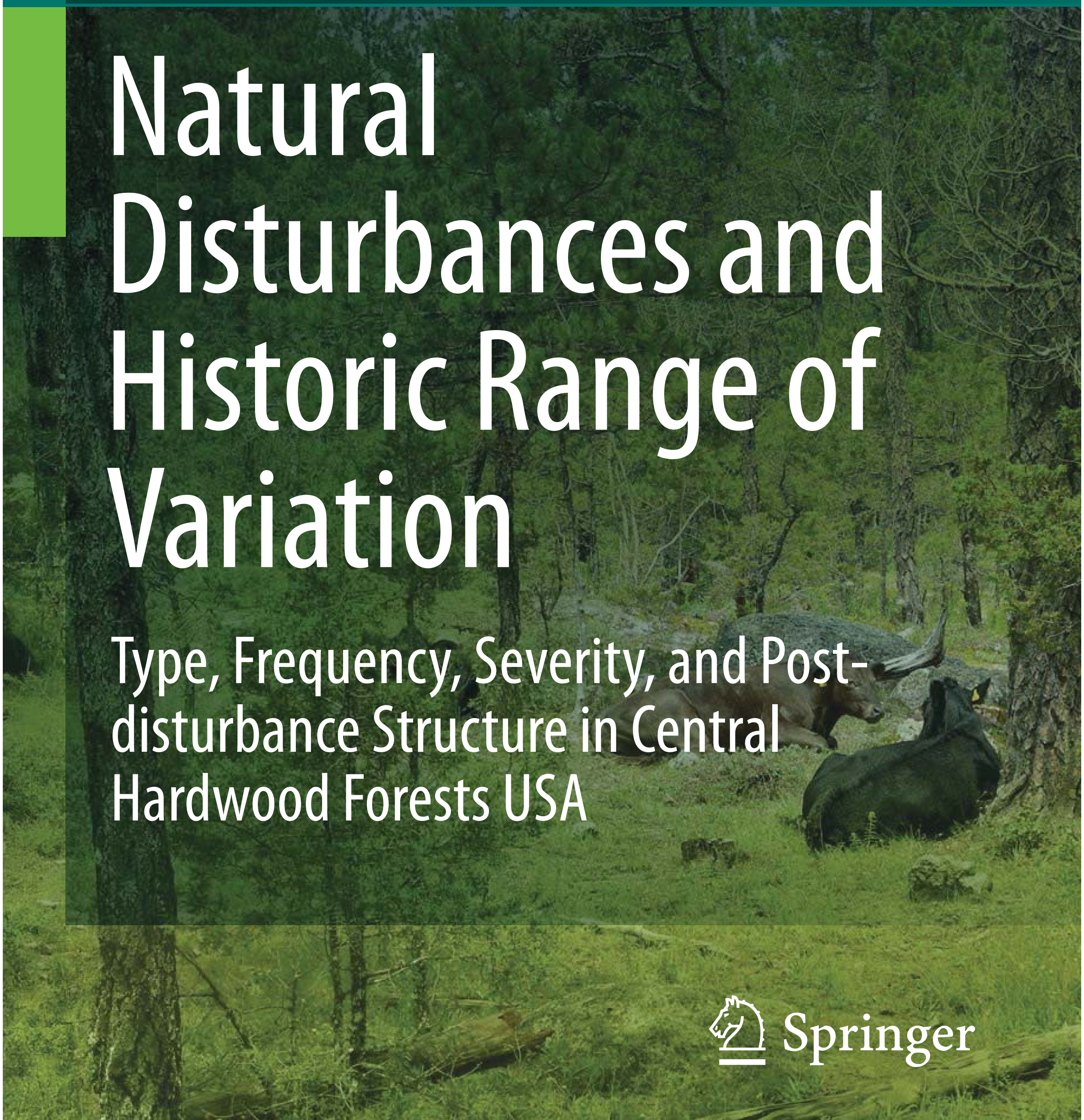
Managing Forest Ecosystems

Cathryn H. Greenberg
Beverly S. Collins *Editors*

Natural Disturbances and Historic Range of Variation

Type, Frequency, Severity, and Post-disturbance Structure in Central Hardwood Forests USA



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Aims & Scope

Well-managed forests and woodlands are a renewable resource, producing essential raw material with minimum waste and energy use. Rich in habitat and species diversity, forests may contribute to increased ecosystem stability. They can absorb the effects of unwanted deposition and other disturbances and protect neighbouring ecosystems by maintaining stable nutrient and energy cycles and by preventing soil degradation and erosion. They provide much-needed recreation and their continued existence contributes to stabilizing rural communities

Forests are managed for timber production and species, habitat and process conservation. A subtle shift from *multiple-use management* to *ecosystems management* is being observed and the new ecological perspective of *multi-functional forest management* is based on the principles of ecosystem diversity, stability and elasticity, and the dynamic equilibrium of primary and secondary production.

Making full use of new technology is one of the challenges facing forest management today. Resource information must be obtained with a limited budget. This requires better timing of resource assessment activities and improved use of multiple data sources. Sound ecosystems management, like any other management activity, relies on effective forecasting and operational control.

The aim of the book series ***Managing Forest Ecosystems*** is to present state-of-the-art research results relating to the practice of forest management. Contributions are solicited from prominent authors. Each reference book, monograph or proceedings volume will be focused to deal with a specific context. Typical issues of the series are: resource assessment techniques, evaluating sustainability for even-aged and uneven-aged forests, multi-objective management, predicting forest development, optimizing forest management, biodiversity management and monitoring, risk assessment and economic analysis.

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Preface

This edited volume addresses the historic range of variation (HRV) in types, frequencies, severities, and scales of natural disturbances, and how they create heterogeneous structure within upland hardwood forests of Central Hardwood Region (CHR). The idea for this book was partially in response to a new (2012) forest planning rule which requires national forests to be managed to sustain 'ecological integrity' and within the 'natural range of variation' of natural disturbances and vegetation structure. This new mandate has brought to the forefront discussions of HRV (e.g., what is it?) and whether natural disturbance regimes should be the primary guide to forest management on national forests and other public lands. Natural resource professionals often seek 'reference conditions,' based on HRV, for defining forest management and restoration objectives. A large body of literature addresses changes in forest structure after natural disturbance, but most studies are limited to a specific site, disturbance event, forest type, or geographic area. Several literature reviews address a single natural disturbance type within a limited geographic area (often not the CHR), but do not address others or how their importance may differ among ecoregions. Synthesizing information on HRV of natural disturbance types, and their impacts on forest structure, has been identified as a top synthesis need.

Historically, as they are today, natural (non-anthropogenic) disturbances were integral to shaping central hardwood forests and essential in maintaining diverse biotic communities. In addition to a 'background' of canopy gaps created by single tree mortality, wind, fire, ice, drought, insect pests, oak decline, floods, and landslides recurringly or episodically killed or damaged trees, at scales ranging from scattered, to small or large groups of trees, and across small to large areas. Additionally, some animals, such as beavers, elks, bisons, and perhaps passenger pigeons, functioned as keystone species by affecting forest structure and thus habitat availability for other wildlife species. Prehistoric anthropogenic disturbances – fire and clearing in particular – also influenced forest structure and composition throughout much of the CHR and therefore the distribution of disturbance-dependent wildlife species. The spatial extent, frequencies, and severities differed among these natural disturbance types and created mosaics and gradients of structural conditions and canopy openness within stands and across the landscape.

A full-day symposium, organized by the editors, at the 2014 Association of Southeastern Biologists conference in Spartanburg, South Carolina, was the basis for this book. Our goal was to present original scientific research and knowledge synthesis covering major natural disturbance types, with a focus on forest structure and implications for forest management. Chapters were written by respected experts on each topic with the goal of providing current, organized, and readily accessible information for the conservation community, land managers, scientists, students and educators, and others interested in how natural disturbances historically influenced the structure and composition of central hardwood forests and what that means for forest management today.

Chapters in this volume address questions sparked by debated and sometimes controversial goals and 'reference conditions' in forest management and restoration, such as the following: What was the historic distribution, scale, and frequency of different natural disturbances? What is the gradient of patch sizes or level of tree mortality conditions created by these disturbances? How do gradual disturbances such as oak decline, occurring over a long period of time and across a broad landscape, differ in effects from discrete disturbances such as tornadoes? How does topography influence disturbance regimes or impacts? How do native biotic (insects or fungi, keystone wildlife species) and abiotic (precipitation, drought, temperature, wind, and soil) agents interact to alter disturbance outcomes? What was the diversity of age classes and gradient of forest structure created by natural disturbances alone? How might disturbance-adapted plants and animals have fared in the hypothetical historic absence of anthropogenic disturbances? How might climate change alter disturbance regimes and structure of upland hardwood forests in the future? And finally, should, and how, can land managers manage these forests within the HRV of natural disturbance frequencies, spatial extents, and gradient of conditions they create?

We sincerely thank all those who encouraged and aided in the development of this book. Each chapter was peer-reviewed by at least two outside experts and both coeditors, and we thank these colleagues for their useful suggestions: Chris Asaro, Robert Askins, Francis Ashland, Bart Cattanach, Steven Croy, Kim Daehyun, Dianne DeSteven, Chris Fettig, Mark Harmon, Matthew Heller, Louis Iverson, John Kabrick, Tara Keyser, Scott Lecce, William MacDonald, Henry McNab, Manfred Mielke, Billy Minser, Scott Pearson, Duke Rankin, Jim Rentch, John Stanturf, Scott Stoleson, Ben Tanner, and Thomas Wentworth. We also thank the Association of Southeastern Biologists for allowing us to host a conference symposium on this important topic, and the National Forests of North Carolina for assistance with travel costs for speakers. We especially thank each author for contributing, and for timely chapter revisions, which made this book possible.

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Chapter 4
Southern Pine Beetles in Central Hardwood Forests: Frequency, Spatial Extent, and Changes to Forest Structure

John T. Nowak, Kier D. Klepzig, David R. Coyle, William A. Carothers, and Kamal J.K. Gandhi

Abstract The southern pine beetle (SPB) is a major disturbance in pine forests throughout the range of southern yellow pines, and is a significant influence on forests throughout several Central Hardwood Region (CHR) ecoregions. At endemic levels, SPB colonizes individual stressed or lightning-struck trees, acting as a natural thinning agent. During outbreaks, tree mortality from SPB may impact CHR forests by indirectly converting stands to other species types, or changing the stand age and structure. Southern pine beetle can also create disturbance in stands by causing mortality in large clusters of pine trees or by hastening the succession from pine-hardwood forests to late-successional forest by killing single or groups of overstory pine trees. Populations are cyclical and have traditionally impacted CHR forests every 7–25 years, depending on location. The most significant outbreaks in this region in the past 65 years occurred 1974–1976 and 1997–2003, with the most recent impacting more than 405,000 ha, and caused an estimated economic loss of more than $1 billion across six states. In this chapter we examine the spatial extent and frequency of SPB outbreaks in the CHR. We also discuss the severity of disturbance caused by SPB to forests in this region over the past 65 years; how this

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disturbance has altered the resultant forests; and the potential impacts of changes in climate and anthropogenic effects on preventing infestations and reducing levels of tree mortality attributed to SPB.

Keywords Southern pine beetle • Table mountain pine • Restoration • Southern pine beetle outbreak history • Shortleaf pine

4.1 Introduction

Southern pine beetle (*Dendroctonus frontalis*; SPB) (Fig. 4.1a) is a native bark beetle species (Coulson and Klepzig 2011) that has long been a significant disturbance factor across the Central Hardwood Region (CHR) and all of the southern USA. Trees most commonly used as hosts are loblolly pine (*Pinus taeda*) and shortleaf pine (*P. echinata*), but this insect can also colonize and kill all of the southern yellow pines (Fig. 4.1b). This insect has also been known to kill eastern white pine (*P. strobus*), Table Mountain pine (*P. pungens*), pitch pine (*P. rigida*), Virginia pine (*P. virginiana*), red spruce (*Picea rubens*), and others during significant SPB outbreaks (Chamberlin 1939). This insect can be found from New Jersey south to central Florida and west to eastern Texas. There are also disjunct populations in Arizona and Central America (Thatcher and Barry 1982; Clarke and Nowak 2009). Significant economic losses (an average of $43 million per year; Pye et al. 2011) and ecological impacts, including changes in the physical environment, hydrology, wildlife, forest structure, and more (Tchakerian and Coulson 2011), can occur during SPB outbreaks. In this chapter we examine the spatial extent, frequency, and severity of disturbance that this insect has caused to forests in the CHR over the past several decades and how this disturbance has altered these forests. We will also discuss silvicultural techniques (e.g., thinning to reduce basal area) used to prevent and mitigate the impacts of this insect (Fig. 4.1c).

4.2 SPB Biology

Southern pine beetle is a small bark beetle, ranging from 2.2 to 4 mm long (Chamberlin 1939; Coulson and Witter 1984). This tree-killing species relies on mass attack to overcome tree defenses. An individual SPB eating (or tunneling) its way into a host tree is initially met with an exudation of oleoresin. If enough resin flows from the initial attack wound, the attacking beetle may be ‘pitched out,’ or sealed in a globule of crystallized pine resin. If, however, the attacking female is able to continue tunneling to cambium, she releases an aggregation pheromone which attracts other beetles (including males which mate with the attacking females in ‘nuptial chambers’). In this way, a mass attack of a single tree is orchestrated through chemical communication (Klepzig and Hofstetter 2011). While the exact

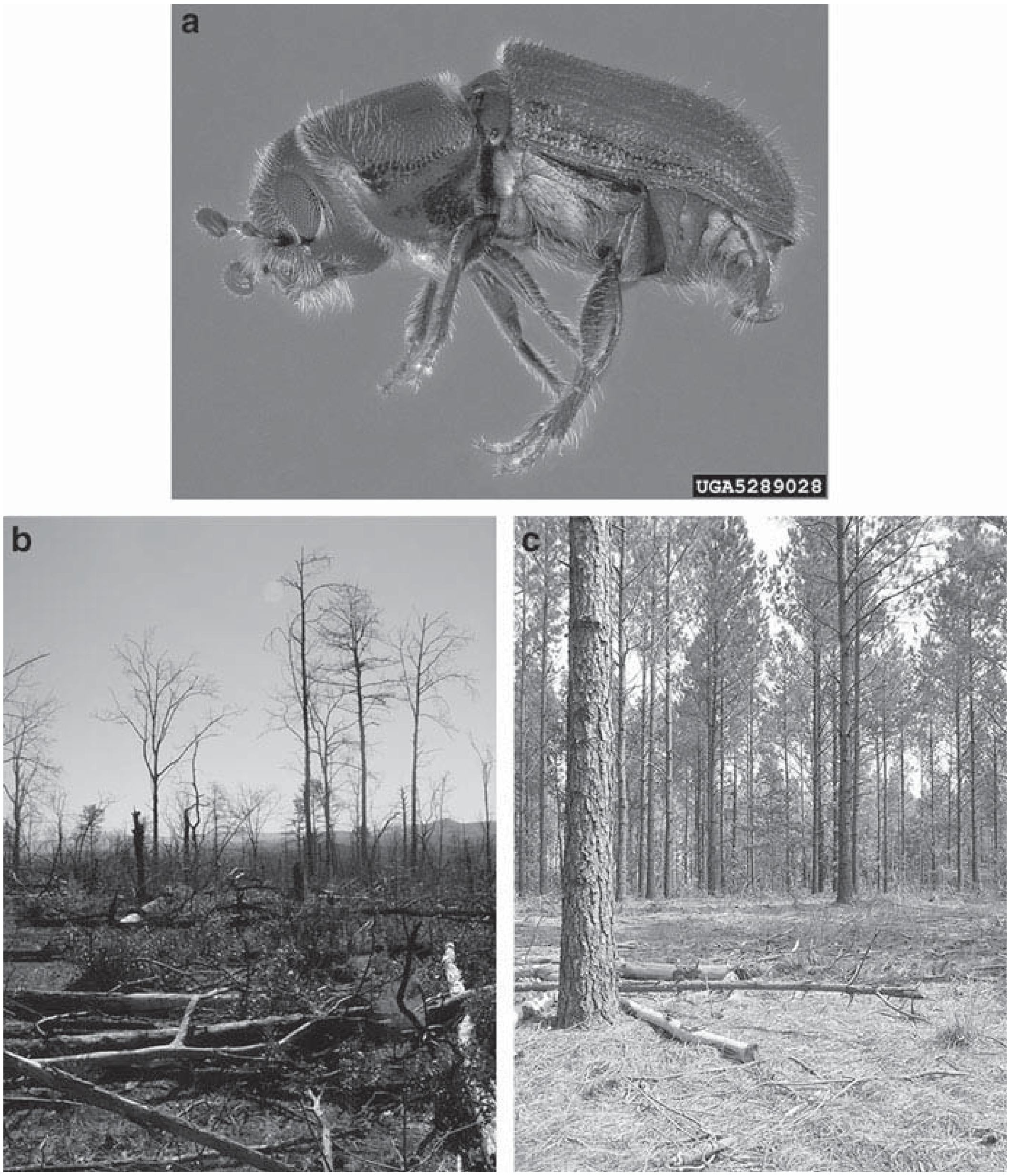


Fig. 4.1 (**a**) Adult southern pine beetle (Photo by Erich Vallery USDA Forest Service, www.bugwood.org); (**b**) shortleaf pine forest heavily impacted by southern pine beetle; and (**c**) and low hazard stand conditions – basal area lower than 21 m^2 per ha and open understory stand conditions

role of the fungi they carry continues to be debated, maternal females next tunnel through the phloem, laying eggs and inoculating the phloem and outer xylem with at least three different species of fungi. One of these fungi (a bluestain fungus, *Ophiostoma minus*) may aid the beetle in killing the tree, and the other two fungi are nutritional mutualists without which the hatching larvae cannot complete development (Klepzig and Hofstetter 2011).

Southern pine beetles also carry a number of other microbes and mites (Hofstetter 2011) and are prey for several associated predators and parasitoids (Reeve 2011). As a group, SPB generally first colonize single or small groups of two or three trees that have been struck by lightning (Lorio 1986) or weakened by some other factor. These small infestations may grow, but may not grow, into larger infestations or 'spots.' Most spots range in size from about 4–50 trees attacked, or about 14 trees attacked in an average spot (Ayres et al. 2011) (Fig. 4.1b). However, individual pines within hardwood stands can still be attacked during outbreak periods. Under most circumstance, tree crowns begin to fade to yellow within 2–4 weeks of attack, indicating the trees were successfully colonized and have died. Tree crowns then turn red and bark begins to slough. By this point SPB have completed their life cycle, emerged, and will have likely attacked adjacent trees. SPB can complete its life cycle within 30 days during summer months and have four to eight overlapping generations per year, depending on local climatic conditions (Hopkins 1909; Coulson and Witter 1984; Hain et al. 2011).

4.2.1 Susceptible Forest Conditions

Southern pine beetles prefer pine and pine-hardwood forests that are overstocked and more than 12–15 years old, although, pine stands with a greater hardwood component are considered to be less susceptible to SPB (Fettig et al. 2007). Natural and plantation forests that have high basal areas of greater than 28 m^2 per ha are considered highly susceptible to SPB. These higher basal area forest conditions could have been created by planting too high density if the stand was artificially regenerated or by the natural succession process with hardwoods growing up under the pine overstory. The standard recommendation for decreasing a forest's susceptibility to SPB is to reduce basal area to 18 m^2 per ha (Fettig et al. 2007; Nowak et al. 2008) (Fig. 4.1c). Trees in overstocked stands are generally less vigorous due to competition for light, water, and nutrients and therefore are considered to have less adequate defense (e.g., resin flow) than trees growing with less competition. The greater spacing between tree also lowers the effectiveness of SPB's pheromone communications system (Gara and Coster 1968; Fettig et al. 2007) due to more airflow and direct sunlight, which can make the pheromone plume more diffuse (Thistle et al. 2004) and less likely to be found by colonizing beetles. Individual tree vigor and more open stand structure are the two main reasons that thinning stands to reduce basal area is recommended to reduce levels of tree mortality attributed to SPB (Nebeker and Hodges 1983; Fettig et al. 2007; Nowak et al. 2008). Prescribed burning in both pine and pine-hardwood stands might also be a tool to create a more open stand and reduce understory competition (Nowak et al. 2008), which can significantly lower a forest's susceptibility to SPB. Thinning from below to retain an overstory of pine and pine-hardwood trees and maintaining low understory competition through chemical treatment or prescribed fire is the most effective way to protect a forest from SPB spot initiation and spot expansion.

4.3 Historical Outbreaks

The SPB has been documented as a pest since the 1750s and was formally described in 1868 by Zimmermann (1868). In the early 1900s more research was conducted on the biology, behavior, and impact of this insect (Hopkins 1909, 1911; Balch 1928; St. George and Beal 1929; St. George 1930) due to an increase in the understanding of its economic impact. Southern pine beetle populations are considered pulse eruptive (Berryman 1986) with periods of low endemic populations and periods of high epidemic population levels. Return intervals for epidemic populations vary within the CHR. Outbreaks occur every 7–12 years in the Piedmont and every 20–25 years in the more mountainous northern parts of the CHR, including the Blue Ridge Mountains and Southwestern Appalachians. At low populations, SPB are generally confined to weak and dying trees, particularly trees struck by lightning, but they can kill more vigorous trees during outbreaks. SPB outbreak periods usually last 1–3 years and populations fluctuate due to a variety of abiotic and biotic factors, such as climate, natural enemies, and host condition and abundance (Birt 2011).

Several SPB outbreaks have occurred in the CHR as noted in the literature since the 1850s. It should be noted that for this discussion on outbreak history we did not include SPB outbreak occurrences in the Piedmont because significant literature on that exists (Coulson and Klepzig 2011) and SPB activity in the Piedmont is more similar to the Coastal Plain than other regions of the CHR. Balch (1928) reported several instances of SPB outbreaks and severe pine mortality in the CHR, including 1890–1893 in Virginia and North Carolina, 1902–1905 in western North Carolina, and 1910–1915 in the Blue Ridge Mountains of Alabama, Georgia, North Carolina, South Carolina, Tennessee, and Virginia. In more recent times and since better records have been maintained, SPB has occurred in much of the CHR every decade since 1950, with the most significant outbreaks in the CHR occurring in 1974–1975 and 1997–2003 (Table 4.1). There has been a noticeable absence of even slight to moderate SPB activity in the CHR since 2002–2003. Concentrating on the mountainous regions of the CHR (Blue Ridge Mountains, Central Appalachians, Ridge and Valley, and Southwestern Appalachians), most of the SPB activity during the past 60 years occurred in the Blue Ridge Mountains and the Southwestern Appalachians in Alabama. There has been limited SPB activity in the Central Appalachians and Ridge and Valley only during the most severe outbreaks in the region (Table 4.1).

The most recent outbreak in the CHR occurred on state land, private farms, industry land, national forests, and national parks from 1997 to 2003 (Nowak et al. 2008), and was considered one of the most impactful SPB outbreaks of all time. This is outbreak was similar in scope and intensity to the 1974–1976 outbreak, and likely earlier outbreaks described by Balch (1928), which also impacted several ecoregions and states in the CHR (Table 4.1). However, record keeping for the past outbreaks was not as thorough as was done for the 1997–2003 outbreak. We will use the 1997–2003 outbreak as an example of the type of ecological and economic impact

Table 4.1 SPB activity in the CHR (excluding the Piedmont) 1950–2014

Ecoregion	State	No. years	Outbreak years	No. spots	Est. no. trees killed
Blue Ridge Mountains	GA	24 years	1959	Significant activity reported, but no quantification of activity	
			1968	4,539	29,276
			1974	2,204	102,051
			1988	330	176,760
			1995	264	202,658
			2000–2001	1,100	252,800
	NC	22 years	1953–1957	General reference to outbreak, but no quantification	
			1974–1975	713	58,589
			1990	427	114,265
			2000–2001	1,450	122,450
	SC	23 years	1968–1969	2,393	2,572 trees/ 405 ha surveyed
			1971–1976	4,337	10,750 trees/ 405 ha surveyed
	TN	14 years	1969–1972	5,236	177,304
			1975–1977	786	179,951
			2000–2001	982	150,554
Central Appalachians	KY	1 year	1997	32	3,130
	VA	3 years	1975–1976	101	3,682
Ridge and Valley	AL	4 years	1964	320	5,542
	GA	11 years	1987–1988	407	38,691
			2000	500	349,800
	VA	8 years	1975–1976	1,657	200,979
			1994	229	54,044
Southwestern Appalachians	AL	25 years	1953–1956	15,000+	No estimates of trees available
			1968–1969	3,861	16,343
			1996	1,151	94,382
			1999–2000	1,043	135,380
	KY	4 years	1975–1976	451	8,081
			2000–2001	4,200	500,000+

Data collected from USDA Forest Service Forest Health Protection Asheville Field Office reports
No. Years = number of years with any SPB activity; outbreak years = years of an outbreak with at least 1 spot per 405 ha host; No. Spots = number of SPB spots during outbreak years

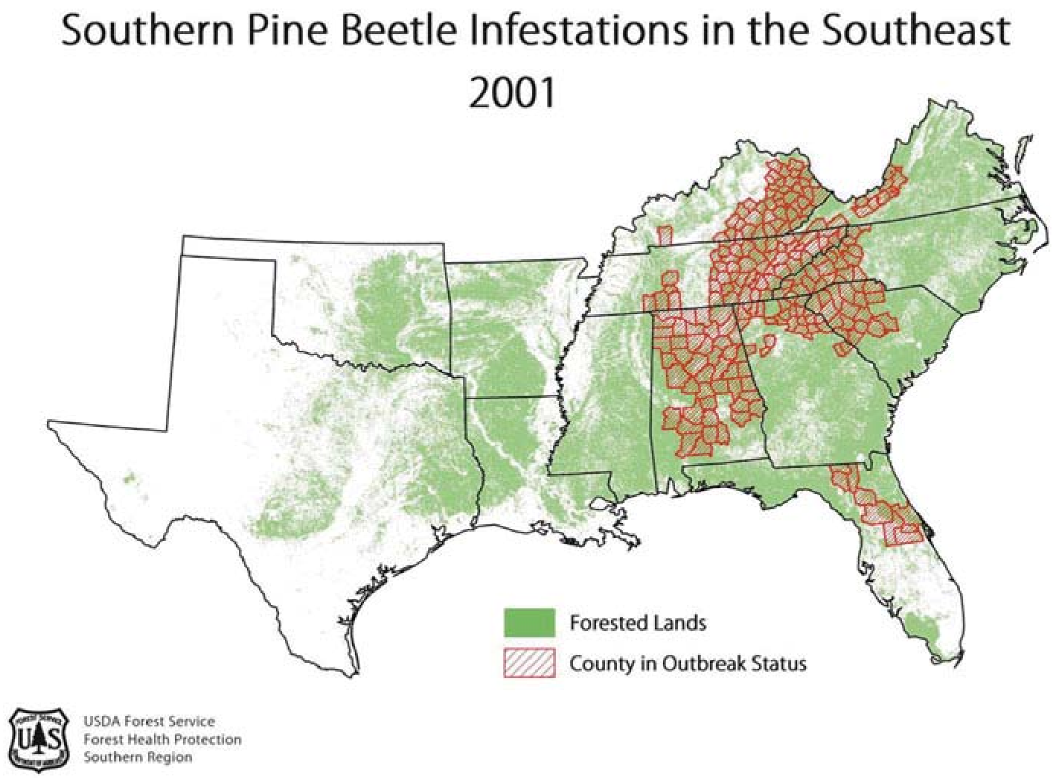


Fig. 4.2 Counties impacted by the 1997–2003 SPB outbreak

during severe SPB outbreaks in the CHR. The outbreak caused more than $1 billion in losses and impacted more than 405,000 ha across six states: Alabama, Georgia, Kentucky, North Carolina, South Carolina, and Tennessee (Fig. 4.2). This outbreak affected multiple ecoregions in the CHR, including the Piedmont, Blue Ridge Mountains, Ridge and Valley, Southwestern Appalachians, Central Appalachians, and Interior Plateau (for ecoregion map, see Greenberg et al. Chap. 1, Fig. 1.1).

Outbreak status is defined as one SPB spot per 405 ha of host type (Price et al. 1998) with high outbreak status at three spots per 405 ha of host type. During the 1997–2003 outbreak, multiple counties in multiple states exceed 20 SPB spots per 405 ha of host type. This level was rarely seen prior to this outbreak and is considered unprecedented although good records do not exist prior to 1990 (Pye et al. 2008). In 2001, 187 counties were in outbreak status and 126 exceeded the high outbreak level (Table 4.2). At the peak of the infestation in 2001, there were approximately 57,000 spots reported in 310 counties.

4.3.1 *Need for Restoration*

Widespread SPB outbreaks, like the 1974–1976 and 1997–2003 outbreaks, alter forest structure, species composition, and function of pine and pine-hardwood forests. Table 4.3 shows the number of individual spots across the region. Each of

Table 4.2 Extent of SPB outbreak in six states (Alabama, Georgia, Kentucky, North Carolina, South Carolina, Tennessee) (Adopted from Pye et al. (2008)). Outbreak status is defined as 1 SPB spot per 405 ha of host type (Price et al. 1998) with high outbreak status at 3 spots per 405 ha of host type

Year	Outbreak (# of counties)	High outbreak (# of counties)	Dollar amount
1997	7	0	$8,615,789
1998	16	1	$18,407,434
1999	41	17	$78,268,052
2000	150	69	$362,574,736
2001	187	126	$233,890,952
2002	147	96	$328,136,210
2003	30	7	$9,707,703

Table 4.3 Total number of SPB spots in six states at the peak of the 1997–2003 outbreak

State	2000	2001
Alabama	26,407	11,945
Georgia	2,582	2,604
Kentucky	1,664	3,513
North Carolina	2,219	3,860
South Carolina	12,996	22,149
Tennessee	9,352	12,766
Totals	58,839	60,628

these spots would have altered stand structure and species composition and all of these stands were in need of restoration after the outbreak in order to return the forest to previous conditions of pine dominated forests. Southern pine beetle can create disturbance in stands by causing mortality in large clusters of pine trees or by hastening the succession from pine-hardwood forests to late-successional forest by killing single or groups of overstory pine trees. Economic impacts are the easiest to quantify because the trees have an easily identifiable value (value per metric ton, population suppression and stand restoration costs, and reduction in market value due to sudden increase in wood supply).

Ecological impacts of SPB are more difficult to quantify, but are no less important. Negative ecological impacts can include loss of habitat for wildlife species that require pine forests (see Greenberg et al. Chap. 12). One particularly significant impact concerns the red-cockaded woodpecker (*Picoides borealis*; RCW) in Kentucky where, due to the 1997–2003 SPB outbreak, there are currently no known individuals living in the state. Before the SPB outbreak in 2001, RCW occupied several locations on the Daniel Boone National Forest in multiple counties– Laurel, McCreary, Pulaski, and Whitley counties. Because of widespread habitat destruction by the SPB, all of the remaining RCW were moved to South Carolina in 2001 by the USDA Forest Service (AWAKE 2014). Other ecological impacts may not be considered negative. As mentioned earlier, SPB is a native species that acts as a natural landscape scale disturbance factor, much like fire. SPB disturbance can include changes in nutrient cycling, hastening of forest succession, and changes in species composition (Tchakerian and Coulson 2011). SPB outbreaks create single and multi-tree gaps and stand level disturbance.

Pine forests in the CHR are also impaired by years of wildfire suppression and significant reductions in human-caused fires (see also Greenberg et al. Chap. 1; Grissino-Mayer Chap. 6; Greenberg et al. Chap. 12) and the forest communities that returned naturally after the 2001 SPB outbreak are different than previous conditions. Pine is not coming back in these areas without intervention (Elliott et al. 2012) because pine are early successional, shade-intolerant species that usually require bare mineral soil and abundant light in order to regenerate. Forest management intervention is needed in order to restore the unique pine and mixed pine-hardwood communities in the CHR (Elliott et al. 2012; Xi et al. 2012). There is also a desire by the USDA Forest Service and several partners to restore fire-adapted communities and increase heterogeneity on the landscape (CFLRP 2012) because, with the lack of landscape level fires and significant SPB outbreaks, there has been a trend towards pine overstory mortality and a lack of oak and pine regeneration (Elliott et al. 1999). Forest restoration efforts have focused on shortleaf and Table Mountain pine stands (Elliott et al. 2012).

Because of the significant economic and ecological impacts of the 1997–2003 SPB outbreak, there was a strong desire by the Southern Group of State Foresters and the USDA Forest Service (Southern Research Station and Forest Health Protection) to restore some of the pine and pine-hardwood forests impacted by SPB (Nowak et al. 2008). This outbreak spurred Congressional approval and appropriation for the SPB Initiative and the SPB Prevention and Restoration Program (Nowak et al. 2008; USDA Forest Service 2014). A central focus of the initiative and program was to provide the necessary funding to restore these pine communities and to enhance our knowledge of the impact on these communities and the most efficacious restoration practices. Additionally, a main objective of the SPB Prevention and Restoration Program is to create more open stands consisting of the appropriate tree species for the site, particularly shortleaf pine, and shortleaf pine-hardwood stands. These treatments include forest thinning, prescribed burning and restoring native pine forests. These activities have multiple benefits beyond protecting forests from SPB, such as increasing stand resiliency to changes in climate and pressure from invasive species, improving wildlife habitat and reducing fire risk. This program has treated nearly 500,000 ha, mostly through landowner cost-share programs that have directly benefitted more than 15,000 landowners. Approximately 25 % of the on-the-ground accomplishments and funding have gone towards restoring pine forests (Nowak et al. 2008; USDA Forest Service 2014).

In order to examine the efficacy of the SPB Prevention and Restoration Program treatments on reducing stand susceptibility to future outbreaks of SPB, plots were established in stands that had received either precommercial thinning or prescribed burns under the SPB Prevention and Restoration Program in the Piedmont ecoregion of the CHR in Georgia and South Carolina. Three plots were established in each stand, each separated by at least 50 m. Measurements included live crown

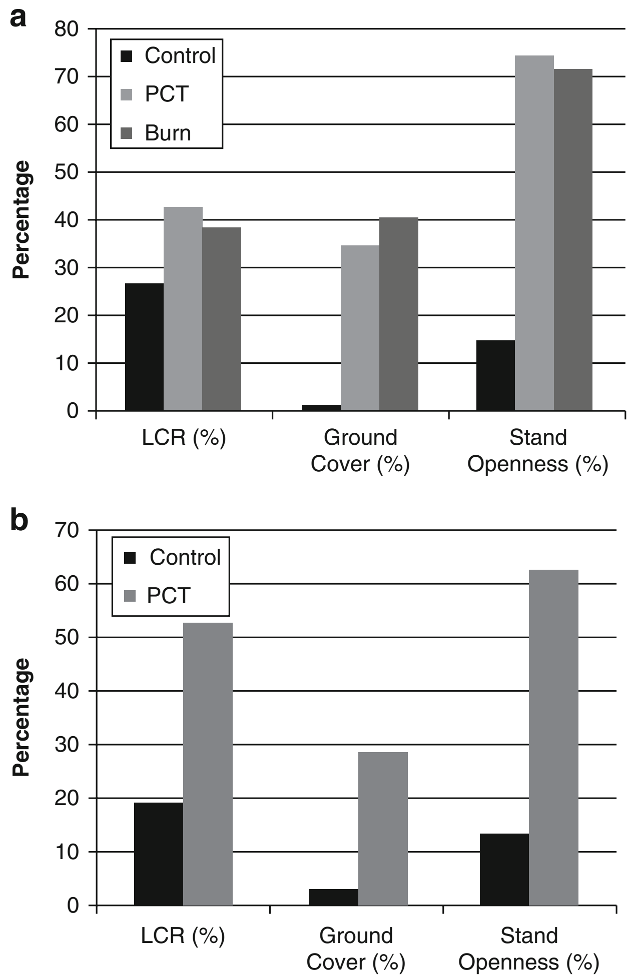
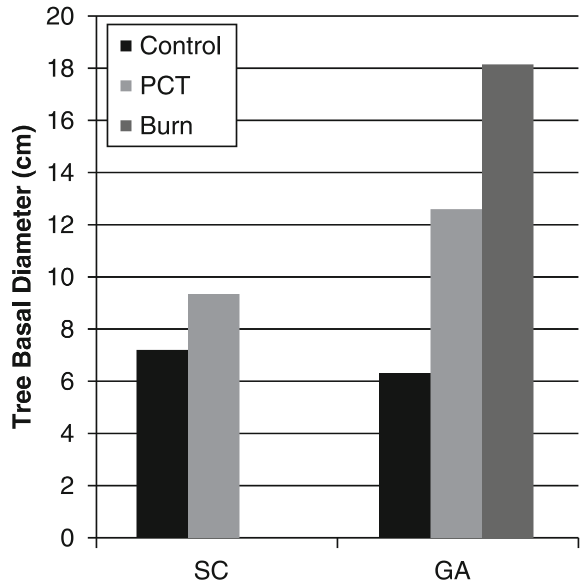


Fig. 4.3 Study in Piedmont ecoregion of (**a**) South Carolina; and (**b**) Georgia, evaluating SPB prevention and restoration treatments on percent live crown ratio, ground cover, and stand openness

ratio, percent ground cover, stem openness at 2 m (estimated percentage of area without stems in view) (Fig. 4.3), and stem diameter at breast height (Figs. 4.4 and 4.5). Overall, treatments by the SPB Prevention and Restoration Program had a very positive effect on stand characteristics, and may greatly reduce stand susceptibility to future SPB outbreaks. Thinning loblolly pine stands in South Carolina reduced stem density by over 390 %, but increased average stem diameter by nearly 40 %. Burning has been shown to lower a stands susceptibility to SPB (Nowak et al. in press) perhaps by creating a more open understory and promoting air movement (Fig. 4.6) (Thistle et al. 2004; Fettig et al. 2007). In the study examining the SPB Prevention and Restoration plots in both Georgia and South Carolina, burned and precommercially thinned treatments had higher live crown ratios, stand openness (Fig. 4.3), and stem diameter (Fig. 4.4).

Fig. 4.4 Study in Piedmont ecoregion of South Carolina and Georgia evaluating SPB prevention and restoration treatments on stem diameter



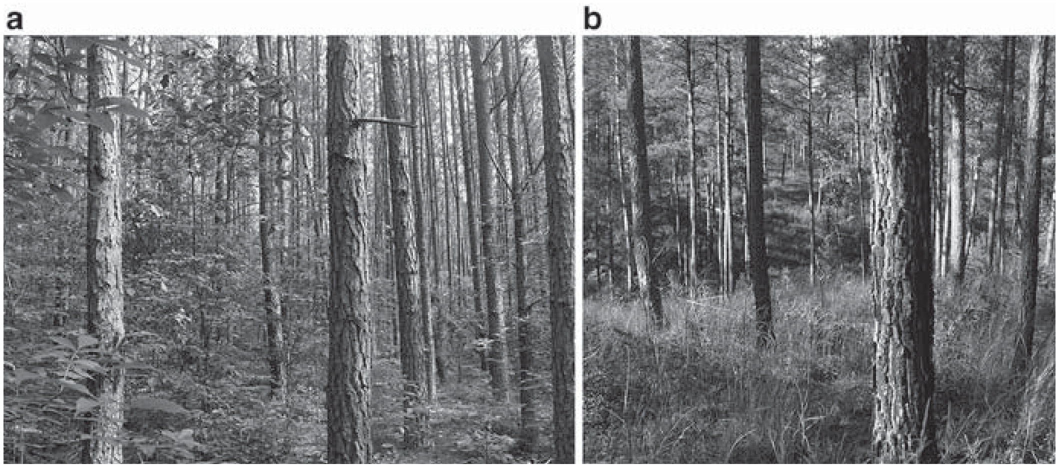


Fig. 4.5 (**a**) Unthinned pine stand with significant understory competition; and (**b**) thinned and burned stand showing greater openness and less competition

Fig. 4.6 (**a**) Mature restored (thinned and burned) shortleaf pine – chalky bluestem forest on the Ouachita National Forest; and (**b**) planted shortleaf pine in stand impacted by SPB (Both projects are partially funded by the SPB Prevention and Restoration Program)

4.3.2 Shortleaf Pine Restoration

Shortleaf pine has a natural range of over 688,000 km^2 in more than 22 states (Lawson 1990) covering much of the CHR and having a similar range as SPB (Clarke and Nowak 2009). Shortleaf pine grows best on a variety of soils, but especially on deep, well-drained soils. In the CHR, shortleaf pine is associated with two cover types, Shortleaf Pine and Shortleaf Pine-Oak (Lawson 1990). Shortleaf pine is important from a variety of commercial, ecological, and wildlife perspectives. Commercially, it is used primarily as lumber, plywood, logs, other structural material, and pulpwood. Ecologically, the diversity and integrity of shortleaf pine stands and related ecosystems has led to interest in restoration efforts on many public lands. Shortleaf pine is also a source of food, shelter, and nesting habitat for small mammals and birds such as the RCW. Shortleaf pine stands grown on appropriate sites with low basal area and open stand conditions would have low susceptibility to SPB (Fig. 4.6a).

Shortleaf pine can be regenerated through natural seeding if there is bare mineral soil (Lawson 1986) or by planting seedlings (Fig. 4.6b) (Barnett et al. 1986). Elliott et al. (2012) found that in shortleaf pine stands killed by SPB, burning the sites increased oak seedling density, but without shortleaf pine in the overstory there was no shortleaf pine regeneration. It is recommended that these sites be burned prior to planting seedlings with a follow-up herbicide release in order to achieve the most successful regeneration (Cassidy 2005; Nowak et al. 2008).

4.3.3 Table Mountain Pine Restoration

Table Mountain pine is endemic to the southern Appalachians, and is limited to xeric, rocky sites at high elevations from Georgia to Pennsylvania (Della-Bianca 1990). This species is often associated with red maple (*Acer rubrum*), black gum (*Nyssa sylvatica*), pitch pine, scarlet oak (*Quercus coccinea*), and chestnut oak (*Q. montana*) (Della-Bianca 1990). Table Mountain pine is not known as a significant timber species, but it is an important species from an aesthetic, erosion control and wildlife perspective (Zobel 1969; Della-Bianca 1990).

The 1997–2003 SPB outbreak and changes in fire frequency have severely impacted mature seed-bearing Table Mountain pine. Table Mountain pine need stand and site disturbance, light, and heat for successful regeneration (Della-Bianca 1990). In undisturbed conditions, succession trends toward oaks and red maple (Welch et al. 2000). Table Mountain pine has a bimodal age distribution with most of the trees in large tree age classes and a smaller percentage in the seedling age class (Williams and Johnson 1990). The authors concluded that without a change in the disturbance regime, such as reintroduction of fire, Table Mountain pine would be difficult to maintain because of the lack of younger age classes. Another study found that the SPB outbreak from 1997–2003 removed up to 30 % of the Table Mountain pine basal area in southwestern Virginia (Lafon and Kutac 2003). These authors also found that due to fire suppression, and reductions in the number of human caused fires (see also Chaps. 1, 6, 12), hardwoods had become more prevalent in the Table Mountain pine stands, and without the reintroduction of landscape-level fire Table Mountain pine would be unlikely to return from disturbances such as SPB.

4.4 Effects of Future Change on SPB

The SPB has a wide host range, high genetic plasticity, and an ability to sustain epidemics even in nontraditional hosts. As such, its geographic range is mostly constrained by host availability and climatic conditions (Ungerer et al. 1999). Certainly host material is abundant further north and would not limit the spread of this insect further into the CHR. While extremely hot temperatures can kill SPB (Wagner et al. 1984), an increase in minimum winter temperature of just a few degrees could result in a substantial increase in the geographic range of the SPB (Ungerer et al. 1999; Tran et al. 2007; Hain et al. 2011). However, while warmer winters in the next 50 years might result in increased numbers of SPB generations (Duerr and Mistretta 2013), they could also disrupt the natural timing of adult emergence and new infestations in the spring (Olatinwo et al. 2014). Widely used future climate projection scenarios agree that such a change to warmer temperatures is probable (Wear and Greis 2013; Dale et al. Chap. 13). Therefore it may be prudent to include SPB management considerations into future forest plans even in the northern portions of the CHR (Duerr and Mistretta 2013; Olatinwo et al. 2014). These management strategies have been discussed above, but include appropriate planting and forest composition, as well as preventative thinning and direct suppression of new infestations.

Other anthropogenic factors may influence SPB success in the CHR. Forest fragmentation (or parcelization) may shift age distributions to younger stands, and increase thinning (assuming a more active management approach – perhaps including thinning – in residential areas to meet goals of fire safe landscapes, aesthetic values and selection for bigger, healthy yard trees). This would likely result in lower SPB numbers and less tree mortality (Olatinwo et al. 2014). Conversely, weather extremes (flooding and drought, windstorms) could stress trees, predisposing them to SPB attack. Regardless, maintaining healthy, vigorous forests via appropriate spacing, stand density, and thinning will continue to be the best methods for mitigating SPB impacts to pine forests and ecosystems.

4.5 Summary

SPB is a significant, but natural disturbance factor in the CHR. It has been reported in this region since the nineteenth century, with some of the most significant outbreaks occurring between 1974 and 1976 and from 1997 through 2003. The most recent outbreak impacted pine and pine-hardwood forests across six states in the CHR, leaving behind groups of standing dead trees created by more than 100,000 SPB spots across 405,000 ha. SPB activity in the mountainous areas of the CHR has been conspicuously absent since 2003. We do predict that SPB will return to the CHR within the next 15 years. Without forest management intervention, these forests will likely not return to pine. It is the desire of the Southern Group of State Foresters and USDA Forest Service for many of these stands in southern ecoregions to return to fire adapted forests such as shortleaf- and Table Mountain pine-dominated forests. These pine forests are valuable to the region for the obvious economic reasons, but also for several ecological and aesthetic reasons. The SPB Prevention and Restoration Program was developed in 2003 in order to restore resilient pine stands in areas that had been impacted and to create stands that would be less susceptible to future outbreaks.

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