Plant and Structural Diversity

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FLORAL DIVERSITY FOLLOWING HARVEST ON SOUTHERN APPALACHIAN MIXED OAK SITES

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Abstract—Floral species richness was assessed for three vegetation strata in midsite quality, upland hardwood stands in the Ridge and Valley, and Allegheny Mountain physiographic provinces of Virginia and West Virginia. High variability in richness was found among six study locations which were relatively homogenous in topographic characteristics and overstory species composition and structure. Total species richness of the 35-acre sites ranged from 92 on the Clinch Ranger District of the Jefferson National Forest in Southwestern Virginia to 167 on the Wythe Ranger District. An inverse relationship occurred between overall species richness and percent crown cover in the midstory. Most of the variability was associated with differences in herbaceous communities. The high variability in species richness between sites of apparently similar age, structural, and topographic characteristics could question the validity of the chronosequence approach to the study of species diversity. By using stands of different age but similar site characteristics to represent community development, chronosequence studies are dependent on the assumption that similar sites will develop similar communities. Preliminary comparisons of 1-year post-treatment species richness to pre-treatment values were made for one of six, 35-acre replicates containing seven 5-acre treatment plots. Treatments were clearcut, two-aged (leave tree), two levels of shelterwood, group selection, chemical understory control, and control/undisturbed. Species richness increased for all treatments. The magnitude of increase followed a gradient of canopy disturbance. The clearcut resulted in a net gain of 26 species while the herbicide treatment mimicked the control with a net gain of 2 species.

INTRODUCTION

Concern over changes in the amount of biological diversity has been increasing during past decades. The Forest Management Act of 1976 (FMA) responds to these concerns by mandating that on federal lands “management prescriptions... shall preserve and enhance the diversity of plant and animal communities... so that it is at least as great as that which would be found in a natural forest” (36 CFR 219.27g). Two questions must be answered in order to successfully abide by this mandate: (1) what is the diversity of a natural forest and (2) how will management prescriptions preserve that diversity?

Although the FMA applies to all Federal lands, this study focuses on the Southern Appalachians. Several authors have contributed to answering the first question (Braun 1950, Whittaker 1956, Duffy and Meir 1992, Aulick 1993) for forests in this region; however, these and other efforts have focused primarily on late-successional forests growing on highly productive sites. Little emphasis has been placed on second-growth forests growing on average sites, site index$_{50}$ 60 to 80**,** in the Southern Appalachians. Other studies have attempted to illustrate community response to silvicultural disturbances (Horn 1980, Moriarty and McComb 1985, Duffy and Meir 1992, Gilliam and others 1995). Most have focused on the clearcut regeneration system and have taken chronosequence approaches to represent community development. This approach is dependent on the assumption that there is low variability in species diversity on sites which are apparently uniform with respect to topographic characteristics and overstory species composition.

The need to investigate community response to silvicultural practices is clear, and a long-term study which tracks the development of a set of forest stands would best reflect those changes. The objectives of this paper were to (1) establish baseline data for the characterization of the vascular plant communities found in mixed oak stands of the Ridge and Valley and Allegheny Mountain physiographic provinces of the Southern Appalachians; and (2) make initial comparisons of species richness between a mature community and the communities which developed following various silvicultural disturbances at one study location.

METHODS

Study Area

Eight study sites have been located within the Ridge and Valley and Allegheny mountain physiographic provinces of southwestern Virginia and West Virginia. The Blacksburg and Clinch Ranger Districts of the Jefferson National Forest contain two study sites each (BB1 and BB2 and Clinch 1 and Clinch 2, respectively); one site is located in the Wytheville Ranger District and one in the New Castle District. Two sites were installed on the Westvaco Wildlife and Ecosystem Research Forest (WWERF) near Cassity, WV. Due to incomplete data, WWERF 2 and Clinch 2 were not included in this analyses. Study sites were selected to represent mid-elevation, southern aspect, mixed oak forests in the Southern Appalachians. Site index$_{50}$ ranged from 60 to 77**.** Sites were installed in mature forests that displayed no indications of silvicultural manipulation within the past 15 years. Elevations range from approximately 2,200 feet at BB1 to 3,580 feet at Clinch 1. Mean daily July temperature for BB1 and BB2, Wythe, and New Castle is 72 °F. There are 150 frost-free days and an associated 42 to 48 inches of annual precipitation, with approximately 24 to 32 inches of snow. The WWERF and Clinch sites, located in the Allegheny Mountains, are slightly cooler and moister, with only 130 frost-free days and a mean daily July

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temperature of 65 °F. They receive 50 to 55 inches of precipitation, including 48 to 64 inches of snowfall (Smith 1995). The soils dominating these sites are derived predominantly from sandstone and shale parent material.

Study Design

A randomized complete block design was used, in which each 35-acre site contains seven 5-acre treatment blocks. Treatments include clearcut, group selection, leave tree, two types of shelterwood, chemical understory control, and a control. Due to area constraints, the WWERF 1 site was limited to five treatments, and did not contain the light shelterwood or chemical understory control. Clearcuts consisted of the removal of all merchantable stems and the felling of all residuals greater than 1-inch d.b.h. Group selections were installed such that opening diameters would be no more than twice the height of the adjacent canopy trees with three openings per block. Leave tree and shelterwood cuts contain 10 to 20 sq. ft., 20 to 30 sq. ft., and 50 to 60 sq. ft. of residual basal area per acre, respectively. Chemical control consisted of a basal spray application of Garlon 4 to undesirable midstory woody vegetation in order to promote development of oak advance regeneration.

To qualify floral structure, vegetation was divided into three strata based on height: the tree stratum consisted of vegetation greater than 48.75 feet (5 m) in height, the shrub stratum was woody vegetation greater than 3.25 feet (1 m) but less than 48.75 feet (5 m), and the herb stratum was woody and herbaceous vegetation less than 3.25 feet (1 m) in height. The tree stratum was measured on three, randomly located, 78- by 78-foot (24- by 24-meter) plots per treatment block. The species of each tree within the plot was recorded and its diameter (d.b.h., 1.37 m aboveground) measured to the nearest 0.04 inches using a metal diameter tape. Sixteen 19.5- by 19.5-foot (6- by 6-m) subplots were established within each tree plot. Three subplots were randomly selected for measurement of the shrub stratum. Numbers of individuals per species and their respective crown diameters were measured and recorded to the nearest 0.3 foot on each plot. A further division of the 78- by 78-foot plots was used to establish eight 3.25- by 3.25-foot (1- by 1-m) plots on which the herb stratum was sampled for numbers of individuals and percent cover by species. Percent cover was visually estimated using a pretransformed scale (Little and Hills 1978). In addition to measurements taken on sampling plots, a walkthrough of each 5-acre treatment plot was done in an attempt to account for all woody and nonwoody vascular plant species present. All strata were sampled mid-May to early June and additional sampling of the herbaceous stratum and walkthroughs occurred in mid-August. The two herbaceous samplings were used to account for within-growing season-variation of the herb stratum. A single data set was created by combining the data from each sample period. All species sampled were included in the combined data set. If a species occurred in both samplings, the larger value for each variable measured was used. The study is designed so that these communities will be periodically sampled throughout an 80-to 100-year rotation. Posttreatment sampling of BB1 occurred one complete growing season after harvest. Future inventories will occur at 3, 5, and 10 years, and at 10-year intervals thereafter until rotation age.

DATA ANALYSES

Pretreatment site characterization was based on both topographic and vegetative characteristics. Average slope percent and aspect were calculated from measurements taken at each tree plot center. Overstory basal area, quadratic mean diameter, and percent crown cover of the shrub stratum were estimated based on d.b.h. and crown radii measurements taken from the tree and shrub plots, respectively. ANOVA and Tukeys multiple comparison tests were used to test for between-site differences in tree basal area and percent cover in the shrub stratum. Pretreatment values for each stratum were calculated by averaging across the appropriate plots at each site. Total species richness and numbers of unique woody and herbaceous species were determined for each site, based on the compilation of treatment plot walkthroughs. In calculating richness values, some species (*Carex*, *Aster*, *Solidago*, *Desmodium, Poa, Panicum, Lespedeza,* and *Rubus*) were grouped by genus to account for variations in identification specificity among sampling crews.

In addition to pretreatment comparisons, changes in total richness following treatments were determined for BB1 based on 1-year posttreatment data.

RESULTS

Site Comparisons

The six study sites evaluated represented aspects between 149 and 270 degrees azimuth with consistent within-site variation (table 1). Slopes ranged from 12 to 38 percent with moderate oak site indexes (table 1). Overstory composition and structure were consistent among the sites with various *Quercus* species dominating between 42 to 84 percent of the basal area (table 2). Overstory richness, compiled from all 78- by 78-foot plots, averaged 26.8 species per site. The greatest richness occurred on BB1 and Clinch 1 with 33 species at each site. WWERF had the least number of overstory species with 22 (table 3). Slightly more variation

Table 1—Site characteristics and associated 95 percent confidence intervals of six study sites located in the Ridge and Valley and Allegheny Mountain physiographic provinces of Southwestern Virginia and West Virginia, based on measurements taken at plot center of 78- by 78-foot tree stratum plots

Site	Aspect	Slope	Oak site index$_{50}$	Age
BB1	153 ± 19	16 ± 3	n/a	n/a
BB2	151 ± 14	21 ± 3	72	99
Clinch	149 ± 20	30 ± 3	60	111
NC	150 ± 16	12 ± 4	60	62
Wythe	163 ± 39	18 ± 4	72	70
WWERF	270 ± 7	38 ± 3	77	73

Table 2—Basal area (square feet per acre) of the overstory stratum of six study sites located in the Ridge and Valley and Allegheny Mountain physiographic provinces of Southwestern Virginia and West Virginia, based on measurements taken on 78- by 78-foot plots (totals followed by the same letter were not significantly different at ∂ = 0.05)

Species	Study site BB1	BB2	Clinch	NC	WWERF	Wythe
Quercus prunus	18.8	49.6	16.6	33.1	20.8	24.9
Q. alba	34.3	12.8	16.9	11.4	0	15.0
Q. velutina	14.5	14.8	2.0	6.7	0	22.3
Q. coccinea	13.2	0.5	0.4	49.6	0	12.2
Q. rubra	2.3	4.3	38.0	0	44.8	1.2
Acer rubrum	8.5	7.9	19.5	3.5	33.1	7.4
Liriodendron tulipifera	4.7	9.1	1.0	0	15.3	1.7
Oxydendron arboreum	0.7	7.4	8.8	3.8	1.2	15.6
Others	19.0	12.6	26.8	11.9	40.8	11.7
Total	116a	119a	130ab	120a	156b	112a

occurred in the shrub stratum, with respect to cover and composition. The 48-percent crown cover at Wythe was significantly lower than the 78.3- and 73.7-percent at BB1 and New Castle, respectively (table 4). However, the 30 shrub species occurring at Wythe was higher than the average midstory richness (table 3). Average midstory richness for all sites, 26.2 species, was just less than overstory richness but had greater variation. It must be noted that these richness values for the tree and shrub strata are not directly comparable since sampling area was more than five times greater for the tree stratum. However, with a larger sample area yielding less than one more species on average in the tree stratum, this suggests that midstory richness is greater than overstory richness across these sites. Also, richness values for WWERF are based on a smaller area sampled, due to there only being five treatment plots.

Table 3—Overstory, midstory, and overall species richness values of six study sites located in the Ridge and Valley and Allegheny Mountain physiographic provinces of southwestern Virginia and West Virginia[a]

Site	Overstory	Midstory	Overall Woody	Overall Herbaceous
BB1	33	33	50	75
BB2	25	29	50	80
Clinch	33	24	44	54
NC	24	17	41	51
Wythe	24	30	52	115
WWERF	22	24	47	88
Average	26.8	26.2	45.7	77.2

[a]Numbers of species in the overstory and midstory are based on the compilation of twenty-one 78- by 78-foot plots and sixty-three 19.5- by 19.5-foot plots respectively at each site. Overall richness is based on compilation of treatment plot walkthroughs at each site.

The overall species richness was quite variable among the locations sampled (table 3). New Castle had only 92 species while 167 species were found at Wythe. Since the number of woody species found at each site was relatively homogeneous, herbaceous vegetation accounted for most of the variability in richness values. Of the 275 species identified across all pretreatment communities, 80 herbaceous and 18 woody species were found at only one site (table 5). Wythe and WWERF had not only the greatest richness values, but also had the greatest number of unique (found only at one site) species, with 30 and 27, respectively. Species richness was inversely related to percent cover in shrub stratum. Those sites with high richness had correspondingly low shrub cover, while those with low richness had high coverage in the shrub stratum. No relationship was found between richness and overstory basal area.

Blacksburg 1 Pretreatment versus Posttreatment

As was expected, overall species richness at BB1 increased dramatically following treatment. Total species richness increased from 125 to 291 species, with herbaceous species representing 91 percent of the 166 invading species. Evaluation of the data collected on the 3.25- by 3.25-foot plots illustrates the variability of the effects of individual treatments (table 6). The clearcut resulted in a net increase in richness of 26 species, while only 2 species were gained by the herbicide and control treatments. The magnitude of change in species richness occurred along a gradient of canopy disturbance. As canopy disturbance increased, more species were lost from the community. For example, 26 species were lost following clearcutting but 42 new species were gained. Treatments with less canopy disturbance, such as the shelterwoods, resulted in species losses equal to those in the control. Considering the group selection as a high-disturbance treatment, the seven species lost in this treatment were an exception to the pattern. The indistinguishable difference between the group selection and control in numbers of species lost could be attributed to the nonuniform effects of the treatment creating refugia for species which would be otherwise negatively impacted by canopy removal. Since groups were installed without consideration of plot location, a portion of the plots were located in group openings while the remaining plots were in the residual stand. Therefore, species invading group openings and species in the residual stand were represented in the sample plots.

Table 4—Percentage of cover of the shrub stratum of six study sites located in the Ridge and Valley and Allegheny Mountain physiographic provinces of southwestern Virginia and West Virginia[a]

	Study site					
Species	BB1	BB2	Clinch	NC	WWERF	Wythe
Acer pennsylvanicum	0.1	0	8.1	0	11.4	1.8
A. rubrum	24.5	13.8	9.3	0	7.9	1.8
Betula lenta	0	0.5	2.1	0	5.7	0.2
Castanea dentata	0.4	1.0	10.4	3.5	0.1	8.2
Cornus florida	12.5	6.0	0	4.0	0	1.6
Fagus grandifolia	0	0	0.1	0	19.7	0
Lindera benzoin	3.8	0	0	0	0.4	0
Magnolia frazieri	0	0	7.4	0	0.5	0
Nyssa sylvatica	10.3	7.7	0.8	58.7	0.3	3.3
Oxydendrum arboreum	0.4	6.2	1.9	3.1	0.7	12.1
Pinus strobus	1.7	2.5	0	0	0	4.9
Others	24.6	25.3	25.9	4.4	12.4	6.6
Total	78.3a	63.0ab	66.0ab	73.7a	59.1ab	44.1b

[a] Species listed were chosen on the basis that they represent the four most common at each site. Totals followed by the same letter were not significantly different at ∂ = 0.05.

Table 5—Number of woody and herbaceous species unique to a site (sites are located in the Ridge and Valley and Allegheny Mountain physiographic provinces of southwestern Virginia and West Virginia)

	Species	
Site	Woody	Herbaceous
BB1	2	9
BB2	2	12
Clinch	3	0
NC	4	9
Wythe	2	28
WWERF	5	22
Total	18	80

The numbers of woody species lost and gained remain fairly constant across treatments. Herbaceous plants account for the patterns of species invasion. The increase in available light associated with the clearcut and other high canopy disturbing treatments created conditions which favored these species.

Table 6—Changes in woody and herbaceous species richness for seven treatments installed in a mixed oak forest near Blacksburg, VA (values are based on the species represented on eighteen 3.25- by 3.25-foot plots per treatment)

	Number of species lost		Number of species gained		
Treatment	Woody	Herb	Woody	Herb	Total change
Clearcut	3	13	2	40	+26
Group selection	2	5	5	22	+20
Leave tree	7	13	5	28	+13
Shelterwood 20-30	2	7	7	19	+17
Shelterwood 50-60	5	4	6	14	+11
Herbicide	2	4	4	4	+2
Control	5	4	3	8	+2

CONCLUSION

There was high variability in species richness among communities which appeared homogenous when compared on the basis of topography and overstory species composition and structure. The number one assumption for chronosequence studies is that if site factors and overstory species composition are similar between sites which are at various stages of successional development, then those sites can be used to represent the development of a community through time following a given disturbance. The high variability in species richness found in this study questions the validity of chronosequence studies for comparing treatment effects on species diversity in Southern Appalachian mixed oak forests.

Consistent with Gilliam and Turrill (1993), these data suggest that overall species richness is negatively correlated with percent cover in the shrub stratum. Further analyses of these data will attempt to quantify that correlation and investigate other possible variables which can be used to predict species richness.

For all treatments at BB1, herbaceous layer species richness increased following treatment. The magnitude of increase followed a gradient of canopy disturbance. Herbaceous species were the majority of those lost and gained. Future work with these data will compare the shifts in species relative abundance and investigate relationships among the species lost and gained as a result of treatment and associated site variables. Future inventories representing the redevelopment of these communities will be necessary to reveal any differences or similarities between treatments.

If silvicultural practices are to survive the scrutiny of current legislation and the environmental community, then scientific documentation of the effects of those practices on all members of a given plant community must be established. Although research such as this can help to identify the changes that take place following silvicultural activities, in no way can we identify which aspects of diversity hold the most ecological significance. Without that ability, societal values will ultimately decide which practices are acceptable, and as scientists we can only provide society with sound information for the basis of that decision.

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LITERATURE CITED

Aulick, S.D.S. 1993. Vascular flora of three watersheds in the Fernow Experimental Forest and factors influencing species composition of the herbaceous layer. Marshall University. Huntington, WV: Thesis.

Braun, E.L. 1950. Deciduous forests of eastern North America. New York: Hafner.

Code of Federal Register. 1982. Part 36. Section 219.27(g)

Duffy, C.D.; Meir, A.J. 1992. Do Appalachian herbaceous understories ever recover from clearcutting? Conservation Biology. 6(2): 196- 201.

Gilliam, F.S.; Turrill, N.L. 1993. Herbaceous layer cover and biomass in a young versus a mature stand of a central Appalachian hardwood forest. Bull. Tor. Bot. 120(4): 445-450.

Gilliam, F.S.; Turrill, N.L.; Adams, M.B. 1995. Herbaceous-layer and overstory species in clear-cut and mature central Appalachian hardwood forests. Ecol. Appl. 5(4): 947-955.

Horn, J.C. 1980. Short-term changes in vegetation after clearcutting in the Southern Appalachians. Castanea. 45: 88-96.

Little, T.M.; Hills, F.J. 1978. Agricultural experimentation. New York: John Wiley & Sons.

Moriarty, J.J.; McComb, W.C. 1985. Plant community composition and structure in a Central Appalachian forest stand 25 years after timber stand improvement. Castanea. 50: 32-42.

Smith, D. Wm. 1995. The Southern Appalachian hardwood region. In: Barrett, J.W. 1995. Regional silviculture of the United States. 3rd ed. New York: John Wiley & Sons.

Whittaker, R.H. 1956. Vegetation of the Great Smokey Mountains. Ecol. Mon. 26: 1-80.

INFLUENCE OF HARVEST VARIABLES ON WOODY PLANT DIVERSITY IN UPPER COASTAL PLAIN MIXED TYPES

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Abstract—Commercial clearcuts and silvicultural clearcuts were applied to upper coastal plain mixed stands in the dormant season and early growing season. The commercial clearcutting removed only merchantable material. The silvicultural clearcutting included felling of all stems 1 inch d.b.h. or larger in addition to the removal of merchantable material. Each combination of season and type of cut was replicated six times. The stands were allowed to regenerate naturally via seedlings and sprouts with no further treatments. Harvest treatments significantly influenced pine seedling proportions and the diversity of foliage cover in the regenerated stands.

INTRODUCTION

This paper presents preliminary results of a study being conducted on the U.S. Department of Energy, Savannah River Site, in the upper coastal plain of South Carolina. Earlier research in the upper piedmont of Georgia demonstrated that harvest season and intensity, alone, could strongly influence the relative species composition of naturally regenerated stands in the oak-pine type (McMinn 1992). However, the degree to which such influences may generalize to other mixed types is open to question. The objective of this study is to determine the influence of harvest season and intensity on subsequent woody species composition and diversity in upper coastal plain mixed hardwood-pine stands.

METHODS

Mixed hardwood-pine stands of from 20 to 60 acres in size were each partitioned into four equal areas to accommodate the four combinations of commercial clearcutting and silvicultural clearcutting in the dormant season and early growing season. The commercial clearcuts removed only merchantable material. The silvicultural clearcuts included felling of all stems 1 inch d.b.h. or larger in addition to the removal of merchantable material. The experimental design is a randomized complete block in which each of six stands constitutes a block. Three blocks were harvested in each of 2 consecutive years, but year was not included as a variable in the analyses reported here. All results reported here are from data within 5 years after treatment. The response variables are number of seedlings by species and foliage coverage by species. Woody plants less than 4.5 feet tall were considered seedlings and were counted on hundredth-acre circular plots spaced at 150 feet on a square grid network. Foliage cover of woody plants at least 4.5 feet tall was estimated from a system of 150-foot line transects. To characterize the general influence of treatments on species composition, species were grouped by oaks, other hard hardwoods, pines, soft hardwoods, shrubs, and miscellaneous species. In the oak group, southern red oak (*Quercus falcata* Michx.), water oak (*Q. nigra* L.), willow oak (*Q. phellos* L.), post oak (*Q. stellata* Wangenh.), and black oak (*Q. velutina* Lam.) were prevalent; turkey oak (*Q. laevis* Walt.) and blackjack oak (*Q. marilandica* Muenchh.) were common; and scarlet oak (*Q. coccinea* Muenchh.), bluejack oak (*Q. incana* Bartr.) and laurel oak (*Q. laurifolia* Michx.) were occasional. Other hard hardwoods included hickories (*Carya* spp.), dogwood (*Cornus florida* L.), persimmon (*Diospyrous virginiana* L.), and American holly (*Ilex opaca* Ait.). Pines included loblolly (*Pinus taeda* L.) and longleaf (*P. palustris* Mill.). Among the soft hardwoods, sweetgum (*Liguidambar styraciflua* L.), blackgum (*Nyssa sylvatica* Marsh.), and black cherry (*Prunus serotina* Ehrh.) were prevalent; winged elm (*Ulmus alata* Michx.) was common; and red maple (*Acer rubrum* L.), sycamore (*Platanus occidentalis* L.), and willow (*Salix* spp.) occurred occasionally. In the shrub group, plum (*Prunus* spp.) and sumac (*Rhus* spp.) were prevalent; blueberry (*Vaccinium* spp.) was common; and occasionals were chinkapin (*Castanea* spp.), viburnum (*Viburnum* spp.), waxmyrtle (*Myrica cerifera* L.), yaupon (*Ilex vomitoria* Ait.), St. Johnswort (*Hypericum* spp.), and privet (*Ligustrum* spp.). The miscellaneous group included apple (*Malus* spp.) and sassafras [*Sassafras albidum* (Nutt.) Nees]. Species diversity and evenness for seedlings and foliage cover were calculated using Shannon's indexes (Magurran 1988).

RESULTS AND DISCUSSION

Significant difference by block in the proportion of seedlings or foliage cover for a given species group reflects only a difference among initial stands in the prevalence of that species group (table 1). The only statistical difference attributable to harvest treatment is the proportion of total seedlings that are pine. For the commercial clearcuts, dormant season harvests produced almost nine times the pine proportion as early growing season harvests (table 2). For the silvicultural clearcuts, dormant season harvests resulted in over six times the pine proportions as early growing season harvests. The strong seasonal influence on pine proportions is consistent with results in the upper piedmont (McMinn 1992). Larger proportions of the "other" species following early growing season harvests compared to dormant season harvests are also consistent with the piedmont study.

As with seedling and foliage proportions, significant differences in diversity and evenness by block reflect variability among initial stand characteristics (table 3).

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Table 1—Analysis of variance results for seedling proportion and foliage cover proportions of selected species groups

Species groups	Block	Season	Cut	Season x cut
	Source of variation			
Seedling proportion:				
Oak	-	-	-	-
Hard hardwood	-	-	-	-
Pine	*[a]	*	-	-
Soft hardwood	*	-	-	-
Foliage cover proportion:				
Oak	*	-	-	-
Hard hardwood	-	-	-	-
Pine	-	-	-	-
Soft hardwood	-	-	-	-

[a] Asterisk denotes significance at the .05 level.

Table 2—Seedling percentages by species group and harvest treatment

Treatment Season	Treatment Cut	Oak	H. hwd.	Pine	S. hwd.	Other[a]
		Species group				
		Percent				
Dormant	Comm	15.9	10.9	16.1	8.3	48.8
Dormant	Silv.	9.7	16.6	15.3	11.8	46.6
Growing	Comm	6.9	9.5	1.8	8.7	73.1
Growing	Silv.	9.8	12.7	2.4	6.4	68.7

Comm = commercial clearcuts, silv = silvicultural clearcuts.
[a] Includes shrubs and miscellaneous species.

Foliage cover diversity differed by type of cut and the interaction between type of cut and season. Species richness ranged only from 26 to 28 for seedlings and from 18 to 21 for foliage cover, so any differences among treatments in diversity are attributable to differences in evenness (table 4). The high foliage cover diversity following growing season commercial clearcuts compared to growing season silvicultural clearcuts is likely due to a degree of species stability provided by the residual stand. The low foliage cover diversity following growing season silvicultural clearcuts is likely due to differential sprouting among hardwood species from the harvest timing that introduces the greatest stress. Higher seedling diversity following silvicultural clearcuts compared to commercial

Table 3—Analysis of variance results for woody species diversity and evenness of seedlings and foliage cover

Trait	Block	Season	Cut	Season x cut
	Source of variation			
Seedlings:				
Diversity	*[a]	-	-	-
Evenness	-	-	-	-
Foliage Cover:				
Diversity	**	-	*	*
Evenness	-	-	-	-

[a] Single asterisk denotes significance at the .05 level, double asterisk at the .01 level.

Table 4—Woody species diversity by harvest treatment for seedlings and foliage cover

Treatment Season cover	Treatment Cut	Trait Seedlings	Trait Foliage
Dormant	Commercial	1.77	1.60
Dormant	Silvicultural	1.92	1.67
Growing	Commercial	1.77	1.89
Growing	Silvicultural	1.94	1.44

clearcuts could, perhaps, be due to differential response of various seedling species to competition from residuals in the commercial cuts.

CONCLUSION

These results suggest that harvest season and intensity do have the potential to influence species composition and woody plant diversity in upper coastal plain mixed types.

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LITERATURE CITED

Magurran, Anne E. 1988. Ecological diversity and its measurement. Princeton, New Jersey: Princeton University Press. 179 p.

McMinn, James W. 1992. Diversity of woody species 10 years after four harvesting treatments in the oak-pine type. Canadian Journal of Forest Research. 22: 1179-1183.

CHANGES IN WOODY STEM SPECIES DIVERSITY OVER TIME FOLLOWING CLEARCUTTING IN THE MISSISSIPPI RIVER FLOODPLAIN

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Abstract—Woody species diversity is an important ecological aspect of bottomland hardwood stands in the Mississippi River floodplain. A chronosequence study was established with stands which had been clearcut during different years on Delta National Forest. The stands were chosen with respect to site similarity and ranged in topographical position from ridges to high flats. Stand age ranged from 1 to 50 years since harvest. Several older, adjacent stands were utilized as controls. No intermediate silvicultural treatments had been implemented on these stands. Woody stem measurements were recorded during the summer of 1996 and data were analyzed to determine when woody species diversity reached a maximum following clearcutting. Analysis of three canopy levels showed a peak in total woody species diversity (relative dominance and relative frequency) during the first 10 years following clearcutting. Trends were observed relating to changes in diversity over time. When diversity of the three canopy levels was examined separately, the upper and lower canopy levels exhibited a peak during the first 10 years. The lower canopy level actually exhibited two peaks in the chronosequence, one at 10 years and the other at 35 years. In the lower canopy level, the highest diversity occurred at 35 years. Diversity of the midcanopy level did not peak until 15 years following clearcutting. The diversity of all three canopy levels decreased with age after 45 years.

INTRODUCTION

Within the Mississippi River floodplain, the land area occupied by the bottomland hardwood forest type has diminished drastically during the last 100 years. Land clearing for agriculture has been the primary reason for the decrease (Hodges 1993). Remnants of the original forest are left in areas that are either too wet for efficient farming or were in public ownership. These forested sites were subjected to repeated diameter-limit harvests which were in actuality high grading, and left them in disparaging condition with respect to timber quality and species composition.

During the 1960's, the USDA Forest Service began implementing operational clearcuts on the Delta National Forest in a manner that could be construed as a regeneration technique rather than simply a harvesting procedure (Hurst and Bourland 1978). In southern floodplain forests, clearcuts are an effective regeneration technique and will promote a more desirable species composition if advance regeneration is present. From this time period to the present, an abundance of even-aged stands were created.

Another event that transpired during this time period was the designation of three research natural areas on the Delta National Forest. These areas were originally set aside to preserve important communities, maintain genetic diversity, and serve as reference areas for the study of succession, and are still in existence today (USDA 1976). The research natural areas are valuable examples of areas where mature, unmanaged stands exist, although additional stands may be found throughout the forest that were last harvested during the 1910's and 1920's.

Concerns over the perceived loss of biodiversity are of great importance to scientists, natural resource managers, and the general public (Westman 1990). There is a need to explore the effects of clearcutting on plant diversity and to examine natural succession in bottomland hardwood stands. The objectives of this paper are to examine changes in woody plant diversity over time and to ascertain when, following clearcutting, woody species diversity is most similar to that of mature, unmanaged stands.

STUDY SITE

The Delta National Forest is approximately 59,000 acres in size and is located in Sharkey County at the southern end of the Mississippi River alluvial plain in Mississippi. Both the Little Sunflower River and the Big Sunflower River as well as countless sloughs and bayous run through the forest. The forest is actually within the Yazoo Basin and the elevation is about 90 to 95 feet above sea level (USDA 1976). The forest is subjected to spring flooding from the Mississippi River backwater as well as rainwater runoff from surrounding areas.

Due to differences in hydrology between the north and south ends of the forest, the stands used for this study were all located on the north end of the forest. The south end of the forest is flooded for a longer duration during the growing season. The stands on the north end of the forest occupy high flats or low ridge sites or gradations between the two topographic positions.

Forest types are typical of southern bottomland hardwood forests (Johnson and Shropshire 1983). Commercially important species include Nuttall oak (*Quercus nuttallii* Palmer), willow oak (*Q. phellos* L.), overcup oak (*Q. lyrata*

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Walt.), green ash (*Fraxinus pennsylvanica* Marsh.), sugarberry (*Celtis laevigata* Willd.), sweetgum (*Liquidambar styraciflua* L.), eastern cottonwood (*Populus deltoides* Marsh.), sweet pecan [*Carya illinoensis* (Wang.) K. Koch], water hickory [*C. aquatica* (Michx. f.) Nutt.], boxelder (*Acer negundo* L.), red maple (*Acer rubrum* L.), American elm (*Ulmus americana* L.), and persimmon (*Diospyros virginiana* L.). Important midstory and understory species are rough-leaved dogwood (*Cornus drummondii* Meyer), swamp privet [*Forestiera acuminata* (Michx.) Poir.], buckthorn bumelia [*Bumelia lycioides* (L.) Pers.], deciduous holly (*Ilex decidua* Walt.), hawthorn (*Crataegus viridis* L.), and water elm [*Planera aquatica* (Walt.) Gmelin]. All Latin names follow those used by Smith (1988).

METHODS

Inventory Procedures

Stands were grouped in 5-year age classes based on time since harvest. Thirty-five stands were used with approximately four stands in each of nine age classes. Upper, mid, and lower canopy levels were measured using a nested plot design with 10 plots per stand randomly located. Diameters at breast height (d.b.h.) were measured for all living trees or shrubs in the upper canopy level within a 10-acre fixed area circular plot. Diameters at breast height were measured for all living trees or shrubs in the midcanopy level within a 100-acre fixed area circular plot. The same 100-acre plot used for the midcanopy level was also utilized for the lower canopy level, where diameters at the root collar were measured for all living trees or shrubs. Each individual tree or shrub was identified to species.

Canopy level differentiation was based on the authors' observations rather than an arbitrary height. The midcanopy level was not distinguishable as a separate entity in any of the stands until 10 years of age. The basis for separation of the canopy levels was the presence of a distinguishable layer of foliage. Due to the various stages of succession of the stands, the use of a particular size class for a member of a particular canopy level was impractical.

Data Analysis

Relative density by species for each stand was used to calculate Shannon's diversity index. Shannon's diversity index is a measure of the uncertainty of the ability of predicting the species of an individual when that individual is picked at random from the community (Hair 1980). The many stands used for the chronosequence were assumed to represent an average stand on ridge or flat topography in the Delta National Forest. Comparisons were made between the clearcut stands and the mature unmanaged stands. In this case, Sorenson's similarity index was used to compare relative species dominance and relative frequency. Sorenson's index was expressed as a percentage with respect to differences between the clearcut stands and the mature unmanaged stands.

RESULTS

Woody species diversity in the upper canopy level reached a peak during the first 10 years following clearcutting. It had a Shannon's diversity index value of 2.44. This peak was actually somewhat higher than that found in the mature unmanaged stands. Shannon's diversity for the older unmanaged stands was 1.37. The diversity levels of the lower strata seemed to be adversely affected by the diversity levels of the upper canopy in stands older than 10 years of age. As species richness increased in the upper canopy, diminished species richness in the lower strata was a result (fig. 1). Although many species which are generally considered to be midstory species were present in these younger stands, they occupied either lower or upper canopy positions.

Woody species diversity in the lower canopy level actually peaked twice during the chronosequence. The first peak was at about 10 years, with a Shannon's diversity index

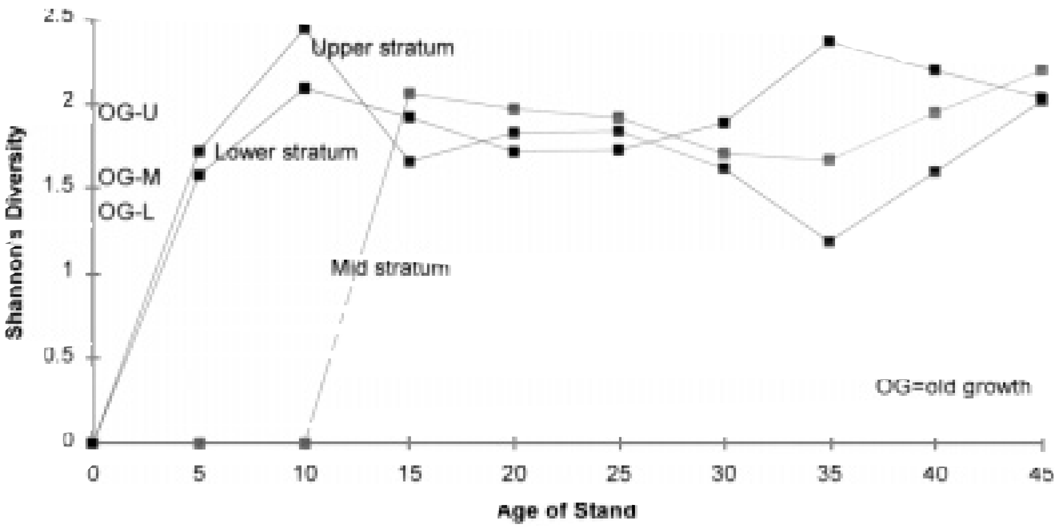

Figure 1—Shannon's diversity index for each canopy strata for stands between 1 and 45 years of age.

value of 2.09. The second peak occurred at about 35 years, with a Shannon's diversity index value of 2.37.

The diversity levels of the three strata were averaged together according to 5-year age intervals to determine total site woody plant diversity. Again, diversity levels were highest in the 5- to 10-year age interval (fig. 2). The high diversity levels during this time period were attributed to the high diversity of the lower stratum. Diversity levels in the lower stratum are very dynamic, responding to flood events with drastic decreases. This leads to decreased stability with respect to species diversity within this stratum (May 1977). When the diversity of the lower stratum was excluded from the analysis, diversity was highest during the 10- to 15-year age interval (fig. 3). The lower stratum was the only canopy level that had its highest peak in stands over 25 years of age.

Clearcut stands were compared with older unmanaged stands. After 10 years, diversity in the clearcut stands and diversity in older unmanaged stands was very similar. Ten- to 15-year old stands and 20- to 25-year old stands were 73 percent similar to older stands, and 15- to 20-year old stands were 71 percent similar to their more mature counterparts. Stands between the ages of 25 and 40 ranged from 63 percent to 69 percent similar to the older stands (table 1). Similarity increased steadily after 45 years.

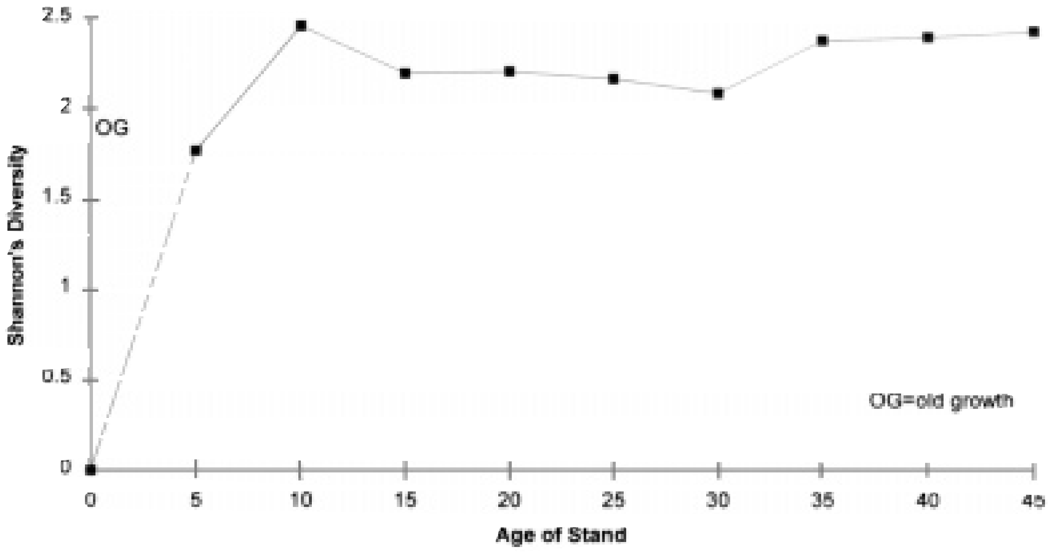

Figure 2—Shannon's diversity index for the average of all canopy strata for stands between 1 and 45 years of age.

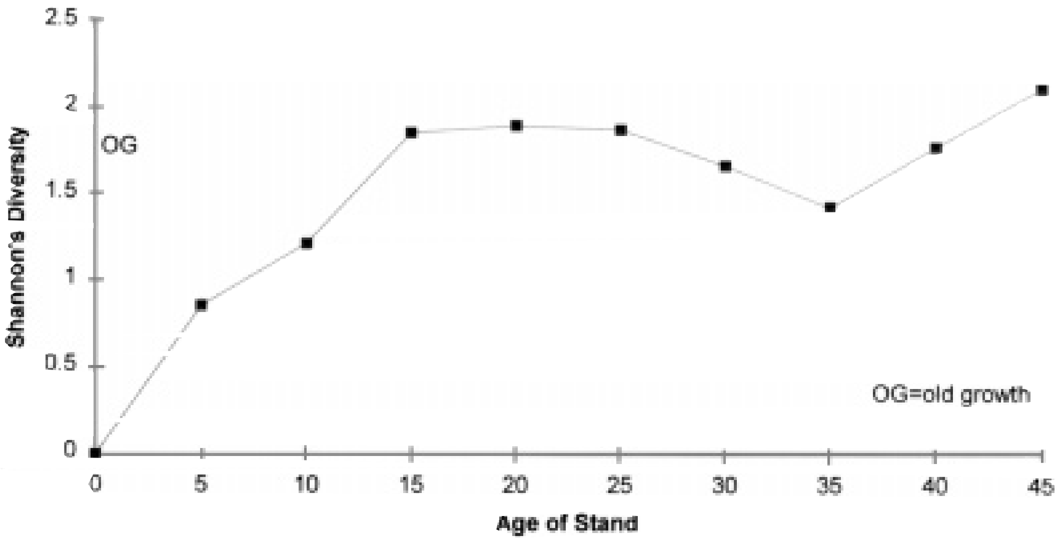

Figure 3—Shannon's diversity index for the average of the mid and upper canopy strata for stands between 1 and 45 years of age.

Table 1—Similarity between clearcuts less than 45 years of age and older unmanaged stands for all canopy levels

Age class of clearcut	Similarity to older stands
	------------*Percent*------------
5	46
10	73
15	71
20	73
25	65
30	63
35	69
45	71

DISCUSSION

Oliver and Larson (1990) best describe the stand development patterns that are taking place in this chronosequence. According to their treatment the four stages of stand development are stand initiation, stem exclusion, understory reinitiation, and old growth.

The rapid increase in diversity during the first 10 years was expected. Biological legacies from the preceding stand and pioneer species infiltrating the new stand lead to a rapid increase in the species richness on the site. The diversity peak during the stand initiation stage and subsequent drop in diversity during the stem exclusion stage is well documented (Bormann and Likens 1979, Hibbs 1983, Schoonmaker and McKee 1988). The stands which were examined in this study had higher levels of diversity at about 10 years of age than did stands over 95 years of age. The reduced amounts of light reaching the forest floor and the exclusion of many of the pioneer species found in the younger stands are a probable cause of the differences in levels of diversity (Spurr and Barnes 1964).

Another possible explanation for the reduction in main canopy diversity involves predetermined species stature. Some species exist in the main canopy of the young stand that do not have the capacity to grow to tree stature. These species are excluded in the older stand when they become overtopped by the larger species.

Canopy closure in these stands leads to the stratification of the midcanopy level. At this stage of succession the faster growing species behave like emergents in older stands. They are beginning to form the upper canopy level but have not yet reached upper stratum canopy closure. As the upper stratum approaches canopy closure the less shade tolerant species, which occupy the midstrata position, are lost and a decrease in midstratum diversity occurs. At the end of the stem exclusion stage the upper stratum has increased somewhat in diversity and has become more efficient in its canopy structure and a subsequent loss in lower stratum diversity occurs.

The lower stratum in the southern floodplain forests is very dynamic and subject to more rapid change due to flooding. Flooding is an excellent seed dispersal mechanism for many plant species (Weaver and Clements 1938), and water stress on upper stratum species allows more sunlight to infiltrate the canopy, creating a favorable environment for plant growth when the flood waters recede. The flood effect phenomenon causes fluctuation in the diversity levels of the lower stratum for individual stands and may be confounding this study, where a spatial chronosequence was developed using many stands of varying ages in a single year. The lower canopy level did react to fluctuations in diversity in the upper stratum, decreasing in diversity when the upper stratum increased.

The lower canopy level exhibited its greatest diversity between 30 and 35 years following clearcutting. Thirty to 35 years marks the beginning of the understory reinitiation stage (Oliver and Larson 1990). The lower stratum at this time is composed of both shrub species and regeneration from tree species which persists in the lower stratum but does not exhibit rapid growth. The 30- to 35-year period is also the time in which similarity to older natural stands begins to increase.

The diversity level fluctuations observed during certain successional stages are not unique to southern floodplain forests. Schoonmaker and McKee (1988) observed similar patterns in the Cascade Mountains of Oregon. They found that old-growth forests had lower diversity levels than did early successional stands following clearcutting. They also observed two peaks in diversity during the early stages of succession, as was found in this study.

Spatial chronosequence studies which examine many stands of different ages during a short time period are difficult to implement with great accuracy (Oliver 1982). Factors such as site differences and preharvest stand conditions are extremely relevant (Blackman 1978). Many stands must be sampled to obtain an accurate representation of an average stand on a particular topographic position. The characteristics described in this paper are not meant to describe a particular stand but should prove beneficial in developing a representation of changes in diversity over time in a lower Mississippi River valley bottomland hardwood ecosystem.

ACKNOWLEDGMENT

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LITERATURE CITED

Blackman, B.G. 1978. Site-species relationships for southern hardwoods. SAF proceedings: 354-359.

Borman, F.H.; Likens, G.E. 1979. Pattern and process in a forested ecosystem. New York: Springer-Verlag. 253 p.

Hibbs, D.E. 1983. Forty years of forest succession in central New England. Ecology. 64: 1394-1401.

Hodges, J.D. 1993. Ecology of bottomland hardwoods. Workshop to resolve conflicts in the conservation of migratory landbirds in bottomland hardwood forests: 5-11.

Hurst, G.A.; Bourland, T.R. 1978. Hardwood density and species composition in bottomland areas treated for regeneration. Southern Journal of Applied Forestry: 122-127.

Johnson, R.L.; Shropshire, F.W. 1983. Bottomland hardwoods. In: Burns, R.M.,ed. Silvicultural systems for the major forest types of the United States. Agric. Handb. 445. Washington, DC: U.S. Department of Agriculture: 175-178.

May, R.M. 1977. Thresholds and breakpoints in ecosystems with a multiplicity of stable states. Nature. 267: 471-477.

Oliver, C.D. 1982. Stand development-its uses and methods of study. In: Means, J.E., ed. Forest development and stand development research in the Northwest. Corvallis, OR: Oregon State University, Forest Research Laboratory: 100-112.

Oliver, C.D.; Larson, B.C. 1990. Forest stand dynamics. U.S.: McGraw-Hill, Inc. 467 p.

Schoonmaker, P.; McKee, A. 1988. Species composition and diversity during secondary succession of coniferous forests in the Western Cascade Mountains of Oregon. Forest Science. 34(4): 960-979.

Smith, E.B. 1988. An atlas and annotated list of the vascular plants of Arkansas. 2d. ed. Fayetteville, AR: Edwin B. Smith. 489 p.

Spurr, S.H.; Barnes, B.V. 1964. Forest ecology. 2d. ed. New York: The Ronald Press Company. 571 p.

USDA. 1976. Final environmental statement and unit plan, Delta National Forest. U.S. Department of Agriculture, Forest Service, National Forests in Mississippi, Jackson, MS. 196 p.

Weaver, J.E.; Clements, F.E. 1938. Plant ecology. New York: McGraw-Hill, Inc. 601 p.

Westman, W.E. 1990. Managing for biodiversity: unresolved science and policy questions. Bioscience. 40 (1): 26-33.

DYNAMICS OF COARSE WOODY DEBRIS IN UNMANAGED PIEDMONT FORESTS AS AFFECTED BY SITE QUALITY

John B. Crooks III, Thomas A. Waldrop, David H. Van Lear, and Victor B. Shelburne[1]

Abstract—Although coarse woody debris (CWD) is accepted as an important component of forest ecosystems, little is known about the changes in CWD loads that might be expected over long periods. This study documents CWD loadings across a chronosequence of 100+-years on three different site types in the piedmont of South Carolina. Seven age classes (0 to 3, 4 to 7, 8 to 15, 16 to 25, 26 to 50, 51 to 100, 100+) and three site types (subxeric, intermediate, and submesic) were sampled on the Clemson Experimental Forest. CWD loads were measured in five stands of each age-site combination using a modified Versiolanar Intersect Method. These data were used to test the FORCAT gap model which was modified to predict CWD accumulation. Field observations showed that CWD loadings varied by age class with smaller amounts found in stands between ages 8 and 25, but the loadings did not vary among landscape ecosystem classification (LEC) units. When comparing FORCAT to field observations, the patterns of CWD loading were similar. However, the model overpredicted CWD loads throughout the simulation period. Predicted CWD loads may have been too high because inputs from harvesting were too high and simulated decomposition rates were too low.

INTRODUCTION

As a forest ecosystem matures and progresses from one successional stage to another, many changes occur. One of these changes is the mortality of trees, which provides coarse woody debris (CWD) to the ecosystem. Nonliving woody material, i.e., any dead fallen or standing tree stem or limb material greater than 3 inches, is called CWD. CWD provides several ecosystem functions such as seed germination sites, reservoirs of moisture during droughts, sites of nutrient exchange for plant uptake, and critical habitat for a number of forest organisms (Harmon 1982, Van Lear 1996, Van Lear and Waldrop 1995). Because of these important functions, CWD should be considered by land managers and incorporated into management regimes. The amount of CWD present on a site may be affected by the type and intensity of silvicultural practices performed on that site.

Although CWD is recognized as an important structural component of forest ecosystems (Harmon and others 1986), little is known about the changes in CWD loads that might be expected over long periods. Most studies in the Southeast have provided short-term “snapshots” of CWD at specific successional stages (MacMillan 1988, Smith and Boring 1990, Muller and Liu 1991). Research on CWD accumulation over time is very limited, with most of it being conducted in the Pacific Northwest (Spies and others 1988, Spies and Cline 1988, Harmon and Hua 1991). However, Hedman and others (1996) have documented loading patterns of CWD by species over a 300+-year chronosequence in Southern Appalachian streams.

Essentially no information is available for long-term CWD dynamics in terrestrial southeastern ecosystems. One study (Waldrop 1996) used the FORCAT gap model (Waldrop and others 1986) to simulate CWD dynamics in mixed-species forests on xeric and mesic sites in east Tennessee. CWD accumulation on the two sites remained relatively low for the first 30 years after clearcutting. From years 30 to 75, there was a rapid increase in CWD accumulation. CWD continued to accumulate after year 75, but at a slower rate, and it reached a maximum at age 90. For the remainder of the 200-year simulation period, decomposition exceeded accumulation and CWD loads gradually decreased. CWD loading on the mesic site decreased much more rapidly than on the xeric site.

The objective of this study was to document CWD accumulation in a chronosequence of selected stands in the piedmont of South Carolina and to examine the effects of stand age and site on CWD accumulation. The results were then used as a rough test to verify the accuracy of the FORCAT CWD model.

METHODS

The study used five replications of a 7*3 factorial arrangement of a completely random design. Factors included stand age (using seven age classes) and site (three site types). Age classes were suggested by the results of Spies and Cline (1988) and Waldrop (1996) and included 0 to 3, 4 to 7, 8 to 15, 16 to 25, 26 to 50, 51 to 100, and over 100 years. All sampled stands in the 100+ age class were between 100 and 125 years of age according to Clemson University records. However, some of these stands were uneven-aged and may have cohorts of trees as old as 175 years. Sites were identified by using the Landscape Ecosystem Classification (LEC) system for the upper piedmont as described by Jones (1991). This system identifies each site type by a combination of soil type, slope position, and aspect. Site types in the upper piedmont include xeric, subxeric, intermediate, submesic, and mesic. In this study, CWD accumulation was measured on subxeric, intermediate, and submesic site types. Differences of CWD accumulation between age classes and site types were detected by analysis of variance (alpha = 0.05) using Duncan’s Multiple Range Test for mean separation.

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CWD accumulation was measured in selected stands on the Clemson Experimental Forest in Pickens, Oconee, and Anderson Counties of South Carolina. Sample stands were selected that have similar histories and with little or no management after stand regeneration. Each stand was regenerated (natural or artificial) to pure pine, although—as a result of no management activities—a significant hardwood component was expected in the older stands and on the better sites. Sample stands have no evidence of burning, thinning, herbicide application, or any management activity that would affect natural CWD loadings. Stands to be selected within the younger age classes (0 to 15 years) had no site preparation other than broadcast burning. Although site preparation burning reduces CWD loading to some degree, most of the consumed biomass is smaller than the 3-inch minimum diameter to be considered as CWD (Van Lear 1996). The method of site preparation was not a limitation for selecting sample stands older than 15 years. CWD decomposes rapidly in the Eastern United States and, therefore, CWD loading in stands older than 15 years should not be affected by site preparation technique (Waldrop 1996).

Sample stands for the subxeric, intermediate, and submesic site units were selected by estimating site units on the basis of slope position and aspect (Jones 1991). Once each stand was selected, a more accurate assessment of its site unit classification was made by using methods described by Jones (personal communication). This method uses three measurements to classify site types: soil rating, exposure rating, and terrain surface rating. The soil rating is affected by the depth to the B-horizon and the highest percent clay found in that horizon. The exposure rating is a result of the landform index average of eight clinometer readings from plot center to skylight, and the plot's aspect. Terrain surface rating measures the convexity or concavity of the plot. Measurements in each of these rating areas are taken on each plot, and values from 0 to 10 are assigned in each area with low values associated with mesic indicators and high values associated with xeric indicators. The three values are then summed to produce an environmental score which identifies the plot along an environmental gradient from mesic to xeric. All 105 sample plots were therefore evaluated and assigned a site unit from mesic to xeric.

One sample plot was randomly placed in each stand. Sample plots were square and 1/10 acre in size. Each side was 66 feet long. The corners and the plot center were marked with flags at each point. CWD accumulation in each sample plot was measured by the planar intersect method (Brown 1974). This technique was developed for conditions in the Western United States, but it was adapted for sites on the Clemson Experimental Forest by Sanders and Van Lear (1988). Five 50-foot-long planar transects were located within each 1/10-acre sample plot, one on each side and one along the diagonal between the fourth and second corners.

The diameter and decomposition stage (Maser and others 1979) of each piece of woody material that crossed the plane defined by the 50-foot transect was measured and recorded. The diameter of pieces of CWD larger than 1.0 inch was measured and recorded along the entire 50-foot transect. CWD that was less than 1.0 inch in diameter was measured for only the first 15 feet of the 50-foot transect. It was assumed that all CWD lies in a horizontal plane. A correction factor was used for slopes over 10 percent (Brown 1974). These CWD data were converted to biomass to produce an estimate of total amount of CWD loading.

If any snags were present in the plot, their diameter, decay class, and species type (pine or hardwood) were recorded. The condition of each snag was determined by placing it into one of five different decay classes (Maser and others 1979), where Class I is a recently dead tree and Class V is a stump with little woody material remaining. These data were then entered into diameter regression equations obtained from Clark and others (1986) and Van Lear and others (1984) to produce biomass values.

RESULTS AND DISCUSSION

A total of 105 stands (seven age classes * three site types * five replications) was selected for this study. Because of the strict criteria placed on stand selection, only 82 of these sample stands were found on the Clemson Experimental Forest. The number of sample stands was evenly distributed across age classes and site units, except for the 4 to 7 age class which had only four plots (table 1). Because the sample size for the 4- to 7-year age class was too small, these plots were excluded from the study. Therefore, this study uses only 6 age classes and a total of 78 sample plots.

Figure 1 shows loadings of CWD for each site unit within each age class. CWD loading of the 1- to 3-year age class is very high, due to logging slash left from clearcutting the previous stand. This slash decomposes very rapidly and the loading decreases to a minimum somewhere between 8 and 25 years of age. CWD loading began to increase in 16- to 25-year-old stands, probably due to canopy closure and increased mortality. This increase in loading is seen throughout the final three age classes, reaching a peak in

Table 1—Number of sample plots by age class and landscape ecosystem classification unit

	Landscape ecosystem classification unit		
Age class	Subxeric	Intermediate	Submesic
Years			
0 to 3	5	5	2
4 to 7	1	2	1
8 to 15	5	5	3
16 to 25	5	5	4
26 to 50	3	5	3
51 to 100	5	5	5
100+	4	4	5

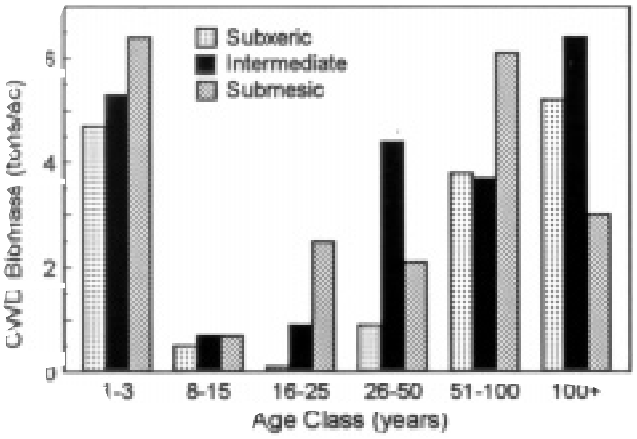

Figure 1—Coarse woody debris accumulation by age class and landscape ecosystem classification unit.

the 100+ age class. Analysis of variance showed that the loadings of the 8 to 15 and 16 to 25 age classes were significantly lower (F= 5.05, P= 0.0006) than those of the other four age classes, suggesting that CWD loadings do vary with stand age.

Differences in CWD loadings among LEC units were compared across the age classes. We expected that submesic sites would have the highest CWD loading, due to higher site productivity, and that subxeric sites would have the lowest. This expected trend held true for the first three age classes, but not for all six. A possible explanation for this trend is the different rates of succession that occur among LEC units. Because these stands are unmanaged, species composition changes over time. This change could alter the physical characteristics of the stand, the amount of debris accumulating on the site, and the decomposition rates of the debris. CWD loading may be higher on the more productive sites immediately after disturbance. However, after these periodic heavy loads decompose, the relatively high inputs on productive sites may be balanced by faster decomposition rates.

One of the objectives of this study was to use these field observations as a validation for the FORCAT model. To make a more equal comparison between the two studies, only the subxeric site field observations were used because the FORCAT model simulated loadings on xeric sites. When these two graphs (fig. 2) were compared, there were both similarities and differences. The two curves had the same general pattern of CWD loading over time. Both had a U-shaped pattern for the first half of the time period and a bell-shaped curve for the remainder. The major difference in the two curves is that the magnitude of CWD loading in the FORCAT predictions was greater than that of the field observations. This could be due to the differences in the locations of the two study sites. Since the FORCAT simulations were based on conditions of the Cumberland Plateau of east Tennessee and the field observations were obtained from the piedmont of South Carolina, differences in soil, topography, elevation, and species composition may affect the amount of CWD accumulating on the sites.

The differences in the magnitude of CWD loading could also be caused by an overestimation of the biomass of postharvest slash made by the FORCAT model. To test this possibility, field measurements from a fuel loading study (Scholl 1996) on sites in South Carolina were substituted for those estimated by FORCAT. The result (fig. 3) was that loadings became very similar to the observed field measurements of this study for the first few years after harvest. Therefore, the FORCAT model seems to overestimate postharvest CWD loadings and the need for modifications is indicated.

When comparing the two curves as in figure 2, their minimums are somewhat different. The FORCAT model predicts the minimum about 10 years later than the actual field observations. This difference may indicate that the decomposition rates of the model were too low. If the decomposition rates were too low, then the debris would be on the site longer; therefore, shifting the curve to the right. To test for this assumption, the simulated decomposition rate was increased from 6 percent to 12 percent. With this adjustment, the two curves (fig. 3) become very similar throughout the simulation period. A 12 percent decomposition rate may be unrealistically high for subxeric

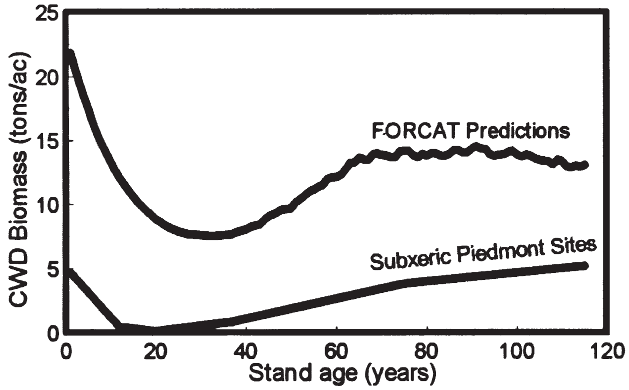

Figure 2—Comparison of coarse woody debris loadings between FORCAT predictions (using a 6 percent decomposition rate) and subxeric piedmont sites.

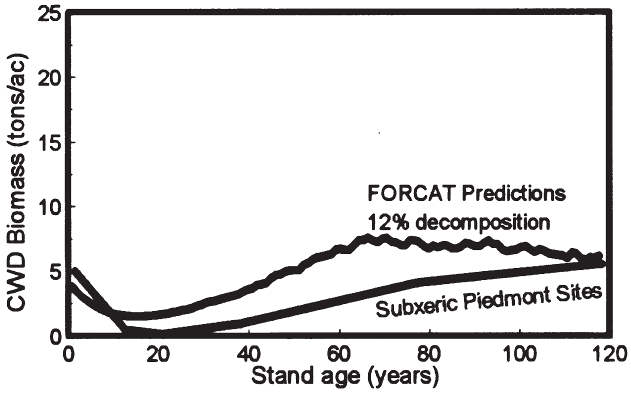

Figure 3—FORCAT loading predictions adjusted to a 12 percent decomposition rate and compared to subxeric piedmont sites.

sites, but its use here for preliminary model testing emphasizes the ability of FORCAT to predict the pattern of CWD loading.

A limitation to the FORCAT simulation study was the lack of accurate mortality and decomposition rates for each species and site type. Since the FORCAT model was tailored for a specific area and not verified by field observations, adjustments would have to be made so that it could be used for other areas. If a database of site characteristics and conditions could be established for a range of site units in different geographic areas, this model could then be adjusted to accurately represent CWD loadings on a wide array of sites.

CONCLUSIONS

For piedmont sites in South Carolina, observed field observations showed that the loading of CWD varied by age class with smaller amounts found in stands between ages 8 and 25. Consequently, these CWD loadings did not vary among LEC units. The pattern of CWD accumulation over time within the study stands was similar to a chronosequence reported by Spies and Cline (1988) and model predictions reported by Waldrop (1996).

When looking at the comparison between the FORCAT model projections and this study's observed field observations, it is seen that the FORCAT model successfully predicts the patterns of CWD accumulation over time, but it overpredicts the postharvest CWD levels after clearcutting. Also, FORCAT predicted the magnitude of CWD loading to be somewhat higher than the field observations. These predictions may have been too high because simulated decomposition rates were too low.

This study is only a preliminary validation for the FORCAT model. The model is not ready to be used immediately as a management tool, but this study does show that there is potential for its use in the future. The key to the model's success is that more research has to be conducted in the areas of decomposition rates and CWD inputs over time for various site types. If a database could be established that incorporates various site characteristics for different geographic areas, models such as FORCAT could be adjusted to compensate for these variables. This would then enable land managers to use these model projections to help determine how to alter the level or timing of their activities to enhance CWD loading on both a stand and a landscape scale.

LITERATURE CITED

Brown, J.K. 1974. A planar intersect method for sampling fuel volume and surface area. Forest Science. 17:96-102.

Clark, Alexander, III; Phillips, Douglas R.; Frederick, Douglas J. 1986. Weight, volume, and physical properties of major hardwood species in the Upland-South. Res. Pap. SE-257. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station. 55 p.

Harmon, Mark E. 1982. Decomposition of standing dead trees in the Southern Appalachian Mountains. Oecologia. 52:214-215.

Harmon, M.E.; Franklin, J.F.; Swanson, F.J. [and others]. 1986. Ecology of coarse woody debris in temperate ecosystems. Advances in Ecological Research. 15:133-302.

Harmon, M.E.; Hua, C. 1991. Coarse woody debris dynamics in two old-growth ecosystems. Bioscience. 41:604-610.

Hedman, Craig W.; Van Lear, David H.; Swank, Wayne T. 1996. In-stream large woody debris loading and riparian forest seral stage associations in the Southern Appalachian Mountains. Canadian Journal of Forest Research. 26(7):1218-1227.

Jones, S.M. 1991. Landscape ecosystem classification for South Carolina. In: Mengel, D.L.; Tew, D.T., eds. Ecological land classification: applications to identify the productive potential of southern forests. Proceedings of a symposium; 1991 January 7-9; Charlotte, NC. Gen. Tech. Rep. SE-68. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station: 59-68.

MacMillan, P.C. 1988. Decomposition of CWD in an old-growth Indiana forest. Canadian Journal of Forest Research. 18(11):1353- 1362.

Maser, C.; Anderson, R.G.; Cromack, K., Jr. [and others]. 1979. Dead and down woody material. In: Thomas, J.W., tech. ed. Wildlife habitats in managed forests: the Blue Mountains of Oregon and Washington. Agric. Handb. 553. Washington, DC: U.S. Department of Agriculture: 78-95.

Muller, R.N.; Liu, Yan. 1991. Coarse woody debris in an old-growth deciduous forest on the Cumberland Plateau, southeastern Kentucky. Canadian Journal of Forest Research. 21:1567-1572.

Sanders, B.M.; Van Lear, D.H. 1988. Photos for estimating residual loadings before and after burning in Southern Appalachian mixed pine-hardwood clearcuts.Gen. Tech. Rep. SE-49. U.S. Department of Agriculture, Forest Service. 21 p.

Scholl, Eric. 1996. Fuel classification for prescribed burning of longleaf and loblolly pine plantations in the Upper Coastal Plain. Clemson University, Clemson, SC. 62 p. M.S. thesis.

Smith, R.N.; Boring, L.R. 1990. *Pinus rigida* coarse woody debris inputs and decomposition in pine beetle gaps of the Southern Appalachians. Bulletin of the Ecological Society of America, program and abstracts, 75th annual ESA meeting; 1990 July 29-August 2; Snowbird, Utah. Supplement to vol. 71, No.2. Tempe, AZ: The Ecological Society of America: 329.

Spies, T.A.; Cline, S.P. 1988. Coarse woody debris in forest and plantations of Coastal Oregon. Chapter 1. In: Maser, C.; Tarrant, R.F.; Trappe, J.M.; Franklin, J.F., tech. eds. From the forest to the sea: a story of fallen trees. Gen. Tech. Rep. PNW-GTR-229. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station.

Spies, T.A.; Franklin, J.F.; Thomas, T.B. 1988. Coarse woody debris in Douglas-fir forests of western Oregon and Washington. Ecology. 69(6):1689-1712.

Van Lear, David H. 1996. Dynamics of coarse woody debris in southern forest ecosystems. In: McMinn, James W.; Crossley, D.A., Jr., eds. Biodiversity and coarse woody debris in southern forests, proceedings of the workshop on coarse woody debris in southern forests: effects on biodiversity; 1993 October 18-20; Athens, GA. Gen. Tech. Rep. SE-94. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 146 p.

Van Lear, David H.; Waide, J.B.; Teuke, M.J. 1984. Biomass and nutrient content of a 41-year-old loblolly pine (*Pinus taeda* L.) plantation on a poor site in South Carolina. Forest Science. 30(2):395-404.

Van Lear, David H.; Waldrop, Thomas A. 1995. Coarse woody debris considerations in southern silviculture. In: Edwards, M. Boyd, comp. Proceedings of the eighth biennial southern silvicultural research conference; 1994 November 1-3; Auburn, AL. Gen. Tech. Rep. SRS-1. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 633 p.

Waldrop, Thomas A. 1996. Dynamics of coarse woody debris—a simulation study for two southeastern forest ecosystems. In: McMinn, James W.; Crossley, D.A., Jr., eds. Biodiversity and coarse woody debris in southern forests, proceedings of the workshop on coarse woody debris in southern forests: effects on biodiversity; 1993 October 18-20; Athens, GA. Gen. Tech. Rep. SE-94. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 146 p.

Waldrop, T.A.; Buckner, E.R.; Shugart, H.H., Jr.; McGee, C.E. 1986. FORCAT: A single tree model of stand development following clearcutting on the Cumberland Plateau. Forest Science. 32(2):297-319.

EFFECTS OF SITE, DIAMETER, AND SPECIES ON EARLY DECOMPOSITION OF WOODY DEBRIS

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Abstract—Bolts of loblolly pine (*Pinus taeda* L.) 1 meter long by 20 to 30 centimeters in diameter were placed in contact with the ground on xeric, mesic, and hydric upper coastal plain sites in late April and early May. In addition, bolts of loblolly pine 4 to 6 centimeters and 40 to 50 centimeters in diameter and southern red oak (*Quercus falcata* Michx.) 20 to 30 centimeters in diameter were placed on mesic sites only. There are three replicates of each site and enough bolts to sample periodically for several years. Bolts were destructively sampled at 0, 6, 12, and 18 months to determine wood density as an indicator of decomposition. For the period sampled there was no difference in decomposition among sites, but substantial differences by size in loblolly pine, and between loblolly pine and southern red oak.

INTRODUCTION

Woody debris, comprised of standing dead snags and forest floor material, was probably a significant component of native southern forest communities. Although largely undocumented and untested, there is evidence that woody debris may be a key resource in enhancing local diversity of forest flora and fauna through various mechanisms (Maser and others 1988, Spies and Franklin 1988, McMinn and Crossley 1996). One of the primary consequences of forest management is a reduction in the size, species composition, and quantity of material. The critical role of woody debris clearly depends upon the dynamics of recruitment and decomposition. While there is significant information on the generation of woody debris, there is little on the dynamics of decomposition and how to characterize the state of the material in terms of its functional contribution. The study objectives are to assess the role of primary environmental conditions (landscape position or site) and initial conditions (size and species) on decomposition rates. This paper reports preliminary results through the first 18 months following establishment.

METHODS

The study is being conducted on the U.S. Department of Energy's Savannah River Site in the upper coastal plain of South Carolina. Effects of landscape position are being tested on the basis of three general site moisture categories: xeric, mesic, and hydric. These correspond to the following landscape ecosystem classes after Jones (1991): "xeric to subxeric," "submesic to mesic," and "well-drained bottomland," respectively. Each site category is replicated three times.

Fresh-cut bolts of loblolly pine 1 meter (m) long by 20 to 30 centimeters (cm) in diameter were placed in contact with the ground on all sites in late April and early May of 1995. In addition, fresh-cut bolts of loblolly pine 4 to 6 cm and 40 to 50 cm in diameter, and southern red oak 20 to 30 cm in diameter were placed on mesic sites only. Sufficient bolts were prepared to remove three bolts for each comparison from each experimental unit twice a year for 5 years. At 6-month intervals, bolts are destructively sampled to determine density on the basis of oven-dry weight and water-displacement estimates of volume (Barber and Van Lear 1984). As the study progresses, relative density will be used as an indicator of decay state at each sampled point in time after Christensen (1984).

RESULTS AND DISCUSSION

After 18 months, there were negligible differences among the three sites in decomposition of loblolly pine (table 1). Wood density on all sites was about 90 percent of the initial density. Different temporal patterns and overall decomposition did occur by size of material (table 2). Small material exhibited no detectable decomposition the first 6 months and increasing rates thereafter. This is attributed to rapid drying initially and an early tendency for wood moisture to follow weather patterns up to a certain stage of colonization by decay organisms. Early decomposition of large material is attributed to lower initial drying rates, but after the large bolts dry decomposition rates tend to stabilize for a period. Medium size bolts exhibited a relatively constant rate of decomposition over the observation period. Overall, after 18 months small, medium, and large loblolly pine material was 72, 90, and 80 percent, respectively, of initial density. Southern red oak was denser than loblolly pine initially and exhibited a more rapid decomposition rate (table 3). After 18 months southern red oak specific gravity was 78 percent of the initial value compared to 90 percent for loblolly pine.

Table 1—Specific gravity of loblolly pine wood initially and after 18 months on three site types

	Site		
Sample time	Xeric	Mesic	Hydric
Months	---------- Specific gravity (cm) -----------		
0	.473	.489	.482
18	.426	.438	.432

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Table 2—Wood specific gravity of three sizes of loblolly pine bolts through 18 months on mesic sites

	Bole diameter		
Sample time	4-6	20-30	40-50
Months	------- Specific gravity (cm) -----------		
0	.476	.489	.525
6	.476	.471	.462
12	.447	.460	.459
18	.343	.438	.418

Table 3—Wood specific gravity of medium size loblolly pine and southern red oak bolts through 18 months on mesic sites

	Species	
Sample time	Loblolly pine	Southern red oak
Months	---------- Specific gravity (cm) -----------	
0	.489	.576
6	.471	.548
12	.460	.523
18	.438	.451

CONCLUSIONS

For the 18-month observation period there is no evidence that site moisture influences the rate of wood decomposition. There is, however, evidence that different decomposition rates can be expected by size of woody debris and by species.

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LITERATURE CITED

Barber, B.L.; Van Lear, D.H. 1984. Weight loss and nutrient dynamics in decomposing woody loblolly pine logging slash. Soil Science Society of America Journal. 48:906-910.

Christensen, Ole. 1984. The states of woody litter determined by relative density. Oikos. 42:211-219.

Jones, Steven M. 1991. Landscape ecosystem classification for South Carolina. In: Mengel, Dennis L.; Tew, D. Thompson, eds. Ecological land classification: application to identify the productive potential of southern forests: Proceedings of a symposium. Gen. Tech. Rep. SE-68. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station: 59-62.

Maser, Chris; Tarrant, Robert F.; Trappe, James M.; Franklin, Jerry F., tech. eds. 1988. From the forest to the sea: a story of fallen trees. Gen. Tech. Rep. PNW-GTR-229. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 153 p.

McMinn, James W.; Crossley, D.A., eds. 1996. Biodiversity and coarse woody debris in southern forests. Proceedings of a workshop; 1993 October 18-20; Athens, GA. Gen. Tech. Rep. SE-94. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 146 p.

Spies, Thomas A.; Franklin, Jerry F. 1988. Old growth and forest dynamics in the Douglas-fir region of western Oregon and Washington. Natural Areas Journal. 8:190-201.

EFFECT OF SITE ON BACTERIAL POPULATIONS IN THE SAPWOOD OF COARSE WOODY DEBRIS

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Abstract—Coarse woody debris (CWD) is an important structural component of southeastern forest ecosystems, yet little is known about its dynamics in these systems. This project identified bacterial populations associated with CWD and their dynamics across landscape ecosystem classification (LEC) units. Bolts of red oak and loblolly pine were placed on plots at each of three hydric, mesic, and xeric sites at the Savannah River Station. After the controls were processed, samples were taken at four intervals over a 16-week period. Samples were ground within an anaerobe chamber using nonselective media. Aerobic and facultative anaerobic bacteria were identified using the Biolog system and the anaerobes were identified using the API 20A system. Major genera isolated were: *Bacillus, Buttiauxella, Cedecea, Enterobacter, Erwinia, Escherichia, Klebsiella, Pantoea, Pseudomonas, Serratia,* and *Xanthomonas*. The mean total isolates were determined by LEC units and sample intervals. Differences occurred between the sample intervals with total isolates of 6.67, 13.33, 10.17, and 9.50 at 3, 6, 10, and 16 weeks, respectively. No significant differences in the numbers of bacteria isolated were found between LEC units.

INTRODUCTION

Coarse woody debris (CWD) may influence a site for several decades in the form of snags, logs, chunks of wood, large branches, or coarse roots. Snags create habitat favorable for cavity nesting birds and animals. Logs in contact with ground may serve as habitat for plants, animals, fungi, and other microorganisms. The degradation process recycles nutrients in the soil and creates a fine-textured material which enhances soil nutrient and energy content, thus creating richer soils for tree growth (Harmon and others 1986, Maser and others 1988, Spies and Cline 1988). Mortality and breakage of living trees add CWD to an ecosystem while fire may remove or transform it (Van Lear 1996). Addition of CWD may occur over time as trees age and die or it may occur sporadically due to disturbances such as hurricanes, tornadoes, and insect and disease epidemics.

At a recent workshop on CWD in southern forests (McMinn and Crossley 1996), emphasis was placed on the need to manage CWD to preserve ecosystem function and health. A conclusion of the workshop was that a lack of knowledge of CWD dynamics is a major limitation to managers. With a better understanding of the CWD loads that could be expected at each stage of forest succession, managers may be able to increase loading during critical periods. Research on CWD dynamics in southern forests is limited to one study (Waldrop 1996) which used a forest-succession model to predict loading. That study suggested that CWD dynamics could be strongly influenced if inputs (limbfall or tree mortality) and outputs (decomposition) of CWD were varied between different types of forest sites. Abbott and Crossley (1982) suggested that decomposition rates vary by site quality.

Decomposition of CWD is a relatively slow process. Factors that control the rate of decay involve temperature, moisture, oxygen, carbon dioxide, and substrate quality (Harmon and others 1986). All of these factors affect the organisms that cause decomposition of CWD. Major microbial organisms that cause decomposition include fungi and bacteria. Insects grind woody components into smaller pieces but do not chemically decompose the wood. Microorganisms degrade cell contents of recently dead woody cells (sugars, starches, proteinaceous materials) and cell wall components (lignin, pectin, hemicellulose, and alpha-cellulose) (Harmon and others 1986). Another population of microorganisms develops to live on the degradation products of these dead microorganisms. Fungi have long been given credit for the majority of the decomposition of wood, but bacteria may also play an important role in the primary breakdown of some woody cell wall components and facilitate the entry of fungi to begin the major part of the decay of CWD.

This study examines the populations of bacteria that occur in CWD placed across three forest site types. These sites were defined using the landscape ecosystem classification (LEC) approach developed by Barnes and others (1982) for forests in Michigan and applied to the South Carolina upper coastal plain by Jones (1991).

This study was performed as part of a long-term, larger research project which examines the decomposition of CWD by site class and species across an environmental gradient at the Savannah River Station (SRS), Aiken, SC. The present study specifically determined the bacterial populations present during the decomposition of sapwood of CWD, by site class and species, within the first 16 weeks following placement of freshly cut wood bolts on the sites.

METHODS

Preparation of Bolts

Boles of loblolly pine (*Pinus taeda* L.) and red oak (*Quercus* spp.) between 20 and 30 centimeters (cm) in

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diameter were cut into half-meter-long bolts. The loblolly pine was collected from the Savannah River Station (Barnwell County, SC) and placed on the study plots within 2 days after cutting; the red oak was collected from the Clemson Experimental Forest (Pickens County, SC) and transported to the Savannah River Station the same day and placed on the study plots the next day.

Location of Study Sites

The USDA Forest Service provided the study plots which were located on the Savannah River Station. They were part of a larger study by McMinn (1997) on the decomposition of CWD by site class and species across a landscape environmental gradient. LEC units were chosen based on their average soil moisture regime and associated understory flora (Jones and others 1984). Three sites of each were selected and included: xeric, mesic, and hydric. The xeric sites were located in pine plantations with little or no undergrowth. The mesic sites were also located in pine plantations; however, there was more undergrowth and debris present on these sites. The hydric sites were located in mixed overstory species stands with dense understories. These latter sites were also located near streams and the soil was very moist at all times. These sites were classified and used in studies on CWD decomposition as affected by microarthropods (Bailey 1994) and fungi (Hare 1992). On each LEC unit, a square plot was established and 11 sample bolts of each species were placed on each plot.

Sample Collection

The sample bolts were collected at 3, 6, 9, and 16 weeks after placement. As a control, a separate bolt was processed immediately after the trees were felled. A randomized system for bolt selection was created for the collection of two bolts of each species from each site during the different sampling periods. The bolts were collected and taken to Clemson University for sampling, breakdown, and analysis. Overall, 162 bolts of wood were sampled.

For bacterial isolations, only the face of each bolt in contact with the ground was sampled. After washing the debarked area with ethanol, a sterile increment borer was used to extract approximately 4 to 5 grams (g) of sapwood. The cores were placed in sterilized, preweighed, screw cap tubes that were continuously filled with nitrogen. The tubes were then weighed to obtain the weight of the core sample. After all cores were collected, they were transferred to an anaerobe chamber.

Culture Preparation

Each core was ground for 5 minutes in 20 milliliters (mL) of anaerobic LPBB (Zeikus and others 1979) using a Sorvall Omni Mixer. With a sterile syringe and needle, 1 mL of this ground material was used to inoculate 9 mL of an anaerobic THAM broth tube (Schink and others 1981). Then, successive transfers were made to dilute the original inoculum. This was done in triplicate. Then, 10 microliters (µL) and 100 µL of the inoculum were used to inoculate anaerobically conditioned THAM agar culture plates. The inoculum was removed from the anaerobic chamber and used to inoculate aerobic dilution tubes of THAM broth and THAM agar plates. After all the samples were processed, the dilution tubes were incubated for 7 days at 30 °C. The anaerobic plates, which were placed in a GasPak jar, and the aerobic plates were incubated for 24 hours (hr) at 30 °C.

At 24 hrs the THAM agar plates were viewed for isolate selection which were then streaked to fresh THAM agar plates in order to obtain pure cultures. At 7 days the dilution tubes were observed for growth and the most probable number (MPN) was calculated. The last positive tube was then streaked for isolation to THAM agar plates.

Identification of Isolates

The aerobic isolates from the MPN tubes were identified using the Biolog Microstation System (Biolog Inc., Hayward, CA, 1993). Biolog is capable of identifying Gram negative, Gram positive, lactic acid bacteria, and yeasts. All isolates were grown and prepared as per Biolog instructions and incubated in microplates for 4 and 24 hours at 30 °C before readings were taken. The anaerobic isolates were identified using the Analytical Profile Index (API) 20A system (BioMerieux Vitek, 1991) which is based on 21 different biochemical reactions. All isolates were grown and prepared as per API instructions and incubated in ampules of API basal medium.

Statistical Analysis

Because several dilutions were missed on the MPN, the bacterial populations were estimated and averaged over LEC unit and sample interval and over tree species and sample interval. No statistical analysis could be performed on these results. An analysis of variance was performed to compare the number of isolates by LEC unit and tree species. Mean separation was by Duncan's Multiple Range Test. Differences were significant at $\partial = 0.05$.

RESULTS AND DISCUSSION

Aerobic Isolations

All bacteria that were isolated under aerobic conditions and taken from the last positive tube in a serial dilution are summarized in table 1. Overall, 272 organisms were isolated aerobically from the dilution tubes. Of these, 69 were omitted from the identification process for various reasons: 28 did not grow on the medium required for identification, 26 were yeasts, 1 was a fungus, 1 was fastidious, 1 was an actinomycete, and 11 were duplicates. Therefore, 203 isolates were obtained for identification.

Anaerobic Isolations

Only 67 isolates were obtained from the anaerobic dilution tubes. These isolates were also taken from the last positive tube of the dilution series. No strict anaerobes were isolated, only facultative anaerobes similar in genus to the aerobic isolates (table 2). Unlike the aerobic isolates, no yeasts were found. However, *Enterobacter* consisted of 31.7 percent of the anaerobic isolates, with *Erwinia* at 20.6 percent and *Serratia* at 17.5 percent. The only strict anaerobes found in the bolts were those taken from anaerobically conditioned plates of THAM inoculated with the original inoculum of ground core sample in LPBB under

Table 1—Totals by genus of all aerobic isolates and percentages overall of the 257 isolates and the 203 isolates identified by Biolog

Group	Total	Overall	Biolog
		---------*Percent* ---------	
Enterobacter spp.	47	18.3	23.2
Serratia spp.	21	8.2	10.3
Xanthomonas spp.	16	6.2	7.9
Klebsiella spp.	15	5.8	7.4
Erwinia spp.	15	5.8	7.4
Cedecea spp.	13	5.1	6.4
Pantoea spp.	13	5.1	6.4
Pseudomonas spp.	11	4.3	5.4
Buttiauxella spp.	8	3.1	3.9
Bacillus spp.	7	2.7	3.4
Escherichia spp.	7	2.7	3.4
Curtobacterium spp.	4	1.6	2.0
Other	8	3.1	3.9
Unidentified	18	7.0	8.9
Yeast	26	10.1	n/a
THAM	28	10.9	n/a

Table 2—Totals by genus of the anaerobic isolates and percentages overall of 63 isolates

Group	Total isolates	Percentage
Enterobacter	20	31.7
Erwinia	13	20.6
Serratia	11	17.5
Buttiauxella	3	4.8
Pantoea	3	4.8
Bacillus	2	3.2
Clavibacter	2	3.2
Cardiobacterium	1	1.6
Cedecea	1	1.6
Escherichia	1	1.6
N/I	6	9.5

N/I = Biolog could not identify.

anaerobic conditions (table 3). Only 13 anaerobes were isolated and it could not be determined if these bacteria were present in high numbers in the wood. These bacteria were identified using API 20A and only 4 of the 13 could not be matched to the database.

Differences Between LEC Units

To determine if site type affected the bacterial populations, first the total number of organisms was calculated. From serial dilution tubes, the MPN per gram of core sample was calculated for both the aerobic and anaerobic dilutions. Since many of the dilutions were missed, the estimation of bacterial populations should be higher. Using the MPN/g values, calculated averages by sample period and site type were plotted logarithmically (fig. 1). On all sites, the MPN increased dramatically between week 6 and week 10. However, this pattern and the actual MPN values were nearly identical among LEC units. Analysis of variance showed no differences between the LEC units in total isolates (table 4). The only significant difference occurred between the control and the rest of the LEC units with means of 16.5 for the control, 11.0 for mesic, 9.5 for hydric, and 9.3 for xeric. When comparing the tree species, there was no significant difference between the mean total number of isolates and LEC units. For the xeric and mesic sites the p-values are p>0.7018 and p>0.8312, respectively.

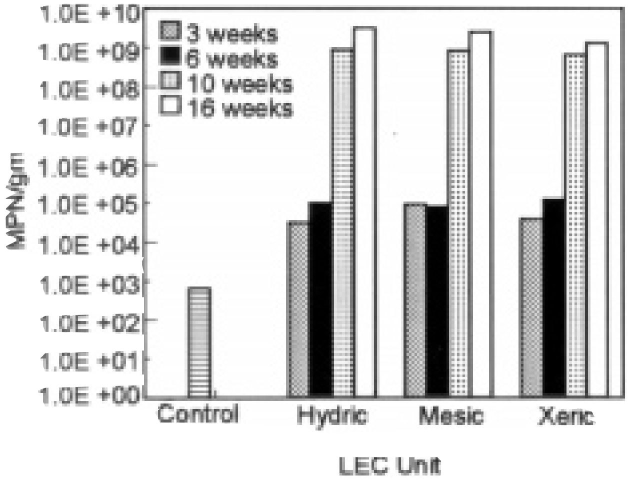

Figure 1—Averages of the most probable number of bacteria per gram of cores sampled from sapwood of red oak and loblolly pine over sample interval and LEC unit.

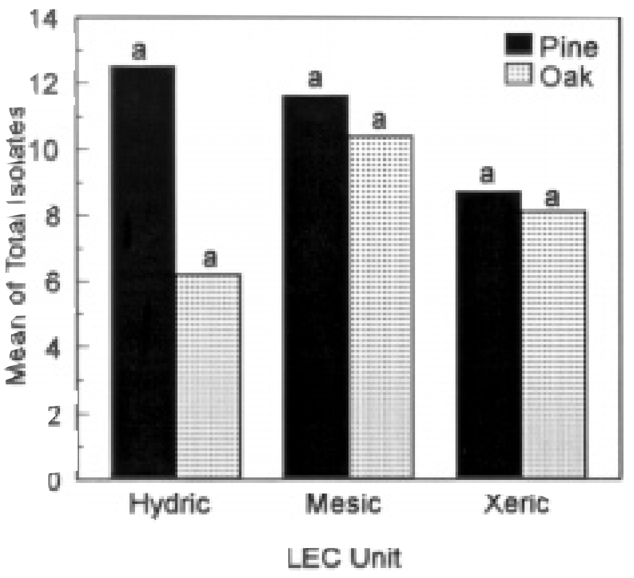

Figure 2—Means of total isolates from the sapwood of red oak and loblolly pine over LEC units. Means within a pair of bars with the same letter are not significantly different at ∂=0.05.

Table 3—Bacteria isolated anaerobically from plated samples of red oak and loblolly pine using API 20

Isolate number	Tree number	Tree species	Sample group	Site type	Bacterial organism
AA1	O-2	Oak	0	Control	*Bacteroides oralis*
AA2	O-4	Oak	0	Control	*Bacteroides oralis*
AA3	814B	Pine	1	Mesic 2	*Bifidobacillus adolescentis*
AA4	815B	Pine	1	Mesic 2	N/I
AA5	833	Pine	1	Mesic 3	*Bifidobacillus adolescentis*
AA6	975	Oak	2	Hydric 1	*Streptococcus intermedius*
AA7	867	Pine	3	Hydric 1	N/I
AA8	956A	Oak	3	Xeric 3	*Bacteroides oralis*
AA9	873	Pine	4	Hydric 1	*Clostridium beijerinckii*
AA10	895	Pine	4	Hydric 3	N/I
AA11	898	Pine	4	Hydric 3	*Actinomyces israelii*
AA12	813	Pine	4	Mesic 2	N/I
AA13	825	Pine	4	Mesic 3	*Clostridium beijerinckii*

N/I = API 20A could not identify.

Table 4—Mean total number of isolates by LEC unit

LEC unit	Number	Mean
Control	2	16.500 a
Mesic	8	11.000 b
Hydric	8	9.500 b
Xeric	8	9.250 b

Note: Means with the same letter are not significantly different at ∂=0.05.

Over the hydric sites the p-value is $p>0.0728$, indicating that there is, likewise, not a difference. However, when comparing the mean values over LEC unit the oak is nearly double that of the pine (fig. 2).

If the isolates are grouped by genus, *Erwinia* and *Xanthomonas* are prevalent in the controls (fig. 3). However, *Enterobacter* was the most abundant in all the LEC units, with the largest number isolated from the mesic site. *Serratia* was also prevalent throughout all LEC units, again with most isolates found on mesic sites. This same trend also pertained to the yeasts that were isolated.

CONCLUSIONS

Previous studies suggested that decomposition varies by site quality, possibly due to different populations or numbers of microorganisms (Abbott and Crossley 1982, Bailey 1994, Hare 1994). In this study, however, bacterial populations were not found to vary, suggesting that they play similar roles in CWD decomposition on hydric, mesic, and xeric LEC units. These results may suggest that bacteria play a minor role in CWD decomposition or that they are highly adaptable to different forested environments. This study identified a large number of facultative anaerobic bacteria from the sapwood of loblolly pine and red oak and created an extensive database of bacteria that inhabit decaying wood of these two tree species.

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LITERATURE CITED

Abbott, D.T.; Crossley, D.A., Jr. 1982. Woody litter decomposition following clearcutting. Ecology. 63:35-42.

Bailey, M.L. 1994. Soil microarthopods in pine plantations: effects of landscape ecosystem classification site unit, root mat, litterfall, moisture and nutrient additions on abundance. Athens, GA: University of Georgia. 121 page M.S. thesis.

Barnes, B.V.; Pregitzer, K.S.; Spies, T.A.; Spooner, V.H. 1982. Ecological forest site classification. Journal of Forestry. 80:493-498.

Hare, V.A. 1994. Factors influencing microbial degradation along an environmental gradient. Clemson, SC: Clemson University. 51 page M.S. Thesis.

Harmon, M.E.; Franklin, J.F.; Swanson, F.J. [and others]. 1986. Ecology of coarse woody debris in temperate ecosystems. Advances in Ecological Research. 15:133-302.

Jones, S.M. 1991. Landscape ecosystem classification for South Carolina. In: Mengel, Dennis L.; Tew, D. Thompson, eds. Proceedings of a symposium: ecological land classification: applications to identify the productive potential of southern forests. 1991 January 7-9; Charlotte, NC. Gen. Tech. Rep. SE-68. Asheville, NC:

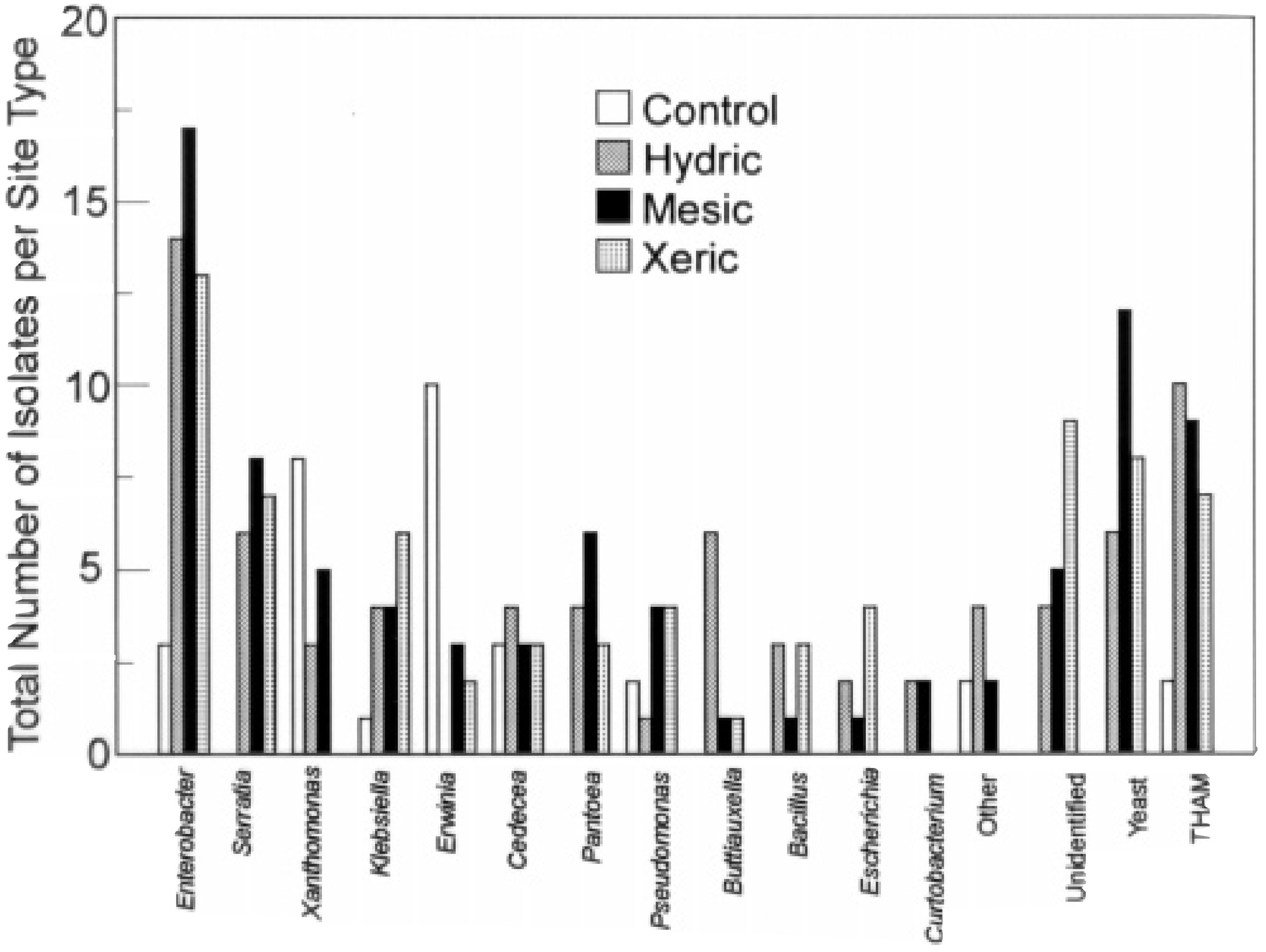

Figure 3—Total number of isolates per LEC unit by genus from the sapwood of red oak and loblolly pine over 16 weeks.

U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station: 59-68.

Jones, S.M.; Van Lear, D.H.; Cox, S.K. 1984. A vegetation-landform classification of forest sites within the upper coastal plain of South Carolina. Bulletin of the Torrey Botanical Club. 11:349-360.

Maser, C.; Cline, S.P.; Cromack, K., Jr. [and others]. 1988. What we know about large trees that fall to the forest floor. In: From the forest to the sea: A story of fallen trees. Maser, C.; Tarrant, R.F.; Trappe, J.M.; Franklin, J.F., eds. PNW-GTR-229: 25-45.

McMinn, J.W. 1997. Effects of site, diameter, and species on early decomposition of woody debris. [These proceedings.]

McMinn, J.W.; Crossley, D.A., Jr. 1996. Proceedings of biodiversity and coarse woody debris in southern forests; 1993 October 18-20; Athens, GA. Gen. Tech. Rep. SE-94. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station. 146 p.

Schink, B.; Ward, J.C.; Zeikus, J.G. 1981. Microbiology of wetwood: Role of anaerobic bacteria populations in living trees. Journal of General Microbiology. 123:313-322.

Spies, T.A.; Cline, S.P. 1988. Coarse woody debris in forests and plantations of coastal Oregon. In: From the forest to the sea: A story of fallen trees. Maser, C.; Tarrant, R.F.; Trappe, J.M.; Franklin, J.F., eds. PNW-GTR-229: 5-25.

Van Lear, D.H. 1996. Dynamics of coarse woody debris in southern forest ecosystems. In: McMinn, J.W.; Crossley, D.A., Jr., eds. Biodiversity and coarse woody debris in southern forests; 1993 October 18-20; Athens, GA. Gen. Tech. Rep. SE-94. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station: 10-17.

Waldrop. T.A. 1996. Dynamics of coarse woody debris—a simulation study of two southeastern forest ecosystems. In: McMinn, J.W.; Crossley, D.A., Jr., eds. Biodiversity and coarse woody debris in southern forests; 1993 October 18-20; Athens, GA. Gen. Tech. Rep. SE-94. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station: 18-24.

Zeikus, J.G.; Hegge, P.W.; Anderson, M.A. 1979. *Thermoanaerobium brockii* gen. nov. and sp. nov., A new chemoorganotrophic, caldoactive, anaerobic bacterium. Archives of Microbiology. 122:44-48.

IMPACT OF ECOSYSTEM MANAGEMENT ON STANDS IN THE PIEDMONT

Alexander Clark III and James W. McMinn[1]

Abstract—An ecosystem approach to managing national forests is being used to conserve biodiversity, improve the balance among forest values, and achieve sustainable conditions. This paper reports on a study established to identify the implications of ecosystem management strategies on natural pine stands in the piedmont. The impact of partial cuts, group selections, seed tree cuts, and no human disturbance on species composition, wood properties, and tree quality of pine stands in the piedmont is described.

INTRODUCTION

In the early 1990's an ecosystem approach to managing national forests was introduced to conserve biodiversity, improve the balance among forest values, and achieve sustainable, healthy conditions while retaining the esthetic, historic, and spiritual qualities of the land. Under ecosystem management, pine and pine/hardwood stands on national forests in the piedmont are being converted from even-aged monocultures to uneven-aged or two-aged pine and mixed species stands.

This paper describes a study established to monitor the effects of ecosystem management strategies on species composition, tree growth, wood properties, and tree quality of natural pine stands on national forests in the piedmont. The paper also presents initial results on species composition, wood properties, and tree quality.

METHODS

A series of permanent measurement plots have been established in loblolly (*Pinus taeda* L.) and shortleaf pine (*P. Echinata* Mill.) stands on the Oconee, Sumter, and Uwharrie National Forests to monitor the response of these stands to a range of ecosystem management practices. The management practices included: (1) partial cuts, (2) group selection cuts, (3) seed tree cuts, and (4) reserve areas. Included in the partial cuts are single tree selection, salvage cuts, stand improvement cuts, and shelterwood cuts. Reserve areas are stands in which no human disturbance is planned. Each monitoring plot is a cluster group (CG) consisting of three 1/5-acre circular plots and is randomly located within each stand selected for monitoring. Cluster groups were inventoried at establishment, then prior to and following harvest treatments. Cluster groups will also be inventoried every 5 years and after any natural disturbances. Cluster groups were established in stands representative of five 20-year age classes (1, 20, 40, 60, and 80 years) and two broad site-index (SI) classes (SI <80 and SI ≥80) for each management practice. Table 1 shows the distribution of the 49 cluster groups established in the piedmont by management practice, age class, and SI class.

Table 1—Number of cluster groups[a] established in pine and pine/hardwood stands in the piedmont by management practice, site index class, and age class

	Management practice							
	Partial cut		Group selection		Seed tree		Reserve	
Age class	SI <80	≥80	SI <80	≥80	SI <80	≥80	SI <80	≥80
Years								
20	2	3		2				
40	2	2	2		1	1		
60	5	5		2	2	3	1	1
80+	3	2		2	3	3	1	1

[a]Cluster group = three 1/5 acre-plots.

On each 1/5-acre plot, all trees ≥5.0 inches in diameter at breast height (d.b.h.) were located by azimuth and distance from plot center. Species, d.b.h., total height, merchantable height, crown class, tree grade, and defect indicators were recorded for each live and dead tree. Five 1/300-acre microplots were located 30 feet from plot center at 72° intervals within each 1/5-acre plot to tally seedlings and saplings. Seedlings (trees up to 1.0 d.b.h. inches) were tallied by species count. Saplings (trees 1.0 to 4.9 inches d.b.h.) were tallied by species, d.b.h., and total height.

Softwoods 5.0 to 8.9 inches d.b.h. and hardwoods 5.0 to 10.9 inches d.b.h. were classified as pole timber. Softwoods ≥9.0- and hardwoods ≥11.0-inches d.b.h. were classified as sawtimber if they contained one or more 16-foot saw log. Softwood sawtimber trees were placed into one of three tree grades based on a pine tree grading system for natural pine developed by Clark and McAlister (1997) and hardwood sawtimber trees were graded using USDA Forest Service hardwood tree grades (Hanks 1976).

Increment cores [5 millimeter (mm) in diameter] for wood properties analysis were extracted from bark to pith at 4.5 feet aboveground from four trees in each 1/5-acre plot. The largest diameter pine at 90° intervals in each plot was selected for boring. Increment cores were analyzed to determine tree age, earlywood and latewood annual radial

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growth, and amount of juvenile wood, sapwood, and heartwood at breast height.

Species diversity and evenness were calculated using Shannon's indexes of diversity and evenness based on species stem counts. (Magurran 1988).

RESULTS

Species Composition

Table 2 shows the average number of stems per acre, species richness, diversity, and evenness by management practice for the natural pine stands sampled. Average number of stems per acre for seedlings, saplings, and pole and sawtimber was highest in the reserve stands and lowest in the seed tree cuts. Species richness and diversity for sapling and pole and sawtimber trees were also highest in the reserve stands and lowest in the seed tree cuts, as would be expected. In the natural stands, the three most abundant species in the seedling class were red maple (*Acer rubrum* L.) (18 percent), loblolly pine (15 percent), and sweetgum (*Liquidambar styraciflua* L.) (13 percent). In the sapling class, the most abundant species were sweetgum (25 percent), loblolly pine (18 percent), and dogwood (*Cornus florida* L.) (11 percent). The most abundant species in the pole and sawtimber class were loblolly pine (57 percent), sweetgum (13 percent), and shortleaf pine (6 percent).

Number of stems per acre in the seedling class in the younger natural pine stands was lower than that found in older stands (table 3). This occurs because the young pine stands have a closed canopy and little sunlight reaches the forest floor. However, when these young stands are thinned, the crown canopy is opened and the number of stems in the seedling class increases significantly. Species diversity and evenness for seedlings, however, decreases in the older stands (table 3). Species diversity and evenness for pole and sawtimber trees was lowest in the young stands and highest in the oldest stands that have been thinned several times.

Wood Properties

The increment cores were analyzed to determine various wood properties. On national forests managing for red-cockaded woodpecker (*Picoides boreales*) (RCW) habitat the effect of management practice, site productivity, and tree characteristics on heartwood formation are important. One result of this long-term study will be increased knowledge on how and where forest managers can expect to find increased heartwood for RCW cavity habitat. The

Table 2—Average stems per acre, species richness, diversity, and evenness for seedlings, saplings, and pole and sawtimber by management practice for natural stands in the piedmont

Characteristic	Management practice: Partial cut[a]	Group selection[a]	Seed tree	Reserve
	------------	*Number*	------------	
Stands sampled	28	2	7	4
Seedlings				
(trees <1.0 in. d.b.h.)				
Stems/acre	14,992	12,210	10,674	28,960
Richness	19	16	18	20
Diversity	1.9	1.8	2.0	1.6
Evenness	0.7	0.7	0.7	0.6
Saplings				
(trees 1.0-4.9 in. d.b.h.)				
Stems/acre	446	510	346	630
Richness	5	4	3	7
Diversity	1.2	0.7	0.6	1.4
Evenness	0.8	0.6	0.5	0.7
Pole and sawtimber				
(trees ≥5 in. d.b.h.)				
Stems/acre	153	115	31	190
Richness	8	6	4	10
Diversity	1.2	1.1	0.8	1.4
Evenness	0.8	0.6	0.5	0.6

[a] Stand conditions before harvest.

Table 3—Average stand characteristics and species richness, diversity, and evenness for seedlings, saplings, and pole and sawtimber by stand-age class for partial cut and group selection natural pine stands

Characteristic	Stand-age class 20 years	40 years	60 years	80 years
	Number			
Stands sampled	2	4	16	8
Seedlings				
(trees <1.0 in. d.b.h.)				
Stems/acre	5,060	13,295	16,716	14,180
Richness	17	19	20	19
Diversity	2.3	1.9	2.0	1.8
Evenness	0.8	0.7	0.7	0.6
Sapling				
(trees 1.0-4.9 in. d.b.h.)				
Stems/acre	550	725	310	570
Richness	4	7	4	7
Diversity	1.2	1.5	1.0	1.4
Evenness	0.9	0.8	0.8	0.8
Pole and sawtimber				
(trees ≥5 in. d.b.h.)				
Stems/acre	287	174	122	161
Basal area/acre (ft^2)	77	90	97	113
Richness	5	8	8	10
Diversity	0.2	1.2	1.3	1.5
Evenness	0.1	0.6	0.6	0.7

RCW requires a minimum of 5 inches of heartwood at cavity height to envelop the cavity.

A regression equation developed by Clark (1994) was used to estimate heartwood diameter at 22 feet based on d.b.h., tree age, and heartwood diameter at 4.5 feet. Initial study results confirm that the diameter of heartwood at cavity height (22 feet) increases not only with tree age but with site productivity. The proportion of trees bored that had ≥5 inches of heartwood at 22 feet was higher for pines growing in stands with SI ≥80. This indicates that managers should concentrate RCW recruitment activities not only in older stands but also on high-productivity sites.

Tree Quality

The impact of harvest operations on volume of trees cut and removed, cut and not removed, accidentally downed, and the health of the residual stand are important. After monitoring plots were established in the mature pine stands, seven of the stands were harvested using some type of partial cut, two harvested using group selection, and four harvested using seed tree cuts. The monitoring plots were remeasured following tree-length harvesting. In the partial cut stands only 1 percent of the basal area marked was cut and not harvested, 1 percent was pushed down during harvest, 65 percent was left standing healthy, but 7 percent was left standing with logging damage (table 4). In the group selection cuts, 5 percent of the initial stand

Table 4—Proportion of initial stocking[a] harvested[b], cut and not removed, down during harvest, residual standing healthy, and standing with logging damage by type of management practice for natural pine stands in piedmont

Stocking before harvest	Trees harvested	Trees cut not removed	Trees down in harvest	Residual standing healthy	Standing logging damage
BA/A ft[b]	*Percent*				
Partial cut (N=7)					
105	26	1	1	65	7
Group selection (N=2)					
98	90	5	2	3	0
Seed tree (N=4)					
109	67	0	3	25	5

[a] Trees ≥ 5 in. d.b.h.
[b] Tree length logging.

basal area marked was cut but not removed. This increase in volume of trees cut and not removed was because the feller-buncher felled all the trees in the 1 1/2-acre openings into a crisscross pile and then the skidder removed the felled trees. When operating in these small openings, it appears best to cut and skid a portion of the stand at a time.

In the stands marked for seed tree cuts, all of the marked trees were removed, 3 percent of the original basal area was pushed over during the harvest, and 25 percent of the initial basal area was left standing and healthy. However, 5 percent of the initial basal area left standing contained logging damage.

SUMMARY

This paper describes a study established to monitor the effects of ecosystem management practices on species composition, wood properties, and tree quality of natural loblolly/shortleaf pine stands in the piedmont. Variation in species diversity, evenness, and richness by stand-age class and management practice are described. The effect of tree age and site productivity on heartwood formation for RCW cavity habitat is discussed. The impact for various management practices and tree length logging on proportion of basal area cut and not harvested, pushed over during harvest, and left standing with logging damage is discussed.

LITERATURE CITED

Clark, A., III. 1994. Heartwood formation in loblolly and longleaf pine for red-cockaded woodpecker nesting cavities. In: Jones, E.J., ed. Proceedings of 46th annual conference of southeastern association of fish and wildlife agencies; 1992 October 25-28; Corpus Christi, TX. 79-87.

Clark, A., III; McAlister, R.H. [In press]. A visual tree grading system for estimating southern pine lumber yield. Forest Products Journal.

Hanks, L.F. 1976. Hardwood tree grades for factory lumber. Res. Pap. NE-333. Upper Darby, PA. U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 81 p.

Magurran, A.E. 1988. Ecological diversity and its measurement. Princeton, NJ: Princeton University Press: 145-148.

DESCRIPTION OF A MONTANE LONGLEAF PINE COMMUNITY ON FORT McCLELLAN, ALABAMA

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Abstract—A montane longleaf pine (*Pinus palustris* Mill.) community on the Main Post of Fort McClellan, AL, was described and characterized. Choccolocco Mountain and its foothills contain relatively undisturbed disjunct longleaf pine populations, shaped by past logging episodes and sporadic presence of fire. Canopy and tree regeneration layers were sampled and overstory pines were aged. Aging indicated historic stand dominance by longleaf pine. Fire suppression and selective logging occurred about 80 years ago. This allowed other pine species to establish. Remnant longleaf pine maintained a seed source, resulting in a longleaf pine-dominated stand with establishment of loblolly (*P. taeda* L.), shortleaf (*P. echinata* Mill.), and Virginia pines (*P. virginiana* Mill.) and scattered large oaks and hickories. Infrequent fires allowed a vigorous hardwood component to establish in the understory. Recruitment of longleaf pine regeneration into the canopy apparently has decreased due to absence of growing season burns.

The Fort McClellan longleaf pine community represented an excellent but deteriorating remnant of the montane longleaf pine ecosystem. Timely implementation of growing season burns should provide for maintenance and restoration of a longleaf pine-dominated stand with scattered hardwoods and other pine species.

INTRODUCTION

Longleaf pine (*Pinus palustris* Mill.) forests have declined from 37.2 million hectares (ha) (ca. 1600 AD) to less than 1.3 million ha today (Landers and others 1995). Historically these communities were maintained by natural and anthropogenic fire. Fire suppression and harvesting, with conversion to agriculture or other tree species [e.g., loblolly pine (*P. taeda* L.)], have led to the widespread decline of this forest type. Currently, this type is considered threatened or endangered and restoration efforts have been initiated (Bridges and Orzell 1989, Landers and others 1995, Means and Grow 1985).

Longleaf pine ecosystems are found throughout the Coastal Plain and Piedmont, and extend into the Ridge and Valley and Mountain Provinces of the Southeastern United States (Boyer 1990). Freezing temperatures and heavy clay soils limit northern extent. Mortality can occur in adult trees that are damaged by ice storms (Lipps 1966), and fall-germinated seedlings can be killed by frost-heaving on heavier soils (Croker 1979). Longleaf pine inhabits the highest elevations (ca. 600 meters) of its range in the Blue Ridge Physiographic region (fig. 1).

Little information exists on the unique montane longleaf pine type, which occurs in the Blue Ridge and Ridge and Valley Physiographic regions of northwest Georgia and northeast Alabama. Longleaf pine occurs on ridgetops and south/southwest slopes in these Appalachian disjunct populations. It occurs in single-species or mixed stands with shortleaf (*P. echinata* Mill.) or Virginia pine (*P. virginiana* Mill.) and blackjack (*Quercus marilandica* Muenchh.) and chestnut oak (*Q. prinus* L.) (Harper 1905, Mohr 1901). Presence is related to fire and edaphic conditions.

Range limits of many Appalachian, Coastal Plain, and Piedmont plants interface in this region, resulting in unique biotic assemblages (Jones 1974, Lipps and DeSelm 1969, Mohr 1901). Plants more aligned with Appalachian or more northern regions include scarlet (*Q. coccinea* Muenchh.) and chestnut oak (Howell 1921), mountain blueberry (*Vaccinium vacillans* Torrey) (Clark 1971), and Virginia pine (Harper 1928). Coastal elements include poison oak (*Rhus toxicodendron* L.), foxglove [*Aureolaria pectinata* (Nuttall) Pennell], sensitive briar [*Schrankia microphylla (*Solander ex Smith.) McBride], and turkey oak (*Q. laevis* Walter) (Anon 1994). These species reach their regional limits by adaptations that allow survival in a more fire-prone habitat, centered around a longleaf pine-dominated overstory.

A study was initiated in summer 1994 to examine the montane longleaf pine community of Fort McClellan, AL (Maceina, 1997). Previous overview surveys indicated that this area contained an excellent but deteriorating remnant of this forest type.

STUDY OBJECTIVES

(1) Research the literature regarding the history of montane longleaf pine in the Blue Ridge and Ridge and Valley Physiographic Provinces of northwest Georgia and northeast Alabama.

(2) Determine the status of the montane longleaf pine ecosystem in Alabama, with emphasis on the community of Fort McClellan Main Post (MP), located in central northeast Alabama.

(3) Describe a montane longleaf pine community on MP. Aspects were: overstory tree composition—species and diameter at breast height (d.b.h.); overstory pine age structure; and regeneration tree species composition.

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Figure 1—Location of Blue Ridge disjuncts in northwest Georgia and northeast Alabama. Fort McClellan is in central northeast Alabama. (Adapted from Miller and Robinson 1994.)

HISTORICAL BACKGROUND OF MONTANE LONGLEAF PINE

History of Georgia Montane Longleaf Pine

Occurrence of longleaf pine in Georgia's highlands was documented by early botanists. They noted that harvesting activities were initiated prior to their observations, but that ridgetop areas were not yet exploited.

> "Some of the long-leaf pines there (Bartow County, GA) are over 2 feet in diameter, and but for their inaccessibility they would probably have been cut long ago." (Harper 1905, p. 57).

The demise of the Georgia montane longleaf pine community was tied to European settlement beginning in the 1830's (Wharton 1978). Timber harvesting took most of the longleaf pine. Fire suppression or annual burning eliminated the conditions necessary for successful regeneration.

> "Notwithstanding the abundance of long-leaf pine in the region under consideration (northwest Georgia), it seems to be very little used for lumber, and not at all for turpentine. A part of the charcoal which is made in considerable quantities in Bartow, Floyd, and Polk counties to supply iron furnaces in the vicinity doubtless comes from this species, but in Haralson and Carroll counties the only evidence I saw of its being used in any way was a few logs at a small sawmill in Bremen. It is probably not abundant enough in these highlands to make its exploitation profitable at present in competition with the much greater supply in the coastal plain. A great deal of it was doubtless destroyed in clearing the land for agricultural purposes before its timber was as much in demand as it is now." (Harper 1905, p. 59-60).

> "The southern slopes (of Floyd County, GA, the inland and upland range limit of longleaf pine) are covered with the remains of great forests of this valuable timber, interspersed with various hardwood trees and with shortleaf pines. They have been repeatedly cut for lumber and burned over by >ground fires= started in spring by farmers to provide a free range for their cattle, but the longleafs continue to reproduce themselves with a pertinacity which, if not too diligently thwarted by the blundering incompetence of county officials and the shortsighted greed of ignorant timber cutters, will in the course of a generation or two repopulate the southern mountain slopes with a new forest growth sprung from the old stock." (Andrews 1917, p. 498).

Today, relict trees are found throughout the Georgia montane longleaf pine region, remnants of a longleaf pine forest that was replaced by less fire-tolerant pine and hardwood species. Establishment of these trees changed the forest composition to a mixed pine-hardwood type.

History of Alabama Montane Longleaf Pine

In Alabama, the Blue Ridge Physiographic Province is surrounded by the Ridge and Valley and Piedmont Physiographic Provinces (fig. 1). These areas had vigorous longleaf pine forests that extended to steep ridges and slopes. Periodic fires maintained the species dominance.

> "Our long-leaf pine forests must have been burned originally at least 5 years out of 10, but very likely at irregular intervals. A fire every year on every acre might make it very difficult for any seedlings to survive, but a fire-free period for 2 or 3 years once or twice in a century might be sufficient to give a new crop of trees a start." (Harper 1943, p. 34).

Lower elevation areas were settled, and lands cleared for farming and grazing. Alabama montane longleaf pine stands were not exploited due to their remoteness and inaccessibility on steep slopes (Harlin and others 1961, Harper 1913).

> "At least 90 percent of the Blue Ridge region has never been cleared, the ground being too steep and rocky to offer much attraction to the farmer." (Harper 1913, p. 66).

Contrasted with the loss of much of Georgia's montane longleaf pine, Alabama still contains representative stands of the montane longleaf pine ecosystem. Remnant montane longleaf pine communities in Alabama approach elevational limits of the type, and occur on Shoal Creek and Talladega Ranger Districts of Talladega National Forest (TNF), Cheaha State Park, and Choccolocco Mountain. Shoal Creek is 10 kilometers east, Choccolocco Mountain forms the eastern part of MP, and Cheaha State Park and Talladega Ranger District are south of MP.

STATUS OF ALABAMA MONTANE LONGLEAF PINE

Each of the three main Alabama regions of montane longleaf pine have different management histories and land use plans. TNF ranger districts had a history of timber harvest and species type conversion. Cheaha State Park received fire protection, while MP was subjected to frequent fires.

Cheaha State Park was spared of timber harvesting activities, but fire suppression was initiated with park creation in 1927. Present ridge species are mostly Virginia pine. Upper slopes had longleaf pine pockets, but focus to maintain these stands is weak. Most prescribed fires have been fuel reduction burns in Virginia pine stands.

The montane longleaf pine forest on Shoal Creek Ranger District is now managed for red-cockaded woodpecker (RCW) (*Picoides borealis*) habitat. In the 1960's, restoration of logged ridges and slopes to longleaf pine began. Hardwoods, except flowering dogwood and persimmon, were removed and a fire program was initiated. Intense hardwood control and summer fires produced open, two-layered stands with rich herbaceous growth. Transition from ridge longleaf pine to bottom hardwood was defined by hydric conditions that limited fire extent downslope, allowing development of a dynamic ecotone between pines and hardwoods.

The Talladega Ranger District had logged much of its original longleaf pine, and replanted with loblolly pine. In the past 5 years, stand conversion to longleaf pine postharvest was mandated, with 120-year rotations to provide RCW habitat. Prescribed fires were mostly during the winter, with growing-season burns planned. Logistical and personnel concerns limited burning.

Fort McClellan Military Reservation was established in 1917. Much of the MP montane longleaf pine stands of Choccolocco Mountain and its foothills were in old-growth age structure when acquired. By the 1960's, logging had removed much of the accessible old-growth longleaf pine, but uncut pockets may remain in more inaccessible areas of MP.

The surviving MP longleaf pine forest contained an excellent remnant of a contiguous montane pine woodland system in Alabama's Blue Ridge region (Anon 1994). Military maneuvers perpetuated fire, which maintained a longleaf pine overstory and provided the conditions necessary for longleaf pine regeneration. Fires also maintained a diverse herbaceous community. Primary MP forest uses were for military maneuvers, watershed protection, and hunting.

METHODS

Stand selection on MP was accomplished with reconnaissance of longleaf pine areas on Choccolocco Mountain and its foothills. A 25-ha midslope tract was selected for detailed study.

Overstory trees were sampled in a systematic grid of circular plots, with 0.08-ha pine and nested 0.04-ha hardwood plots. All pines $1.5 centimeters (cm) d.b.h. and all hardwoods $5.1 cm d.b.h. were included in the overstory sample on 29 plots. Relative density (RD), relative basal area (RBA) and importance value (IV), [(RD + RBA)/2] were determined for overstory trees for each plot. Stand structure was evaluated using diameter distributions of species and species groups across the study area. Pine species were aged using increment cores. Species age-d.b.h. relationships were evaluated to determine stand history.

Four, 0.0002-ha nested subplots in each overstory plot sampled the tree regeneration. Maximum size for inclusion in the regeneration sample was <1.5 cm d.b.h. for pine species and <5.1 cm d.b.h. for hardwood species.

Sapling and seedling size/age groups were defined by ability to survive fire. Pine seedlings were defined as fire-tolerant if ground line diameter (GLD) was $0.8 cm or d.b.h. <1.5 cm, and hardwoods if d.b.h. $2.5 cm. Smaller pine and hardwood seedlings were considered fire susceptible.

Regeneration RD was determined for each species by the total regeneration population and by sapling size groups. These values were summed over each set of four subplots per overstory plot.

Community successional status was evaluated by comparison of overstory with sapling and total regeneration groups. Results provided the basis for formulating a fire management plan to restore and maintain the montane longleaf pine community on MP.

RESULTS AND DISCUSSION

Overstory

Stand history was revealed from the coring of pine trees. Ring counts ranged from 9 to 104, with maxima of 104 for longleaf, 70 for shortleaf, 64 for loblolly, and 40 for Virginia pines (fig. 2). The number of individuals per species cored was based on its percentage of total density. Trees were cored across all diameter classes. Of the 298 cores, 201 were longleaf, 43 were loblolly, 32 were shortleaf, and 22 were Virginia pines.

Longleaf pine had a wide range of diameters and ages with many trees in the older age classes, indicating historic dominance. Shortleaf and loblolly pine established on the stand 60 to 80 years ago. This timing coincided with logging on MP and the emergence of a fire-suppression policy which began in the 1920's. Openings and infrequent fire facilitated survival of these less fire-tolerant pine species. Virginia pine, the most fire-sensitive of the pine species on the tract, established about 40 years ago. Under a sporadic fire regime, with frequency >10 years, all pine species could survive (Chapman 1932), as evidenced by the tract's mixed pine overstory.

The pine species diameter size class distribution (fig. 3) illustrates the pattern of pine species establishment and survival on the tract. The size class distribution was a reverse j-shaped curve, with a deficiency in the 2.5 cm diameter class. Longleaf pine regeneration and recruitment into larger size classes is a concern as it represents a potential shift in dominance to other species. The low number in the 2.5 cm class indicated lack of recent recruitment by this species. About half of this size class was made of other pine species, suggesting initial species shifts from longleaf to other pines, under a sporadic and winter fire regime.

Hardwoods <5.1 cm d.b.h. were not sampled in the overstory plots, but the diameter size class distribution was consolidated into the overstory sample. Most

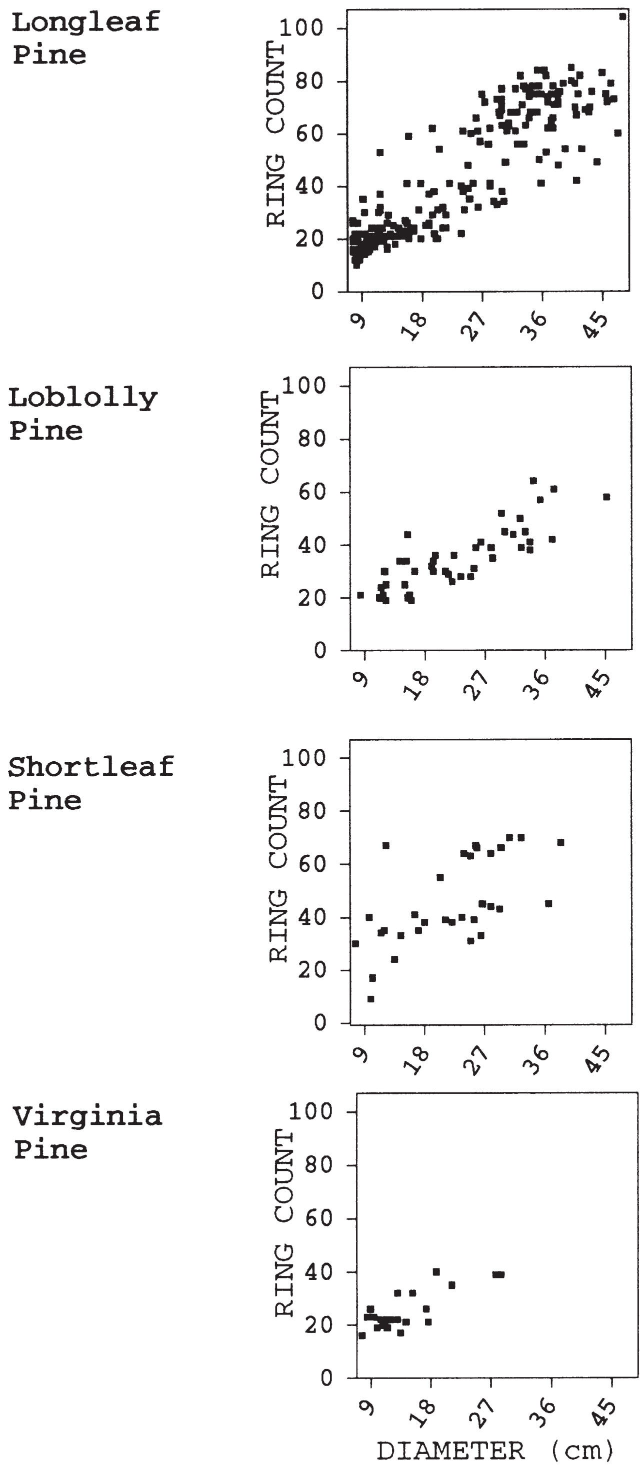

Figure 2—A comparison of diameter-ring count patterns of the four pine species on the study area.

hardwoods were in the 5 and 8 cm diameter classes (fig. 3), inferring recent establishment dates and initial species compositional shifts. Presence was related to irregular and dormant-season fires (Boyer 1990, Brender and Cooper 1968, Heyward 1939, Waldrop and Lloyd 1990).

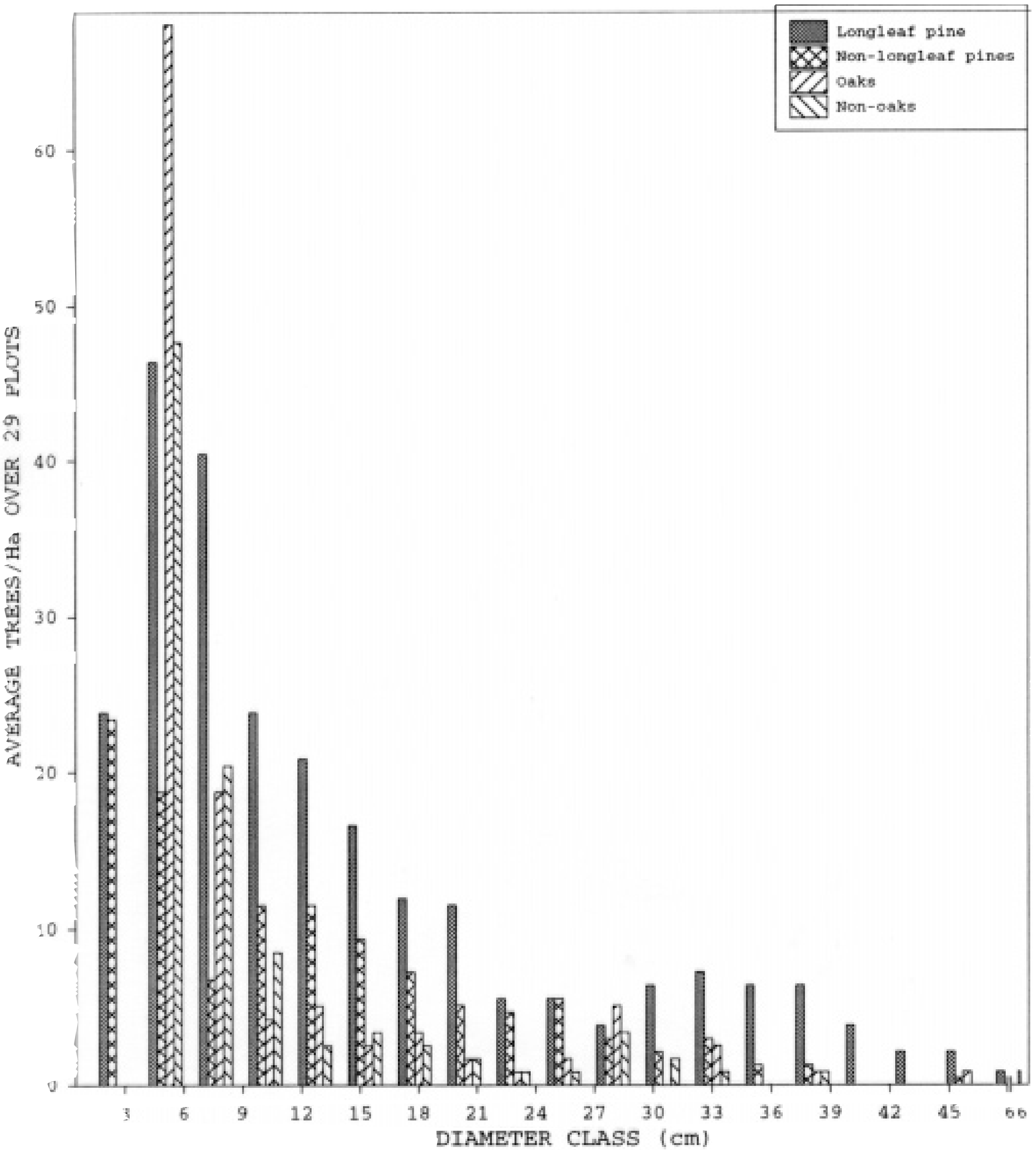

Figure 3—Composite pine and hardwood diameter size class distribution, presented in 2.5-cm increments. Pines were sampled from the 2.5-cm size class, and hardwoods from the 5.1-cm size class. The largest tree was a 65.3-cm mockernut hickory.

Understory

Sapling and overstory species compositions were compared, providing indications of successional status if the current periodic fire regime was continued or increased. Overstory longleaf pine dominated 27 plots, compared with 17 plots for the sapling size class. The longleaf pine component was declining over the tract, measured by comparison of overstory and regeneration species composition. Twelve plots with a longleaf pine-dominated overstory had no saplings in sample subplots (fig. 4). Longleaf pine saplings

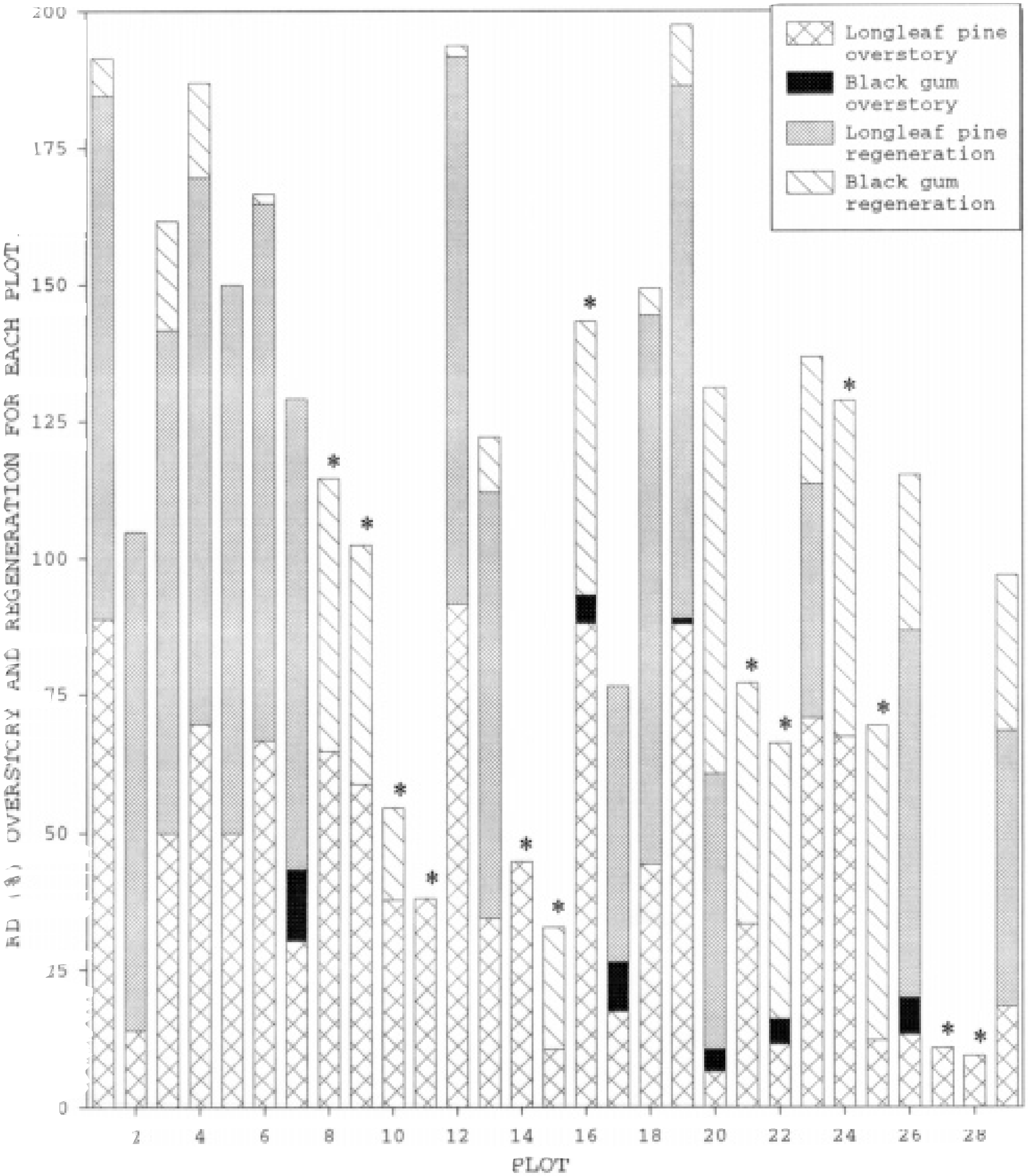

Figure 4—Longleaf pine and blackgum relative density by overstory and regeneration over the 29 plots on the study area.

strongly dominated 16 plots with ≥50 percent sapling RD, ranging from 50 to 100 percent, and 1 plot had 42.9 percent. In the sapling size class, longleaf pine saplings dominated 11 plots, 7 plots had shared pine-hardwood species dominance, and hardwood species dominated 8 plots.

When all regeneration sizes were summarized, results were very different from the sapling size class and revealed not only more plots with other pine species, but large increases in number and variety of hardwoods. Species compositional changes were suggested with sapling data but very apparent with total regeneration RD's, which included smaller sizes. This analysis indicated future overstory composition under an infrequent or dormant season fire regime.

Blackgum regeneration was observed in 25 plots compared with overstory trees in only 7 plots (fig. 4). McGee (1990) reported that blackgum responded vigorously to release, quickly colonizing openings, could grow in shade, and was susceptible to fire. Trees also propagated by root suckers. Seedling establishment on 25 plots indicated past recent sprouting or lack of recent fires.

Species shifts shown by comparisons of overstory and regeneration samples were evidenced by the decline in longleaf pine and the strong emergence of blackgum. This illustrated initial species shifts that resulted from a decrease in fire frequency and a shift to dormant season prescribed fire. The sapling regeneration population has potential to maintain the longleaf pine component over the tract, if increased frequency of fire in the growing season is initiated. The regeneration sample indicated the beginning of a species change from a pine-dominated to a hardwood-dominated forest type.

CONCLUSIONS

Longleaf pine was the dominant overstory species on Fort McClellan's Main Post. With fire suppression and logging, other pine species, followed by hardwoods, were able to establish and grow into the overstory. Evaluation of overstory species composition and size class distributions revealed initial species shifts from longleaf pine to other pine and hardwood species. Comparisons of overstory with regeneration by species composition in sapling and total regeneration size classes showed a decrease in number of plots with longleaf pines. There was an increase in other pine and hardwood species. This species shift was indicated by the strong emergence of blackgum in the regeneration sample.

Prescribed fire or other disturbance will be required to maintain the longleaf pine type and retard hardwood succession in this forest type. Fires should be of low intensity to protect present longleaf pine regeneration, but conducted at frequent intervals during the growing season to control more fire-sensitive species. Without hardwood-controlling fire or other disturbance, the longleaf pine type will not maintain its historic dominance in this system.

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LITERATURE CITED

Andrews, E.F. 1917. Agency of fire in propagation of longleaf pines. Botanical Gazette. 64:497-508.

Anonymous. 1994. Natural heritage inventory of Fort McClellan Main Post: Federal endangered, threatened, candidate species and State-listed species. Submitted to: U.S. Army Corps of Engineers, Mobile District and Fort McClellan, Anniston, AL 36201-5080. Alabama Natural Heritage Program, Alabama Dept. of Conservation and Natural Resources, State Lands Division, Montgomery, AL. 191 p.

Boyer, William D. 1990. *Pinus palustris* Mill. (longleaf pine). In: Silvics of North America I. Conifers. Agric. Handb. 654 Washington, DC: U.S. Department of Agriculture: 405-412.

Brender, Ernst V.; Cooper, Robert W. 1968. Prescribed burning in Georgia's piedmont loblolly pine stands. Journal of Forestry. 66:31- 36.

Bridges, Edwin L.; Orzell, Steve L. 1989. Longleaf pine communities of the west gulf coastal plain. Natural Areas Journal. 9:246-262.

Chapman, Herman H. 1932. Is the longleaf pine type a climax? Ecology. 13(4):328-334.

Clark, Ross C. 1971. The woody plants of Alabama. Annals of the Missouri Botanical Garden. 58:99-242.

Croker, Thomas C., Jr. 1979. Longleaf: the longleaf pine story. Journal of Forest History. 23:32-43.

Harper, Roland M. 1905. Some noteworthy stations for *Pinus palustris.* Torreya. 5(4):55-60.

_____. 1913. Economic botany of Alabama - Part 1: Geographical report on forests. Monograph 8.Tuscaloosa, AL: Geological Survey of Alabama. 228 p.

_____. 1928. Economic botany of Alabama - Part 2: Catalogue of the trees, shrubs and vines of Alabama, with their economic properties and local distribution. Monograph 9. Tuscaloosa, AL: Geological Survey of Alabama. 357 p.

_____. 1943. Forests of Alabama. Monograph 10. Tuscaloosa, AL: Geological Survey of Alabama, University of Alabama. 230 p.

Heyward, Frank. 1939. The relations of fire to stand composition of longleaf pine forests. Ecology. 20(2):287-304.

Howell, A.H. 1921. A biological survey of Alabama: 1. Physiography and life zones; II The mammals. North American fauna: Number 45. Washington, DC: U.S. Department of Agriculture, Bureau of Biological Survey. 89 p.

Jones, Samuel. B., Jr. 1974. The flora and phytogeography of the Pine Mountain region of Georgia. Castanea. 39(2):113-148.

Landers, J. Larry; Van Lear, David H.; Boyer, William D. 1995. The longleaf pine forest of the southeast: requiem or renaissance? Journal of Forestry. 93(11):39-44.

Lipps, E.L. 1966. Plant communities of a portion of Floyd County, Georgia- especially the Marshall Forest. Knoxville, TN: The University of Tennessee, Knoxville. 207 p. dissertation.

_____ ; **DeSelm, H.R.** 1969. The vascular flora of the Marshall Forest, Rome, Georgia. Castanea. 34:414-432.

Maceina, Edelgard C. 1997. Characterization of a montane longleaf pine community on Fort McClellan, Alabama: Community structure within pine-hardwood forest type. Auburn, AL: Auburn University, Auburn. 180 p. thesis.

McGee, C.E. 1990. *Nyssa sylvatica* Marsh. (Black tupelo). In: Silvics of North America. 2. Hardwoods. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture: 482-489.

Means, D. Bruce; Grow, Gerald. 1985. The endangered longleaf pine community. ENFO Report. 4:1-12.

Miller, Jim H.; Robinson, Kelly S. 1994. A regional perspective of the physiographic provinces of the southeastern United States. In: Edwards, M. Boyd, comp, proceedings of the eighth biennial southern silvicultural research conference. 1994 Nov. 1-3; Gen. Tech. Rep. SRS-1. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 633 p.

Mohr, Charles. 1901. Plant life of Alabama. U.S. Department of Agriculture, Department of Botany. Vol. 6.

Waldrop, Thomas A.; Lloyd, F. Thomas. 1990. Forty years of prescribed burning on the Santee fire plots: effects on overstory and midstory vegetation. In: Nodvin, Stephen C.; Waldrop, Thomas A., eds. Fire and the environment: ecological and cultural perspectives: Proceedings of an international symposium. 1990 March 20-24; Knoxville, TN. Gen. Tech. Rep. SE-69. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station: 45-50.

Wharton, Charles H. 1978. The natural environments of Georgia. Atlanta, GA: Georgia Department of Natural Resources, Geology and Water Resources Division and Resource Planning Section, Office of Planning and Research. 227 p.