in January and July (2020 and 2060) in the Southern United States, projected using an ensemble of selected general circulation models (CGCM3, GFDL, CCSM3, and HadCM3) and assuming three Intergovernmental Panel on Climate Change (2007) emissions storylines; (a) A1B, (b) A2, and (c) B1 (see text and Chapter 2 for storyline descriptions).

and Mistretta in press). Highly favored host species include sweetgum (*Liquidambar styraciflua*), northern red oak (*Q. rubra*), and American basswood (*Tilia americana*). Species of limited suitability include pines, ashes, maples (*Acer* spp.), American beech (*Fagus grandifolia*), and black cherry (*Prunus serotina*). Species that are not favored or are avoided include blackgum (*Nyssa sylvatica*), yellow poplar (*Liriodendron tulipifera*), black locust (*Robinia pseudoacacia*), bald cypress (*Taxodium distichum*), southern magnolia (*Magnolia grandiflora*), and tupelo (Duerr and Mistretta 2011, in press).

Impacts of climate change—Duerr and Mistretta (in press) suggested that regardless of changes in climate, the range of gypsy moth infestation is expected to expand, and at a faster rate than can be attributed to any potential climate change. Logan et al. (2007) noticed that the climatic suitability for the gypsy moth in the Western United States could improve rapidly, due to a general warming trend that began few decades ago and still continues today. In the southeast, the gypsy moth will likely encounter lower concentrations of oak and cove hardwoods as it moves south and west, and forest susceptibility will decrease in some areas. Therefore, a significant portion of the South could experience widespread gypsy moth infestations over the next 50 years. Mixtures of oak and hickory (*Carya* spp.) could displace the boreal forests at higher elevations in the South, and a drier climate could increase host stress and reduce buildup of the gypsy moth's fungal predator, which thrives during wet spring weather. Future outbreaks will depend on the current management efforts to slow,

suppress, and eradicate infestations; the abundance and health of hardwood forests that are subject to infestation; and the resilience of the gypsy moth, its natural enemies, and its host trees to changes in temperature and moisture (Duerr and Mistretta in press).

The pattern of potential suitable habitat for the gypsy moth in January would be similar for all three storylines in the 2020 projections. Much of the South is potentially suitable, with the exception of a few counties in southern Florida, western Oklahoma, and Texas. The suitable habitat in July could become bigger under the B1 storyline compared to A1B and A2. Reduction in July suitable habitat is likely by 2060 under the A1B storyline. The A2 storyline predicts a slight northward shift in 2060 (Figure 6.7).

Management—Human population growth could also increase the spread of gypsy moths through increases in recreation and transportation. Therefore, continued vigilance and the application of early detection methods are needed to reduce infestations and help prevent rapid spread in the South. Logan et al. (2007) demonstrated the use of a risk assessment system for predicting establishment of the gypsy moth. Several methods are available for managing the spread of the gypsy moth, including insecticides (*Bacillus thuringiensis* or diflubenzuron), or pheromone flakes to disrupt mating (Duerr and Mistretta in press). According to Sharov et al. (2002), the multi-agency pilot project called “Slow the Spread Program” reduced the rate of spread from approximately 25 miles a year to 7–10 miles per year. Continuation of these programs means that the gypsy moth would

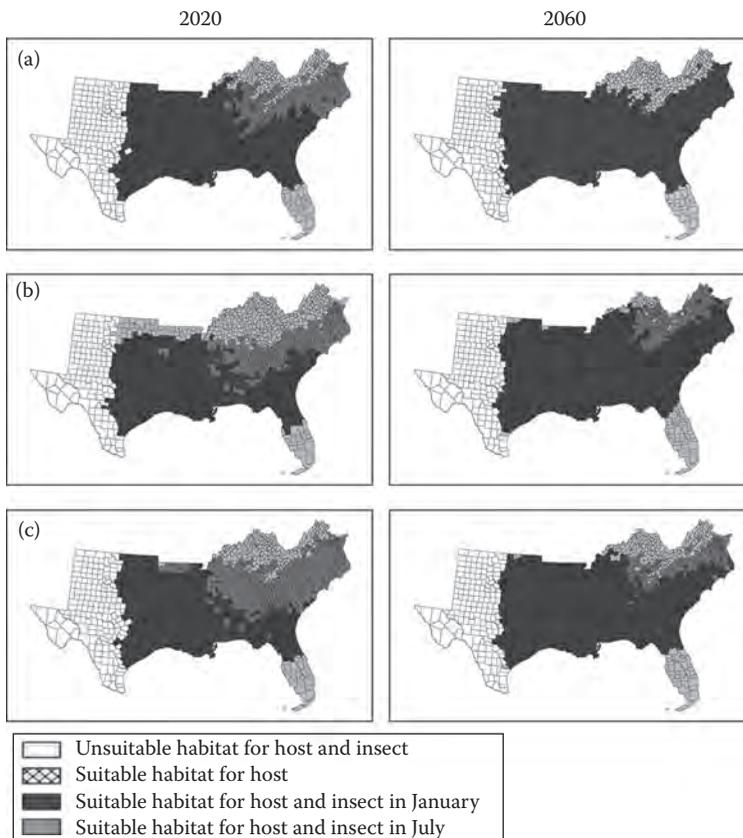


FIGURE 6.7 Suitable habitat for the gypsy moth in January and July (2020 and 2060) in the Southern United States, projected using an ensemble of selected general circulation models (CGCM3, GFDL, CCSM3, and HadCM3) and assuming three Intergovernmental Panel on Climate Change (2007) emissions storylines; (a) A1B, (b) A2, and (c) B1 (see text and Chapter 2 for storyline descriptions).

take the next 50 years to move 350–500 miles farther south, compared to 25–30 years without the program (Duerr and Mistretta in press).

HEMLOCK WOOLLY ADELGID

The hemlock woolly adelgid is native to Asia. In North America, this destructive invasive pest of ornamental and forest hemlock trees (*Tsuga canadensis*) was first reported in the western Canadian province of British Columbia in the 1920s (Canadian Food Inspection Agency 2011), and was initially thought to have crossed the U.S. border in Virginia years later. However, Havill et al. (2006) show they were very distinct genetically. Heavy infestations in the Eastern United States have killed host trees in as little as four years, although some trees have survived infestations for more than a decade (McClure et al. 2001). Hemlock mortality can occur quickly and uniformly throughout a stand (Fajvan and Wood 2010), and periods of low winter temperatures have been thought to cause high hemlock woolly adelgid mortality, thereby hindering population establishment and spread (Evans and Gregoire 2007; Morin et al. 2009; Parker et al. 1999; Shields and Cheah 2005). Hemlock woolly adelgid is widely distributed among hemlock growing areas, where minimum winter temperatures stay above -28.8°C . Its cold hardiness depends on geography and season, experiencing a gradual loss of cold tolerance as winter progresses (Skinner et al. 2003).

Impacts of climate change—McClure and Cheah (2002) indicated that the northerly spread and future range of hemlock woolly adelgid could be hindered by the duration, severity, and timing of minimum winter temperature. Paradis et al. (2008) found that an average winter temperature of -5°C is required to prevent hemlock woolly adelgid populations from expanding and spreading. Warmer winters would likely increase survival, fecundity, population growth, and extent of infestations (Pontius et al. 2002, 2006), which could lead to more outbreaks in its current range (Parker et al. 1997), as well as a northward expansion in range (Paradis et al. 2008). Morin et al. (2009) reported that the abundance of host hemlock was a major factor in the spread rate of the hemlock woolly adelgid with hemlock abundance negatively correlated to temperature. Widespread mortality of hemlocks following adelgid outbreaks would change forest composition, structure, nutrient cycling, surface water quality, and populations of associated wildlife (Ford et al. 2012; Jenkins et al. 1999; Kizlinski et al. 2002; Ross et al. 2003; Spaulding and Rieske 2010; Tingley et al. 2002; Vose et al. 2013; Webster et al. 2012). Historical spread of hemlock woolly adelgid was estimated at 3.6 km/year (± 0.2 km) from east to west and 5.8 km/year (± 0.28 km) from north to south (U.S. Department of Agriculture Forest Service 2011a).

The projection maps indicate a likely shrinkage in potential suitable July habitats under the A1B storyline by 2060. The B1 storyline will likely have larger suitable July habitat by 2020 compared to the other storylines (Figure 6.8). A marked northward shift in suitable habitat in January is likely under the A2 storyline by 2060. The distribution of hemlock woolly adelgid habitat could be limited to ecosystems in upper areas of the region.

Management—Hemlock woolly adelgid management involves the integrated use of cultural, chemical, biological, and genetic approaches to reduce adelgid populations. The cultural management approach for hemlock woolly adelgid includes preventing movement of infested plant materials to noninfested locations, especially between March and June when adelgid eggs and crawlers are abundant and seek out stress resistant cultivars of hemlock. Cowles et al. (2006) found that application of the systemic insecticide imidacloprid to soil was effective in eliminating the adelgid for a few years. Application of chemical insecticides is particularly useful for managing hemlock woolly adelgid in urban landscapes, and has been used extensively in select priority forest settings (McClure et al. 2001).

Hemlock woolly adelgids do not have host resistance or natural enemies in eastern areas of North America, although local arthropod predators help regulate their populations in their native ecosystem (McClure et al. 2001). *Laricobius nigrinus* Fender (Coleoptera: Derodontidae), native to western North America, has also been observed as a predator of hemlock woolly adelgids (Mausel et al.

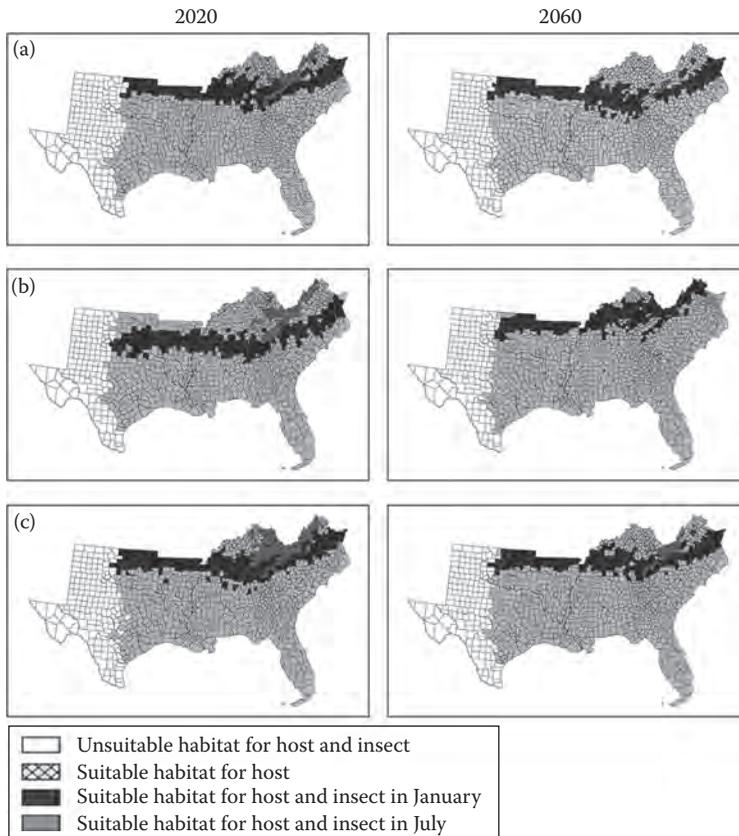


FIGURE 6.8 Suitable habitat for the hemlock woolly adelgid in January and July (2020 and 2060) in the Southern United States, projected using an ensemble of selected general circulation models (CGCM3, GFDL, CCSM3, and HadCM3) and assuming three Intergovernmental Panel on Climate Change (2007) emissions storylines; (a) A1B, (b) A2, and (c) B1 (see text and Chapter 2 for storyline descriptions).

2010). Predatory beetles imported from China and Japan have shown promising results (McClure et al. 2001; Onken and Reardon 2011). The widespread mortality seen in hemlock could mean replacement by American beech, northern red oak, sugar maple (*Acer saccharum*), yellow birch (*Betula alleghaniensis*), and red maple (*Acer rubrum*). Vose et al. (2013) identified needs for restoration efforts to include novel approaches, such as the introduction of nonnative species, facilitating movement of native species to new habitats (e.g., white pine), and aggressive management of existing species (e.g., rhododendron) with mechanical removal, fire, or chemicals. According to Jonas et al. (2012), in addition to encouraging natural regeneration of hemlock in damaged forest stands, growing offsite stocks of hemlock seedlings for replant, creating hemlock woolly adelgid-resistant hybrid hemlocks, and replanting using already resistant western hemlock or Chinese hemlock are useful genetic management strategies against hemlock woolly adelgid.

EMERALD ASH BORER

The emerald ash borer is a devastating, wood-boring beetle first discovered in 2002 near Detroit and across the Canadian border in Windsor, Ontario (Kovacs et al. 2010). More than 50 million ash trees (*Fraxinus* spp.) have died, are dying, or are heavily infested in Michigan, Indiana, and Ohio (Smith et al. 2009), with additional infestations reported in 12 other states (Illinois, Iowa,

Kentucky, Maryland, Minnesota, Missouri, New York, Pennsylvania, Tennessee, Virginia, West Virginia, and Wisconsin) as of early 2011 (U.S. Department of Agriculture Animal and Plant Health Inspection Service 2011b). Ash trees are an important part of the rural and urban forests of the United States, valued at >\$282 billion (U.S. Department of Agriculture Animal and Plant Health Inspection Service 2003). Unfortunately, all 16 species of ash tree in North America appeared to be susceptible to the emerald ash borer, which has significantly damaged the ecology and economy of infested areas (Duerr and Mistretta in press). Keena et al. (2009) reported that the optimum temperature for egg hatching is approximately 25°C, and that adult longevity decreases as temperature increases from 20°C to 30°C. Sobek et al. (2009) noted that the emerald ash borer larvae cannot survive temperatures below -30.6°C.

Impacts of climate change—Although susceptible species, such as green ash (*Fraxinus pennsylvanica*) and white ash (*Fraxinus americana*), have wide distributions in the South, they are not a dominant component of southern forests but a common component of riparian forests in this region. Over the next 50 years, the emerald ash borer could possibly infest and kill many, if not most, of the ash trees in the South. Millions of ashes could die from extreme drought, rapid insect population growth, and multiple infestation outbreaks, ultimately shrinking their southern range (Duerr and Mistretta in press). Unfortunately, at present, little information is available about the life cycle, flight capabilities, host preferences, natural enemies, or management methods.

The potential suitable habitat for the emerald ash borer in July is predicted to be significantly larger in 2020 compared to 2060 for all three emissions storylines. In 2020, the largest suitable habitat in July would be under B1, and A1B and A2 would likely be similar in distributions but smaller than B1. A slight northward shift in 2060 is possible under the A2 storyline at the southern edge of the range (Figure 6.9).

Management—The ability to detect early infestations will be crucial in preventing the spread of the emerald ash borer in the South (Duerr and Mistretta in press). Efforts currently geared toward controlling and eradicating the emerald ash borer—imposing quarantines, conducting surveys, drawing boundaries around areas that are adjacent to confirmed infested sites, removing ash trees, and developing information—would support management efforts. Lack of effective survey and control technology has made containment efforts challenging in some areas where the emerald ash borer has been found. A practical option could be to delineate and protect small pockets of exceptional ash resource as “ash conservation areas.” Currently available chemical treatments are not cost effective for large-scale implementation. Mercader et al. (2011) identified management options with insecticides that significantly reduced the spread of *A. planipennis* when treatments were applied 1–4 years after infestations were initiated. Also, some parasitoids of larva and egg are being investigated for use as biological control agents (U.S. Department of Agriculture Animal and Plant Health Inspection Service 2011b). If effective, these control agents could help mitigate the spread of emerald ash borer populations, but they are not expected to completely stop the spread (U.S. Department of Agriculture Animal and Plant Health Inspection Service 2011b) with future climate change.

PINE ROOT COLLAR WEEVIL

The pine root collar weevil is a native insect pest of pine trees in north central and northeastern areas of the United States. The larval stage of the weevil causes major injury to host trees by feeding below ground in the root collar, root crown, and large roots (Eliason and McCullough 1995; Wilson and Schmiede 1975). Native and nonnative species of pines are susceptible to the weevil, which attacks both young and old pines. Young trees <4 inches in diameter are more vulnerable to attacks than larger trees, and can be killed in a single year by as few as two to five larvae. Attacks by weevils generally reduce growth rate, limit transport of nutrients, and increase susceptibility of the host trees to heavy wind, snow, and secondary pests. Adult feeding also girdles small shoots and branches, causing them to die and turn red. Infestations are more pronounced in pine

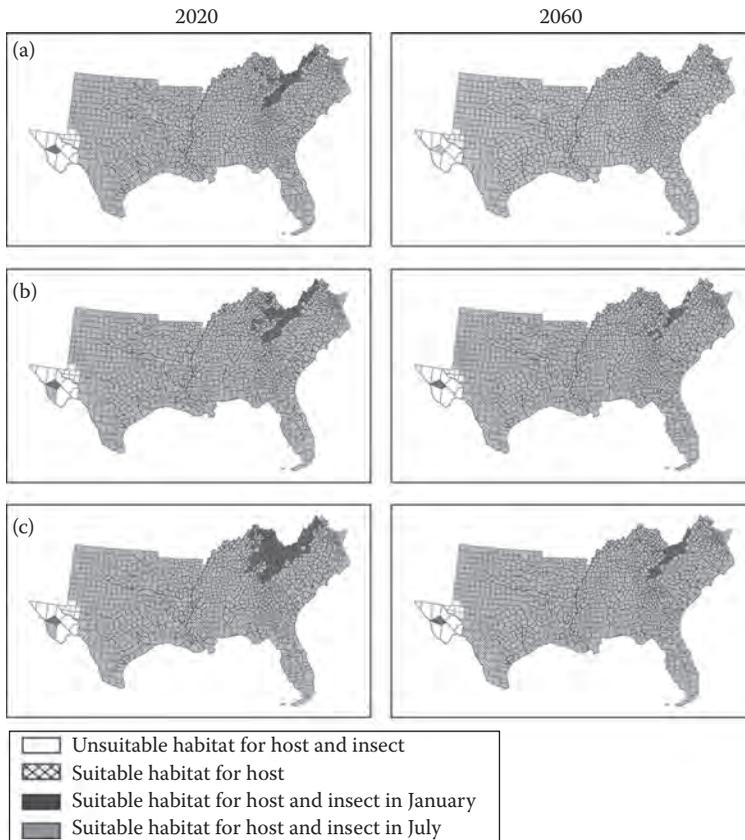


FIGURE 6.9 Suitable habitat for the emerald ash borer in January and July (2020 and 2060) in the Southern United States, projected using an ensemble of selected general circulation models (CGCM3, GFDL, CCSM3, and HadCM3) and assuming three Intergovernmental Panel on Climate Change (2007) emissions storylines; (a) A1B, (b) A2, and (c) B1 (see text and Chapter 2 for storyline descriptions).

plantations and windbreaks (Eliason and McCullough 1995; Wilson and Schmiege 1975). This weevil has also been implicated as a contributor to red pine (*Pinus resinosa*) decline, and was among other insects found to be significantly more abundant in declining stands than in healthy red pine (Klepzig et al. 1991).

Impacts of climate change—Excessive rainfall can cause mortality of pine root collar weevil pupae. Pruning lower branches of host trees can also expose larvae and adults to higher temperatures, causing disruptions in developmental activities (Wilson and Schmiege 1975). Temperatures $<8.5^{\circ}\text{C}$ during the egg developmental period are insufficient to support increases in weevil populations. Therefore, future change in temperatures will potentially influence the northern extent of the weevil's range. Egg development is very sensitive to temperature changes $<12^{\circ}\text{C}$, but only slightly affected by temperature $\geq 18^{\circ}\text{C}$. A prolonged cooling of the root collar area can greatly influence egg development. At about 18°C , a temperature increase of 1°C decreases the incubation period by about a day (Wilson and Millers 1982). Hotter and wetter future conditions would disrupt egg development and spread of infestation. The projection maps indicate no suitable habitat for the pine root collar weevil because red-pine distribution in the South is limited to Virginia and North Carolina (Figure 6.10).

Management—Currently, several management practices can help reduce the likelihood of pine root collar weevil outbreaks. Some management practices suggested by Wilson and Millers (1982) are to (1) plant highly resistant pine species in areas with reported incidences of weevils, (2) avoid

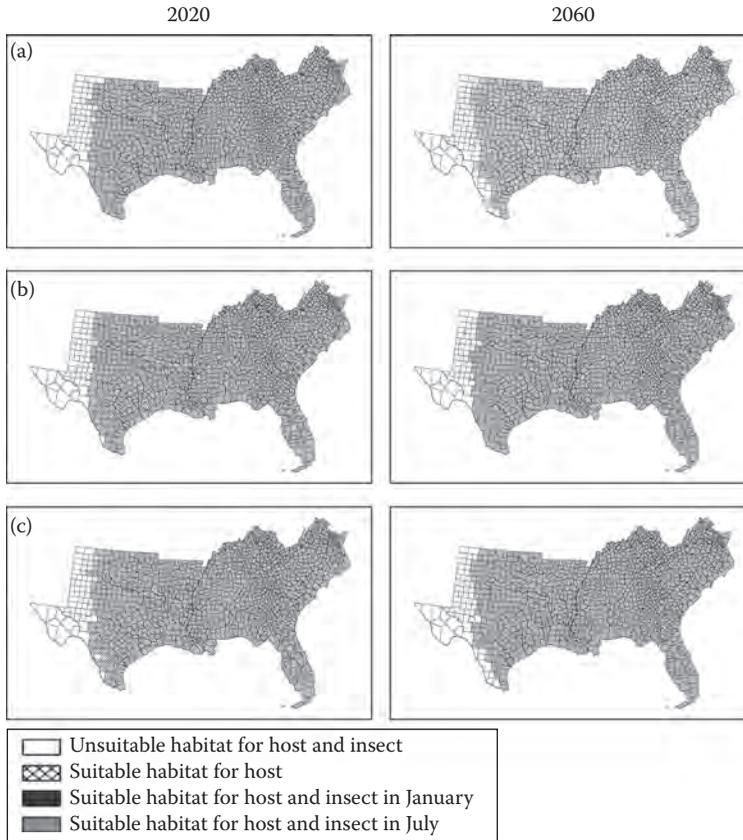


FIGURE 6.10 Suitable habitat for the pine root collar weevil in January and July (2020 and 2060) in the Southern United States, projected using an ensemble of selected general circulation models (CGCM3, GFDL, CCSM3, and HadCM3) and assuming three Intergovernmental Panel on Climate Change (2007) emissions storylines; (a) A1B, (b) A2, and (c) B1 (see text and Chapter 2 for storyline descriptions).

planting mixtures of susceptible and resistant species, (3) cut and destroy the root collars of open-grown, older susceptible pines before planting new ones, (4) maintain a fully stocked stand, because closed stands retard buildup of weevil populations, and (5) prune the lower 2–4 feet of branches from standing pines and rake away the duff beneath infested trees, thereby hindering the normal daytime activities of the adults by depriving them of a cool, shaded hiding place. Planting a resistant species is important for effective management. Pitch pine (*Pinus rigida*) is one of the least susceptible hosts. Eastern white pine (*Pinus strobus*) is rarely attacked unless adjacent to or planted with the highly susceptible species (Wilson and Schmiege 1975).

SIREX WOODWASP

The Sirex woodwasp is native to Europe, Asia, and northern Africa. It was first discovered in North America in 2004 (Hoebeke et al. 2005); its first known establishment in North America was reported in the Canadian province of Ontario and across the U.S. border in New York in late autumn of 2005 (Rabaglia and Lewis 2006). In the southern hemisphere, the species has caused significant mortality, especially in Monterey pine (*Pinus radiata*) and loblolly pine (*Pinus taeda*) plantations (Hopkins et al. 2008; Rabaglia and Lewis 2006). Slash pine (*Pinus elliottii*), shortleaf pine (*Pinus echinata*), ponderosa pine (*Pinus ponderosa*), lodgepole pine (*Pinus contorta*), and jack pine (*Pinus*

banksiana) are also susceptible (Haugen and Hoebeke 2005; Rabaglia and Lewis 2006). The estimated annual value of southern softwood logs, pulpwood, lumber, and veneer is >\$8 billion. If the level of attacks by *Sirex* woodwasps is as aggressive as in South America and Australia, Duerr and Mistretta (in press) speculated that many important southern pine species could be vulnerable to high losses.

Sirex woodwasps usually complete a single generation per year; however, it could take up to two years for them to develop fully in the colder parts of their range. Adults emerge from July to September, with the peak around mid-August (Pollard et al. 2006). The symbiotic fungus (*Amylostereum areolatum*) that is vectored by *Sirex* grows rapidly and excretes wood-digesting enzymes into the sapwood (Haugen and Hoebeke 2005), causing foliage to wilt and yellow and disrupting water movement, which often results in the death of the host (Haugen and Hoebeke 2005; International Union for Conservation of Nature Invasive Species Group 2007b).

Impacts of climate change—Loblolly pine, a major pine species in the South, would be a suitable host if *Sirex* woodwasps were to move into the South (Duerr and Mistretta in press). Although human-aided infestations or natural spread could introduce *Sirex* woodwasps into southern forests within the next 50 years, potentially resulting in significant ecological and economic losses, the complexity of southern forests (their mixed stands, high biodiversity, and abundance of possible competitors, predators, and parasitoids) would tend to reduce the level of vulnerability compared to areas where damages have been reported in monoculture pine plantations (Dodds et al. 2007). However, the recent higher frequency of tropical storms and hurricanes in the South has left some forests damaged and vulnerable to *Sirex* infestations, increasing the risk to economic vitality of softwoods (Duerr and Mistretta in press). Rabaglia and Lewis (2006) predicted that the spread of *Sirex* could take 42 years to reach the Gulf of Mexico, but their prediction does not appear to have considered the impact of projected climate change.

Projections under the B1 storyline (Figure 6.11) indicate a larger suitable July habitat for *Sirex* woodwasp by 2020 compared to any other storylines and a significant reduction in potential suitable July habitat under A1B storyline by 2060. Expansion and northward shift of suitable January habitat are likely under the A2 storyline by 2060.

Management—The management approaches currently available for *Sirex* woodwasps include prevention and suppression techniques such as thinning to increase growth and vigor (Haugen et al. 1990). Destruction of infested trees can also lower abundance of *Sirex* woodwasps in newly infested areas (Hurley et al. 2007). Biological control agents, including a parasitic nematode (*Deladenus siricidicola*) that infests *Sirex* woodwasp larvae and ultimately sterilizes adult females, have been successful in other countries. Successful evaluation of this nematode for use in U.S. forests should provide a potential control option for southern landowners and land managers against future threats (Fernández-Arhex and Corley 2005; Haugen and Hoebeke 2005). However, the existence in the South of numerous competitors (such as bark beetles, wood borers, and fungi) for dead and dying pines, may very well complicate or even preclude the establishment of *Sirex*, the use of biological control to manage it, or both.

OUTLOOK FOR NONNATIVE PLANTS

GARLIC MUSTARD

Introduced from Europe in the 1800s initially for medicinal purpose, garlic mustard is a cool-season biennial forb that grows often in colonies in a variety of habitats. Seeds can be dispersed by humans, animals, and water and can remain dormant up to six years. Garlic mustard produces allelopathic chemicals that can inhibit the regeneration and growth of nearby native plants.

Impacts of climate change—By 2020, the habitat range for garlic mustard in the South is projected to be largest under the A2 storyline, with ranges very similar for A1B and B1 (Figure 6.12). The projected habitat is expected to shift northward by 2060 under all three storylines, with the largest shift

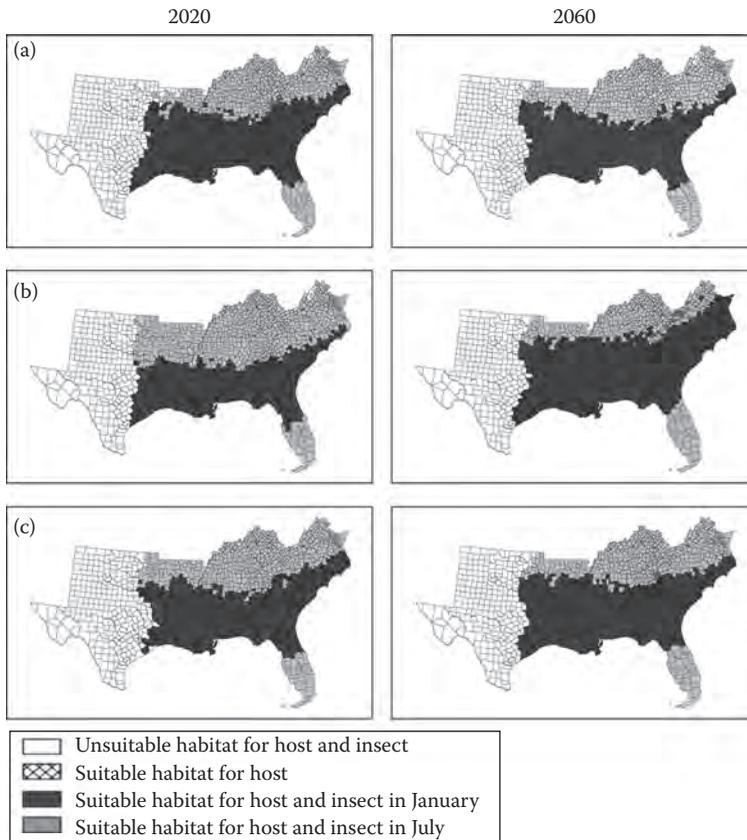


FIGURE 6.11 Suitable habitat for the *Sirex* woodwasp in January and July (2020 and 2060) in the Southern United States, projected using an ensemble of selected general circulation models (CGCM3, GFDL, CCSM3, and HadCM3) and assuming three Intergovernmental Panel on Climate Change (2007) emissions storylines; (a) A1B, (b) A2, and (c) B1 (see text and Chapter 2 for storyline descriptions).

under A1B and smallest under B1. Because garlic mustard is a cool-season forb, the northward shift is not surprising with warmer temperatures, especially at the southern edge of its range.

Management—The current distribution of garlic mustard is mainly concentrated in the northern portions (Kentucky, Tennessee, and Virginia) of the region. Based on the predicted major northward shift under future climate predictions, a regionally coordinated approach that includes early detection and rapid response to newly established populations could be the key to preventing the southward spread of this species. Management strategies such as minimizing disturbances near known infestations, removing newly infested populations, and minimizing seed dispersal by equipment and vehicles could greatly mitigate the spread of garlic mustard.

COGONGRASS

Introduced from Asia in the early 1900s initially for soil erosion control, cogongrass is now widely distributed in Africa, Australia, southern Asia, the Pacific Islands, southern Europe, the Mediterranean, the Middle East, Argentina, Chile, Colombia, the Caribbean, and the Southern United States. It is a highly aggressive, colony-forming perennial grass that invades a wide range of habitats and can cause widespread fires. Worldwide, cogongrass is most invasive in wet tropical and subtropical areas that receive 750–5000 mm of rainfall annually. It spreads by both seed and rhizomes, can tolerate hot weather, but is sensitive to cold.

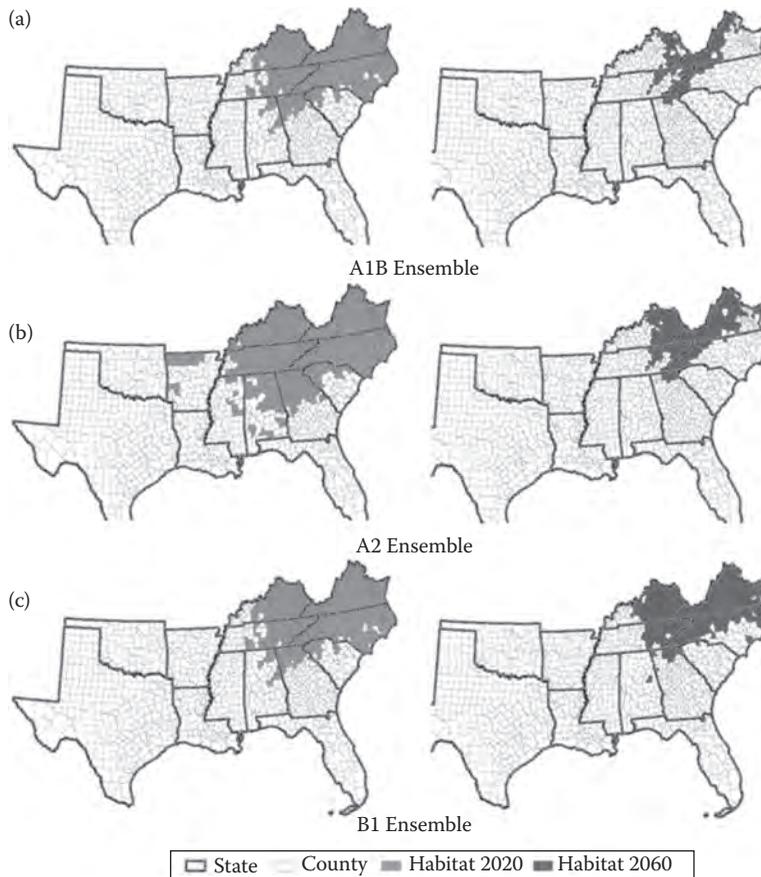


FIGURE 6.12 Suitable habitat for garlic mustard in 2020 and 2060 in the Southern United States, projected using an ensemble of selected general circulation models (CGCM3, GFDL, CCSM3, and HadCM3) and assuming three Intergovernmental Panel on Climate Change (2007) emissions storylines; (a) A1B, (b) A2, and (c) B1 (see text and Chapter 2 for storyline descriptions).

Impacts of climate change—In general, cogongrass is favored by a warming climate. All three climate change storylines predict that potential habitat for cogongrass will cover the majority of the South in 2020 (Figure 6.13); exceptions are western Texas (constrained by low precipitation) and southern Florida (constrained by high summer temperatures). Predicted potential cogongrass habitats in 2060 are very similar to the 2020 predictions, with only minor discrepancies at the western and southern edges of the range.

Management—Cogongrass is currently widely distributed in the lower portion of the South. The predicted climate change would favor its further invasion into the upper portion, making the management of this species more difficult. A complete eradication at this stage is nearly impossible. However, common management practices, such as practicing proper equipment sanitation and raising public awareness, would greatly assist the mitigation of this invasive grass species.

CHINESE PRIVET

Introduced from China in the early 1800s as an ornamental plant, the Chinese privet reproduces and colonizes through both sprouts and seeds. It often grows to form dense thickets thus shading and outcompeting many native species with severe ecological and economic consequences.

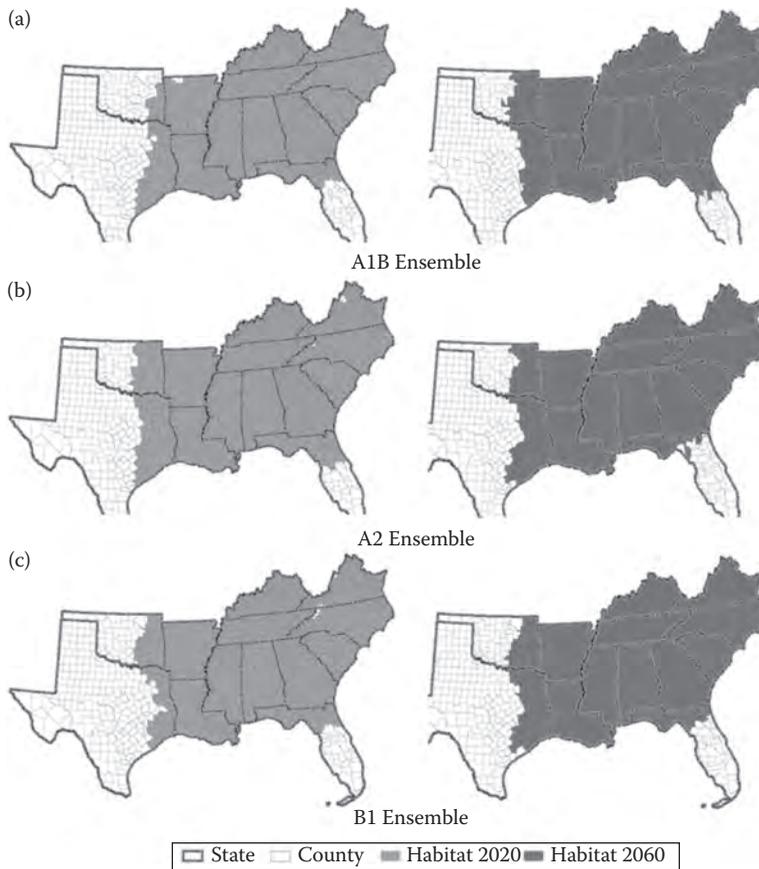


FIGURE 6.13 Suitable habitat for cogongrass in 2020 and 2060 in the Southern United States, projected using an ensemble of selected general circulation models (CGCM3, GFDL, CCSM3, and HadCM3) and assuming three Intergovernmental Panel on Climate Change (2007) emissions storylines; (a) A1B, (b) A2, and (c) B1 (see text and Chapter 2 for storyline descriptions).

Impacts of climate change—The potential habitat for Chinese privet is predicted for all of the southern states except Louisiana and Oklahoma in 2020 based on the A2 ensemble climate change storyline (Figure 6.14), and less so under B1. The A1B-based storyline predicts the fewest suitable areas. As with garlic mustard, prominent northward shift is predicted from 2020 to 2060. By 2060, the suitable Chinese privet habitat is predicted to be concentrated in the upper portions of the South (Kentucky, Virginia, Tennessee, and North Carolina).

Management—Chinese privet currently has a wide distribution across the entire South. Although higher-latitude regions may have increased the risk of Chinese privet invasion, the threat from this species could be alleviated with climate change, especially in the lower portions of the South. Management practices such as prevention of seed spread and dispersal and early detection/removal can mitigate the impacts of this invasive shrub species.

KUDZU

A high-climbing perennial vine initially introduced from Japan in 1876 (and later from China, possibly multiple times), kudzu has been a strong symbol of plant invasions in the United States, especially in the South. The species has been mostly used as a medicinal and is regarded as having

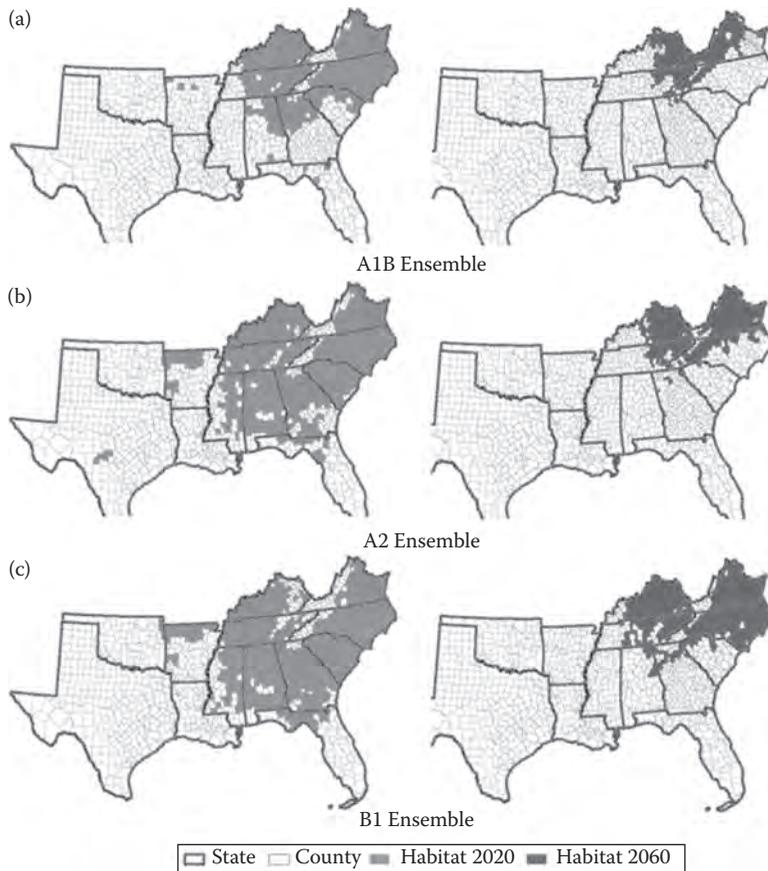


FIGURE 6.14 Suitable habitat for Chinese privet in 2020 and 2060 in the Southern United States, projected using an ensemble of selected general circulation models (CGCM3, GFDL, CCSM3, and HadCM3) and assuming three Intergovernmental Panel on Climate Change (2007) emissions storylines; (a) A1B, (b) A2, and (c) B1 (see text and Chapter 2 for storyline descriptions).

great economic values in its native lands (Li et al. 2011). In countries where it has been introduced, such as the United States, it was first promoted as an ornamental plant, then for erosion control, forage for domestic animals, and soil improvement. However, with time, the species has escaped and now grows in extensive habitats and high abundance. Although it grows over the tops of native forest species, thereby killing them through shading, it is mostly found along the roads, forest edges, and internal forests that have been disturbed. Many treatment methods have been attempted. Some control methods are available and effective but very costly.

Impacts of climate change—For 2020, potential kudzu habitats are predicted to be similar under the A2 and B1 storylines; the A1B predicts the least coverage, mainly along its western edge. A notable shift to the northeast is likely by 2060 with the smallest shift predicted under B1 (Figure 6.15).

Management—Kudzu is widely distributed across the entire South and in lower portions farther north, where the risk of invasion would increase as temperatures increase. In the South, the threat from this invasive vine at the western and southern edges of its range (especially in Texas, Oklahoma, Louisiana, and Florida) could be alleviated; for the western states, the main reason for kudzu decline would be the predicted decrease in precipitation. Management practices such as herbicide applications, grazing, and prescribed fire can help eradicate and contain the kudzu invasion.

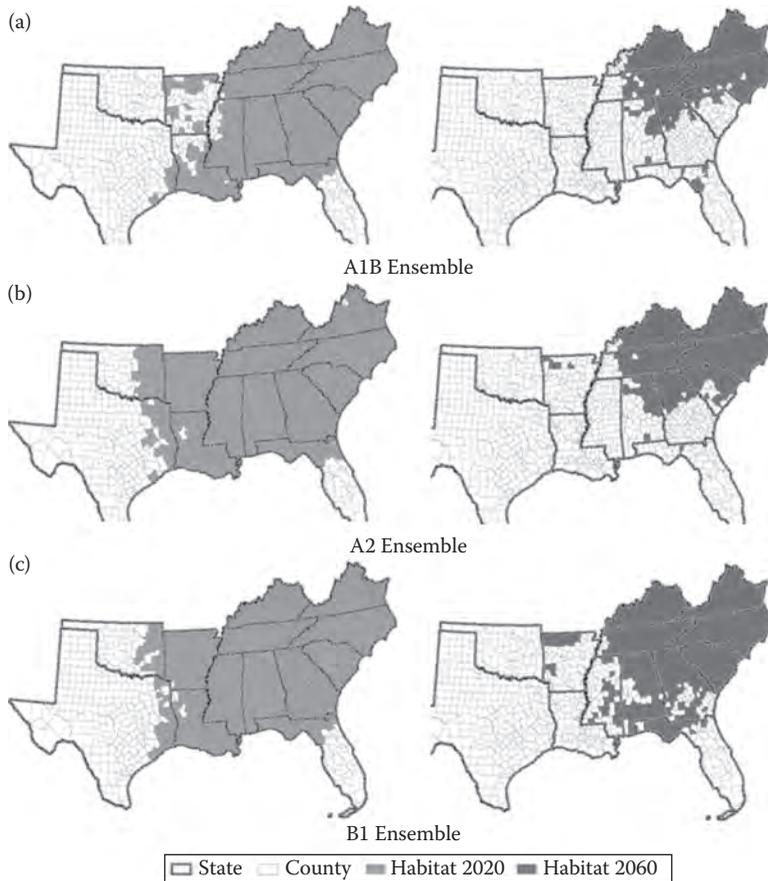


FIGURE 6.15 Suitable habitat for kudzu in 2020 and 2060 in the Southern United States, projected using an ensemble of selected general circulation models (CGCM3, GFDL, CCSM3, and HadCM3) and assuming three Intergovernmental Panel on Climate Change (2007) emissions storylines; (a) A1B, (b) A2, and (c) B1 (see text and Chapter 2 for storyline descriptions).

CHINESE TALLOWTREE

Introduced from China in 1700s as an ornamental plant, Chinese tallowtree is found throughout southeastern Asia (Pattison and Mack 2009). As with kudzu, the species has always been considered as having high economic values for production of “tallow oil” and thus has been widely planted in its native lands (Zhang and Lin 1991). In the lands where it has been introduced, tallowtree grows in relatively wet areas although it can grow well in uplands if the infestation becomes large enough. It has also been shown to have allelopathic effects on many native species. Chinese tallowtree is a serious threat because of its ability to invade high quality, undisturbed forests.

Impacts of climate change—Unlike the other species discussed, the habitat for Chinese tallowtree is predicted to concentrate in the central areas of the South in 2020, with the widest coverage to be found under the A2 storyline. A significant northward shift is predicted from 2020 to 2060 (Figure 6.16), as well as a shift eastward, especially dramatic under the A2 storyline.

Management—The current distribution of Chinese tallowtree is mainly in the coastal areas of the South. Climate change would likely facilitate its range into the central portion by 2020 and into the central–northern portion by 2060, rendering the currently invaded areas as unfavorable. From a management prospective, climate change can effectively confine Chinese tallowtree to a

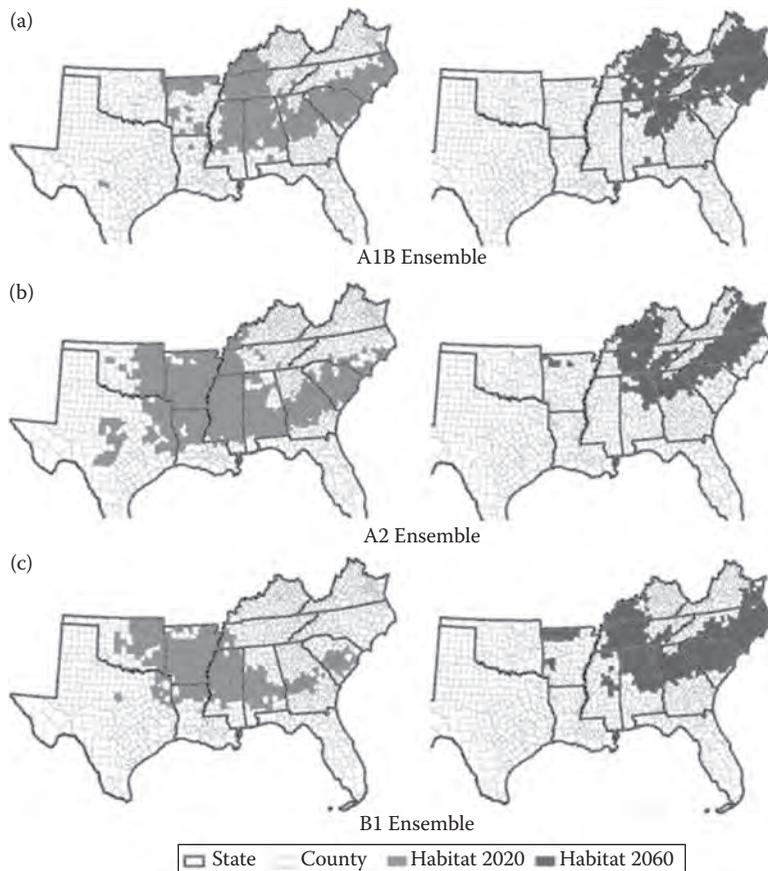


FIGURE 6.16 Suitable habitat for Chinese tallowtree in 2020 and 2060 in the Southern United States, projected using an ensemble of selected general circulation models (CGCM3, GFDL, CCSM3, and HadCM3) and assuming three Intergovernmental Panel on Climate Change (2007) emissions storylines; (a) A1B, (b) A2, and (c) B1 (see text and Chapter 2 for storyline descriptions).

small area if the species can be prevented from spreading northward. As with other species, removing new infestations and minimizing seed dispersal can greatly mitigate the spread of this invasive tree species.

POTENTIAL HABITATS

Our projections show that all five species exhibited some degree of northward (or poleward) shift in latitude in 2020 and even more in 2060. Clear indications of sensitivity to climate change are the variability, both in the species' distribution shifts and in the two forecasting periods (ending in 2020 and 2060). For example, for projections in 2020, garlic mustard exhibited a small northward shift in its ranges under the A2 storyline; however, in the 2060 projections, there was a greater shift under A1B than under B1. In 2020, cogongrass also shifted northward under both A1B and B1 but the northward shift in 2060 was smaller. In 2020, Chinese privet showed a similar northward shift under both storylines; and, by 2060, it had a similar shift to garlic mustard. For kudzu, in 2020 and under A2, the northward shift was small but its western edge shifted eastward under A1B; similarly, in 2060, kudzu exhibited trends similar to garlic mustard. Of the five species, Chinese tallow tree exhibited the smallest northward shift in both 2020 and 2060 although under A1B; its western edge also shifted eastward.

MITIGATION STRATEGIES

In the past, short-term weather variability has created extreme events such as droughts, heavy storms, flooding, and wildfires with long-term devastating impacts on forest ecosystems, particularly in the United States. Extreme droughts often lead to forest fire, an important forest disturbance linked to a warming climate (Westerling et al. 2006), in a cycle of events that can reduce the resistance of surviving trees to attacks by bark beetles and other insects, subsequently increasing fuel loads from dead tree hosts and increasing the frequency of forest fires. Integrated management strategies are preferred for mitigating losses from insect attacks such as those caused by southern pine beetles (Billings et al. 2004). Typically, the approach involves prevention, detection, and control. Preventive measures may include thinning to reduce stand density; removing infested, damaged, and weakened trees; and harvesting before trees become overly mature. Once an outbreak begins, the attention usually shifts to prompt detection and suppression of individual infestations thereby substantially reducing loss of forest resources (Clarke and Billings 2003). Replanting using seedling from a variety of resistant families would be an important component for long-term management of negatively impacted forest ecosystems.

Some recommendations are already in place to help land managers effectively prepare for climate change: (1) reduce current environmental stresses and increase resistance against climate-exacerbated disturbances such as wildfire or insect outbreaks by thinning and fuel abatement treatments at landscape scales and strategically placed area treatments; (2) apply early detection and rapid response systems to identify plant, animal, and ecosystem responses to climate change; (3) plan for higher elevation insect outbreaks, species mortality events, and altered fire regimes; (4) facilitate natural selection by enhancing disturbances that initiate increased seedling development and genetic mixing; (5) reduce homogeneity of stand structure and synchrony of disturbance patterns by promoting diverse age classes, species mixes, and genetic diversity; (6) cultivate an increased understanding of climate change; and (7) expand partnerships and capitalize on potential new ecosystem services opportunities under climate change; and (8) increase collaboration among federal agencies (U.S. Department of Agriculture Forest Service 2009).

Furthermore, the Southern Forest Futures Project, initiated by the Forest Service, is an ongoing effort geared toward in-depth documentation of potential future threats to southern forests, with the goal of developing products that will guide land managers in implementing appropriate adaptation and mitigation management strategies.

WHAT CAN THE PAST TELL US ABOUT MANAGING IN THE FUTURE?

We expect that most of the measures mentioned above and other management strategies currently available for addressing short-term climate variability impacts will be useful in managing long-term climate change impacts, perhaps with some modifications. However, maintaining a healthy forest into the next century will require adaptation and mitigation strategies as well as adequate planning to identify vulnerabilities. Because the scope of likely impacts is so broad, managing for them all within the forest ecosystem would be impossible. However, implementing management options that enable natural adaptation of the forests to climate change will be critical.

Future adaptation strategies focus on reducing the risk of negative impacts while capitalizing on any potential opportunities brought by climate change. Mitigation strategies can focus on the pace and extent of climate change impacts and increasing carbon sequestration through healthy forests. The overall goal would involve short-term management options that forestall impacts and long-term options for anticipating and managing changing forest ecosystem.

WHAT CAN BE DONE NOW?

Some existing strategies could be effective for managing short-term, climate variability-related events and disturbances, while others would be more useful for managing long-term climate change

effects. Overall, early planning of management options offers a good opportunity to identify vulnerabilities, the magnitude and timing (now or later) of potential impacts, the likelihood of impacts and confidence in estimates, and potential strategies for adaptation and mitigation (U.S. Department of Agriculture Forest Service 2009).

DISCUSSION

In this chapter, we defined vulnerability as the susceptibility of a forest ecosystem to stress or disturbance that invokes an undesirable change in structure or function. Vulnerability of an ecosystem to climate change is a function of (1) exposure, or the degree to which the ecosystem has been exposed to climate change and variability, (2) sensitivity to these changes, and (3) adaptive capacity of the ecosystem to respond to these changes. Therefore, developing strategies to help the ecosystem adapt is critically important (Intergovernmental Panel on Climate Change 2007; Williams et al. 2008).

The most vulnerable ecosystems will have low resistance and resilience to climate change, because they cannot cope with short- and long-term exposures to change, are highly sensitive to change, and lack adaptive capacity to withstand change. Vulnerability assessments generally focus on quantifying relative exposures (such as short- or long-term changes in precipitation and temperature) and sensitivity to climate changes (determined by ecological, genetic, and physiological traits), as well as the adaptive capacity of species or ecosystems, which can help direct or prioritize management efforts toward addressing future threats (DeWan et al. 2010).

Species or ecosystems with a high degree of adaptive capacity are subject to fewer risks from climate change impacts compared to species or ecosystems with relatively low adaptive capacity (DeWan et al. 2010). However, most species may lack the capacity to adapt given the rate and magnitude of projected changes (Bradshaw and Holzapfel 2006; Williams et al. 2008). Evidence suggests that entire species tend to shift their geographical distributions in response to climate change, rather than undergoing the major evolutionary adaptations needed for conservation of the original range (Parmesan 2006; DeWan et al. 2010).

As shown in [Table 6.3](#), several physiological and life-history traits can influence species' vulnerability and response to climate change disturbances (DeWan et al. 2010). Therefore, the species most at risk tend to have a narrow range of physiological tolerance to factors such as temperature, water availability, and fire; low genetic variability; long generation times and long time to sexual maturity; specialized requirements (hosts, vector, symbionts, or habitat) that may be restricted; poor dispersal (adaptive capability); and a narrow geographic range.

Conversely, species that are least at risk to climate change are those with a broad range of physiological tolerance to environmental factors including temperature, precipitation, and fire. Species with a high degree of phenotypic plasticity and genetic variability, short generation time (rapid life cycles), and short time to reach sexual maturity, high fecundity, low specificity in habitat requirements (such as food sources and sites), good dispersal capability, and broad geographic ranges will be the least at risk (Steffen et al. 2009).

We anticipate an increase in the level of insect attacks (both native and nonnative) in response to climate change. We also anticipate that projected increases in temperature would increase the frequency of drought stress on host trees. Some southern tree species are likely to shift or shrink in geographical range. Cascading effects could include a more vulnerable forest ecosystem with fewer stands, causing reduced value and insufficient genetic diversity to withstand further changes. Current biological systems and social activities tied to the ecosystem could also become more vulnerable, and others could cease to exist in the future. Attacks of dead and dying host trees by invasive species have already resulted in the destruction of habitats, economic losses in the forest products sector, and the loss of trees in cities, towns, and natural areas that had once supported thriving forests.

TABLE 6.3
Parameters for Evaluating Adaptive Capability and Vulnerability
of Species to Climate Change

Vulnerability Parameter	Least at Risk	Most at Risk
Physiological tolerance	Broad	Narrow
Genetic variability	High	Low
Generation/maturity	Short	Long
Fecundity/sporulation	High	Low
Specialized requirements	No	Yes
Dispersal capability	Good	Poor
Geographic range	Broad	Narrow
Host diversity	Broad	Narrow

Source: Adapted from Steffen, W. et al. 2009. Australia's biodiversity and climate change: A strategic assessment of the vulnerability of Australia's biodiversity to climate change. A report to the Natural Resource Management Ministerial Council commissioned by the Australian government. CSIRO Publishing.

AREAS OF GREATEST CONCERN

Diseases and insects—Thus far, much of the emerald ash borer infestation has been concentrated in northeastern areas of the United States; however, in approximately 50 years (2060), southern forests are expected to lose millions of ash trees from emerald ash borer outbreaks triggered by projected extreme drought and rapid insect development. This could eventually shrink the range of ash trees in the region (Duerr and Mistretta in press).

The spread of the hemlock woolly adelgid (U.S. Department of Agriculture Forest Service 2011a) will likely continue from westward and southward. A warmer and drier climate would increase the likelihood of southern pine beetle activity across southern forests, especially in dense unthinned plantations and in pine acreage of highly susceptible species. A northward spread of southern pine beetles beyond the Appalachian-Cumberland highlands would likely increase and infestations would be more frequent.

Under a warmer and drier climate, the complex interaction between the redbay ambrosia beetle and fungal symbionts may be limited to the southern Coastal Plain (Koch and Smith 2008); however, it is possible that laurel wilt could spread to sassafras farther north since sassafras ranges extend to south Canada. Although redbay are restricted to certain parts of the southern geographic range, key areas with high concentrations of redbay have yet to be invaded, while others are imminently threatened (Koch and Smith 2008).

With the increasing possibility of *Sirex* woodwasp introduction in the South, the extent of climate change impacts on *Sirex* and host pine trees, which are uniformly distributed in the South, still remains uncertain. The likelihood that tropical storms and hurricanes will increase in the South may increase the likelihood of significant damage to southern forests, making them even more vulnerable to *Sirex* woodwasp attacks and infestations. According to Rabaglia and Lewis (2006), the spread of *Sirex* woodwasp would take 10 years to reach Virginia and 42 years to reach the Gulf of Mexico in Alabama, even without climate change.

With the current rate of spread, the gypsy moth could encounter lower concentrations of susceptible hosts as it moves south and west. Although overall forest susceptibility will likely decrease in most areas, the large distribution ranges of potential hosts could allow the spread to persist, raising the possibility of a significant widespread infestation in some southern landscapes within the next 50 years (Duerr and Mistretta in press).

Warmer or mild winters will likely extend the northward range of the pine root collar weevil. Because adults overwinter in the soil or in bark crevices before resuming activities in the spring, warmer winters would allow prolonged activity and an increase in the number of generations per year; at the same time, a decrease in precipitation may reduce the weevil's activity. Healthy forest stands are therefore critical for limiting infestations in the next 50 years. Inadequately managed forest stands will likely experience an increase in attacks and damage caused by the weevil.

Nonnative plants—Although the extent of northward (and sometimes eastward) shifts varies considerably among species and emissions storylines, all five species shift their distributions northward (poleward) in the 2020 and 2060 projections (Guo et al. 2012b). However, note that the vastly increased distributions and spread expected to result from climate change are not documented in this chapter because they are predicted to occur outside the Southern United States. In addition, the northward shifts of the five nonnative plants does not necessarily mean fewer threats to forest ecosystems, as other resident or newly introduced invasives would occupy these habitats.

Knowing where introduced species, especially those that are invasive, are spreading and at what pace is critical for invasive species management and conservation policy and planning. The five species used as examples of dominant invasives in the South for this chapter have continuously increased in occupied areas (Guo et al. 2006), causing drastic ecological and economical damage (Miller et al. in press). For forest and invasive species management, the areas that are subject to climate-induced introductions would experience fewer and less severe invasions if disturbances were kept to a minimum (Guo and Norman 2012). For the areas where a particular species retreats, the negative effects of the former resident could persist, thus requiring continued monitoring and management. Monitoring is also needed to update predictions about which species are likely to invade and at what rate, so that plans are in place to avoid potential damages.

Projections show that some species (such as cogongrass) that occur in much of Florida would have less coverage in 2020 and even less in 2060. In reality, speculating that an invasive species could completely emigrate from any area would be inaccurate, simply because many species are likely to find suitable microhabitats—somewhat similar to the northward migration during the glacier retreat when many species remained at lower latitudes but moved to higher elevations. This is a limitation of our modeling effort, as our projections do not capture the microenvironments that are still suitable for some species during their poleward migration or spread.

As with many studies using climate/land-use change projections for future species distribution forecasts, our projections also have several limitations. First, they do not account for possible biotic interactions (such as competition, predation, and transmission of diseases) among species in projected areas; that is, interactions with resident species and other new species that may invade. In reality, most species could actually spread to smaller areas than those projected because factors other than climate are also at work. Second, the climate variables used for native species are not definitive, simply because of the difficulty in identifying (without experimental support) which climate factor is critical (for temperature alone, options include average annual temperature, warmest/coldest monthly temperature, days when temperature is below a certain level, or the highest or lowest temperature). Third, the five species represent only a small fraction of broadly defined life/growth forms.

For these reasons, caution is advised when interpreting these results. What is needed is more information regarding how species might respond to climate change, in addition to information on how species respond to environmental factors such as temperature and moisture. This information combined with emerging techniques in modeling/simulation would be useful for evaluating responses of individual species (Albright et al. 2010) in an environment or interactions with other species in a given environment.

MANAGEMENT OPTIONS FOR REDUCING VULNERABILITY

Diseases and insects—An integrated approach for managing anticipated spread of invasive species (such as the redbay ambrosia beetle) would necessarily consider trends in human population growth

and recreational activities in areas where the beetles have recently become a major problem. A combination of elimination, proper disposal, and quarantine of infested plant materials is an effective management strategy, because employing most existing measures individually is inadequate against laurel wilt infections—though it might slow down the spread, it is impossible to cover such a large area. Results from fungicide applications have been promising, although they are limited to prevention of laurel wilt in high-value trees.

Most management options currently available for emerald ash borer are not very effective. However, future management efforts in response to climate change should include survey and quarantine of infested plant materials and sites, removal of weak and vulnerable ash trees, and establishment of “ash conservation areas” to protect small pockets of exceptional ash resource. Biological control methods using parasitoids of larvae and eggs combined with limited chemical treatments in conserved areas could play a useful role in reducing the spread of the emerald ash borer in the South.

Increased human populations could increase the spread of gypsy moth by increasing the number of parks and recreational activities. Therefore, continued vigilance and the use of early detection methods will be vital in reducing the risk of gypsy moth epidemics in the South. A combination of aerial applications of biological control agents, treatment with pesticides, deployment of pheromones traps, and continuation of programs such as “Slow the Spread” would help protect southern landscapes for several years.

Future management options for the hemlock woolly adelgid include restricting movement of infested plant materials to noninfested locations and planting new stands of stress-resistant cultivars. Hardwoods that are not susceptible to the hemlock woolly adelgid could replace hemlocks in places where the adelgid has caused widespread mortality.

Future outbreaks of southern pine beetles could be minimized through early detection and monitoring of new infestation spots, followed by timely control measures to suppress establishment of active spots. Lowering stand densities and thinning vulnerable pine stands would increase vigor and resiliency to southern pine beetle attacks.

Implementation of phytosanitary management practices would reduce the risk of pine root collar weevil infestations in vulnerable host stands. Measures include avoiding planting mixtures of susceptible and resistant species together, destroying the root collars of older susceptible pines, maintaining a fully stocked stand, pruning lower branches, and clearing beneath infested trees. These practices will likely reduce further spread of the pine collar weevil.

In addition to prevention and suppression management techniques to combat the threats posed by *Sirex* woodwasp, introducing biological enemies (such as parasitic nematodes) may become an effective tool for lowering the population below damaging threshold levels. Biological control methods will provide added options to landowners and land managers against future threats. Prevention measures such as thinning of susceptible or stressed pines would also increase overall stand vigor, and elimination of infested trees would help keep *Sirex* woodwasp populations low.

Restoring the less susceptible longleaf pine should be considered in areas where annosus root disease has caused significant damage to the forest stands. In response to likely shifts in geographic range from climate change, management options for fusiform rust would include deploying screened, resistant seedlings in areas with a history of high rust incidence, avoiding over-fertilization of seedlings in nurseries, and maintaining healthy saplings through effective silvicultural methods.

Some adaptive management strategies may be useful; examples include prescribed burning, restoring tree species, enabling migration of new tree species, instituting measures to enable migration for species that cannot migrate fast enough to keep up with climate change, and managing invasive species. However, these strategies are sometimes difficult to implement and limited in effectiveness (Frelich and Reich 2009). Overall, the forecasts and management options discussed in this chapter would help strengthen the resilience of southern forests in withstanding possible

adverse impacts of future climate change. Although several uncertainties are inherent in models and projections used in this chapter, we believe that optimizing current models and taking advantage of additional management options as new technologies and adaptive strategies emerge would do much to address the threats of the future.

Nonnative plants—Options to reduce the vulnerability of forests to nonnative invasive plants include implementing management strategies that minimize disturbances near known infestations, removing newly infested populations, and minimizing seed dispersal by equipment and vehicles. These strategies could greatly mitigate the spread of garlic mustard. Although a complete eradication of cogongrass is nearly impossible at this stage, common phytosanitary management practices and greater public awareness would significantly enhance mitigation efforts. Other management practices such as preventing seed spread and dispersal (e.g., by monitoring possible seed-carrying vehicles and travelers) and implementing early detection and removal measures can mitigate the impacts of invasive shrub species. Management practices that involve herbicides, grazing, and prescribed fire can help eradicate and contain kudzu invasions. Interestingly, climate change may serve as a natural management option by confining the spread of Chinese tallowtree to a smaller area and preventing or slowing its spread northward. As with other nonnative plant species, removing newly infested populations and the reducing seed dispersal would greatly mitigate the spread of this invasive tree species.

CURRENT KNOWLEDGE GAPS/AREAS OF FUTURE RESEARCH

The suitable habitat projections that were developed using the ensemble of selected general circulation models and the three emissions storylines (A1B, A2, and B1) have limitations. They lack several environmental parameters that could be critical in biological modeling: daily temperature parameters required for monitoring insect development or relative humidity for monitoring diseases such as fusiform rust.

Figure 6.17 shows the importance of daily data, including average, minimum, and maximum temperatures—in examining the potential number of insect generations (populations) through the season under different climate change storylines. Produced from a regression relationship between accumulated degree-days from the daily dynamic downscaling data and the monthly ensemble general circulation model data, this analysis emphasizes both the need for daily weather data and the importance of accumulated degree-days in predicting the potential suitable habitat for pests such as gypsy moths.

The critical gaps in our current knowledge of climate effects on various species that we have discussed in this chapter bring a high level of uncertainty to the projections of consequences for the incidence or severity of forest diseases, invasive insects, and nonnative plants. Often, limited information is available about the life cycle of an insect or pathogen, or about insect–fungal interactions, especially in situations where new interactions may develop.

Existing scenarios and attempts to model and predict outcomes that are based on a limited number of parameters (temperature or precipitation) are less valuable as decision-making tools than as a validation for continuing sound forest management practices to control the specific species discussed in this chapter, and as tools for exposing critical research needs. The information presented simply highlights the need for much more basic research on virtually all ramifications of climate change as it affects diseases, invasive insects, and nonnative plants, with the long-term goal of strengthening adaptation, restoration, and mitigation strategies, and incorporating them into management of southern forests (Guo and Norman 2012).

The management options presented in this chapter are specific to the individual representative species discussed. These options are not applicable to other species in the same class, because of differences in biology and unique interactions between species and their hosts. As such, they serve as examples of what is available and as guidance on what to consider when addressing nonnative and invasive species.

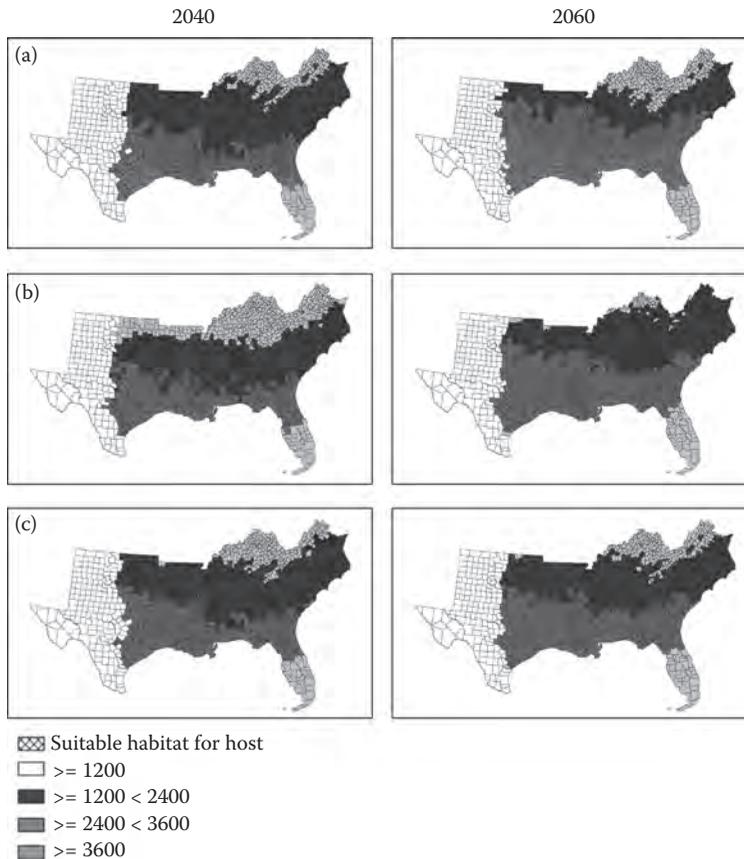


FIGURE 6.17 Accumulation of gypsy moth generations from March to August in 2020 and 2060, projected assuming a baseline count of 1200 per generation at 10°C and three Intergovernmental Panel on Climate Change (2007) emissions storylines; (a) A1B, (b) A2, and (c) B1 (see text and Chapter 2 for storyline descriptions). This figure was produced using a regression equation from the relationship between monthly average temperature and monthly accumulation of degree-days (days of sufficiently warm/cold temperatures needed for physiological processes such as egg hatching and first flight) based on the A2 (2010 data) and future temperature (2020 and 2060) projection data provided by Yongqiang Liu.

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