Attachment 28

Effects on carbon storage of conversion of old-growth forests to young forests

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Article

Combustion of Aboveground Wood from Live Trees in Megafires, CA, USA

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Abstract: Biomass combustion is a major biogeochemical process, but uncertain in magnitude. We examined multiple levels of organization (twigs, branches, trees, stands, and landscapes) in large, severe forest fires to see how combustion rates for live aboveground woody parts varied with tree species, size, and fire severity in Ponderosa pine (*Pinus ponderosa* Dougl. ex Laws.) and mixed conifer-dominated forests of the Sierra Nevada, California, USA. In high severity fire patches, most combustion loss was from branches < 2 cm diameter; in low to moderate severity patches, most was from bole charring. Combustion rates decreased as fire severity declined and with increasing tree size. *Pinus* species had little branch combustion, leading them to have ≈50% the combustion rate of other taxa. Combustion rates could be 100% for small branch segments and up to 57% for small tree aboveground woody biomass in high severity fire patches. However, combustion rates are very low overall at the stand (0.1%–3.2%) and landscape level (0.6%–1.8%), because large trees with low combustion rates comprise the majority of biomass, and high severity fire patches are less than half of the area burned. Our findings of low live wood combustion rates have important implications for policies related to wildfire emissions and forest management.

Keywords: bole combustion; branch combustion; fire severity; mixed conifer forests; multi-level analysis; Sierra Nevada Mountains; wildfire combustion rates; wildfire effects; wildfire emissions

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1. Introduction

Combustion of biomass is a major biogeochemical process by which carbon is returned to the atmosphere from terrestrial ecosystems [1,2]. Fire has been an important process on Earth from at least the Silurian period over 400 million years ago [3]; since that period, this process has influenced the evolution of organisms [4,5], the successional state of ecosystems [6], and biogeochemical cycles including the concentration of oxygen and carbon dioxide in the atmosphere [7]. Although combustion is clearly a natural process that has occurred for much of Earth's history, there is also a current concern that the incidence of wildfires, due to changes in fuel loads and climate, has and will continue to increase in the future [8,9]. Increased combustion could thus serve as a positive feedback to climate change. However, there is also a concern that past estimates of vegetation combustion have been biased upward, with one study indicating that the amount of carbon released by live tree combustion has likely been overestimated by 59% to 83% [10].

The fraction of a fuel combusted in wildfire depends on the interaction of particle size, packing, moisture, distance from heat source, and rate of energy release [11–16]. Although most studies of fuel combustion are on dead biomass, theoretically they can be applied to live fuels and suggest the following mechanisms. As fire intensity (i.e., rate of energy release) increases, so do the temperatures, combustion, and woody plant mortality rates associated with fires. Fine twigs are in close proximity to leaves and other twigs (i.e., within tens of centimeters), leading to more exchange of energy than more distant plant parts;

hence, they are likely to be combusted. Moreover, their small diameter increases their surface area-to-volume ratio, allowing them to dry out and absorb energy faster than larger branches and stems. In contrast, large branch segments are not in such close proximity (i.e., separated by 10 to >100 cm), have a lower surface area-to-volume ratio, dry more slowly, and are less effective at absorbing energy from other burning particles. Tree stems or boles are separated by even more distance (i.e., often >1 m) and have a very low surface area-to-volume ratio when compared to twigs or branches; hence, they are the least likely to be combusted. As a result, the smallest diameter woody fuels are theoretically the most likely to be combusted in a wildfire [17].

Empirical estimates of the fraction of aboveground live woody vegetation combusted (as opposed to killed) by wildfires are highly uncertain, ranging from very little [10,17] to the majority [18]. It is unclear just how much of this variability in estimates is related to methodological differences as opposed to natural variation. We posit that smaller trees would have higher combustion rates than larger ones; moreover, high severity fire areas should have more combustion than low severity fire areas. Thus, natural variation in both vegetation structure and fire severity could lead to a large range in the combustion rate. However, when visual (either ground-based or remote) estimates of combustion after the fire are used, variation either among observers or methods can also introduce uncertainty.

Direct destructive methods are not practical, given the size of woody vegetation such as trees, and because combustion estimates require comparing biomass before and after a fire. Furthermore, there are hazards in working near an advancing fire front. Therefore, the most practical ways to estimate combustion rates are ground-based, post-fire estimates that can then be used to adjust existing biomass equations at the tree level (e.g., Miesel et al. [17]), make stand-level estimates of the proportion combusted (e.g., Campbell et al. [19]), and create similar types of estimates at larger spatial scales (e.g., Knorr et al. [1,2]). One challenge in making these estimates is to reconstruct the plant parts that were combusted. Another is to relate combustion estimates at one level (e.g., branches) to others (trees, stands, and landscapes); although there clearly has to be a relationship among these levels, most published estimates neither explicitly explore nor use this relationship.

Wildfires in California such as the Rim and Creek Fires have been used as an example of “fires of the future” in which wildfires become more intense, severe, and larger [20]. One of the many concerns related to such fires is the amount of carbon added to the atmosphere via combustion. Published estimates of combustion rates of aboveground live biomass from the Rim Fire, based on remote sensing, vary from 16% for low severity patches to 85% for high severity patches [18], whereas ground-based estimates from other fires [10], theoretical considerations (Figures S1 and S2), and ground-based observations (Figure S3) suggest much lower combustion rates. Therefore, our objective was to use ground-based methods to assess the fraction of aboveground woody carbon that is lost via combustion in low, moderate, and high severity patches in large fire complexes, and at multiple levels (twigs, branches and boles, individual trees, stands of trees, and the entire area in which a fire occurs), to test the following five hypotheses:

1. Smaller woody structures (e.g., twigs) would more likely to be completely combusted than larger ones (e.g., boles);
2. Combustion loss rates at the individual tree level would increase with fire severity and decrease with tree size (i.e., diameter and height);
3. High combustion rates in small trees would be countered by lower rates in the largest trees, reducing stand-level average combustion rates, because at the stand-level, larger trees contribute substantially more biomass than small ones;
4. Low and moderate severity patches are proportionally large enough and their combustion rates sufficiently low enough to reduce a fire’s average combustion rate at the landscape-level relative to that estimated for high severity patches.
5. Given Hypotheses 1–4, aboveground woody combustion at the stand to landscape levels would be an order of magnitude less than previously estimated at the Rim Fire (e.g., Garcia et al. [18]).

To address these hypotheses, we developed a more robust ground-based method to determine the fraction of live aboveground woody biomass combusted during wildfire based on a reconstruction of the biomass that was combusted from branches and from the bole (i.e., main stem), as evidenced by charring. We examined a range of fire severities and tree sizes to determine the proportional loss of branch and bole carbon. We then developed species- and size-specific models of combustion rates and applied them to a range of stand diameter distributions and fire severity spatial databases to estimate carbon losses at the stand and landscape level for multiple large fires that have been described as either high severity or catastrophic.

2. Materials and Methods

2.1. Study Area

Our field study areas were composed of mixed-conifer and *Pinus ponderosa* (Dougl. ex Laws.), which dominated forests in the Rim Fire of 2013 and the Creek Fire of 2020 within the central and southern Sierra Nevada Mountains of California, USA (Figure 1). The 104,176 ha Rim Fire was active from 17 August through 4 November of 2013 on the Stanislaus National Forest, some private lands, and the western portion of Yosemite National Park. The 153,738 ha Creek Fire was active from 4 September through 24 December of 2020, mainly on the Sierra National Forest, but including some private lands. Our field data collection locations ranged from 1370 to 1490 m elevation in the Rim Fire, and from 1520 to 2180 m elevation in the Creek Fire.

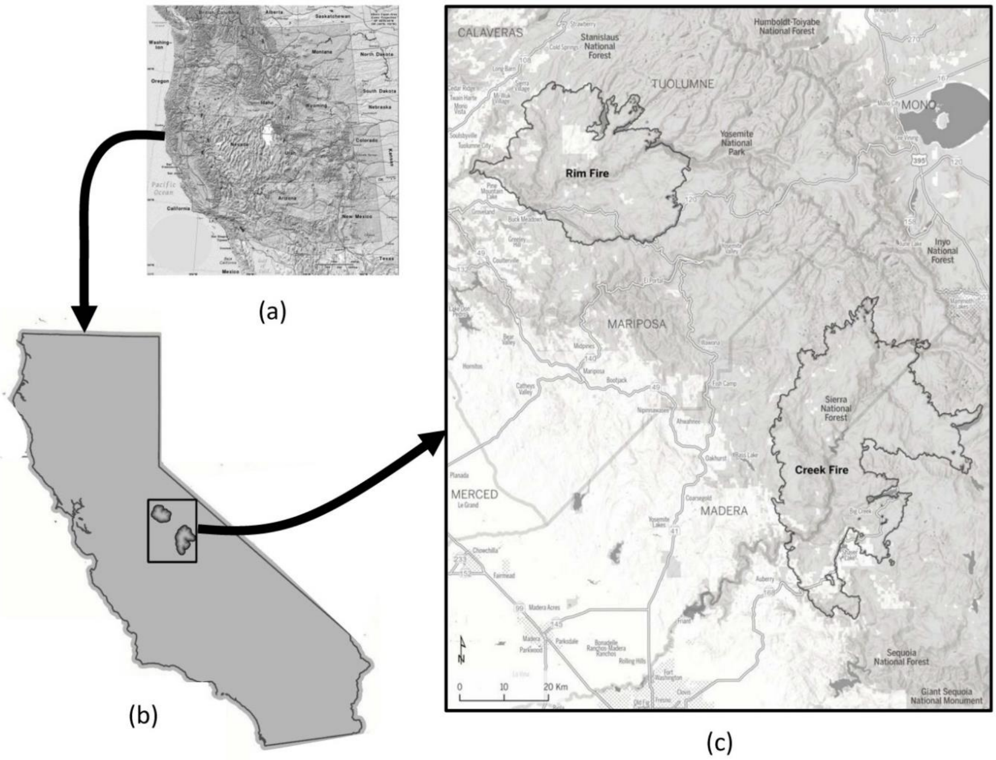


Figure 1. Location of study area in (**a**) western United States, (**b**) California, and (**c**) perimeters of 2020 Creek and 2013 Rim Fires in the Sierra Nevada Mountains, California.

At the lower elevations, these forests primarily consist of Ponderosa pine (*P. ponderosa*), white fir (*Abies concolor* (Gord. & Glend.) Lindl. ex Hildebr.), incense-cedar (*Calocedrus decurrens* (Torr.) Florin), Douglas-fir (*Pseudotsuga menziesii* (Mirbel) Franco), sugar pine (*P. lambertiana* Dougl.), and California black oak (*Quercus kelloggii* Newb.), with shrubs mainly consisting of mountain whitethorn (*Ceanothus cordulatus* Kellogg), deer brush (*C. integerrimus* Hook. & Arn.), and greenleaf manzanita (*Arctostaphylos patula* Greene). At the higher elevations in our study sites, the forests primarily consist of Jeffrey pine (*P. jeffreyi* Grev. & Balf.) and *A. concolor*, with some *C. decurrens* and occasional red fir (*A. magnifica* A. Murr.) at the highest elevations, and shrubs comprising mainly *C. cordulatus*.

2.2. Branch Models

Our branch models were based on branch orders, with first order branches representing the smallest twigs and second order branches representing branches where two first order branches joined, etc. (Figure S4). The benefit of this system is that it can be used to reconstruct the branch orders missing from those that remain given that higher order segments are more likely to remain than lower order ones.

2.2.1. Field Data for Branch Models

There were two phases of field data collection in the development of the branch models. First, branches from recently downed live trees or live tree tops of the main species present in the Rim and Creek Fires (*A. concolor*, *C. decurrens*, *P. lambertiana*, *P. ponderosa*, and *P. menziesii*) were dissected to determine the number, diameter, and length of the branch orders on individual branches. Mid-point diameter and the length were determined on a subsample of up to 10 of each branch order segments of each dissected branch; when there were <10, all segments were measured.

Second, we visually examined intact, standing trees of each of the five species listed above in each of six diameter at breast height (DBH) size classes (1.0–4.9, 5.0–9.9, 10.0–24.9, 25.0–49.9, 50.0–99.9, >100 cm) to determine branch structure, as defined by the highest branch order present on a tree, the maximum branch diameter, length of the highest branch order segment, and the total branch length of the highest branch order. Maximum branch order was determined by starting with the outmost twigs and visually assessing where they joined to form a second order, and where second orders joined to form third orders, and so on. Maximum branch diameter and lengths were estimated visually; however, these estimates were repeatedly checked against a caliper and meter stick when branches were low enough on the tree to be measured directly.

2.2.2. Branch-Level Models

The average maximum branch order, diameter, and length data, combined with the average bifurcation ratio (i.e., the number of lower order branches divided by the number of branches for the next higher order; see Figure S4), as well as segment diameter and length from the branch dissections, were used to develop tree species- and DBH size class-specific branch-level models. For lengths and diameters of orders not dissected, we interpolated between the highest order dissected and the highest order visually estimated. The volume of each branch order was calculated as the product of the branch segment area (as determined from the mid-length diameter), the length, and the number of segments. The total branch volume was determined by summing up all the branch order volumes and the relative branch order contribution was calculated as the ratio of the branch order volume to the total branch volume.

2.2.3. Tree-Level Branch Models

At the tree-level we developed three sets of models, with different weighting of the branches (Figure S5 The simplest used the relative contribution of orders found on the highest order branches of each species and DBH size class). We termed this the maximum branch order model. However, trees also contained branches that terminated at a lower order than those closest to the ground. We accounted for this by either using the total estimated volume of each branch for each DBH size class as a weighting factor (i.e., the volume-weighted model) or using an additional weighting factor related to the bole length each branch represented (i.e., the volume and length-weighted model).

2.3. Tree-Level Combustion Indicator Sampling

We collected field data in the spring of 2018 and 2021 for the Rim and Creek Fires, respectively. In both fire areas, we gathered data in each fire severity patch type (low, moderate, high) at five points separated by 50 m as determined by a laser hypsometer along straight-line transects through the middle of the patches. To determine the location of the fire severity categories, we used the U.S. Forest Service's "Rapid Assessment of Vegetation Condition after Wildfire" (RAVG) fire severity mapping system (https://fsapps.nwcg.gov/ravg/data-access, accessed on 1 May 2021). If we encountered a point location along transects that was a different fire severity category than targeted in the specific sample area, we skipped that location and proceeded to the next location, 50 m further along. Additionally, if the transect came to the end of a patch within a particular fire severity category, we changed the direction of the transect 90 degrees to stay within the patch.

At each point location, we recorded the GPS coordinates and data on individual conifer trees in six size classes (see above), using the closest tree from each size class to the point location. This theoretically resulted in 30 trees being sampled per transect. However, in some cases, certain DBH size classes were not present near a plot center. To replace these "missing" trees, we sampled others as we traveled between points within the fire severity patch. Although we did not sample by species per se, we generally found the full range of DBH size classes for the main species we encountered. In the case of *P. ponderosa,* our method did not provide sufficient numbers of small DBH trees in high severity fire patches. We therefore sampled an additional transect in a high severity patch that was dominated by *P. ponderosa* in small DBH size classes to supplement the database. In addition, because trees >100 cm DBH were uncommon, some of our point locations did not include any trees of this DBH size class. In such cases, for the Creek Fire, we sampled >100 DBH trees as we traveled between point locations.

For each tree in the Rim fire, we recorded species, DBH, height, live/dead status, distance from point location, char depth of bole, maximum bole char height, whether there was any consumption of branches (yes or no), percentage of crown length killed, diameter of the smallest branches remaining, diameter of the largest branches consumed, and percentage consumption of each order of branches. The diameter of each tree was determined using either a DBH tape or a large caliper to the nearest 1 cm. The heights of DBH size classes 3–6 were visually estimated (with periodic checks using a clinometer and tape or hypsometer), whereas those of the smaller DBH size-classes were estimated using a meter stick. The maximum height or relative height (depending on the distance) of charring was determined, as we had assumed there would be a fixed relationship between the maximum and minimum char height. The char depth on the bole was visually estimated after a subset of trees had been examined by cutting into them with either a knife or hatchet. If crown consumption was noted we visually estimated the smallest branch diameter remaining and the largest one combusted, periodically checking these estimates with a caliper. The fraction of each branch order combusted was visually estimated when crown combustion had occurred as 5%, 10%, 25%, 50%, 75%, 90%, 95%, or 100% combustion. Given that the Rim Fire occurred several years before sampling, we distinguished between branches lost via decomposition-related fragmentation and combustion (Figure S6).

In the Creek Fire, we generally followed these same methods at each point with some changes to improve precision. At the Creek Fire, all tree heights were determined either using a clinometer and hypsometer or, if they were short enough, with a meter stick. We determined the minimum and maximum char height (percentage of tree height), given our observations on the Rim Fire that the char height often varied substantially on one side of the tree versus the other side. A detailed examination of char depth at the Rim Fire indicated that the char depth was variable enough that it needed to be determined on each tree. Therefore, at the Creek Fire we used a hatchet to chop into each bole on at least two places to determine the depth of char (generally in 0.5 cm increments).

2.4. Estimate of Wildfire Intensity

We used van Wagner's scorch height model [21] to estimate I, the fire line intensity (kcals m^{-1} s^{-1}) of the sampled areas:

$$\text{Scorch height} = 0.385 \times I^{0.66} \quad (1)$$

with the maximum height of bole char (m) serving as the scorch height. For high severity fire patches, we used the maximum height observed on transects to indicate fire intensity. We did this for moderate and low severity fire patches as well; however, also we calculated the average fire intensity using all the trees in those transects because for these two severities it may be more representative than the maximum.

2.5. Tree-Level Combustion

The Jenkins et al. [22] general softwood equations were used to predict the proportion of aboveground woody mass in boles versus branches as a function of DBH (Figure S1). Data on the proportion of branch volume combusted were then combined with this to determine the fraction of each tree's total aboveground woody biomass lost to branch combustion.

We also used the data on bole char height and depth to estimate proportion of aboveground woody biomass lost to combustion. For the Rim Fire we made several estimates, since only the maximum height of bole char was noted. First, we assumed that the minimum and maximum char heights were equal, giving a maximum bole combustion estimate. Second, we used the relationship between the minimum and maximum bole char height found for a subsample of trees examined in detail on the Rim Fire (Figure S15). This gave the minimum bole combustion estimate. The data on bark char depth collected at each fire were then used to estimate the volume of char, assuming that the char depth declined as one went up the tree, reaching zero where the char ended. We assumed that 46% of the bark's C was lost via pyrolysis [23] during bole charring, based on the average of Czimczik et al. [24].

Individual tree estimates of the fraction combusted for each fire intensity class were used to develop non-linear regression models using SAS procedure NLIN with the Gauss method and parameter bounds [25] that would predict the proportion of branches, boles, and aboveground woody biomass consumed as a function of tree DBH. We estimated models for each fire severity class and species group (i.e., all species together, *Pinus* species, and other species). Species groups were based on a preliminary analysis, which indicated that species in the genus *Pinus* appeared to have substantially lower combustion rates (particularly of branches) than the other conifer species. Although we analyzed all six sets of estimates based on the branch and bole scorch models used, we largely present results for the estimates based on volume weighting of branches and maximum bole char height models.

The tree-level combustion models used were interrelated and based on negative exponential functions with multiple components. The most complicated was a negative exponential model with an asymptote that separated the response into three components:

$$\text{Combustion}_{\text{DBH}} = \text{Combustion}_1\ e^{-k1} + \text{Combustion}_2\ e^{-k2} + \text{Combustion}_3 \quad (2)$$

where Combustion_1, Combustion_2, and Combustion_3 pertain to the combustion maximums (percentage) for component 1, 2, and 3, respectively; $k1$ and $k2$ (cm^{-1}) determine the rate the first two components decline with increasing DBH. We also ran simpler models including a dual component negative exponential (i.e., Combustion_3 was zero):

$$\text{Combustion}_{\text{DBH}} = \text{Combustion}_1\ e^{-k1} + \text{Combustion}_2\ e^{-k2} \quad (3)$$

a single component, negative exponential decline to an asymptote (i.e., $k2$ was zero):

$$\text{Combustion}_{\text{DBH}} = \text{Combustion}_1\ e^{-k1} + \text{Combustion}_2 \quad (4)$$

and a single component negative exponential (i.e., Combustion_2 was zero)

$$\text{Combustion}_{\text{DBH}} = \text{Combustion}_1\ e^{-k1} \tag{5}$$

As Proc NLIN automatically reduced the number of parameters when a simpler model was sufficient to fit the data, the Akaike Information Criteria (AIC) was not necessary to select the simplest model. Therefore, our primary method to identify the "best" model was goodness of fit, which was calculated as:

$$r^2 = 1 - \text{SS}_{\text{error}}/\text{SS}_{\text{corrected total}} \tag{6}$$

where SS is the sums of squared deviations for either the error term or the corrected total. We also examined residuals to determine if there was a bias at any point in the DBH range examined and selected the model with the minimum bias.

2.6. Stand-Level Combustion

We estimated the total stand-level combustion losses for each of the fire severity classes from our combustion regression models and published data on DBH size class structure for mixed conifer forests in the California Sierra Mountains [20,26–29]. While some of the publications reported DBH size structure for species or *Pinus* versus other species, some did not [28,29]. However, the latter did report the relative contribution of species to basal area. We therefore apportioned the number of stems in each DBH size class by the proportion of basal area of species.

Total aboveground biomass was estimated from DBH using equations for species groups developed by Chojnacky et al. [30]. Biomass in branches and boles were calculated from total aboveground biomass using the Jenkins et al. [22] general softwood equations, predicting the proportion of aboveground woody mass in boles versus branches as a function of DBH. To calculate the combustion loss for each tree in a fire severity class (e.g., low severity) the appropriate biomass term (e.g., branch) was multiplied by the combustion rate, predicted from the corresponding combustion–DBH model (e.g., branch–low severity). We also estimated the fraction of combustion coming from branches versus boles.

2.7. Landscape-Level Combustion

To explore the consequences of the distribution of fire severities at the landscape-level (i.e., the entire area burned) we weighted the losses for each fire severity class at the stand-level by the abundance of the severity class on the landscape for the five largest Sierra Nevada fires occurring during 2011–2020 in conifer forests on national forest lands that were composed >50% by *P. ponderosa*, *P. jeffreyi*, and mixed-conifer forest types. In addition to the Creek Fire and Rim Fire, this included the 61386 ha Rough Fire of 2015, the 129068 ha Bear Fire of 2020, and the 70487 ha Castle Fire of 2020, all within the Sierra Nevada region. The Rapid Assessment of Vegetation Condition after Wildfire (RAVG) dataset (https://fsapps.nwcg.gov/ravg/data-access, accessed 15 November 2021) was used to determine distributions for conifer forests on national forest lands in three severity classes: low (<25% basal area mortality), moderate (25%–75% basal area mortality), and high (>75% basal area mortality).

3. Results

3.1. Tree Mortality

A total of 511 trees were examined on transects: 177 from the Creek Fire and 334 from the Rim Fire. For the high severity fire patches sampled, 99.4% of the 175 trees examined were killed by fire. For moderate severity fire patches, 68.4% of 165 the trees examined were killed, and for low severity fire patches, 47.9% of 171 trees were killed (mostly in the smallest size classes). Mortality rates, based on tree numbers, were quite similar between the two fires, with the maximum difference for low severity fire patches: 38–43% mortality of stems on the Creek Fire and 39%–52% on the Rim Fire (Table 1).

Table 1. Mortality, maximum scorch and tree height, and estimated fire intensities for Creek and Rim fires in California's Sierra Mountains.

Severity Class	Transect	Mortality [1] (%)	Maximum Scorch Height (m)	Maximum Tree Height (m)	Fire intensity Maximum (kcals m^{-1} s^{-1})	Average (kcals m^{-1} s^{-1})
			Creek Fire			
High	1	100	53	53	≥1740	NA [2]
High	2	100	45	45	≥1358	NA
Moderate	1	63	25	31	557	71
Moderate	2	72	32	61	826	156
Low	1	38	25	56	557	79
Low	2	43	25	53	557	83
			Rim Fire			
High	1	100	30	30	≥735	NA
High	2	96	30	30	≥735	NA
High	3	100	39	43	1081	NA
High	4	100	40	40	≥1136	NA
Moderate	1	68	22	31	459	87
Moderate	2	75	18	33	339	80
Moderate	3	64	10	30	139	47
Moderate	4	58	13	50	207	38
Low	1	39	10	36	139	26
Low	2	52	10	39	139	27
Low	3	50	17	38	311	30
Low	4	50	15	40	257	34

Notes: [1] On a percentage of trees examined basis; [2] An average is not applicable for high severity fire patches because scorch height is limited by tree height; hence, only the tallest trees can indicate the scorch height.

3.2. Estimates of Fire Intensity

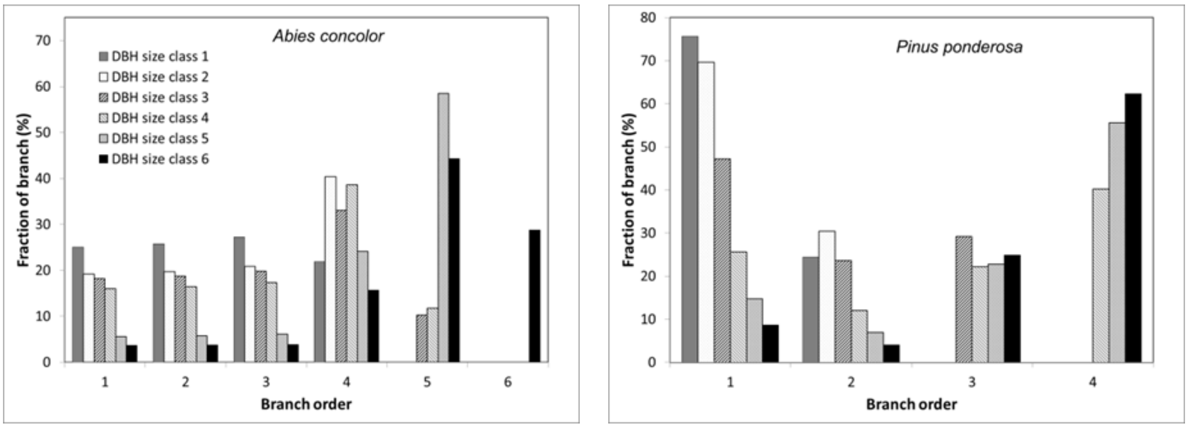
High severity fire patches had at least an order of magnitude higher intensity than moderate and low severity fire patches; however, for the most severe fire patches, the intensity estimate was limited by the heights of trees present at a site (Table 1; Figure S13). Specifically, for high severity fire patches, fire intensity estimates ranged from 735 to 1740 kcal m^{-1} s^{-1}, but for most of these patches the maximum tree height and maximum scorch height were the same. Since maximum tree height ranged from 30 to 53 m, it is likely that fire intensity was considerably higher for some of the high severity patches that we examined. Fire intensity estimates based on maximum tree height ranged from 139–826 kcal m^{-1} s^{-1} to 139–557 kcal m^{-1} s^{-1} for moderate and low severity, respectively. Average fire intensity ranged from 38–156 kcal m^{-1} s^{-1} to 26–83 kcal m^{-1} s^{-1} for moderate and low severity fire patches, respectively.

3.3. Branch Models

Diameter and length of dissected branch segments increased with order for each species (Figure S7). First order branch segment diameters ranged from 2.0 ± 0.1 (mean ± standard error) to 12.3 ± 0.3 mm for *A. concolor* and *P. ponderosa*, respectively (Table S1). Diameters of the highest branch orders on the largest DBH size class ranged from 80 ± 33 to 150 ± 3 mm, for *A. concolor* and *P. menziesii*, respectively (Figure S8b). The length of first order branch segments also varied among species, ranging from 11.2 ± 0.7 (*A. concolor*) to 36.6 ± 2.5 cm (*Pinus*) (Table S1). Average length of the highest branch order segment for the largest two DBH size classes ranged from 110 ± 4 cm for *C. decurrens* to 330 ± 30 cm for *P. lambertiana* (Figure S8d). Based on these dimensions, the highest order branch segments contained 374–62,241 times more volume than first order ones.

The bifurcation ratio on dissected branches varied among branch orders within a species, ranging from 2 to 25 (Table S2). In some species (*P. ponderosa* and *C. decurrens*), the bifurcation ratio decreased as branch order increased; however, in others (*A. concolor*, *P. lambertiana*, and *P. menziesii*), the maximum bifurcation ratio occurred between orders two and three. The highest branch order on a tree significantly varied with DBH size class and species (Figure S8a). The maximum branch order observed also varied, the lowest being 3.8 ± 0.2 for *P. ponderosa* and the highest being 6.4 ± 0.2 for *C. decurrens*. Based on the bifurcation ratios observed, first order branch segments for the largest DBH size class were 52–2755 more numerous than the highest order segments.

The relative contribution of branch orders differed among species and DBH size classes with *A. concolor* and *P. ponderosa* exhibiting the most divergent patterns (Figure 2). In general, as DBH size class increased, the contribution of the highest order branches increased and that of order one decreased. Specifically, branch order one of DBH size class one comprised 25% to 55% of the branches; however, for DBH size class six, branch order one comprised 0.5%–8.6% of the branches.



(**a**) (**b**)

Figure 2. Relative contribution of branch orders to branch volume for (**a**) *Abies concolor* (Gord. & Glend.) Lindl. ex Hildebr. and (**b**) *Pinus ponderosa* Dougl. ex Laws. for range of DBH size classes. These values are from the model that used the highest order branches found on a species-DBH size class combination.

The maximum order method estimated the lowest contribution of branch orders one to three in larger DBH size classes, whereas weighting by volume estimated the highest. However, the way that branches were weighted to produce tree-level branch models had limited effect on the relative contributions of branch orders and did not influence the overall pattern observed (Figure S9). For example, in the case of *C. decurrens* of DBH size class six, order one branch segments comprised 4% of the branches when the maximum order branch present model was used, but 5% when weighted by branch volume or when weighted by branch volume and bole length.

3.4. Branch Segment Sizes Combusted

Observations of the maximum branch segment diameter consumed by fire suggested that only the lowest few orders of branch segments were consumed even in the most severe fire patches, a finding consistent with Hypothesis 1. The largest diameter of branch segments consumed by fire was 2 cm and 1.5 cm at the Rim and Creek fires, respectively. However, the average maximum branch diameter consumed was 0.8 cm and 0.6 cm at the Rim and Creek fires, respectively. These observations of branch consumption were predominately in high severity fire patches (152 of a total of 197 branch consumption observations). Our findings suggest major limits on how much of a tree's aboveground woody biomass can be combusted (Figures S10 and S11).

The smallest diameter branch segments remaining averaged 0.1 cm and 0.3 cm on the Rim Fire and Creek Fire, respectively. As with maximum diameter consumed, the majority of these observations were on high severity fire patches, given that consumption of branches was uncommon in low and moderate severity fire areas. Although very few small diameter branch segments remained on trees in high severity patches, there were always some to be found, particularly on larger trees. For example, in high severity fire patches, 90% of the first order branches of DBH size class one were combusted on average; in contrast, <50% of first order branch segments were combusted for DBH size classes five and six. This indicated that branch combustion was far from complete even in these extreme settings.

3.5. Bole Charring

The maximum char height on tree boles was a function of both DBH and fire severity (Figure S13). For high severity fire patches, the maximum char height generally followed tree height regardless of DBH. For low fire severity, char height was generally <10 m, although some exceptions occurred. This meant that the majority of trees in low severity patches had less than half of their bole length charred.

Although we had initially assumed that fire would have consumed some of the diameter at the trees' base, we rarely saw evidence of this, except on very small diameter trees and a few large *C. decurrens*. Instead, we mostly found evidence of charring. At the Creek Fire, with a few exceptions, the range of char depth was from 0.1 to 2 cm (Figure S14); although the range was generally similar to that at the Rim Fire, more individuals in the low severity patches at the Rim Fire were assigned a char depth of zero. To some degree this may reflect the presence of unburned spots, but it also might reflect a bias caused by not chopping into each tree to distinguish between charring and surficial soot deposition. To estimate bole combustion, we therefore assigned a nominal char depth of 0.1 cm for any tree from the low severity patches of the Rim Fire that had bole scorch.

3.6. Branch Combustion

Combustion of branches declined substantially as fire severity decreased, with the maximum consumption of 100%, 71%, and 45% for the smallest DBH trees in high, moderate, and low severity fire patches, respectively (Figure S12). The average for the smallest DBH size class was lower: 71%, 11%, and 2.6% for high, moderate, and low severity fire patches, respectively. Consistent with Hypothesis 2, branch combustion declined as DBH increased, with the largest DBH size class having an average of 2% branch combustion for high severity fire patches. *Pinus* species had substantially less branch consumption than other species regardless of fire severity; this was likely due to their larger diameter in the lowest branch orders (i.e., 1–3) when compared to the other species.

Nonlinear regressions supported these conclusions, with the sum of the $\text{Combustion}_{1\text{–}3}$ parameters being lower as fire severity decreased and lower for *Pinus* species than other species (Table S3). The combustion maximums of the regressions could be substantially lower than the highest combustion estimates reported above, as not all trees in the smaller DBH's had high combustion rates. Although the majority of non-linear regressions were significant, the goodness of fit declined with fire severity from 0.3 to 0.6 and 0.06 to 0.08 for high and low severity fire patches, respectively. This was likely due to the fact that as fire severity decreased, the number of trees without branch consumption also increased. *Pinus* species in moderate to low severity patches had the only non-significant regressions; however, this was likely due to the fact that few branches on *Pinus* species were consumed at these two fire severity levels. In general, the number of parameters needed to fit the observations declined as fire severity decreased, with high severity best fit by a dual negative exponential, but low severity best fit by a single negative exponential or in some cases a constant.

3.7. Bole Combustion

Consistent with Hypothesis 2, combustion of boles also declined as fire severity decreased, with a maximum estimate of consumption of 24.6%, 13.5%, and 13.5% for the smallest DBH trees in high, moderate, and low severity fire patches, respectively (Figure S12). For the smallest DBH size class, the average bole combustion rate was 9.4%, 2.8%, and 2% in high, moderate, and low severity fire patches, respectively. For the largest DBH size class, bole combustion averaged 1.3% for high severity patches and 0.1% for low severity patches, indicating a decline by roughly an order of magnitude from the smallest to the largest trees. The differences between *Pinus* and other species in bole combustion rates was less evident than for branch combustion rates; however, the maximum bole combustion (as indicated by the sum of Combustion $_{1\text{–}3}$) for *Pinus* species was substantially lower than that of the other species, which was likely related to the thinner depth of char for *P. ponderosa* (Figure S15b).

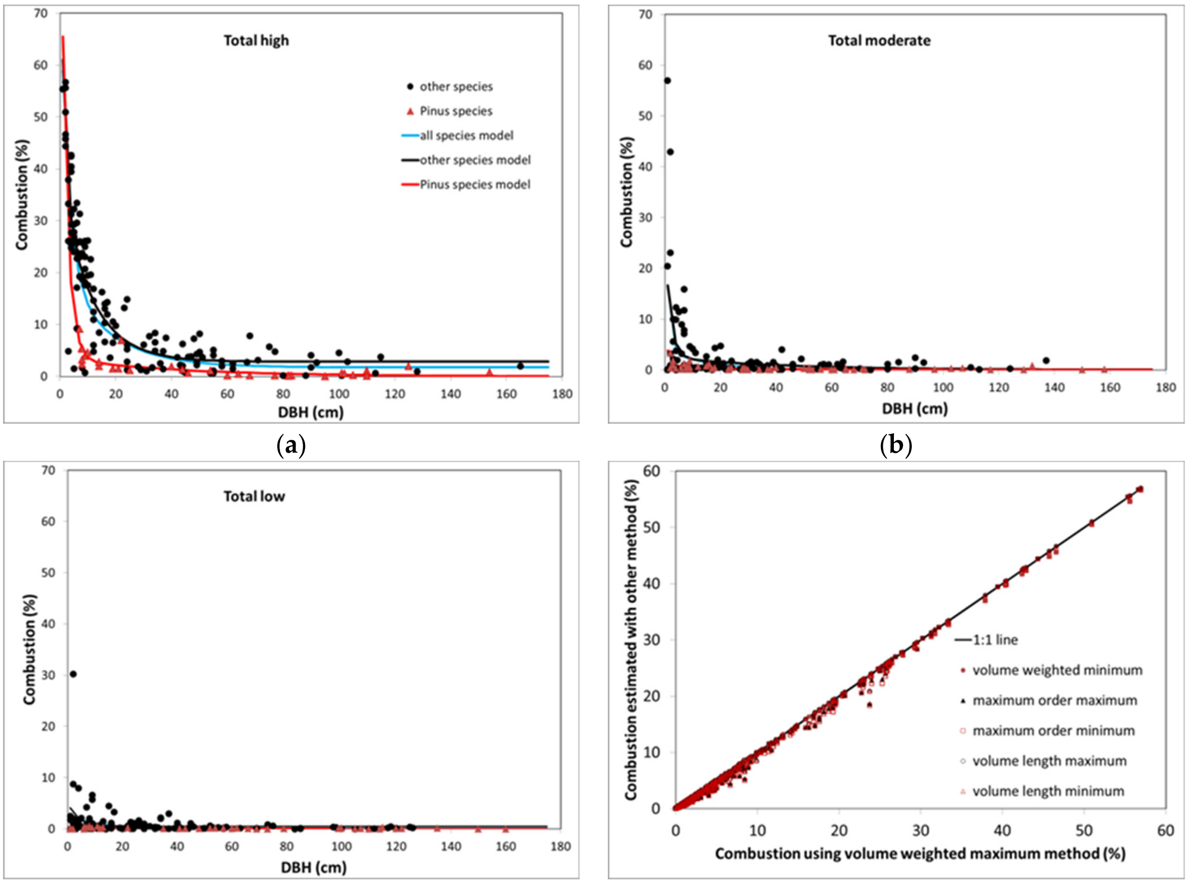
The majority of non-linear regressions for bole combustion were highly significant, but as with branch consumption the goodness of fit declined as fire severity decreased (Table S4). Specifically, the goodness of fit for high severity patches was 0.66–0.75, whereas for low severity patches it was 0.14–0.34. As with branches, consumption was predicted to decrease as DBH increased. *Pinus* species regressions had higher maximum combustion values than the other species; however, the rate that combustion declined with increasing DBH was higher, leading to a prediction of lower combustion rates for *Pinus* species over most of the DBH range.

3.8. Tree-Level Combustion

Consistent with Hypothesis 2, combustion of aboveground woody parts declined as DBH increased and fire severity decreased, with the maximum consumption observed to be 56.7%, 57%, and 30.2% for the smallest DBH trees in high, moderate, and low severity fire patches, respectively (Figure 3). The average, as opposed to the maximum, consumption for the smallest DBH size class was 34.9%, 7.2%, and 2.3% in high, moderate, and low severity fire patches, respectively. As with branches and boles, there was a substantial decline in aboveground woody consumption as DBH increased, so much so that even for high severity patches the largest DBH size class averaged 1.4% consumption. *Pinus* species generally had lower consumption rates than the other tree species, with the largest differences for smaller DBH's in high severity fire patches.

The non-linear regressions for the different species groups and fire severities were highly significant (Table 2). The sum of the Combustion$_{1\text{–}3}$ parameters declined as fire severity declined; in the case of species other than *Pinus* it was 74.3%, 26.5%, and 5.3% for high, moderate, and low severity fire patches, respectively. Although these are higher than reported above, the regression equations predict a maximum of 58.7%, 16.7%, and 4.1% for trees with a DBH of 1 cm. *Pinus* species had lower combustion than the other species, but the Combustion$_{1\text{–}3}$ parameter sums for *Pinus* species sometimes exceeded that of other species. However, *Pinus* species generally had steeper declines with an increase in DBH, leading to a lower predicted consumption rate for most of the diameter range. As with branches and boles, the goodness of fit declined as fire severity decreased; this was specifically from 0.72 to 0.80 and 0.07 to 0.20 for high and low severity fire patches, respectively.

Although Figure 3 displays the results for the volume weighted–maximum bole scorch method, the other five methods produced very similar, if somewhat lower estimates of combustion (Figure 3d). As anticipated, the maximum order–minimum bole scorch method produced the lowest estimates of combustion. The largest difference was 5% lower than the for volume weighted–maximum bole scorch method for intermediate levels of combustion, but for low and high combustion levels it was considerably less (<2%). Therefore, we conclude that the regression models in Table 2 predict the highest level of combustion of any of the methods used.



(**a**) (**b**)

(**c**) (**d**)

Figure 3. Total aboveground consumption as a function of fire severity and diameter at breast height (DBH) at Rim and Creek fires in California's Sierra Mountains: (**a**) high severity; (**b**) moderate severity; (**c**) low severity; and (**d**) estimates of total combustion using other methods when compared to estimates weighting branches by volume and assuming maximum bole scorch.

Table 2. Non-linear regressions predicting aboveground woody consumption in Creek and Rim Fires as a function of DBH for different fire severity classes and species groups.

Species Group [1]	Combustion$_1$	Combustion$_2$	*k*1	*k*2	r^2	DF	*n*	Significance [2]
			High severity					
All	55.4(8.7)	19.2(9.4) 1.7(1.2) [3]	−0.35(0.11)	−0.06(0.03)	0.80	4	173	***
Other	50.6(16.6)	34.8(8.0) 2.8(1.0) [3]	−0.65(0.31)	−0.09(0.02)	0.72	4	139	***
Pinus	100(0)	3.5(1.2)	−0.48(0.05)	−0.02(0.01)	0.72	3	33	***
			Moderate severity					
All	28.8(12.3)	3.7(3.0)	−0.81(0.43)	−0.06(0.06)	0.25	4	162	***
Other	24.0(8.1)	2.5(2.6)	−0.53(0.27)	−0.02(0.04)	0.25	4	110	***
Pinus	8.8(17.7)	0.9(0.3)	−1.07(1.01)	−0.04(0.02)	0.43	4	51	***
			Low severity					
All	2.6(0.7)		−0.09(0.04)		0.07	2	169	***
Other	4.7(2.1)	0.5 (0.3)	−0.28(0.17)		0.09	2	130	**
Pinus	0.2(0.1)		−0.01(0.01)		0.20	2	38	***

Notes: [1] The Pinus species group included Pinus jeffryi, *P. lambertina*, and *P. ponderosa*; the other species included Abies concolor, A. procera, *Calocedrus decurrens*, and *Pseudotsuga menziesii*. [2] Significance levels: NS-not significant; ** $-0.01 > p > 0.001$ >; *** -> $p > 0.001$. [3] This regression involved an additional asymptotic parameter, Combustion$_3$.

3.9. Stand-Level Combustion

Our stand-level results were consistent with Hypothesis 3. Applying the tree level combustion models to published DBH distributions indicated that stands within high severity fire patches combusts 1.2%–3.2% of the aboveground woody biomass (Figure 4a). In contrast, the same stand structure in a low severity fire patch combusted 0.1%–0.2%. The estimates for moderate fire patches were closer to low than high severity, with 0.2%–0.5% combusted. This indicates combustion rates do no correlate linearly with mortality rates.

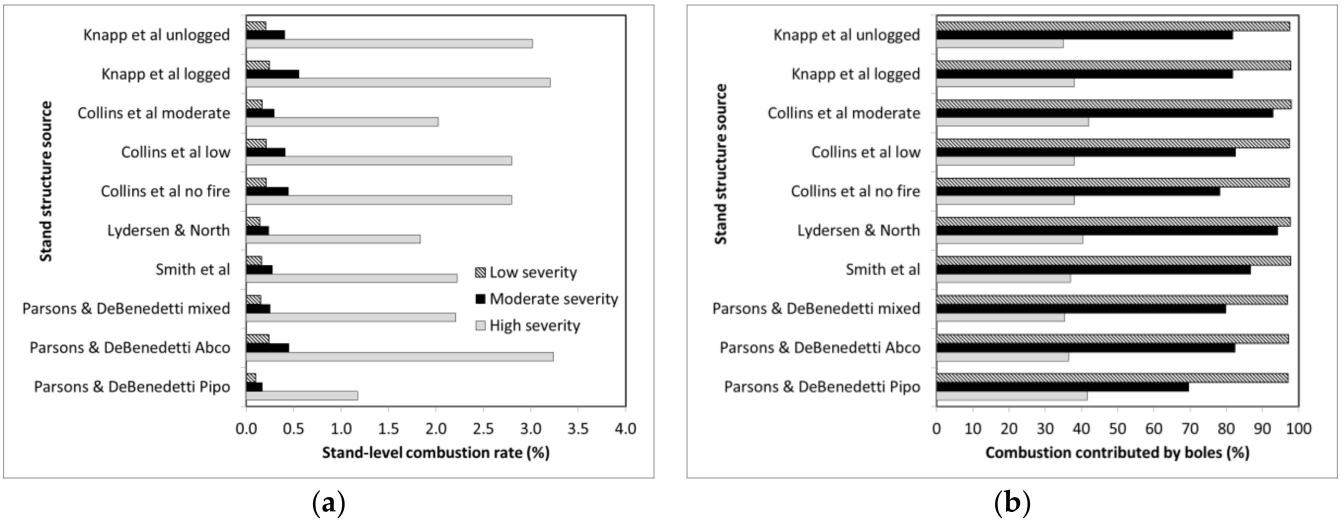


Figure 4. Stand-level combustion for published DBH size distributions for mixed conifer forests in the Sierra Nevada Mountains, California: (**a**) total combustion for high, moderate, and low severity fires; and (**b**) portion of combustion contributed by boles for high, moderate, and low severity fires. Collins et al. moderate and low severity refers to stands than had been disturbed by moderate and low severity fires in the past, respectively.

For moderate and low severity patches the majority of combustion losses (70%–98%) were related to bole combustion (Figure 4b). For high severity patches, combustion losses were predicted to largely come from branch consumption, as bole consumption was 35%–42%. This non-linear transition in combustion sources as one proceeds from high to moderate severity patches is likely the cause of the response of stand-level combustion to changes in fire severity.

3.10. Landscape-Level Combustion

For the five large fires examined RAVG indicated high severity patches comprised 45%–52% of the area, being lowest for the Rim Fire and highest on the Bear Fire (Table 3). Low severity patches (37%–44%) were more abundant than moderate severity ones (11%–16%).

Table 3. Distribution of fire severity classes as determined by RAVG for five large wildfires occurring between 2013 and 2020 in California's Sierra Mountains.

Fire Name	Year	Total Area (ha)	Low Severity [1]	Moderate Severity	High Severity
Rim	2013	104,176	0.436	0.143	0.421
Rough	2015	61,386	0.404	0.161	0.435
Bear	2020	129,068	0.373	0.111	0.516
Castle	2020	70,487	0.366	0.160	0.474
Creek	2020	153,738	0.416	0.132	0.452

Notes: [1] Low severity (<25% basal area mortality), moderate severity (25%–75% basal area mortality), and high severity (>75% basal area mortality) as determined by RAVG.

When these severity distributions were used to estimate the landscape average, we found combustion rates ranged from 0.6% to 1.8% (Figure 5). Within a given fire, the difference between the minimum and maximum combustion rate associated with different DBH structures and species composition was 1.0%–1.1%. This difference increased as the proportion of high severity patches increased. Moreover, the lowest values were for older *P. ponderosa* dominated stands and the highest combustion rate was for logged mixed conifer stands. When the mid-point value was used to represent a mix of species and DBH structures, then the landscape combustion rate would have been 1.0%–1.2%, a result consistent with Hypothesis 4.

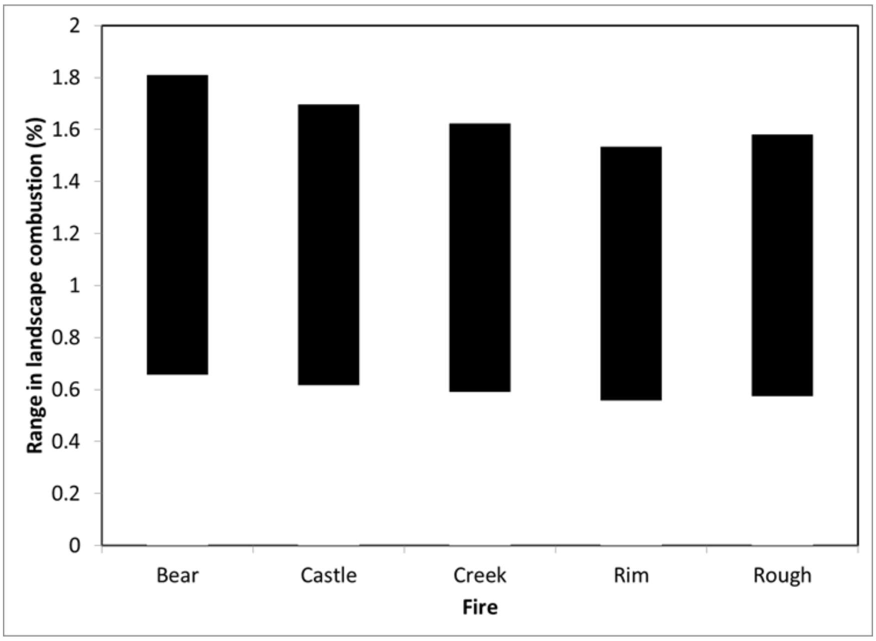


Figure 5. Landscape-level estimates for aboveground woody biomass combustion for five large, severe fires in the California Sierra Mountains. The range presented is for the various stand structures presented in Figure 4.

4. Discussion

4.1. Evaluation of Hypotheses

We employed a novel, field-based method to estimate the fraction of aboveground woody biomass combusted in large wildfires and evaluated five hypotheses that span multiple levels, ranging from tree parts to landscapes. The advantage of our method is that findings at one level can be related to another. For example, although we did find high rates of biomass combustion for some portions of branches and for small trees in high severity fire patches, a number of factors led the average combustion rate at higher levels of organization (i.e., trees, stands, and landscapes) to be substantially lower. Below, we evaluate each of our hypotheses in turn.

4.1.1. Hypothesis 1

Our findings for branches and to some degree for boles were consistent with Hypothesis 1. We did not observe any live branches >2 cm diameter that were fully combusted in even the most severe fire patches; it is likely that 2 cm represents an approximate combustion threshold in terms of the proximity, surface area-to-volume ratio, and drying rates of branch segments. Moreover, the 2 cm diameter limit suggests that for the species we examined, only branch orders one to three are of potentially combustible size. This places significant constraints on how much of a branch or tree can be combusted (Figures S10 and S11).

We found that, while all sizes of trees had losses from bark charring, this loss was proportionally highest for the smallest diameter trees. However, this may have been more related to proximity to ground fuels and the distribution of energy release than diameter per se. This suggests that proximity to heat source is also important to consider.

4.1.2. Hypothesis 2

We found multiple lines of evidence to support Hypothesis 2. Specifically, our branch models predicted that as DBH increased the proportion of branches susceptible to combustion declined. Using *A. concolor* as an example, orders one to three (the most susceptible to combustion) comprise 78% and 11% of branches for DBH size classes one and six, respectively (Figure S10). At the level of trees, the proportion of branches in total aboveground woody biomass declined as DBH increased, which meant that even if the combustion rate of branches was constant, the fraction of total woody biomass combusted would be reduced 60% from the smallest to the largest DBH trees. Combining these two DBH-related effects suggests that 1.8%–31% of aboveground woody mass could be consumed for this species, with the highest value for the smallest trees (Figure S11). These estimates pertain to *A. concolor* in high severity fire patches; very little branch consumption occurred in low and moderate severity fire patches. Specifically, in moderate severity patches, only 17% of the trees had evidence of crown consumption. In low severity patches, even fewer trees (6%) had evidence of crown consumption. This suggests that on moderate and low severity fires, one could expect that 0.3%–5.3% and 0.1%–1.9% of aboveground woody biomass would be consumed at the level of trees, respectively, with the higher values pertaining to the smallest trees.

We found that the rate of bole combustion related to charring also declined as tree size increased. Although char depth increased to some degree with fire severity, the main effect was to char more of the bole length as fire severity increased (Figure S13). In contrast to Hypothesis 2, tree species had a larger effect on bole combustion than tree size. The differences in char depth among species seemed to be related to bark density, with less dense bark exhibiting a deeper char layer (Figure S15b).

4.1.3. Hypothesis 3

At the stand-level, consistent with Hypothesis 3, higher combustion rates (i.e., 34.9% in high severity fire patches) in the smallest trees were strongly offset by the lower ones (i.e., 1.4% also in high severity fire patches) in larger trees because biomass increases exponentially with DBH. Consider that it would take ≈190,000 1 cm DBH trees to equal the biomass of a single 100 cm DBH tree (or ≈30,000 to equal a 50 cm DBH tree). Thus, while small DBH trees might be orders of magnitude more abundant than larger ones, they are not sufficiently abundant to counter the effects that a few larger DBH trees have on stand-level combustion rates. In addition, increasing the proportion of biomass in *Pinus* species greatly decreased stand-level combustion rates, because branches of these species were rarely combusted. This suggests that estimates of stand level combustion rate need to account for differences in taxa as well as tree sizes.

4.1.4. Hypothesis 4

Consistent with Hypothesis 4, we found that, at the landscape level, the very low rates of combustion in low and moderate severity fire patches diluted the effects of higher combustion rates in high severity fire patches. The much lower rates of combustion observed in low and moderate severity fire patches is likely due to the fact that branch combustion in these two severity classes is extremely low. Despite being classified as high severity fires, a large share (48% to 58%) of the fires we examined was of low and moderate severity according to RAVG. Moreover, as noted below, the proportion of moderate severity fire patches is underestimated by RAVG, so it is highly likely that high severity fire patches comprise the minority of even “highly severe” wildfires. Thus, the overall combustion rate in "high" severity fires is likely to be less than half that observed in those portions that were actually high severity.

4.1.5. Hypothesis 5

Our estimates of stand- to landscape-level combustion rates are more than an order of magnitude less than reported by some remote sensing-based studies (Table 4). For example, Garcia et al. [18] estimated that 32%, 52%, and 85% of the aboveground biomass was consumed by the Rim fire in low, moderate, and high severity fire patches, respectively. De Santis et al. [31] estimated that conifers in low, moderate, and high severity fire patches had average biomass combustion rates of 25%, 47%, and 65%, respectively. Applied to the RAVG fire severity distribution for the Rim Fire, these values suggest a landscape average combustion rate of 45%–57% in contrast to the 1%–1.2% we found when our combustion models were applied to various stand structures. For high severity fire patches, we estimated that 1.2%–3.2% of aboveground woody biomass was combusted by fire. Even when foliage was accounted for (see below) our ranges were far below that of Garcia et al. [18] and De Santis et al. [31]. In contrast, our estimates are more in line with the 1%–3% that can be inferred from Miesel et al. [17] for a range of fire severities in mixed conifer forests of California. They are also consistent with field-based research of high-severity fire patches in multiple large fires in the Eastern Cascades of Oregon, where <3% of live conifer tree biomass was consumed [32]. Thus, we cannot reconcile Garcia et al.'s or De Santis et al.'s outcomes with our field data, or with that from other field-based studies, which also find that the vast majority of the carbon remains in trees, even in moderate- and high-severity fire patches in large wildfires [10,19,32,33].

Table 4. Comparison of combustion rates (percentage) at stand- and landscape-level at Rim Fire.

Source	**Fire Severity**			**Landscape**
	Low	**Moderate**	**High**	**Average** [1]
Garcia et al. [18]	32	52	85	57
De Santis et al. [31]	25	47	65	45
This study	0.1–0.2	0.2–0.5	1.2–3.2	1.0–1.2

Notes: [1] We used the distribution of fire severities from RAVG to weight the stand-level combustion rates for different fire severities.

4.2. Evaluation of Uncertainty

Our models predicting the relative volume of branch orders were preliminary and additional efforts should be made to improve them, particularly regarding estimates of bifurcation ratios. However, it is doubtful that these improvements would alter the relative relationships we found, specifically, that as tree DBH increases the maximum order present, diameter, and length of branches increases. Regardless of bifurcation ratios, these trends led to the highest order branch segments to be much larger than the lowest order ones. Hence, for the largest trees, the volume of the highest order branch segment was 374–62,241 times larger than for the first order ones. For the highest order and first order branch segments to have equal volume, then first order branch segments would have to be similarly more abundant than the highest order branch segments. Our models predicted that first order branch segments were less abundant than this by a factor of 5.5–66. It is highly unlikely that our estimates of bifurcation ratios could have been off by that much, but even if they were it may not have made a difference. In the case in which branch segment orders are equally abundant, the relative contribution of each order would be the reciprocal of the maximum order present; with six orders present, the contribution of the first three orders would be 50% to branches or 8% of aboveground woody mass of the largest trees. Therefore, given the limitations of branch mass on trees, even exceedingly high amounts of order one to three branch segments would not allow the majority of aboveground woody biomass to be consumed by fire.

Uncertainty was also introduced by our visual estimates of the fraction of branch orders consumed; however, this was unlikely to have altered our basic findings. In the case of small diameter trees in high fire severity patches, consumption of branch orders one to three were often complete, leading to little uncertainty in estimating the fraction consumed. Similarly, for trees in which no crown consumption occurred (the vast majority of trees), no uncertainty would have been introduced. Therefore, this source of uncertainty is highest for the largest trees in high severity fire patches, but even here it was unlikely to have been substantial. For example, if first order branch segments were 10% of branches and branches were 16% of aboveground woody mass (as is typical for large DBH trees), then estimating 50% consumption of first order branch segments would mean that 0.8% of aboveground woody biomass would have been consumed. Had 25 or 75% of the first order branch segments been consumed it would mean that 0.4% and 1.2% of the aboveground woody mass would have been consumed, respectively. Although these numbers certainly differ, all are small proportions of the aboveground woody biomass.

There were also uncertainties associated with combustion related to bole charring. We assumed that char thickness decreased as one proceeds up the stem. More needs to be learned about the longitudinal pattern of char depth. If the char depth is constant (which seems unlikely), then perhaps bole char losses would have been twice what we estimated. This would have had the largest impact on our estimates of combustion for low and moderate fire severity patches, because bole charring contributed to most of the combustion losses in these areas. However, char height was also limited in these patches. In addition to better understanding longitudinal variation in char depth, the amount of carbon lost via pyrolysis needs to be more precisely determined. Our use of a constant fraction of carbon loss of ≈46% likely overestimated losses from low severity fires and potentially underestimated it for high severity ones. As charring was the most important carbon loss mechanism in low to moderate severity fire patches, it is worthy of further attention.

Our stand-level estimates are preliminary, but additional DBH distribution data would be unlikely to alter our basic finding that combustion losses from the largest trees dominate stand-level calculations (Figure S16). Substantial improvements in stand-level estimates would most likely be related to better estimates of the proportion of stands comprised of *Pinus* versus other species, given that the former have ≈50% the combustion rates of the latter. Additionally, our analysis did not address interactions between stand structure and fire severity; however, while this interaction influences fire severity distributions, it would not change the fundamental relationships between tree size, biomass, and combustion rates.

At the level of entire fires (i.e., landscapes), uncertainty was related to the distribution of fire severity classes used. Specifically, we used RAVG to determine the areal extent of fire severity classes. Although RAVG provides an early (30–60 days) estimate of fire severities, it also tends to classify areas that eventually become moderate severity into the high severity class [34,35]. Therefore, our estimate of average landscape consumption rates may be higher than actually occurred. We can assess the degree this would have influenced our estimates of aboveground woody consumption for the Rim Fire. Potter [36] estimated that 33% of the Rim Fire was high severity; estimates based on MTBS (Monitoring Trends in Burn Severity) indicate that 20% was high severity. Using these fire severity distributions suggests that our landscape range for the Rim Fire would have been 0.5%–1.3% for Potter and 0.3%–0.9% for MTBS versus the 0.6%–1.5% we determined using RAVG.

4.3. Other Combustion Losses

We did not estimate fire consumption of either foliage or roots; however, the former, at least for high severity patches, could be substantial. If we assume that consumption of first order branch segments is the same as foliage consumption (Figure S17), then our stand level estimates of aboveground consumption for high severity patches would have been 3.2%–5.7%. At the landscape level, the average total aboveground consumption rate would have been much lower because there was very little foliage consumption for low and moderate severity patches. Specifically, the total aboveground biomass consumed at the landscape level, adjusted for foliage combustion, would have ranged from 1.4% to 3%. Although some of the fine roots in the lower portions of the organic soil horizons may have been consumed, particularly in high severity fire patches, there was little evidence that coarse roots of live trees were consumed. Assuming that combustion of live roots was minimal, and roots comprised 20% of total live biomass, the total stand-level live biomass combustion for high severity patches would be 2.6%–4.6%. Adjusting the landscape-level combustion rate to account for root biomass gives a range of 1.1%–2.4%. Although these combustion rates are low, they are 1.3–1.9 times our estimate for aboveground woody combustion and suggest that a more detailed accounting of foliage and root combustion is warranted in future studies.

We also did not evaluate combustion losses from either understory plants or aboveground, dead organic matter. During our field work in high severity fire patches, we observed that the forest floor and many downed tree boles, stumps, and associated dead coarse roots were largely consumed. In contrast, sound, large-standing dead trees could be deeply charred, but largely remained even in high severity fire patches. We can use the published distribution of carbon pools (e.g., found in Miesel et al. [17]) combined with our live woody biomass combustion rates, and assume the complete combustion of the understory plants and aboveground dead pools (i.e., coarse woody debris and the forest floor) to estimate pool-specific losses for high severity fire patches. This indicates that combustion of aboveground live woody biomass could account for 4%–10% of the total ecosystem combustion losses. Alternatively, if we assume that 50% of the coarse woody debris was combusted, then live aboveground woody biomass would have contributed 5%–12% of the total ecosystem combustion losses. These estimates contrast markedly with the live tree contribution if the values from Garcia et al. [18] were used: live combustion rates of 85% would mean 74% to 78% of the ecosystem combustion losses would have been associated with live tree combustion.

4.4. Policy Implications

Our findings have significant implications for a wide range of policy analyses related to wildfire combustion. The fraction of live woody biomass combusted varies not only with fire severity level; but also, depending on the organizational level being examined, a number of other factors, including the size structure of woody parts and trees, the species composition of trees present, and mixture of fire severity levels within fires. This suggests that great care must be taken in selecting the combustion rate(s) to be used in either modeling combustion or calibrating remote sensing-based estimates. Furthermore, until an improved multi-level (e.g., branches to landscapes) understanding of combustion rates has been attained, we recommend that such research be tied to field-based, level-relevant data to the extent possible. A key improvement includes determining the limits within which realistic combustion rates can be expected for a given level of analysis.

These findings also suggest that assumptions about combustion in past policy-related analyzes (e.g., [37,38]) should be re-evaluated. Based on our results and those of others [10], combustion-related emissions may have been overestimated. In contrast, fire-related losses via decomposition of fire-killed trees and the amount of fuel generated by fire disturbance may have been underestimated. Hence, the consequences of management actions may not be adequately understood until a more robust understanding of live woody combustion develops.

5. Conclusions

Our field-based examination of the amount of live aboveground woody biomass combusted indicated that while rates for small branch segments can be quite high (i.e., 100%), these rates do not translate in to major losses at the stand or landscape level. This is because high combustion rates in smaller structures are countered by other factors as one proceeds from branches to trees to stands, and to landscapes. The end result in the forests we examined is that even very severe fires combust <2% of live aboveground woody biomass on average. Our work as well as that of others [10] suggests that additional field research is needed to determine how wildfires release carbon to the atmosphere in a wide range of forest structures and fire-weather conditions. We suggest that researchers and policy makers avoid using estimators that are not field-based, because they currently appear to overstate the wildfire emissions used in carbon emissions reporting. As such, they have the potential to misdirect climate mitigation policy. The fact that the vast majority of aboveground woody biomass is not combusted raises the question of when fire-killed trees actually release their carbon. If dead trees are allowed to remain in place, the natural decomposition process could take many decades to centuries to release fire-killed carbon [39]. In contrast, if logged and removed for biomass energy, much of this carbon could be released relatively quickly [40]. Therefore, additional research is also needed to determine the degree that post-fire forest management influences the temporal profile of carbon release.

Supplementary Materials: The following supporting information can be downloaded at: https://www.mdpi.com/article/10.3390/f13030391/s1, Figure S1. Relationship for conifers between the proportion of aboveground woody parts in branches and tree diameter at breast height (based on equations in [22]; Figure S2. Amount of stem volume remaining for a given level of cumulative stem height lost or percentage of diameter lost along boles; Figure S3. Images of the Rim Fire from (a) December 2017 and (b) May 2018 indicate that substantial amounts of aboveground woody biomass remained even in high severity fire patches where all the trees were killed; Figure S4. Schematic depiction of branch orders and bifurcation ratios; Figure S5. Schematic depiction of branch weighting to create a whole tree estimate abundance of branch order segments for trees of different DBH size classes: (a) the maximum order method; (b) the volume weighed method; (c) the volume-length weighted method; and (d) the length weighted method; Figure S6. Examples of characteristics used to separate branches lost via combustion versus decomposition-related fragmentation: (a) branches with rectangular profiles without charring lost via decomposition-related fragmentation; (b) charring of branch stubs that had been broken off prior to fire; (c) thorn-like remnants of branches consumed in fire; Figure S7. Mean dimensions of branch orders of dissected branches for different tree species (a) mid-length diameter; and (b) segment length; Figure S8. Changes in branch structure among species and DBH size classes ((1: 1.0–4.9, 2: 5.0–9.9, 3: 10.0–24.9, 4: 25.0–49.9, 55: 0.0–99.9, 6: >100 cm)) adjacent to the Rim Fire, California: (a) highest branch order; (b) diameter of highest branch order present; (c) total length of highest order branches; and (d) length of highest order branch segment; Figure S9. Comparison of the different models used to estimate branch order contributions for C. decurrens. (a) maximum order model; (b) simple, unweighted average; (c) volume weighted model; and (d) volume and length weighted model; Figure S10 Theoretical maximum branch combustion possible based on: (a) branch orders observed to be consumed or (b) the maximum branch diameters to be consumed; Figure S11. Proportion of tree aboveground woody biomass in: (a) branch order segments 1–3; and (b) branch segments <2 cm diameter; Figure S12. Branch and bole consumption as a function of fire severity and diameter at breast height for the Creek and Rim Fires, California: (a) branch consumption for high severity fire patches; (b) bole consumption for high severity patches; (c) branch consumption moderate severity; (d) bole consumption moderate severity; (e) branch consumption low severity; and (f) bole consumption low severity; Figure S13. Bole char heights on trees compared to total tree height at: (a) Rim; and (b) Creek Fires; Figure S14. Radial char depth at base of tree bole based on: (a) chopping into and measuring char depth on each tree at Creek Fire; and (b) visual estimates from the Rim Fire; Figure S15. Bole char heights and depths for a subsample of trees on the Rim Fire: (a) the maximum versus minimum proportion of the bole charred; and (b) means depth of char at base of trees of three common species; Figure S16. Stand-level combustion for theoretical and published DBH size distributions: (a) total combustion of high severity fires for different DBH distribution types; and (b) total combustion of high severity fires for differing coefficients of variation for normal distribution (e.g., CV100 = coefficient of variation of 100%); Figure S17. Relationship between diameter at breast height (DBH) of trees and the amount of foliage consumed by high severity fire patches in the Rim and Creek Fires; Table S1. Dimensions of dissected branch segments for five species in Rim Fire, Sierra Nevada Mountains, CA; Table S2. Bifurcation ratios for dissected tree branches for five species in Rim Fire, Sierra Nevada Mountains, CA; Table S3. Non-linear regressions predicting branch consumption as a function of DBH for different fire severity classes and species groups; Table S4. Non-linear regressions predicting bole consumption as a function of DBH for different fire severity classes and species groups.

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Attachment 29

Combustion of Aboveground Wood from Live Trees in Megafires, CA, USA

overlapping reflections was very clear. For the powder pattern of $(Mg_{0.81}Fe_{0.19})SiO_3$ tetragonal garnet, on the other hand, line splitting was not clear except for the (400)-(004) doublet and the (240)-(402)-(204) triplet; other overlapping reflections were diffused and looked like one broad peak.

Under the optical microscope, sections of the tetragonal garnet phase exhibited low birefringence. In one of the runs on a starting material of $(Mg_{0.8}Fe_{0.2})SiO_3$ composition, an isotropic phase was optically detected; however, the x-ray diffraction pattern resembled that of tetragonal garnet that was synthesized from the same starting material, showing small splitting of some peaks. An electron microprobe analysis indicated that the chemical composition of this optically isotropic phase was also $x = 0.19(1)$ [where $x = \mathrm{Fe}/(\mathrm{Fe} + \mathrm{Mg})$], $Al_2O_3 \leq 0.1\%$ by weight, with no other contaminants present. The lattice parameters determined by the WPPD method are $a = 11.5323(3)$ Å and $c = 11.4541(4)$ Å, with $R_{wp} = 3.6\%$, which are essentially the values of isochemical tetragonal garnet. In conclusion, this "isotropic" phase is identified as tetragonal garnet. It may appear "isotropic" on account of the fineness of the crystal grain size. The microcrystallinity (<2 μm) is a remarkable microscopic feature of the tetragonal garnets synthesized in the present study.

Kato (*9*) reported in the conclusion of his experimental studies of the $MgSiO_3$-$FeSiO_3$ system that the cubic garnet phase with a normal garnet structure (majorite) is stable in the range of composition $0.2 < x < 0.4$ at 20 GPa and 2000°C, whereas the tetragonal garnet phase is stable for $x < 0.2$. We carried out a series of experiments with a starting composition of $x = 0.3$ as well but could not observe cubic garnet; we observed only a small amount of optically anisotropic tetragonal phase in insufficient proportions for x-ray diffraction analysis. The major proportion of the sample product was an assemblage of spinel and stishovite when experimental temperature was somewhat low and quench crystals from liquid when it was somewhat high. (The experiments were performed several times at temperatures around 2000°C.) Our present observations thus do not suggest the existence of the cubic garnet phase. It is possible that Kato (*9*) might have misidentified "isotropic" tetragonal garnet as "cubic" garnet.

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Effects on Carbon Storage of Conversion of Old-Growth Forests to Young Forests

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Simulations of carbon storage suggest that conversion of old-growth forests to young fast-growing forests will not decrease atmospheric carbon dioxide (CO_2) in general, as has been suggested recently. During simulated timber harvest, on-site carbon storage is reduced considerably and does not approach old-growth storage capacity for at least 200 years. Even when sequestration of carbon in wooden buildings is included in the models, timber harvest results in a net flux of CO_2 to the atmosphere. To offset this effect, the production of lumber and other long-term wood products, as well as the life-span of buildings, would have to increase markedly. Mass balance calculations indicate that the conversion of 5×10^6 hectares of old-growth forests to younger plantations in western Oregon and Washington in the last 100 years has added 1.5×10^9 to 1.8×10^9 megagrams of carbon to the atmosphere.

DEFORESTATION HAS BEEN A source of increasing C in the atmosphere in the last century (*1–9*). However, it has recently been suggested that the CO_2 content of the atmosphere could be reduced if slowly growing, "decadent," old-growth forests were converted to faster growing, younger, intensively managed forests (*10*). Such suggestions may seem reasonable at first glance in that young forests have higher net primary productivity than old-growth forests (*11*). But such reasoning disregards the critical factor, which is the amount of C stored within a forest, not the annual rate of C uptake.

In this report, we explore the effects that conversion of old-growth to younger forests has on atmospheric CO_2 and terrestrial C budgets. We use three lines of evidence: the current disposition of C resulting from cutting old-growth timber, a model of C dynamics in old-growth and second-growth forests, and a comparison of C storage in an old-growth and a young forest by means of simulation.

Approximately 42% of the timber currently harvested in the Pacific Northwest enters long-term storage (products with a life-span of >5 years) in forms such as structural components of buildings (Fig. 1). This level is significantly higher than the historical level, which was as low as 20% in the 1950s (*12*). The long-term average is considerably lower than the current value because 75% of the timber harvested in the last 100 years in Oregon and Washington was cut before 1960 (*13*).

At least 15% of the wood fiber in a typical harvest is left behind as broken or defective (*14, 15*). Some of this material is used for fuel or paper production and is therefore quickly converted to atmospheric CO_2. Of the C removed from the site, 11% is in bark (*16*), which is either burned or composted to form mulch. Most of the tree volume removed from a stand is used in lumber production (*17*). When undecayed harvested wood is converted to boards or plywood, at least 35 to 45% is lost to sawdust or scrap during production (*15*). Some of this waste material is used in particle- and wafer-board production, but most is consumed as fuel or converted to paper. Production of paper, even with recycling, results in a loss of CO_2 to the atmosphere, in that only 46 to 58% of primary paper production is recovered as fiber (*15*) and the residue serves largely as fuel.

The result of all this activity is that, of the 325 Mg of C per hectare harvested from a typical old-growth forest, 187 Mg of C per hectare may be lost to the atmosphere from paper production, fuel consumption, or de-

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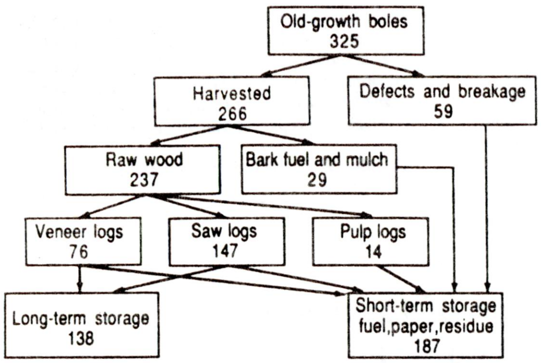

Fig. 1. Flow of C (megagrams per hectare) into long- and short-term storage components after harvest of a 1-ha old-growth forest. Data are from studies on Douglas fir and western hemlock (*14–17*). Boards and plywood are assumed to enter long-term storage (>5 years). Sawdust, scrap, and pulp are assumed to enter short-term storage.

composition (Fig. 1). The proportion of young forests converted to long-term storage is probably even lower than that of old-growth forests because less breakage or defect will be offset by less recovery of boards and plywood (*15*). If we assume that 45% of the boles in a 60-year-old stand is converted to long-term storage, harvest of a 60-year-old forest will still result in a net loss to the atmosphere of 132 Mg of C per hectare. For wood harvested from either old-growth or young-growth forests, the "long-term" storage is perhaps 200 years at most (*18*).

We constructed a computer simulation model to examine the temporal dynamics of C storage in the Douglas fir and hemlock (*Pseudotsuga-Tsuga*) ecosystems common to the Pacific Northwest. This nonlinear difference model with a 1-year time step tracks C storage in the following forest components: foliage, branchwood, boles, coarse roots, fine roots, fine woody debris, forest floor, coarse woody debris, and light and heavy soil C (*19*). Data for the biomass, production, and C turnover of these components were compiled for young and old-growth Douglas fir and hemlock forests growing on the west side of the Oregon and Washington Cascade Mountain Range.

For the purposes of analysis, we adopted several assumptions: that changing climatic conditions and CO_2 concentrations would not affect processing rates; that net production of bole wood and bark for all simulations would peak at 30 years at 8.5 Mg of C per hectare per year (*20, 21*); and that repeated harvesting would not reduce long-term site productivity. The latter assumption is conservative in that repeated harvest may well reduce productivity (*22*) and detrital storage (*23*). In the simulations, we compared and assessed the effect on C storage of (i) a natural disturbance versus timber harvest, (ii) a 50% increase in the decomposition rate after disturbance versus no increase, and (iii) the removal of logging residues versus no removal in repeated harvests on a 60-year rotation.

The simulated biomass accumulation rates matched those for the old-growth condition closely (+2%) but were 25% higher than actual values for natural stands at 60 years (Table 1), matching more closely the values for plantations (*20*). Harvest of old-growth forests reduced C storage for at least 250 years, and, interestingly, a natural disturbance such as fire or windthrow also reduced storage but much less drastically (Fig. 2). Storage declined with harvest both with and without an assumption of increased decomposition with disturbance, although the decrease was larger with this assumption. The decomposition rate of the forest floor has increased with harvest in other forest ecosystems (*24*) and is expected to increase in the Pacific Northwest because sapwood volume is greater in woody detritus from young trees than from old trees (*25*) and leaf-litter decay is greater early in succession (*26*).

Although detrital components store 25 to 30% of the C in Douglas fir and hemlock ecosystems, they can be strongly and negatively affected by management manipulations. Coarse woody debris, for example, virtually disappeared in one simulation of short harvest rotations and intensive utilization (*23*). Soil organic matter, especially the light fraction (*27*), most likely will decrease under intensive management. In simulations of repeated 60-year harvests, the reduction in C storage was stabilized after two rotations (Fig. 3). Increases in living-tree storage brought about by genetic improvement, nutrient fertilization, and CO_2 fertilization (*28*) may offset some of the losses from detrital pools. However, even if coarse woody debris is the only detrital component reduced (with a reduction of 100 Mg of C per hectare), these improvements will need to nearly double the mean annual increment at rotation age to offset the losses.

In a comparison of total C storage, there was 2.2 to 2.3 times as much storage in a 450-year-old *Pseudotsuga-Tsuga* natural stand as in a 60-year-old *Pseudotsuga* plantation (Table 1). However, over a landscape, managed forests in the full range of age classes store less C than a forest of uniform age. With the assumption of a sustained forest yield, one can calculate the mean C stored in a landscape after conversion from the old-growth condition by averaging over the first *t* years of the simulation, where *t* is rotation age. For landscapes with rotations of 50, 75, and 100 years, the C stored would be at most 38, 44, and 51%, respectively, of that stored in the old-growth stand (*29*). As discussed above, these differences are conservative because storage in detrital components would be greatly reduced with repeated harvest.

Conversion of old-growth forests to young plantations invariably reduces C storage, even when structural components in buildings are considered. Comparison of the actual biomass of an old-growth forest and that of a 60-year-old forest of similar site quality indicates that C storage is reduced 350 Mg of C per hectare by conversion, again a conservative estimate because forests continue to lose mass for three decades after disturbance. Model results accounting for this process indicate that C storage is reduced on site by 370 Mg of C per hectare as a result of conversion. However, C stored

Table 1. Carbon (*33*) storage in a 60-year-old *Pseudotsuga* forest and a 450-year-old *Pseudotsuga-Tsuga* forest.

Component	60-year-old forest		450-year-old forest	
	Mg of C per hectare	Reference	Mg of C per hectare	Reference
Foliage	5.5	(*20*)	6.2 −7.0	(*16*) (*40*)
Branchwood	7.0	(*20*)	26.3	(*16*)
Boles (wood and bark)	145	(*20*)	323	(*16*)
Coarse roots	29	(*34*)	71	(*16*)
Fine roots	5.6	(*35*)	5.6	(*16*)
Fine woody debris and forest floor	7.1	(*36*)	26	(*16*)
Coarse woody debris	3.8 −19	(*37*) (*38*)	97	(*25*)
Soil carbon	56	(*39*)	56	(*16*)
Total*	259 to 274		611 to 612	

*Range given because of variation in estimates for foliage and coarse woody debris.

off site in buildings offsets some of the reductions in on-site storage. Given a 42% conversion of the boles to structural components in buildings and a 2% annual replacement of the structures, the conversion of old-growth to younger forests reduces storage by 305 Mg of C per hectare in one 60-year rotation. Unless utilization standards greatly increase and structural components in buildings can be made to have greater life expectancy, it is doubtful that repeated harvests can offset the original losses caused by conversion (*30*).

Conversion of old-growth forests in the Pacific Northwest has been a significant source of C in the atmosphere. In western Oregon and Washington there are 10×10^6 ha of commercial forest land (*31*). If we use as a basis the age-class structure of large, uncut areas, such as those in Mount Rainier and Olympic national parks, we calculate that 7×10^6 ha were probably in an old-growth condition in 1890. Currently, 2×10^6 ha of old growth remain (*31*); thus 5×10^6 ha have been converted. If C storage has been reduced by ~305 to ~370 Mg of C per hectare by the conversion, 1.5×10^9 to 1.8×10^9 Mg of C has been added to the atmosphere in the last century. In reality, the total flux from this region from changes in land use will have been considerably higher because of the harvest of second-growth forest, widespread fires, and the removal of forest land from production by such processes as road construction and urbanization. Given the small area we are considering, a mere 0.017% of the earth's land surface, old-growth forest conversion appears to account for a noteworthy 2% of the total C released because of land use changes in the last 100 years (*6, 7, 32*).

Although reintroducing forests to deforested regions will increase C storage in the biota, conversion of old-growth forests to younger forests under current harvesting and use conditions has added and will continue to add C to the atmosphere. This conclusion is likely to hold in most forests in which the age of harvest is less than the age required to reach the old-growth stage of succession. The amount of C added by conversion will vary among forests, depending on their maximum storage capacity and the difference between the timber rotation age and the age of the old-growth state within the given ecosystem.

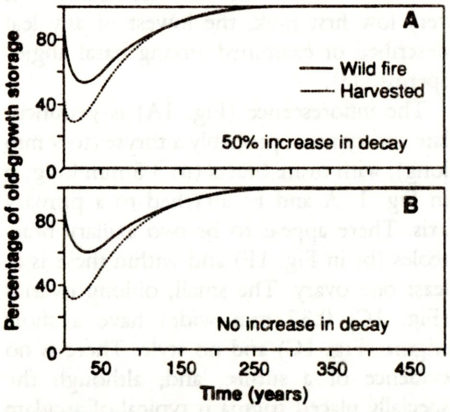

Fig. 2. Carbon storage, expressed as a percentage of old-growth storage, in a simulation of a Douglas fir and hemlock old-growth ecosystem disturbed by fire or timber harvest. The assumptions are that fire used in site preparation will remove 50% of the fine woody debris and forest floor and 25% of the coarse woody debris. The simulation was run with two scenarios: (**A**) disturbance is followed by a 50% increase in the decomposition rate, which decreases 3% annually and reaches old-growth values in 100 years; (**B**) disturbance does not affect decomposition rates.

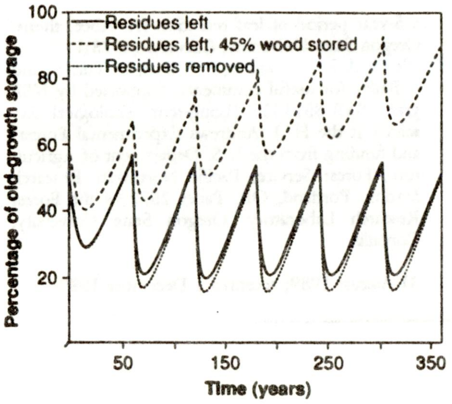

Fig. 3. Carbon storage expressed as a percentage of old-growth storage in a simulation of repeated harvests on a 60-year rotation. The assumptions are that site productivity will not change, that disturbance will initially increase decomposition rates 50%, and that fire used in site preparation will remove 50% of the fine woody debris and forest floor and 25% of the coarse woody debris. Three scenarios were examined: (i) coarse woody debris and residues such as defective boles are left on site; (ii) coarse woody debris is left but other residues are removed; and (iii) all residues are left, but 45% of harvested wood is converted to long-term storage (buildings and other structures) with a 2% annual loss.

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21. The response to climatic change is likely to be complex and was not modeled. Increased temperatures may decrease detrital storage and increase moisture stress and thus are likely to decrease tree growth rates. On the other hand, increased nutrient availability due to increased decomposition rates and increased atmospheric CO_2 are likely to increase tree growth rates. As the model results are expressed relative to old-growth storage, these changes are unlikely to influence the relative difference between young- and old-growth forests.
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31. Report of the Society of American Foresters (SAF) Task Force on scheduling the harvest of old-growth timber (SAF, Washington, DC, 1984).
32. Approximately 100×10^9 Mg of C have been released in the last 100 years because of changes in land use.
33. Assumed organic matter was 50% C.
34. Assumed coarse roots were 20% of bole C, as in Grier and Logan (*16*).
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An Aptian Plant with Attached Leaves and Flowers: Implications for Angiosperm Origin

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Recent phylogenetic studies and fossil finds support a new view of the ancestral angiosperm. A diminutive fossil angiosperm from the Aptian of Australia has attached leaves, with intermediate pinnate-palmate, low-rank venation, and lateral axes bearing pistillate organs subtended by bracts and bracteoles that are the oldest direct evidence of flowers. A variety of data suggests a similar morphology for the ancestral angiosperm. This hypothesis explains similarities between rhizomatous to herbaceous Magnoliidae and basal monocots, scarcity of early angiosperm wood, and lack of recognition of earlier remains.

THE OLDEST UNEQUIVOCAL ANGIOSPERM remains, mostly dispersed organs, are from Lower Cretaceous strata. Fossil pollen is reported from the Hauterivian of England and Barremian of West Africa (*1*), and leaves from the Barremian to Aptian of eastern North America (*2*). Unequivocal angiosperm flowers (*3*) and wood (*4*) first appear during the Albian. These remains show affinities to taxa with diminutive stature and reproductive organs (*2, 5–7*) and to taxa with shrub to tree habit and moderate-sized, complex flowers (*2, 3*). The early and possibly oldest occurrence of the former conflicts with the existing theory that the ancestral angiosperm was a small tree or shrub, with pinnately veined, simple leaves and flowers of moderate to large size with numerous reproductive parts (*8*), though other views have been proposed (*2, 9, 10*).

We recently recognized the angiospermous affinities of a plant described by Drinnan and Chambers as a fern ("Marsileales ? indet?") (*11*) from the Aptian Korumburra Group of the Gippsland Basin at Koonwarra, Victoria, Australia (*11–13*). This fossil has leaves and attached female inflorescences (Fig. 1A), which are the oldest unequivocal angiosperm reproductive structures. The only angiospermous pollen reported from Koonwarra, *Clavatipollenites hughesii* (*12*), is of a type having the earliest range of any flowering plant. Taken together, the fossil evidence and recent phylogenetic analyses of extant plants (*10, 14*) are compatible with a new hypothesis for the ancestral angiosperm.

The fossil has two leaves attached to the axis, which bends sharply to the right at the upper node, and two axillary inflorescences (Fig. 1A). Attachment of the proximal leaf and distal inflorescence is shown by their orientation and similarity to the other clearly attached organs. The inflorescences are masses of overlapping bracts, bracteoles, and ovaries; distinct bracts are noticeable at the apex of the lower inflorescence and along the right side of the upper, where they overlap the distal petiole.

The axis is thin (1.4 mm wide) and exhibits longitudinal ridges, which may be the remains of vascular bundles. Apparent fragility, an apparently dissected stele, and co-occurrence of fully expanded, diminutive leaves with well-developed axillary inflorescences suggest a herbaceous habit. Widely spaced yellow-brown, translucent, discoidal impressions (0.03 to 0.04 mm; Fig. 1G) occurring throughout the fossil may be the remains of ethereal oil cells.

The leaves are alternately arranged (Fig. 1A). The lower (Fig. 1E) has a long petiole that clasps the axis, and a lamina that is apparently folded over distally (Figs. 1E and 2A). Evidence for folding derives from two major veins that extend to the margin and abruptly reverse at the fold; complex, anomalously dense higher venation apparently resulting from superimposition of two levels of veins; and lack of a carbonaceous thickening along the folded margin. The leaf is simple, unlobed, slightly asymmetrical at the base, and broadly ovate, to 10.1 mm wide. The lower laminar margin is darkly stained, suggesting a thickening, and has an inferred incipient sinus (at indentation on left; Fig. 1E). The overfolded upper portion appears to be dissected into three deeply incised dentations. Evidence for dentations, rather than tears, is the symmetry of their outline and vein convergence toward their apices.

A five-stranded vascular trunk emerges into the leaf blade (Figs. 1E and 2A) with the medial strand composed of two bundles. The vein pattern qualifies equally as very loosely and irregularly palinactinodromous or weakly pinnate with three to four pairs of secondary veins. The basal two pairs are crowded proximally and arise as lateral bundles directly from the petiole at an acute angle. The festooned brochidodromous distal secondaries have irregular spacing and angles of origin, branch dichotomously to form loose and irregular loops in at least two series, and are poorly differentiated from the primary and tertiary venation.

Tertiary and higher (to fifth) order veins (Figs. 1E and 2A) form a random reticulum in which vein orders cannot be consistently determined, and the angle of tertiary vein origin is irregular but mostly acute. A fimbrial vein appears to be present. Areolation is apparently incomplete or possibly lacking over some of the leaf. The leaf-rank (*15*) is very low first rank, the lowest of any leaf described or examined among basal angiosperms (*16*).

The inflorescence (Fig. 1A) is pedunculate and cymose, probably a thryse (to 9 mm long), with ovate bracts (to 3.5 mm long; b in Fig. 1, A and F) attached to a primary axis. There appear to be two axillary bracteoles (br in Fig. 1F) and within these is at least one ovary. The small, oblong ovaries (Fig. 1C; 0.57 mm wide) have a short stigma (Fig. 1C) and no style. There is no evidence of a suture, and, although the specially placed stigma is typical of ascidate carpels, the ovary could be syncarpous.

Leaf characters alone reveal the angiospermous affinities of the fossil. Random-reticulate venation with anastomoses at several vein orders, a multistrand splaying out into the laminar base forming an indeterminate actinodromous-brochidodromous venation, and incomplete areoles occur in combination only in angiosperms (*5, 16*). In addition, the morphology of the reproductive

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Attachment 30

Technical Support Document: Social Cost of Carbon, Methane, and Nitrous Oxide Interim Estimates under Executive Order 13990

Technical Support Document: Social Cost of Carbon, Methane, and Nitrous Oxide
Interim Estimates under Executive Order 13990

Interagency Working Group on Social Cost of Greenhouse Gases, United States Government

With participation by

Council of Economic Advisers
Council on Environmental Quality
Department of Agriculture
Department of Commerce
Department of Energy
Department of Health and Human Services
Department of the Interior
Department of Transportation
Department of the Treasury
Environmental Protection Agency
National Climate Advisor
National Economic Council
Office of Management and Budget
Office of Science and Technology Policy

February 2021

Preface

The Interagency Working Group (IWG) on the Social Cost of Greenhouse Gases is committed to ensuring that the estimates agencies use when monetizing the value of changes in greenhouse gas emissions resulting from regulations and other relevant agency actions continue to reflect the best available science and methodologies. This Technical Support Document (TSD) presents interim estimates of the social cost of carbon, methane, and nitrous oxide developed under Executive Order 13990. These interim values are the same as those developed by the IWG in 2013 and 2016. The current IWG will take comment on recent developments in the science and economics for use in a more comprehensive update, to be issued by January 2022, which will more fully address the recommendations of the National Academies of Sciences, Engineering, and Medicine as reported in Valuing Climate Damages: Updating Estimation of the Social Cost of Carbon Dioxide (2017) and other pertinent scientific literature. As a part of that request for comment, the IWG will seek comment on the discussion of advances in science and methodology included in this TSD and how those advances can best be incorporated into the revised final estimates.

Executive Summary

A robust and scientifically founded assessment of the positive and negative impacts that an action can be expected to have on society provides important insights in the policy-making process. The estimates of the social cost of carbon (SC-CO_2), social cost of methane (SC-CH_4), and social cost of nitrous oxide (SC-N_2O) presented here allow agencies to understand the social benefits of reducing emissions of each of these greenhouse gases, or the social costs of increasing such emissions, in the policy making process. Collectively, these values are referenced as the "social cost of greenhouse gases" (SC-GHG) in this document. The SC-GHG is the monetary value of the net harm to society associated with adding a small amount of that GHG to the atmosphere in a given year. In principle, it includes the value of all climate change impacts, including (but not limited to) changes in net agricultural productivity, human health effects, property damage from increased flood risk natural disasters, disruption of energy systems, risk of conflict, environmental migration, and the value of ecosystem services. The SC-GHG, therefore, should reflect the societal value of reducing emissions of the gas in question by one metric ton. The marginal estimate of social costs will differ by the type of greenhouse gas (such as carbon dioxide, methane, and nitrous oxide) and by the year in which the emissions change occurs. The SC-GHGs are the theoretically appropriate values to use in conducting benefit-cost analyses of policies that affect GHG emissions.

Federal agencies began regularly incorporating social cost of carbon (SC-CO_2) estimates in benefit-cost analyses conducted under Executive Order (E.O.) 12866[1] in 2008, following a court ruling in which an agency was ordered to consider the value of reducing CO_2 emissions in a rulemaking process. The U.S. Ninth Circuit Court of Appeals remanded a fuel economy rule to DOT for failing to monetize CO_2 emission reductions, stating that "while the record shows that there is a range of values, the value of carbon emissions reduction is certainly not zero."[2] In 2009, an interagency working group (IWG) was established to ensure that agencies were using the best available science and to promote consistency in the values used across agencies. The IWG published SC-CO_2 estimates in 2010 that were developed from an ensemble of three widely cited integrated assessment models (IAMs) that estimate global climate damages using highly aggregated representations of climate processes and the global economy combined into a single modeling framework. The three IAMs were run using a common set of input assumptions in each model for future population, economic, and GHG emissions growth, as well as equilibrium climate sensitivity (ECS) – a measure of the globally averaged temperature response to increased atmospheric CO_2 concentrations. These estimates were updated in 2013 based on new versions of each IAM. In August 2016 the IWG published estimates of the social cost of methane (SC-CH_4) and nitrous oxide (SC-N_2O) using methodologies that are consistent with the methodology underlying the SC-CO_2 estimates. In January 2017, the National Academies of Sciences, Engineering, and Medicine issued recommendations for an updating process to ensure the estimates continue to reflect the best available science. In March 2017, Executive Order 13783 disbanded the IWG and instructed agencies when monetizing the value of changes

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[1] Under E.O. 12866, agencies are required, to the extent permitted by law and where applicable, "to assess both the costs and the benefits of the intended regulation and, recognizing that some costs and benefits are difficult to quantify, propose or adopt a regulation only upon a reasoned determination that the benefits of the intended regulation justify its costs." As indicated in the discussion above, many statutes also require agencies to conduct at least some of the same analyses required under E.O. 12866, such as the Energy Policy and Conservation Act which mandates the setting of fuel economy regulations.

[2] *Ctr. for Biological Diversity v. Nat'l Highway Traffic Safety Admin.*, 538 F.3d 1172, 1200 (9th Cir. 2008).

in greenhouse gas emissions resulting from regulations to follow the Office of Management and Budget's (OMB) Circular A-4.

On January 20, 2021, President Biden issued E.O. 13990 which re-established the IWG and directed it to ensure that SC-GHG estimates used by the U.S. Government (USG) reflect the best available science and the recommendations of the National Academies (2017) and work towards approaches that take account of climate risk, environmental justice, and intergenerational equity. The IWG was tasked with first reviewing the SC-GHG estimates currently used by the USG and publishing interim estimates within 30 days of the E.O. that reflect the full impact of GHG emissions, including taking global damages into account. In this initial review, the IWG finds that the SC-GHG estimates used since E.O. 13783 fail to reflect the full impact of GHG emissions in multiple ways. First, the IWG found previously and is restating here that a global perspective is essential for SC-GHG estimates because climate impacts occurring outside U.S. borders can directly and indirectly affect the welfare of U.S. citizens and residents. Thus, U.S. interests are affected by the climate impacts that occur outside U.S. borders. Examples of affected interests include: direct effects on U.S. citizens and assets located abroad, international trade, tourism, and spillover pathways such as economic and political destabilization and global migration. In addition, assessing the benefits of U.S. GHG mitigation activities requires consideration of how those actions may affect mitigation activities by other countries, as those international mitigation actions will provide a benefit to U.S. citizens and residents by mitigating climate impacts that affect U.S. citizens and residents. Second, the IWG found previously and is restating here that the use of the social rate of return on capital to discount the future benefits of reducing GHG emissions inappropriately underestimates the impacts of climate change for the purposes of estimating the SC-GHG (see Section 3.1). Consistent with the findings of the National Academies (2017) and the economic literature, the IWG continues to conclude that the consumption rate of interest is the theoretically appropriate discount rate in an intergenerational context (IWG 2010, 2013, 2016). The IWG recommends that discount rate uncertainty and relevant aspects of intergenerational ethical considerations be accounted for in selecting future discount rates.

While the IWG works to assess how best to incorporate the latest, peer reviewed science to develop an updated set of SC-GHG estimates, it is setting interim estimates to be the most recent estimates developed by the IWG prior to the group being disbanded in 2017. The IWG concludes that these interim estimates represent the most appropriate estimate of the SC-GHG until the revised estimates have been developed. This update reflects the immediate need to have an operational SC-GHG for use in regulatory benefit-cost analyses and other applications that was developed using a transparent process, peer-reviewed methodologies, and the science available at the time of that process. Those estimates were subject to public comment in the context of dozens of proposed rulemakings as well as in a dedicated public comment period in 2013.

At the same time, consistent with its continuing commitment to a transparent process and a desire to move quickly to update SC-GHG estimates to better reflect the recent science, the IWG will be taking comment on how to incorporate the recommendations of the National Academies (2017) and other recent science , including the advances discussed in this Technical Support Document (TSD), both during the development of the fully updated SC-GHG estimates to be released by January of 2022 and in subsequent updates. The IWG will soon issue a Federal Register notice with a detailed set of requests for public comments on the new information presented in this TSD, as well as other topics and issues the IWG will address as we develop the next set of updates.

This TSD presents the IWG's interim findings and provides interim estimates of the SC-CO_2, SC-CH_4, and SC-N_2O that should be used by agencies until a comprehensive review and update is developed in line with the requirements in E.O. 13990. The TSD maintains the same methodological approach as has been used for global USG SC-GHG estimation to date. The estimates rely on the same models and harmonized inputs and are calculated using a range of discount rates. At this time, the IWG has determined that it is appropriate for agencies to revert to the same set of four values drawn from the SC-GHG distributions based on three discount rates (2.5 percent, 3 percent, and 5 percent) as were used in regulatory analyses between 2010 and 2016 and subject to public comment. However, as described below, based on the IWG's initial review, new data and evidence strongly suggests that the discount rate regarded as appropriate for intergenerational analysis is lower.

Tables ES-1, ES-2, and ES-3 summarize the interim SC-CO_2, SC-CH_4, and SC-N_2O estimates, respectively, for the years 2020 through 2050. These estimates are reported in 2020 dollars but are otherwise identical to those presented in the previous version of the TSD and its Addendum, released in August 2016. For purposes of capturing uncertainty around the SC-GHG estimates in analyses, the IWG emphasized previously and reemphasizes here the importance of considering all four of the SC-GHG values. In particular, this TSD discusses how the understanding of discounting approaches suggests discount rates appropriate for intergenerational analysis in the context of climate change that are lower than 3 percent. Consistent with the guidance in E.O. 13990 for the IWG to ensure that the SC-GHG reflect the interests of future generations, the latest scientific and economic understanding of discount rates discussed in this TSD, and the recommendation from OMB's Circular A-4 to include sensitivity analysis with lower discount rates when a rule has important intergenerational benefits or costs, agencies may consider conducting additional sensitivity analysis using discount rates below 2.5 percent. Furthermore, the IAMs used to produce these interim estimates do not include all of the important physical, ecological, and economic impacts of climate change recognized in the climate change literature. For these same impacts, the science underlying their "damage functions" – i.e., the core parts of the IAMs that map global mean temperature changes and other physical impacts of climate change into economic (both market and nonmarket) damages – lags behind the most recent research. Likewise, the assumptions regarding equilibrium climate sensitivity and socioeconomic and emissions scenarios used as inputs to the model runs in this TSD will need to be updated. It is the IWG's judgment that, taken together, these limitations suggest that the range of four interim SC-GHG estimates presented in this TSD likely underestimate societal damages from GHG emissions.

Table ES-1: Social Cost of CO_2, 2020 – 2050 (in 2020 dollars per metric ton of CO_2)[3]

	Discount Rate and Statistic			
Emissions Year	5% Average	3% Average	2.5% Average	3% 95th Percentile
2020	14	51	76	152
2025	17	56	83	169
2030	19	62	89	187
2035	22	67	96	206
2040	25	73	103	225
2045	28	79	110	242
2050	32	85	116	260

Table ES-2: Social Cost of CH_4, 2020 – 2050 (in 2020 dollars per metric ton of CH_4)

	Discount Rate and Statistic			
Emissions Year	5% Average	3% Average	2.5% Average	3% 95th Percentile
2020	670	1500	2000	3900
2025	800	1700	2200	4500
2030	940	2000	2500	5200
2035	1100	2200	2800	6000
2040	1300	2500	3100	6700
2045	1500	2800	3500	7500
2050	1700	3100	3800	8200

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[3] The values reported in this TSD are identical to those reported in the 2016 TSD adjusted for inflation to 2020 dollars using the annual GDP Implicit Price Deflator values in the U.S. Bureau of Economic Analysis' (BEA) NIPA Table 1.1.9: 113.626 (2020)/ 92.486 (2007) = 1.228575 (U.S. BEA 2021). Values are the average across models and socioeconomic emissions scenarios for each of three discount rates (2.5%, 3%, and 5%), plus a fourth value, selected as the 95th percentile of estimates based on a 3 percent discount rate. Values of SC-CO_2 are rounded to the nearest dollar; SC-CH_4 and SC-N_2O are rounded to two significant figures. The annual unrounded estimates are available on OMB's website for use in regulatory and other analyses: https://www.whitehouse.gov/omb/information-regulatory-affairs/regulatory-matters/#scghgs

Table ES-3: Social Cost of N_2O, 2020 – 2050 (in 2020 dollars per metric ton of N_2O)

	Discount Rate and Statistic			
Emissions Year	5% Average	3% Average	2.5% Average	3% 95th Percentile
2020	5800	18000	27000	48000
2025	6800	21000	30000	54000
2030	7800	23000	33000	60000
2035	9000	25000	36000	67000
2040	10000	28000	39000	74000
2045	12000	30000	42000	81000
2050	13000	33000	45000	88000

While point estimates are important for providing analysts with a tractable approach for regulatory analysis, they do not fully quantify uncertainty associated with the SC-GHG estimates. Figures ES-1 through ES-3 present the quantified sources of uncertainty in the form of frequency distributions for the SC-GHG estimates for emissions in 2020. The distributions of SC-GHG estimates reflect uncertainty in key model parameters chosen by the IWG such as the equilibrium climate sensitivity, as well as uncertainty in other parameters set by the original model developers. To highlight the difference between the impact of the discount rate and other quantified sources of uncertainty, the bars below the frequency distributions provide a symmetric representation of quantified variability in the SC-GHG estimates for each discount rate. There are other sources of uncertainty that have not yet been quantified and are thus not reflected in these estimates. When an agency determines that it is appropriate to conduct additional quantitative uncertainty analysis, it should follow best practices for probabilistic analysis.[4] The full set of information that underlies the frequency distributions in Figures ES-1 through ES-3 is available on OMB's website[5].

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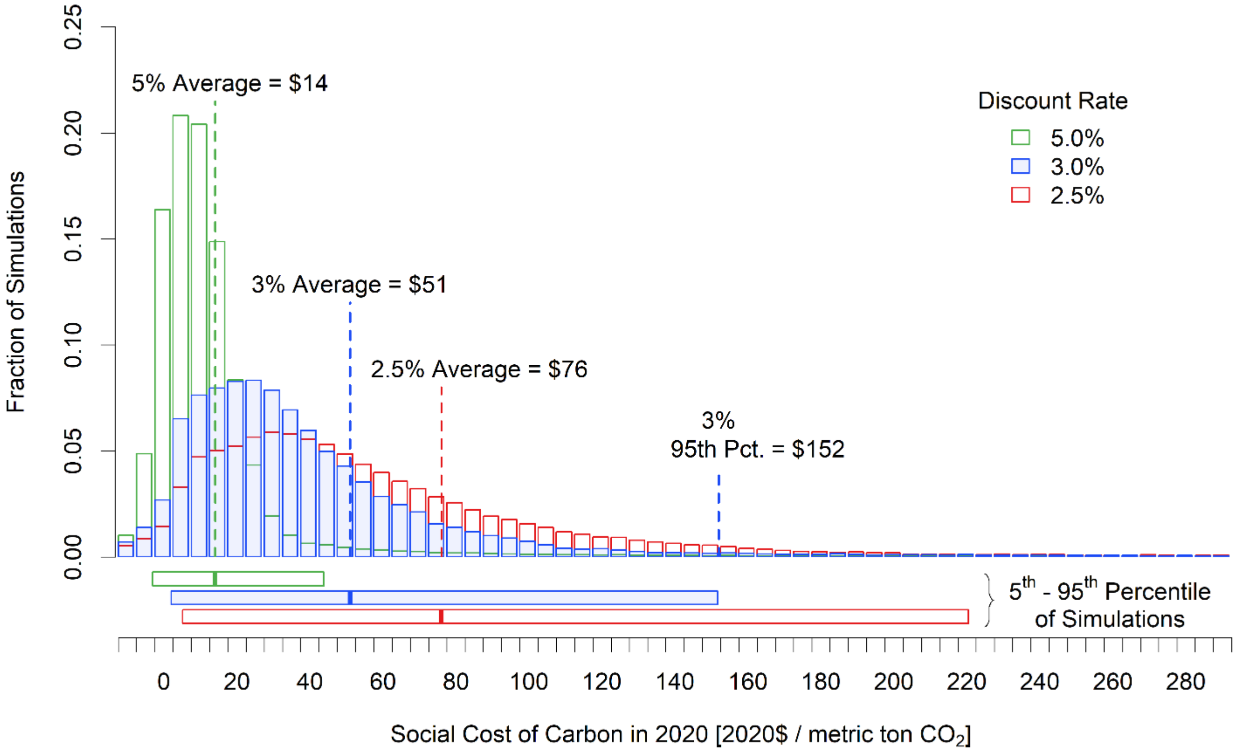
[4] See e.g. OMB's Circular A-4, section on *Treatment of Uncertainty*. Available at: https://www.whitehouse.gov/sites/whitehouse.gov/files/omb/circulars/A4/a-4.pdf.

[5] Available at https://www.whitehouse.gov/omb/information-regulatory-affairs/regulatory-matters/#scghgs

Figure ES-1: Frequency Distribution of SC-CO$_2$ Estimates for 2020[6]



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[6] Although the distributions and numbers in Figures ES-1, ES-2, and ES-3 are based on the full set of model results (150,000 estimates for each discount rate and gas), for display purposes the horizontal axis is truncated with 0.02 to 0.68 percent of the estimates falling below the lowest bin displayed and 0.12 to 3.11 percent of the estimates falling above the highest bin displayed, depending on the discount rate and GHG.

Figure ES-2: Frequency Distribution of SC-CH$_4$ Estimates for 2020

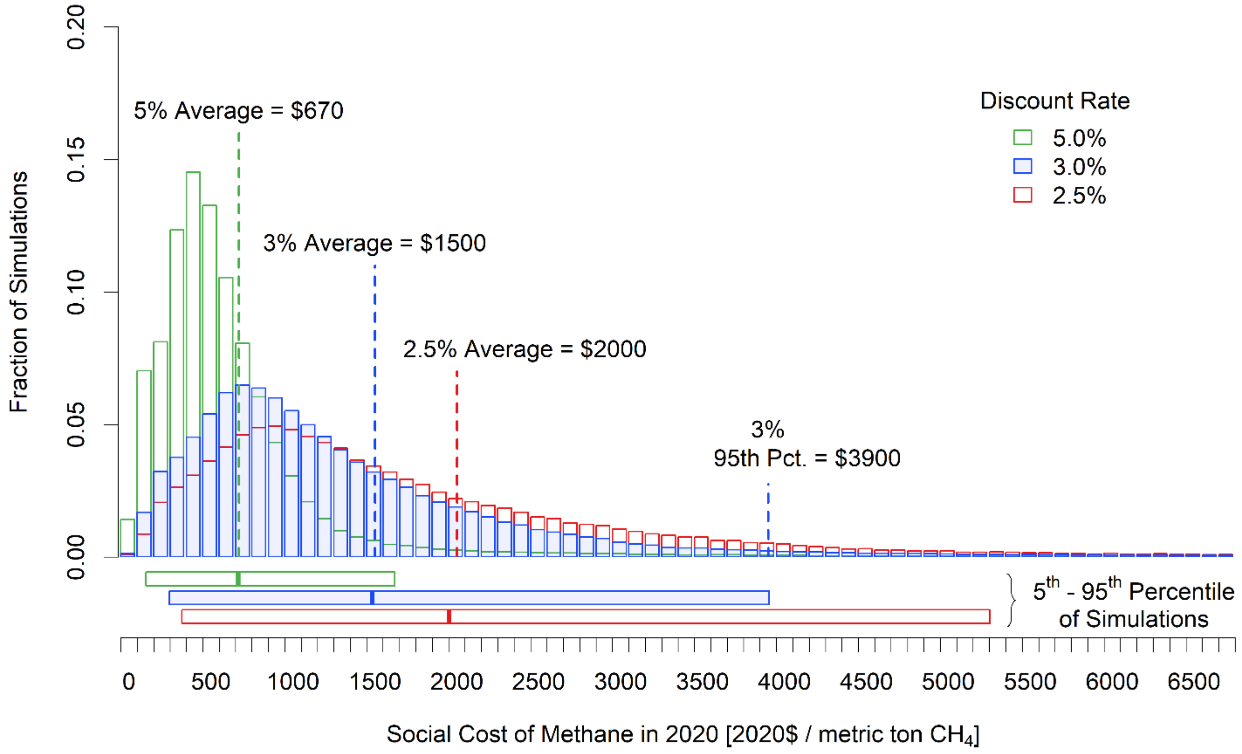
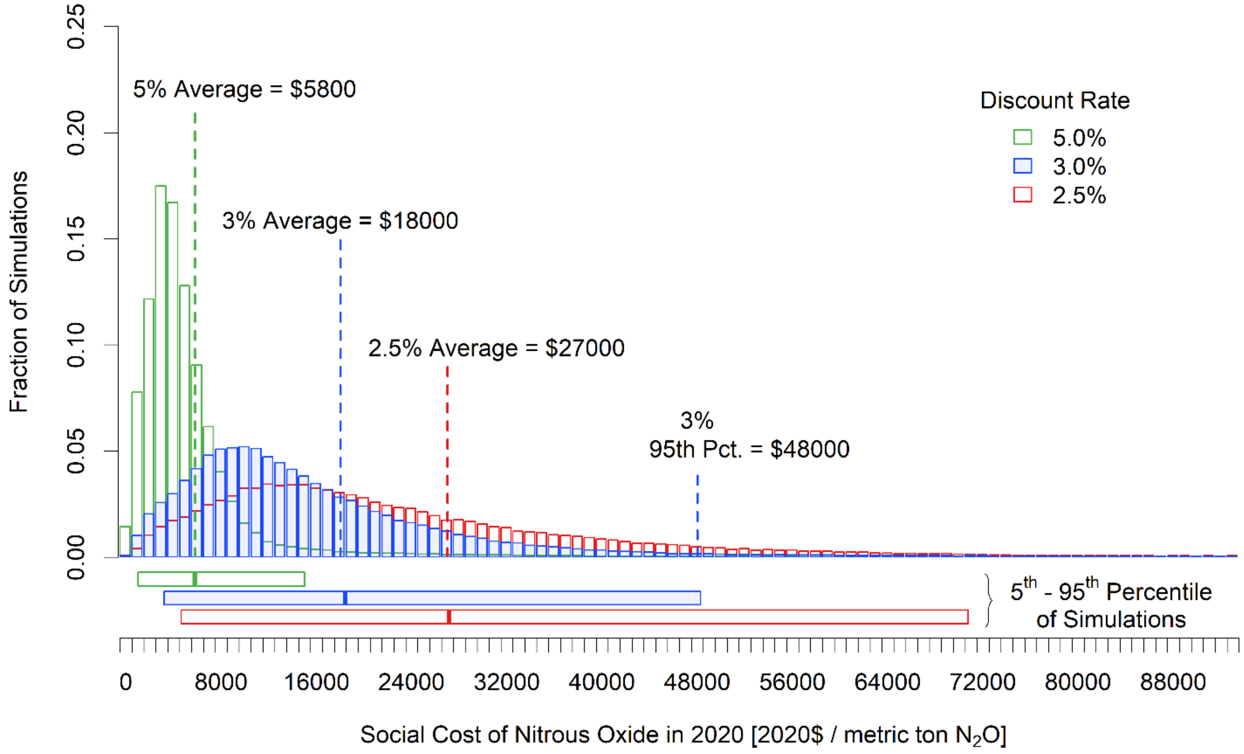


Figure ES-3: Frequency Distribution of SC-N$_2$O Estimates for 2020



1 Background

The estimates of the social cost of carbon (SC-CO_2), social cost of methane (SC-CH_4), and social cost of nitrous oxide (SC-N_2O) presented here allow agencies to incorporate the social benefits of reducing emissions of each of these greenhouse gases, or the social costs of increasing such emissions, in decision making. Collectively, these values are referenced as the "social cost of greenhouse gases" (SC-GHG) in this document. The SC-GHG is the monetary value of the net harm to society associated with adding a small amount of that GHG to the atmosphere in a given year. In principle, it includes the value of all climate change impacts, including (but not limited to) changes in net agricultural productivity, human health effects, property damage from increased flood risk natural disasters, disruption of energy systems, risk of conflict, environmental migration, and the value of ecosystem services. The SC-GHG, therefore, should reflect the societal value of reducing emissions of the gas in question by one ton. The marginal estimate of social costs will differ by the type of greenhouse gas (such as carbon dioxide, methane, and nitrous oxide) and by the year in which the emissions change occurs. The SC-GHGs are calculated along a baseline path and provide a measure of the marginal benefit of GHG abatement. Thus, they are the theoretically appropriate values to use when conducting benefit-cost analyses of policies that affect GHG emissions.[7]

1.1 Overview of U.S. Government SC-GHG Estimates to Date

Estimates of the social cost of carbon and other greenhouse gases have been published in the academic literature for many years. Meta-reviews of SC-CO_2 estimates were available as early as 2002 (Clarkson and Deyes 2002). Federal agencies began regularly incorporating SC-CO_2 estimates in regulatory impact analyses in 2008, following a court ruling in which an agency was ordered to consider the SC-CO_2 in the rulemaking process. The U.S. Ninth Circuit Court of Appeals remanded a fuel economy rule to the Department of Transportation (DOT) for failing to consider the value of reducing CO_2 emissions, stating that "while the record shows that there is a range of values, the value of carbon emissions reduction is certainly not zero."[8]

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[7] These estimates of social damages should not be confused with estimates of the costs of attaining a specific emissions or warming limit. Specifically, there is another strand of research that investigates the costs of setting a specific climate target (e.g., capping emissions or temperature increases to a certain level). If total emissions are capped, IAM models can estimate the costs of limiting emissions or temperature increase to that cap. Similarly, other models simulate market trading in a cap and trade system. The price of a permit to emit one ton of carbon provides a measure of the marginal cost of GHG abatement, which can be useful in evaluating policy cost-effectiveness but is not an alternative way to value damages from GHG emissions in benefit-cost analysis. Moreover, a policy that specifies an environmental target implicitly requires a valuation of damages when setting the constraint even though it is not explicitly modeled or estimated. For example, a target set to keep temperature increases below a certain threshold implicitly places value on damages incurred beyond that threshold. For more on how these concepts (e.g., a predetermined target-based approach and a damage (SC-GHG) based approach) can be used when designing climate policy see, for example, Hansel et al. (2020) and Stern and Stiglitz (2021).

[8] *Ctr. for Biological Diversity v. Nat'l Highway Traffic Safety Admin.*, 538 F.3d 1172, 1200 (9th Cir. 2008).

In 2009, an interagency process was launched, under the leadership of the Office of Management and Budget (OMB) and the Council of Economic Advisers (CEA), that sought to harmonize a range of different SC-CO_2 values being used across multiple Federal agencies. The purpose of this process was to ensure that agencies were using the best available information and to promote consistency in the way agencies quantify the benefits of reducing CO_2 emissions in regulatory impact analyses. This included the establishment of an IWG which represented perspectives and technical expertise from many federal agencies and a commitment to following the peer-reviewed literature. In 2010, the IWG finalized a set of four SC-CO_2 values for use in regulatory analyses and presented them in a TSD that also provided guidance for agencies on using the estimates (IWG 2010). Three of these values were based on the average SC-CO_2 from three widely cited integrated assessment models (IAMs) in the peer-reviewed literature – DICE, PAGE, and FUND[9] – at discount rates of 2.5, 3, and 5 percent. The fourth value was included to represent higher-than-expected economic impacts from climate change further out in the tails of the SC-CO_2 distribution. For this purpose, it used the SC-CO_2 value for the 95th percentile at a 3 percent discount rate.

In May of 2013, the IWG provided an update of the SC-CO_2 estimates to incorporate new versions of the IAMs used in the peer-reviewed literature (IWG 2013). The 2013 update did not revisit other IWG modeling decisions (i.e., the discount rates or harmonized inputs for socioeconomic and emission scenarios and equilibrium climate sensitivity). Improvements in the way damages are modeled were confined to those that had been incorporated into the latest versions of the models by the developers themselves in the peer-reviewed literature.[10] In August of 2016, the IWG published estimates of the social cost of methane (SC-CH_4) and nitrous oxide (SC-N_2O) that are consistent with the methodology underlying the SC-CO_2 estimates (IWG 2016a, 2016b).

Over the course of developing and updating the USG SC-GHG, through both the IWG and individual agencies, there were extensive opportunities for public input on the estimates and underlying methodologies. There was a public comment process associated with each proposed rulemaking that used the estimates, and OMB initiated a separate comment process on the IWG TSD in 2013. Commenters offered a wide range of perspectives on all aspects of process, methodology, and final estimates and diverse suggestions for improvements. The U.S. Government Accountability Office (GAO) also reviewed the development of the USG SC-CO_2 estimates and concluded that the IWG processes and methods reflected three principles: consensus-based decision making, reliance on existing academic literature and models, and disclosure of limitations and incorporation of new information (U.S. GAO 2014).

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[9] The DICE (Dynamic Integrated Climate and Economy) model by William Nordhaus evolved from a series of energy models and was first presented in 1990 (Nordhaus and Boyer 2000, Nordhaus 2008). The PAGE (Policy Analysis of the Greenhouse Effect) model was developed by Chris Hope in 1991 for use by European decision-makers in assessing the marginal impact of carbon emissions (Hope 2006, Hope 2008). The FUND (Climate Framework for Uncertainty, Negotiation, and Distribution) model, developed by Richard Tol in the early 1990s, originally to study international capital transfers in climate policy was widely used to study climate impacts (e.g., Tol 2002a, Tol 2002b, Anthoff et al. 2009, Tol 2009).

[10] The IWG subsequently provided additional minor technical revisions in November of 2013 and July of 2015, as explained in Appendix B of the 2016 TSD (IWG 2016a).

In 2015, as part of the IWG response to the public comments received in the 2013 solicitation, the IWG announced a National Academies of Sciences, Engineering, and Medicine review of the IWG estimates (IWG 2015). Specifically, the IWG asked the National Academies to conduct a multi-discipline, two-phase assessment of the IWG estimates and to offer advice on how to approach future updates to ensure that the estimates continue to reflect the best available science and methodologies. The National Academies' interim (Phase 1) report (National Academies 2016a) recommended against a near term update of the SC-CO_2 estimates within the existing modeling framework. For future revisions, the National Academies recommended the IWG move efforts towards a broader update of the climate system module consistent with the most recent, best available science and offered recommendations for how to enhance the discussion and presentation of uncertainty in the SC-CO_2 estimates. In addition to publishing estimates of SC-CH_4 and SC-N_2O, the IWG's 2016 TSD revision responded to the National Academies' Phase 1 report recommendations regarding presentation of uncertainty. The revisions included: an expanded presentation of the SC-GHG estimates that highlights a symmetric range of uncertainty around estimates for each discount rate; new sections that provide a unified discussion of the methodology used to incorporate sources of uncertainty; detailed explanation of the uncertain parameters in both the FUND and PAGE models; and making the full set of SC-CO_2 estimates easily accessible to the public on OMB's website.

In January 2017, the National Academies released their final report, *Valuing Climate Damages: Updating Estimation of the Social Cost of Carbon Dioxide*, and recommended specific criteria for future updates to the SC-CO_2 estimates, a modeling framework to satisfy the specified criteria, and both near-term updates and longer-term research needs pertaining to various components of the estimation process (National Academies 2017). A description of the National Academies' recommendations for near-term updates are described in Section 1.2 of this document. Shortly thereafter, in March 2017, President Trump issued Executive Order (E.O.) 13783 which called for the rescission and review of several climate-related Presidential and regulatory actions as well as for a review of the SC-GHG estimates used for regulatory impact analysis. E.O. 13783 disbanded the IWG, withdrew the previous TSDs, and directed agencies to ensure SC-GHG estimates used in regulatory analyses are consistent with the guidance contained in OMB's Circular A-4, "including with respect to the consideration of domestic versus international impacts and the consideration of appropriate discount rates" (E.O. 13783, Section 5(c)). Benefit-cost analyses following E.O. 13783 used SC-GHG estimates that attempted to focus on the domestic impacts of climate change as estimated by the models to occur within U.S. borders and were calculated using two discount rates recommended by OMB's Circular A-4, 3 percent and 7 percent.[11] All other methodological decisions and model versions used in SC-GHG calculations remained the same as those used by the IWG in 2010 and 2013, respectively.

On January 20, 2021, President Biden issued E.O. 13990, which re-established the IWG and directed it to ensure that USG SC-GHG estimates reflect the best available science and the recommendations of the National Academies (2017). The IWG was tasked with first reviewing the SC-GHG estimates currently used by the USG and publishing interim estimates within 30 days of the E.O. that reflect the full impact of GHG emissions, including by taking global damages into account. The E.O. instructs the IWG to develop final SC-GHG estimates by January 2022. Section 1.3 describes requirements established by E.O. 13990 in greater detail. In addition, the E.O. instructs the IWG to provide recommendations to the President by

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[11] OMB Circular A-4 (2003) indicates that sensitivity analysis using lower discount rates than 3 percent and 7 percent may be appropriate where intergenerational effects are important. See Section 3 for further discussion.

September 2021, regarding areas of decision-making, budgeting, and procurement by the Federal Government where the SC-GHG should be applied. The SC-GHG has been used previously in non-regulatory Federal analysis, such as in federal procurement,[12] grant programs,[13] and National Environmental Policy Act (NEPA) analysis,[14] as well as in state level applications; the latter is discussed further in Section 5.

1.2 Recommendations from the National Academies of Sciences, Engineering, and Medicine

In 2015, the IWG requested that the National Academies of Sciences, Engineering, and Medicine review and recommend potential approaches for improving its SC-CO_2 estimation methodology. In response, the National Academies convened a multidisciplinary committee, the Committee on Assessing Approaches to Updating the Social Cost of Carbon. In addition to evaluating the IWG's overall approach to SC-CO_2 estimation, the committee reviewed its choices of IAMs and damage functions, climate science assumptions, future baseline socioeconomic and emission projections, presentation of uncertainty, and discount rates.

In its final report (National Academies 2017), the National Academies committee recommended that the IWG pursue an integrated modular approach to the key components of SC-CO_2 estimation to allow for independent updating and review and to draw more readily on expertise from the wide range of scientific disciplines relevant to SC-CO_2 estimation. Under this approach, each step in SC-CO_2 estimation is developed as a module—socioeconomic projections, climate science, economic damages, and discounting—that reflects the state of scientific knowledge in the current, peer-reviewed literature. In the longer-term, it recommended that the IWG also fund research on ways to better capture interactions and feedbacks between these components. In addition, the committee noted that, while the IWG harmonized assumptions across the IAMs for socioeconomic and emission projections, climate sensitivity, and discount rates when estimating the SC-CO_2, using a single climate module in the nearer-term (2-3 years) and eventually transitioning to a single IAM framework will enhance transparency, improve consistency with the underlying science, and allow for more explicit representation of uncertainty. It recommended these three criteria also be used to judge the value of other updates to the methodology. In addition, it recommended that the IWG update SC-CO_2 estimates at regular intervals, suggesting a five-year cycle.

Regarding the key components of the SC-CO_2, the committee recommended the following improvements in the nearer-term:

- Socioeconomic and emissions projections: Use accepted statistical methods and elicit expert judgment to project probability distributions of future annual growth rates of per-capita GDP and

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[12] For example, SC-CO_2 estimates have been used in Domestic Delivery Services contracts for USG parcel shipping (https://westcoastclimateforum.com/sites/westcoastclimateforum/files/related_documents/FedGSA_DDS3_green_features_fact_sheet.pdf).

[13] For example, in 2016 DOT's Transportation Investment Generating Economic Recovery (TIGER) discretionary grant program required a demonstration that benefits justify costs for proposed projects, and the guidance DOT provides to applicants for how to conduct such an analysis specified that they should use the USG SC-CO_2 estimates (https://www.transportation.gov/sites/dot.gov/files/docs/BCARG2016March.pdf).

[14] See Howard and Schwartz (2019) for examples of the use of SC-CO_2 estimates in NEPA analyses.

population, bearing in mind potential correlation between economic and population projections. Then using expert elicitation, guided by information on historical trends and emissions consistent with different climate outcomes, project emissions for each forcing agent of interest conditional on population and income scenarios. Additional recommendations were offered for improving the socioeconomic module centered on four broad criteria: time horizon, future policies, disaggregation, and feedbacks.

- Climate science: Adopt or develop a simple Earth system model (such as the Finite Amplitude Impulse Response (FaIR) model) to capture relationships between CO_2 emissions, atmospheric CO_2 concentrations, and global mean surface temperature change over time while accounting for non-CO_2 forcing and allowing for the evaluation of uncertainty. It also recommended the IWG adopt or develop a sea level rise component in the climate module that: (1) accounts for uncertainty in the translation of global mean temperature to global mean sea level rise and (2) is consistent with sea level rise projections available in the literature for similar forcing and temperature pathways. It also noted the importance of generating spatially and temporally disaggregated climate information as inputs into damage estimation. It recommended the use of linear pattern scaling (which estimates linear relationships between global mean temperature and local climate variables) to achieve this goal in the near-term.
- Economic damages: Improve and update existing formulations of individual sectoral damage functions when feasible; characterize damage function calibrations quantitatively and transparently; present spatially disaggregated damage projections and discuss how they scale with temperature, income, and population; and recognize any correlations between formulations when multiple damage functions are used.
- Discounting: Account for the relationship between economic growth and discounting; explicitly recognize uncertainty surrounding discount rates over long time horizons using a Ramsey-like approach; select parameters to implement this approach that are consistent with theory and evidence to produce certainty-equivalent discount rates consistent with near-term consumption rates of interest; use three sets of Ramsey parameters to generate a low, central, and high certainty-equivalent near-term discount rate, and three means and ranges of SC-CO_2 estimates; discuss how the SC-CO_2 estimates should be combined with other cost and benefit estimates that may use different discount rates in regulatory analysis.

Additional details on each of these recommendations as well as longer term research needs are provided in the National Academies' final report (National Academies 2017).

1.3 Executive Order 13990

On January 20, 2021, President Biden issued E.O. 13990, "Protecting Public Health and the Environment and Restoring Science to Tackle the Climate Crisis." Echoing one of the general principles of E.O. 12866 that an Agency "shall base its decisions on the best reasonably obtainable scientific, technical, economic, and other information", E.O. 13990 states that it is essential for Agencies to account for the benefits of reducing GHG emissions as accurately as possible. It emphasizes that a full global accounting of the costs of GHG emissions "facilitates sound decision-making, recognizes the breadth of climate impacts, and supports the international leadership of the United States on climate issues" (E.O. 13990 2021). Specifically, E.O. 13990 reinstates the IWG as the Interagency Working Group on the Social Cost of Greenhouse Gases, names the Chair of the CEA, Director of OMB, and Director of the Office of Science and Technology Policy (OSTP) as co-chairs of the IWG, and specifies the membership of the IWG to include the following officials, or their designees: the Secretary of the Treasury; the Secretary of the Interior; the Secretary of Agriculture; the Secretary of Commerce; the Secretary of Health and Human Services; the Secretary of Transportation; the Secretary of Energy; the Chair of the Council on Environmental Quality; the Administrator of the Environmental Protection Agency; the Assistant to the President and National Climate Advisor; and the Assistant to the President for Economic Policy and Director of the National Economic Council.

E.O. 13990 tasks the reinstated IWG with the following:

(1) publish an interim update to the SC-GHG (SC-CO_2, SC-CH_4, and SC-N_2O) estimates by February 19, 2021, for agencies to use when monetizing the value of changes in greenhouse gas emissions resulting from regulations and other relevant agency actions until final values are published;

(2) publish a final update to the SC-GHG estimates by no later than January 2022;

(3) provide recommendations, by no later than September 1, 2021, regarding areas of decision-making, budgeting, and procurement by the Federal Government where the SC-GHG estimates should be applied;

(4) provide recommendations, by no later than June 1, 2022, regarding a process for reviewing and, as appropriate, updating the SC-GHG estimates to ensure that these estimates are based on the best available economics and science; and

(5) provide recommendations, to be published with the interim SC-GHG estimates if feasible and by no later than June 1, 2022, to revise methodologies for SC-GHG calculations to the extent that current methodologies do not adequately take account of climate risk, environmental justice, and intergenerational equity.

Finally, the E.O. specifies that in carrying out its activities, the IWG shall consider the recommendations of the National Academies (2017) and other pertinent scientific literature; solicit public comment; engage with the public and stakeholders; seek the advice of ethics experts; and ensure that the SC-GHG estimates reflect the interests of future generations in avoiding threats posed by climate change.

This TSD presents the interim SC-GHG estimates called for in the first of these tasks. It also provides preliminary discussion of how at least one component of SC-GHG estimation, discounting, warrants reconsideration in the more comprehensive update by January 2022 to reflect the advice of the National Academies (2017) and other recent scientific literature.

2 The Importance of Accounting for Global Damages

Benefit-cost analyses of U.S. Federal regulations have traditionally focused on the benefits and costs that accrue to individuals that reside within the country’s national boundaries. This is a natural result of the fact that most regulations have a limited impact on individuals residing outside of the United States and do not reflect any other scientific, legal, or other rationale. According to OMB’s Circular A-4 (2003), an "analysis should focus on benefits and costs that accrue to citizens and residents of the United States."[15] While Circular A-4 does not elaborate, this guidance towards a focus on U.S. populations in domestic policy analysis is broadly consistent with the fact that the authority to regulate only extends to a nation's own residents who have consented to adhere to the same set of rules and values for collective decision-making (EPA 2010; Kopp et al. 1997; Whittington and MacRae 1986). However, guidance towards a focus on impacts to U.S. citizens and residents is different than recommending that analysis be limited to the impacts that occur within the borders of the U.S. Furthermore, OMB Circular A-4 states that when a regulation is likely to have international effects that "these effects should be reported" though the guidance recommends this be done separately. There are many reasons, as summarized in this TSD, why it is appropriate for agencies to use the global value of damages in making decisions that affect, or may be affected by, GHG emissions. Courts have upheld the use of global damages in estimating the social cost of GHGs, in part in recognition of the diverse ways in which U.S. interests, businesses, and residents may be impacted by climate change beyond U.S. borders.[16]

Unlike many environmental problems where the causes and impacts are distributed more locally, climate change is a true global challenge making GHG emissions a global externality. GHG emissions contribute to damages around the world regardless of where they are emitted. The global nature of GHGs means that U.S. interests, and therefore the benefits to the U.S. population of GHG mitigation, cannot be defined solely by the climate impacts that occur within U.S. borders. Impacts that occur outside U.S. borders as a result of U.S. actions can directly and indirectly affect the welfare of U.S. citizens and residents through a multitude of pathways. Over 9 million U.S. citizens lived abroad as of 2016[17] and U.S. direct investment positions abroad totaled nearly $6 trillion in 2019.[18] Climate impacts occurring outside of U.S. borders will have a direct impact on these U.S. citizens and the investment returns on those assets owned by U.S. citizens and residents. The U.S. economy is also inextricably linked to the rest of the world. The U.S. exports over $2 trillion worth of goods and services a year and imports around $3 trillion.[19] Climate impacts that occur outside U.S. borders can thus impact the welfare of individuals and firms that reside in the United States through their effect on international markets, trade, tourism, and other activities. Furthermore, additional spillovers can occur through pathways such as economic and political destabilization and global migration that can lead to adverse impacts on U.S. national security, public health, and humanitarian concerns (DoD 2014, CCS 2018). As described by the National Academies (2017), to correctly assess the total damages to U.S. citizens and residents, one must account for these spillover effects on the United States.

As an empirical matter, the development of a domestic SC-GHG is greatly complicated by the relatively few region- or country-specific estimates of the SC-CO_2 in the literature. At present, the only quantitative

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[15] OMB's Circular A-4 provides guidance to Federal agencies on the development of regulatory analysis conducted pursuant to Executive Order 12866.

[16] *Zero Zone, Inc. v. Dep't of Energy*, 832 F.3d 654, 678-79 (7th Cir. 2016) (rejecting a petitioner's challenge to DOE's use of a global (rather than domestic) social cost of carbon in setting an efficiency standard under the Energy Policy and Conservation Act, holding that DOE had reasonably identified carbon pollution as "a global externality" and concluding that, because "national energy conservation has global effects, . . . those global effects are an appropriate consideration when looking at a national policy.").

[17] U.S. Department of State's Bureau of Consular Affairs.

[18] BEA Direct Investment by Country and Industry 2019, https://www.bea.gov/data/intl-trade-investment/direct-investment-country-and-industry

[19] BEA National Income and Product Accounts Table 1.1.5.

characterization of domestic damages from GHG emissions, as represented by the domestic SC-GHG, is based on the share of damages arising from climate impacts occurring within U.S. borders as represented in current IAMs. This is both incomplete and an underestimate of the share of total damages that accrue to the citizens and residents of the U.S. because these models do not capture the regional interactions and spillovers discussed above. A 2020 U.S. GAO study observed that "[a]ccording to the National Academies, the integrated assessment models were not premised or calibrated to provide estimates of the social cost of carbon based on domestic damages, and more research would be required to update the models to do so. The National Academies stated it is important to consider what constitutes a domestic impact in the case of a global pollutant that could have international implications that affect the United States" (U.S. GAO 2020).

The global nature of GHGs means that damages caused by a ton of emissions in the U.S. are felt globally and that a ton emitted in any other country harms those in the U.S. Therefore, assessing the benefits of U.S. GHG mitigation activities will require consideration of how those actions may affect mitigation activities by other countries since those international actions will provide a benefit to U.S. citizens and residents. A wide range of scientific and economic experts have emphasized the issue of reciprocity as support for considering global damages of GHG emissions (e.g., Kopp and Mignone 2013, Pizer et al. 2014, Howard and Schwartz 2019, Pindyck 2017, Revesz et al. 2017, Carleton and Greenstone 2021). Carleton and Greenstone (2021) discuss examples of how historic use of a global SC-CO_2 may have plausibly contributed to additional international action. Houser and Larson (2021) estimate that under the Paris Agreement, other countries pledged to reduce 6.1 to 6.8 tons for every ton pledged by the U.S. Kotchen (2018) offers a theoretical perspective showing that non-Nash game theoretic behavior can lead countries to optimally chose a social cost of carbon higher than their domestic value to encourage additional reductions from other countries. Using a global estimate of damages in U.S. analyses of regulatory and other actions allows the U.S. to continue to actively encourage other nations, including emerging major economies, to take significant steps to reduce emissions.

The IWG found previously and is restating here that because of the distinctive global nature of climate change that analysis of Federal regulatory and other actions should center on a global measure of SC-GHG. This approach is the same as that taken in regulatory analyses over 2009 through 2016. In the 2015 response to comments, the IWG noted that the only way to achieve an efficient allocation of resources for emissions reduction on a global basis is for all countries to base their policies on global estimates of damages (IWG 2015). Therefore, the IWG continues to recommend the use of global SC-GHG estimates in analysis of Federal regulatory and other actions. The IWG also continues to review developments in the literature, including more robust methodologies for estimating SC-GHG values based on purely domestic damages, and explore ways to better inform the public of the full range of carbon impacts, both global and domestic.

3 Discounting in Intergenerational Analyses

GHG emissions are stock pollutants, where damages are associated with what has accumulated in the atmosphere over time, and they are long lived such that subsequent damages resulting from emissions today occur over many decades or centuries depending on the specific greenhouse gas under consideration.[20] In calculating the SC-GHG, the stream of future damages to agriculture, human health, and other market and non-market sectors from an additional unit of emissions are estimated in terms of reduced consumption (or consumption equivalents). Then that stream of future damages is discounted to its present value in the year when the additional unit of emissions was released. Given the long time horizon over which the damages are expected to occur, the discount rate has a large influence on the present value of future damages. However, the choice of a discount rate also raises highly contested and exceedingly difficult questions of science, economics, ethics, and law.

In 2010, in light of disagreements in the literature on the appropriate discount rate to use in this context, and uncertainty about how rates may change over time, the IWG elected to use three discount rates to span a plausible range of certainty-equivalent constant consumption discount rates: 2.5, 3, and 5 percent per year. The IWG at that time determined that these three rates reflected reasonable judgments under both descriptive and prescriptive approaches to selecting the discount rate.

The 3 percent value was included as consistent with estimates provided in OMB's Circular A-4 (OMB 2003) guidance for the consumption rate of interest. The IWG found that the consumption rate of interest is the correct discounting concept to use when future damages from elevated temperatures are estimated in consumption-equivalent units as is done in the IAMs used to estimate the SC-GHG (National Academies 2017). The upper value of 5 percent was included to represent the possibility that climate-related damages are positively correlated with market returns, which would imply a certainty equivalent value higher than the consumption rate of interest. The low value, 2.5 percent, was included to incorporate the concern that interest rates are highly uncertain over time. It represents the average certainty-equivalent rate using the mean-reverting and random walk approaches from Newell and Pizer (2003) starting at a discount rate of 3 percent. Using this approach, the certainty equivalent is about 2.2 percent using the random walk model and 2.8 percent using the mean reverting approach. Without giving preference to a particular model, the average of the two rates is 2.5 percent. Additionally, a rate below the consumption rate of interest would also be justified if the return to investments in climate mitigation are negatively correlated with the overall market rate of return. Use of this lower value was also deemed responsive to certain judgments based on the prescriptive or normative approach for selecting a discount rate and to related ethical objections that have been raised about rates of 3 percent or higher. Further details about the process for selecting these rates is presented in the 2010 TSD (IWG 2010). Finally, it is important to note that, while the consumption discount rate is the conceptually correct rate for discounting the SC-GHG, and the three rates originally selected were based on this concept, the latest data as well as recent discussion in the economics literature indicates that the 3 percent discount rate used by the IWG to develop its range of discount rates is likely an overestimate of the appropriate discount rate and warrants reconsideration in future updates of the SC-GHG.

This section discusses three issues related to the selected discount rates: (1) why the social rate of return to capital, estimated to be 7 percent in OMB's Circular A-4, is not appropriate for use in calculating the SC-GHG, (2) new evidence on the consumption rate of interest, which may inform the future updates to the SC-GHG, and (3) analytic consistency across discounting within an analysis.

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[20] "GHGs, for example, CO_2, methane, and nitrous oxide, are chemically stable and persist in the atmosphere over time scales of a decade to centuries or longer, so that their emission has a long-term influence on climate. Because these gases are long lived, they become well mixed throughout the atmosphere" (IPCC 2007).

3.1 Social Rate of Return on Capital and Intergenerational Analyses

When analyzing policies and programs that result in GHG emission reductions, it is important to account for the difference between the social and private rate of return on any capital investment affected by the action. Society is not indifferent between a regulation that displaces consumption versus investment in equal amounts. Market distortions, in large part taxes on capital income, cause private returns on capital investments to be different from the social returns. In well-functioning capital markets, arbitrage opportunities will be dissipated, and the cost of investments will equal the present value of future private returns on those investments. Therefore, an individual forgoing consumption or investment of equal amounts as the result of a regulation will face an equal private burden. However, because the social rate of return on the investment is greater than the private rate of return, the overall social burden will be greater in the case where investment is displaced.

OMB's Circular A-4 points out that "the analytically preferred method of handling temporal differences between benefits and costs is to adjust all the benefits and costs to reflect their value in equivalent units of consumption and to discount them at the rate consumers and savers would normally use in discounting future consumption benefits" (OMB 2003). The damage estimates developed for use in the SC-GHG are estimated in consumption-equivalent terms. An application of OMB Circular A-4's guidance for regulatory analysis would then use the consumption discount rate to calculate the SC-GHG, while also developing a more complete estimate of social cost to account for the difference in private and social rates of return on capital for any investment displaced as a result of the regulation. This more complete estimate of social costs can be developed using either the shadow price of capital approach or by estimating costs in a general equilibrium framework, for example by using a computable general equilibrium model. In both cases, displaced investment would be converted into a flow of consumption equivalents.

In cases where the costs are not adjusted to be in consumption-equivalent terms, OMB's Circular A-4 recommends that analysts provide a range of estimates for net benefits based on two approaches. The first approach is based on using the consumption rate of interest to discount all costs and benefits. This approach is consistent with the case where costs are primarily borne as reduced consumption. The second approach, the social opportunity cost of capital (SOC) approach, focuses on the case where the main effect of a regulation is to displace or alter the use of capital in the private sector (OMB 2003). When interpreting the SOC approach from the point of view of whether to invest in a single government project, it is asking whether the benefits from the project would at least match the returns from investing the same resources in the private sector. Interpreting the approach from the standpoint of a benefit-cost analysis of regulation, the approach focuses on adjusting estimates of benefits downward by discounting at a higher rate to offset additional social costs not reflected in the private value of displaced investment.

Harberger (1972) derived a more general version of the social opportunity cost of capital approach, recognizing that policies will most likely displace a mix of consumption and investment and therefore a blended discount rate would be needed to adjust the benefits to account for the omitted costs. In his partial equilibrium approach, the blended discount rate is a weighted average of the consumption interest rate and social rate of return on capital, where the weights are the share of a policy's costs borne by consumption versus investment. This general result has been extended to the general equilibrium context by Sandmo and Drèze (1971) and Drèze (1974) and can be extended to account for changes in foreign direct investment (CEA 2017). This highlights that using the social rate of return for benefits and costs is at best creating a lower bound on the estimate of net benefits that would only be met in an extreme case where regulatory costs fully displace investment. If the beneficial impacts of the regulation induce private investment whose social returns have not been quantified and fully converted to consumption equivalents, then the net benefits calculated using the social rate of return on capital is not even a lower bound.[21] Li and Pizer (2021) further generalize the SOC framework and demonstrate that temporal pattern of benefits is important and that when benefits occur far in the future discounting using the social rate of return on capital again is not even a lower bound on net benefits.

For regulations whose benefits and costs occur over a relatively short time frame, the range of net benefits computed using the two discounting approaches will be relatively narrow. Therefore, there is less risk in maintaining an uninformed prior over the share of regulatory costs that will displace investment and using the potential bounding cases for net benefits. However, for cases where the costs are borne early in the time horizon and benefits occur for decades or even centuries, such as with GHG mitigation, the two estimates of net benefits will differ significantly. In this case, the risk to society of maintaining an uninformed prior over the share of regulatory costs borne by investment is significantly higher. In turn, the preferred approach is to discount benefits using the consumption rate of interest and strive to provide a more complete measure of costs, accounting for displacement of investment whose social rate of return exceeds the private rate of return, either by using a shadow price of capital approach or a general equilibrium framework, like a computable general equilibrium model.

It is important to note that even if an appropriately specified blended SOC rate could be calculated based on the share of regulatory costs that are expected to displace investment that would not obviate the need to carefully consider issues of uncertainty and ethics when discounting in an intergenerational context, pointing to a lower rate.

For these reasons, the IWG is returning to the approach of calculating the SC-GHG based on the consumption rate of interest, consistent with the findings of the National Academies (2017) [22].

3.2 New Evidence on the Consumption Discount Rate

The three discount rates selected by the IWG in 2010 are centered around the 3 percent estimate of the consumption interest rate published in OMB's Circular A-4 in 2003. That guidance was based on the real rate of return on 10-year Treasury Securities from the prior 30 years (1973 through 2002), which averaged 3.1 percent. Over the past four decades there has been a substantial and persistent decline in real interest rates (see Figure 1). Recent research has found that this decline has been driven by decreases in the equilibrium real interest rate (Bauer and Rudebusch 2020).

Re-estimating the consumption rate of interest following the same approach applied in Circular A-4, including using data from the most recent 30 years, yields a substantially lower result. The average rate

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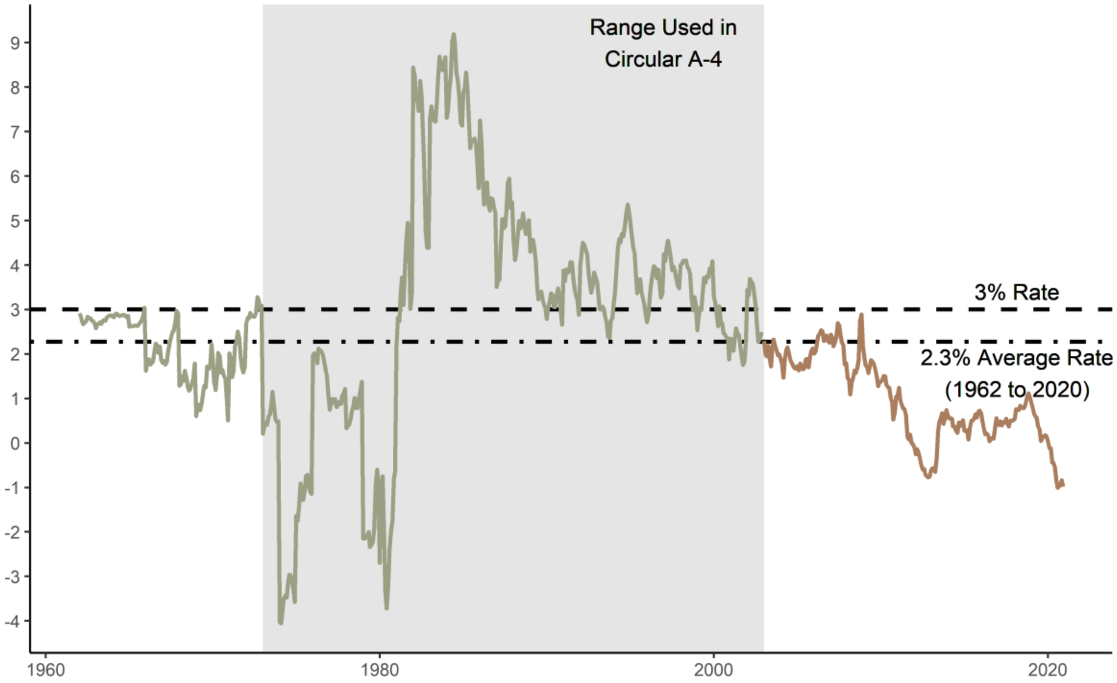
[21] The SOC approach as outlined in OMB's Circular A-4 is most applicable to cases where the benefits are represented as consumption equivalents and costs may not be. If the benefits of the policy include the inducement of new private investment, discounting both benefits and costs at the social rate of return for capital is no longer appropriate. The results of Bradford (1975) show that in a case where regulatory costs are primarily borne through reduced consumption and the beneficial impacts of the policy may induce private investment the appropriate rate under the SOC approach could be below the consumption interest rate.

[22] NAS (2017) stated "The estimates that result from the SC-IAMs are measured in consumption- equivalent units: thus, a discount rate that reflects how individuals trade off current and future consumption is defensible in this setting" (p. 236-7).

of return on inflation adjusted 10-year Treasury Securities over the last 30 years (1991-2020) is 2.0 percent. These rates are not without historic precedent, such that over the last 60 years the inflation adjusted 10-year Treasury Securities is 2.3 percent. Current real rates of returns below 2 percent are expected to persist. The U.S. Congressional Budget Office (CBO) in its September 2020 Long Term Budget Outlook forecasts real rates of return on 10-Year Treasury Securities to average 1.2 percent over the next 30 years (U.S. CBO 2020). This new information suggests that the consumption rate of interest is notably lower than 3 percent. CEA (2017) examined additional forecasts of 10-Year Treasury Securities and data on futures contracts, reaching the conclusion that the appropriate consumption discount rate should be at most 2 percent.

Figure 1: Monthly 10-Year Treasury Security Rates, Inflation-Adjusted[23]



Several surveys have been conducted in recent years to elicit experts' views on the appropriate discount rates to use in an intergenerational context (e.g., Drupp et al. 2018; Howard and Sylvan 2020). For example, Drupp et al. (2018) offers confirming evidence that the economics profession generally agrees that the appropriate social discount rate is below 3 percent as reflected in the recent trends in data. They surveyed over 200 experts and found a "surprising degree of consensus among experts, with more than three-quarters finding the median risk-free social discount rate of 2 percent acceptable" (Drupp et al. 2018).[24]

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[23] Monthly 10-Year Treasury Security returns, adjusted for inflation. Real interest rates prior to 2003 (green line) are calculated by subtracting the annual rate of inflation as measured by the CPI-U from the nominal rate of return on 10-Year constant maturity Treasury Securities. Interest rates from 2003 onwards (brown line) are based on the 10-Year Treasury Inflation-Protected Securities.

[24] For a detailed explanation of discounting concepts and terminology see EPA's *Guidelines for Preparing Economic Analysis* (2010). https://www.epa.gov/environmental-economics/guidelines-preparing-economic-analyses

It is important to note that the new information pointing to a lower consumption rate of interest, lower than 3 percent, does not obviate the need to carefully consider issues of uncertainty and ethics when discounting in an intergenerational context.[25] If 2 percent was used as the consumption interest rate and adjusted for uncertainty using the results of Newell and Pizer (2003) as was done in the 2010 TSD, the process would yield a discount rate lower than 2 percent. Therefore, a consideration of discount rates below 3 percent, including 2 percent and lower, are warranted when discounting intergenerational impacts.

This is consistent with the 2003 recommendation in OMB's Circular A-4 that noted "[a]lthough most people demonstrate time preference in their own consumption behavior, it may not be appropriate for society to demonstrate a similar preference when deciding between the well-being of current and future generations" and found that certainty equivalent discount rates as low as 1 percent could be appropriate for intergenerational problems (OMB 2003). Similarly, if implementing a declining discount rate schedule to account for uncertainty (see next section), an updated consumption rate of interest, based on additional data presented above, may be a starting point for an update.

In light of the evidence and discussion on discount rates presented in this TSD and elsewhere, the recommendation from OMB's Circular A-4 to include further sensitivity analysis with lower discount rates when a rule has important intergenerational benefits or costs, and the direction to the IWG in E.O. 13990 to ensure that the SC-GHG reflect the interest of future generations, the IWG finds it appropriate as an interim recommendation that agencies may consider conducting additional sensitivity analysis using discount rates below 2.5%.

3.3 Analytic Consistency and Declining Discount Rates

While the consumption rate of interest is an important driver of the benefits estimate, it is uncertain over time, as may be observed in Figure 1. Weitzman (1998, 2001) showed theoretically and Newell and Pizer (2003) and Groom et al. (2005) confirmed empirically that discount rate uncertainty can have a large effect on net present values. A main result from these studies is that if there is a persistent element to the uncertainty in the discount rate (e.g., the rate follows a random walk), then it will result in an effective (or certainty-equivalent) discount rate that declines over time. This is because lower discount rates tend to dominate over the very long term (see Weitzman 1998, 1999, 2001; Newell and Pizer 2003; Groom et al. 2005; Gollier 2009; Summers and Zeckhauser 2008; Gollier and Weitzman 2010; Arrow et al. 2013; Cropper et al. 2014; and Arrow et al. 2014).

The proper way to specify a declining discount rate schedule remains an active area of research. One approach is to develop a stochastic model of interest rates that is empirically estimated and used to calculate the certainty equivalent declining discount rate schedule (e.g., Newell and Pizer 2003; Groom et al. 2007). An alternative approach is to use the Ramsey equation based on a forecast of consumption growth rates that accounts for uncertainty (e.g., Cropper et al. 2014; Arrow et al. 2013). If the shocks to consumption growth are positively correlated over time then the result of the Ramsey equation will be a certainty-equivalent discount rate schedule that declines over time (Goiller 2014). Others have argued for a less structural approach to specify a declining discount rate schedule (e.g., Weitzman 2001, the United

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[25] For a more detailed explanation of ethical and uncertainty considerations around discounting see National Academies (2017) and the 2010 TSD (IWG 2010).

Kingdom's "Green Book" for regulatory analysis (HM Treasury 2020), the declining discount schedule in France (Lebègue 2005) and varying the discount rate based on the time period in Germany (Schwermer 2012, U.S. GAO 2020)). This approach uses a higher discount rate initially, like the current estimate of the consumption interest rate, but applies a graduated scale of lower discount rates further out in time.[26]

Instead of explicitly specifying a declining discount rate schedule, the IWG in 2010 elected to use a constant but lower discount rate to capture the directional effect of the literature on discounting under uncertainty. Specifically, the IWG considered two declining discount rate schedules based on the mean-reverting and random walk models from Newell and Pizer (2003) starting at a discount rate of 3 percent. The 2.5 percent discount rate selected by the IWG in 2010 reflected the midpoint between the average certainty equivalent discount rates of both models. The approach of using a lower, but constant, discount rate to capture the effect of uncertainty has led to inconsistency in regulatory analyses, where impacts occurring in a given year are discounted at different rates depending on whether they are related to climate change (Arrow et al. 2014). The National Academies (2017) and EPA's Science Advisory Board (2021) have recommended that the U.S. Government establish an explicit declining discount rate schedule that is applied to all regulatory impacts in an analysis to capture the effect of uncertainty on long-term discount rates, while also maintaining consistency across impact categories in the analysis. The IWG will consider the literature on declining discount rates and the recommendations of the National Academies (2017) and EPA's Science Advisory Board (2021) as it develops future updates to the SC-GHG. In the interim, the IWG is returning to the use of the 2.5, 3, and 5 percent discount rates in calculating the SC-GHG but recommends that agencies describe potential limitations in their analyses to ensure transparency. As noted above, agencies may also consider discount rates below 2.5 percent as part of a sensitivity analysis.

4 Interim Estimates of SC-CO_2, SC-CH_4, SC-N_2O

The interim SC-GHG estimates presented in this TSD rely on the same models and harmonized inputs for the socioeconomic emissions scenarios and equilibrium climate sensitivity distribution used for USG SC-GHG estimates since 2013. Specifically, the SC-GHG estimates rely on an ensemble of three IAMs: Dynamic Integrated Climate and Economy (DICE) 2010 (Nordhaus 2010); Climate Framework for Uncertainty, Negotiation, and Distribution (FUND) 3.8 (Anthoff and Tol 2013a, 2013b); and Policy Analysis of the Greenhouse Gas Effect (PAGE) 2009 (Hope 2013). IAMs are useful because they combine climate processes, economic growth, and feedback between the climate and the global economy into a single modeling framework. They gain this advantage at the expense of a more detailed representation of underlying climatic and economic systems. DICE, PAGE, and FUND all take stylized, reduced-form approaches and have been widely used in the economic and scientific literature since the 1990s. They are periodically updated by the model developers, but as discussed further in Section 5, the versions of the three models used in the 2013 and 2016 TSDs do not reflect the tremendous increase in the scientific and economic understanding of climate-related damages that has occurred in the past decade. The three IAMs

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[26] For instance, the United Kingdom applies a discount rate of 3.5 percent to the first 30 years; 3 percent for years 31 - 75; 2.5 percent for years 76 - 125; 2 percent for years 126 - 200; 1.5 percent for years 201 - 300; and 1 percent after 300 years. As a sensitivity, it recommends a discount rate of 3 percent for the first 30 years, also decreasing over time.

were run using a common set of assumptions in each model for future population, economic, and GHG emissions growth, as well as equilibrium climate sensitivity (ECS) – a measure of the globally averaged temperature response to increased atmospheric CO_2 concentrations. The socioeconomic and emission projections included five reference scenarios based on the Stanford Energy Modeling Forum EMF-22 modeling exercise (Clarke, et al. 2009; Fawcett, et al. 2009). The models were run using a probability distribution for ECS, calibrated to the Intergovernmental Panel on Climate Change's (IPCC) Fourth Assessment Report findings using the Roe and Baker (2007) distribution. Details on these versions of the IAMs and the harmonized inputs are presented in the 2016 TSD and Addendum and 2010 TSD. (IWG 2010, 2016a, 2016b). The 2016 Addendum also describes the methodology used to calculate the SC-CH_4 and SC-N_2O estimates in greater detail.[27] Finally, for the reasons set forth in Section 3 above, the interim estimates were based on three constant discount rates of 2.5, 3, and 5 percent.

The combination of three models and five scenarios produced 15 separate frequency distributions of SC-GHG estimates for each discount rate in a given year, with each distribution consisting of 10,000 estimates based on draws from the standardized ECS distribution (as well as distributions of parameters treated as uncertain in two of the models (FUND and PAGE)). For each discount rate, the IWG combined the distributions across models and socioeconomic emissions scenarios (applying equal weight to each) and then selected a set of four values for use in benefit-cost analyses: an average value resulting from the model runs for each of three discount rates (2.5%, 3%, and 5%), plus a fourth value, selected as the 95th percentile of estimates based on a 3 percent discount rate. The fourth value was included to provide information on potentially higher-than-expected economic impacts from climate change, conditional on the 3% estimate of the discount rate. For this purpose, the SC-GHG value for the 95th percentile at a 3 percent discount rate was presented.[28] For the purposes of capturing the uncertainties involved in analyses, the IWG emphasized previously and emphasizes in this TSD the importance and value of including all four SC-GHG values. In particular, values based on lower discount rates are consistent with the latest scientific and economic understanding of discounting approaches relevant for intergenerational analysis (described in Section 3).

Tables 1-3 show the four selected values for SC-CO_2, SC-CH_4, and SC-N_2O, respectively, in five-year increments from 2020 to 2050. These estimates are reported in 2020 dollars but are otherwise identical to those presented in the previous version of the TSD and its Addendum, released in August 2016.[29] The

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[27] The IWG calculated the SC-CH_4 and SC-N_2O estimates following the approach used in Marten et al. (2015). In order to develop SC-CH_4 and SC-N_2O estimates consistent with the methodology underlying the SC-CO_2 estimates, Marten et al. (2015) needed to augment the IWG modeling framework in two respects: (1) augment the climate model of two of the IAMs to explicitly consider the path of additional radiative forcing from a CH_4 or N_2O perturbation, and (2) add more specificity to the assumptions regarding post-2100 baseline CH_4 and N_2O emissions. See IWG (2016b) for more discussion of these two modeling modifications and the peer review and public comment processes accompanying their development.

[28] A detailed set of percentiles by model and scenario combination and additional summary statistics for the 2020 values is available in the 2016 TSD and Addendum (IWG 2016a, 2016b).

[29] The values in Tables 1-3 are the same as those reported in the 2016 TSD and Addendum adjusted for inflation to 2020 dollars using the annual GDP Implicit Price Deflator values in U.S. Bureau of Economic Analysis (BEA) NIPA Table 1.1.9: 113.626 (2020)/ 92.486 (2007) = 1.228575 (U.S. BEA 2021). Values of SC-CO_2 presented in this TSD are rounded to the nearest dollar; SC-CH_4 and SC-N_2O are rounded to two significant figures. The annual unrounded estimates are available on OMB's website for use in regulatory and other analyses: https://www.whitehouse.gov/omb/information-regulatory-affairs/regulatory-matters/#scghgs.

full set of annual SC-GHG values between 2020 and 2050, calculated using linear interpolation between the numbers shown in Tables 1-3, is reported in the Appendix and the full set of model results are available on the OMB website.[30] The SC-GHG estimates increase over time within the models – i.e., the societal harm from one metric ton emitted in 2030 is higher than the harm caused by one metric ton emitted in 2025 – because future emissions produce larger incremental damages as physical and economic systems become more stressed in response to greater climatic change, and because GDP is growing over time and many damage categories are modeled as proportional to GDP.

Table 1: Social Cost of CO_2, 2020 – 2050 (in 2020 dollars per metric ton of CO_2)[31]

	Discount Rate and Statistic			
Emissions Year	5% Average	3% Average	2.5% Average	3% 95th Percentile
2020	14	51	76	152
2025	17	56	83	169
2030	19	62	89	187
2035	22	67	96	206
2040	25	73	103	225
2045	28	79	110	242
2050	32	85	116	260

Table 2: Social Cost of CH_4, 2020 – 2050 (in 2020 dollars per metric ton of CH_4)

	Discount Rate and Statistic			
Emissions Year	5% Average	3% Average	2.5% Average	3% 95th Percentile
2020	670	1500	2000	3900
2025	800	1700	2200	4500
2030	940	2000	2500	5200
2035	1100	2200	2800	6000
2040	1300	2500	3100	6700
2045	1500	2800	3500	7500
2050	1700	3100	3800	8200

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[30] https://www.whitehouse.gov/omb/information-regulatory-affairs/regulatory-matters/#scghgs

[31] The values reported in this TSD are identical to those reported in the 2016 TSD adjusted for inflation to 2020 dollars using the annual GDP Implicit Price Deflator values in the U.S. Bureau of Economic Analysis' (BEA) NIPA Table 1.1.9: 113.626 (2020)/ 92.486 (2007) = 1.228575 (U.S. BEA 2021). The IWG combined the distributions across models and socioeconomic emissions scenarios for each of three discount rates (2.5%, 3%, and 5%), plus a fourth value, selected as the 95th percentile of estimates based on a 3 percent discount rate. Values of SC-CO_2 are rounded to the nearest dollar; SC-CH_4 and SC-N_2O are rounded to two significant figures. The annual unrounded estimates are available on OMB's website for use in regulatory and other analyses: https://www.whitehouse.gov/omb/information-regulatory-affairs/regulatory-matters/#scghgs.

Table 3: Social Cost of N_2O, 2020 – 2050 (in 2020 dollars per metric ton of N_2O)

	Discount Rate and Statistic			
Emissions Year	5% Average	3% Average	2.5% Average	3% 95th Percentile
2020	5800	18000	27000	48000
2025	6800	21000	30000	54000
2030	7800	23000	33000	60000
2035	9000	25000	36000	67000
2040	10000	28000	39000	74000
2045	12000	30000	42000	81000
2050	13000	33000	45000	88000

Multiplying the SC-GHG in year *t* by the change in emissions in year *t* yields the monetized value of future emission changes from a year *t* perspective. This value must then be discounted to the present before being included in an analysis. For this purpose, the monetized value of future emission changes should be discounted at the same rate used to calculate the initial SC-GHG to ensure internal consistency—i.e., future damages from climate change using the SC-GHG at 2.5 percent should be discounted to the base year of the analysis using the same 2.5 percent rate.

As noted above, to correctly assess the total climate damages to U.S. citizens and residents, an analysis must account for both the impacts that occur within U.S. borders and spillover effects from climate action elsewhere. For the reasons discussed in Section 2 above, estimates focusing on the climate impacts occurring within U.S. borders are an underestimate of the benefits of GHG mitigation accruing to U.S. citizens and residents and, therefore, are not equivalent to a domestic estimate of the SC-GHG. (Section 2 also discusses why analyses should center their attention on a global measure of the SC-GHG). Additionally, models differ in their treatment of regional damages[32] with one of the model developers recently noting that regional damages are "both incomplete and poorly understood" (Nordhaus 2017). The IWG further notes that the domestic focused SC-GHG estimates used under E.O. 13783[33] did not

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[32] Both the PAGE and FUND model contain a U.S. region and so the damages for this region are reported directly for those models. The DICE 2010 model does not explicitly include a separate U.S. region in the model. For the domestic focused SC-GHG estimates used under E.O. 13783, the DICE model damages occurring within U.S. borders were approximated as 10 percent of the global estimate from the DICE model runs, based on the results from a regionalized version of the model (RICE 2010) reported in Table 2 of Nordhaus (2017). Although the regional shares reported in Nordhaus (2017) are specific to SC-CO_2, they were also used in approximating the share of marginal damages from CH_4 and N_2O emissions occurring within U.S. borders. Direct transfer of the U.S. share from the SC-CO_2 likely understate the U.S. share of the IWG global SC-CH_4 estimates based on DICE due to the combination of three factors: a) regional damage estimates are known to be highly correlated with output shares (Nordhaus 2017, 2014), b) the U.S. share of global output decreases over time in all five EMF-22 based socioeconomic scenarios used for the model runs, and c) the bulk of the temperature anomaly (and hence, resulting damages) from a perturbation in emissions in a given year will be experienced earlier for CH_4 than CO_2 due to the shorter lifetime of CH_4 relative to CO_2.

[33] For emissions occurring in 2020, the average estimates of marginal damages occurring within the U.S. borders for CO_2, CH_4, and N_2O emissions across all model runs that were used in 2017-2020 regulatory analyses were $7/mt$CO_2$,

benefit from a consensus-based IWG process, were not documented in a dedicated TSD, subjected to a SC-GHG specific notice and comment period, or considered by National Academies in their 2017 review. The IWG will request public comments on the new information presented in this TSD, as well as other topics and issues the IWG will address as we develop the next set of updates (see Section 6).

4.1 Treatment of Uncertainty

Uncertainty about the value of the SC-GHGs is in part inherent, as with any analysis that looks into the future, but it is also driven by current data gaps associated with the complex physical, economic, and behavioral processes that link GHG emissions to human health and well-being. Some sources of uncertainty pertain to aspects of the natural world, such as quantifying the physical effects of greenhouse gas emissions on Earth systems. Other sources of uncertainty are associated with current and future human behavior and well-being, such as population and economic growth, GHG emissions, the translation of Earth system changes to economic damages, and the potential extent and costs of adaptation. It is important to note that even in the presence of uncertainty, scientific and economic analysis can provide valuable information to the public and decision makers. Such uncertainty should, however, be acknowledged, communicated as clearly as possible, and taken into account in the analysis whenever possible.

The 2016 TSD and the 2017 National Academies report provide detailed discussions of the ways in which the modeling underlying the development of the SC-GHG estimates addressed quantified sources of uncertainty.

In developing the SC-CO_2 estimates, the IWG considered various sources of uncertainty through a combination of a multi-model ensemble, probabilistic analysis, and scenario analysis. For example, the three IAMs used collectively span a wide range of Earth system and economic outcomes to help reflect the uncertainty in the literature and in the underlying dynamics being modeled. The use of an ensemble of three different models is also intended to, at least partially, address the fact that no single model includes all of the quantified economic damages. It also helps to reflect structural uncertainty across the models, which is uncertainty in the underlying relationships between GHG emissions, Earth systems, and economic damages that are included in the models. Bearing in mind the different limitations of each model (discussed in the 2010 TSD) and lacking an objective basis upon which to differentially weight the models, the three IAMs were given equal weight in the analysis.

The IWG used Monte Carlo techniques to run the IAMs a large number of times. In each simulation the uncertain parameters are represented by random draws from their defined probability distributions. In all three models the equilibrium climate sensitivity is treated probabilistically based on the probability distribution described in the 2010 TSD. The equilibrium climate sensitivity is a key parameter in this

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\$190/mtCH_4, and \$2,300/mtN_2O (in 2020 dollars), respectively, using a 3 percent discount rate, and \$1/mtCO_2, \$59/mtCH_4, and \$380/mtN_2O (in 2020 dollars) using a 7 percent discount rate. These values increased over time; for 2050 emissions, the average estimates of marginal damages occurring within the U.S. borders are \$11/mtCO_2, \$380/mtCH_4, and \$4,000/mtN_2O (in 2020 dollars) using a 3% discount rate and \$3/mtCO_2, \$160/mtCH_4, and \$1,000/mtN_2O (in 2020 dollars) using a 7% discount rate. Using the same approach with a 2.5 percent discount rate, the average estimates of marginal damages occurring within the U.S. borders of CO_2, CH_4, and N_2O for emissions in 2020 are \$10/mtCO_2, \$240/mtCH_4, and \$3,300/mtN_2O (in 2020 dollars), respectively; for 2050 emissions, these values increase to \$15/mtCO_2, \$450/mtCH_4, and \$5,300/mtN_2O (in 2020 dollars).

analysis because it helps define the strength of the climate response to increasing GHG concentrations in the atmosphere. In addition, the FUND and PAGE models define many of their parameters with probability distributions instead of point estimates. For these two models, the model developers' default probability distributions are maintained for all parameters other than those superseded by the IWG's harmonized inputs (i.e., equilibrium climate sensitivity, socioeconomic and emissions scenarios, and discount rates). More information on the uncertain parameters in PAGE and FUND is presented in Appendix C of the 2016 TSD (IWG 2016a).

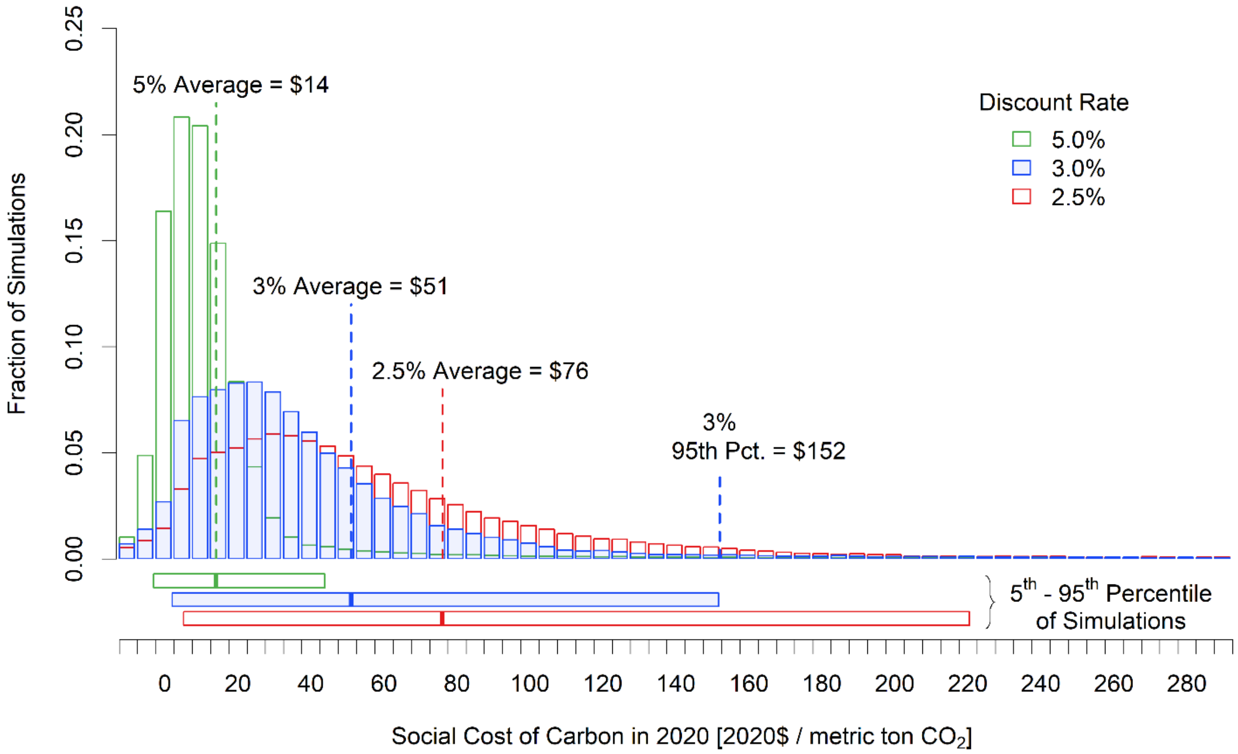
Finally, based on the review of the literature, the IWG chose discount rates that reflect reasonable judgements under both prescriptive and descriptive approaches to intergenerational discounting. As discussed in the 2010 TSD, in light of disagreement in the literature on the appropriate discount rate to use in this context and uncertainty about how rates may change over time, the IWG selected three certainty-equivalent constant discount rates to span a plausible range: 2.5, 3, and 5 percent per year. However, unlike the approach taken for consolidating results across models and socioeconomic and emissions scenarios, the SC-GHG estimates are not pooled across different discount rates because the range of discount rates reflects both uncertainty and, at least in part, different policy or value judgements.

The outcome of accounting for various sources of uncertainty using the approaches described above is a frequency distribution of the SC-CO_2 estimates for emissions occurring in a given year for each of the three discount rates. These frequency distributions reflect the uncertainty around the input parameters for which probability distributions were defined, as well as from the multi-model ensemble and socioeconomic and emissions scenarios where probabilities were implied by the equal weighting assumption. It is important to note that the probability distribution for the SC-GHG calculated using the modeling approach outlined above does not fully characterize uncertainty about the SC-GHG due to impact categories omitted from the models and sources of uncertainty that have not been fully characterized due to data limitations. To name just one example of many known GHG-induced damages omitted in the three IAMs, none of the models include damages associated with ocean acidification, and, therefore, naturally the models do not reflect uncertainty as to the potential severity of those damages.

Figures Figure ***2*** through Figure ***4*** present the frequency distribution of the interim SC-CO_2, SC-CH_4, and SC-N_2O estimates, respectively, for emissions in 2020 and for each discount rate. Each distribution represents 150,000 estimates based on 10,000 simulations for each combination of the three models and five socioeconomic and emissions scenarios. In general, the distributions are skewed to the right and have long right tails, which tend to be longer for lower discount rates. To highlight the difference between the impact of the discount rate on the SC-GHG and other quantified sources of uncertainty, the bars below the frequency distributions provide a symmetric representation of quantified variability in the SC-GHG estimates conditioned on each discount rate. The full set of SC-GHG results through 2050 is available on OMB's website.

As illustrated by the frequency distributions in Figures Figure ***2*** through Figure ***4*** , the assumed discount rate plays a critical role in the ultimate estimate of the SC-GHG. As explained in Section 3, this is because GHG emissions today continue to impact society far out into the future, so with a higher discount rate, costs that accrue to future generations are weighted less, resulting in a lower estimate. As discussed in Section 3.1, new data and evidence strongly suggest that the consumption interest rate is likely to be less than 3, near 2 percent or lower.

Figure 2: Frequency Distribution of SC-CO$_2$ Estimates for 2020[34]



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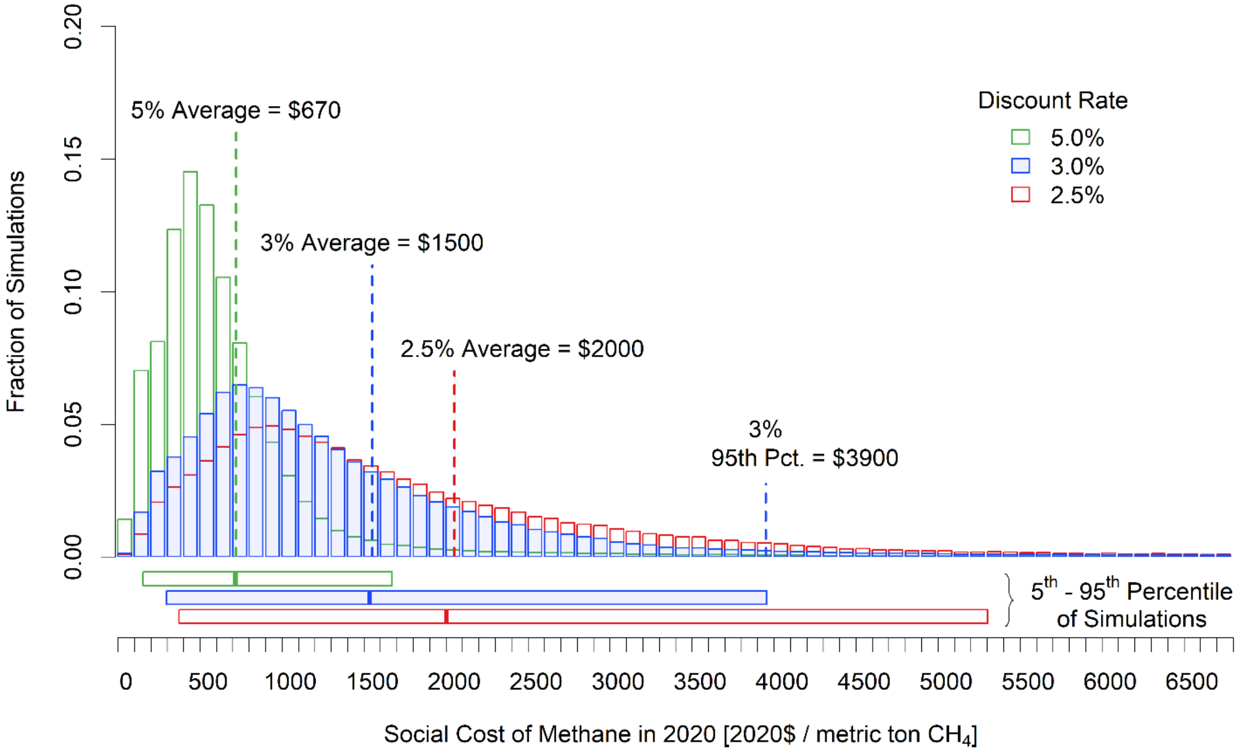
[34] Although the distributions and numbers in Figure 2 are based on the full set of model results (150,000 estimates for each discount rate), for display purposes the horizontal axis is truncated with 0.81 percent of the estimates falling below the lowest bin displayed and 3.56 percent of the estimates falling above the highest bin displayed.

Figure 3: Frequency Distribution of SC-CH₄ Estimates for 2020[35]



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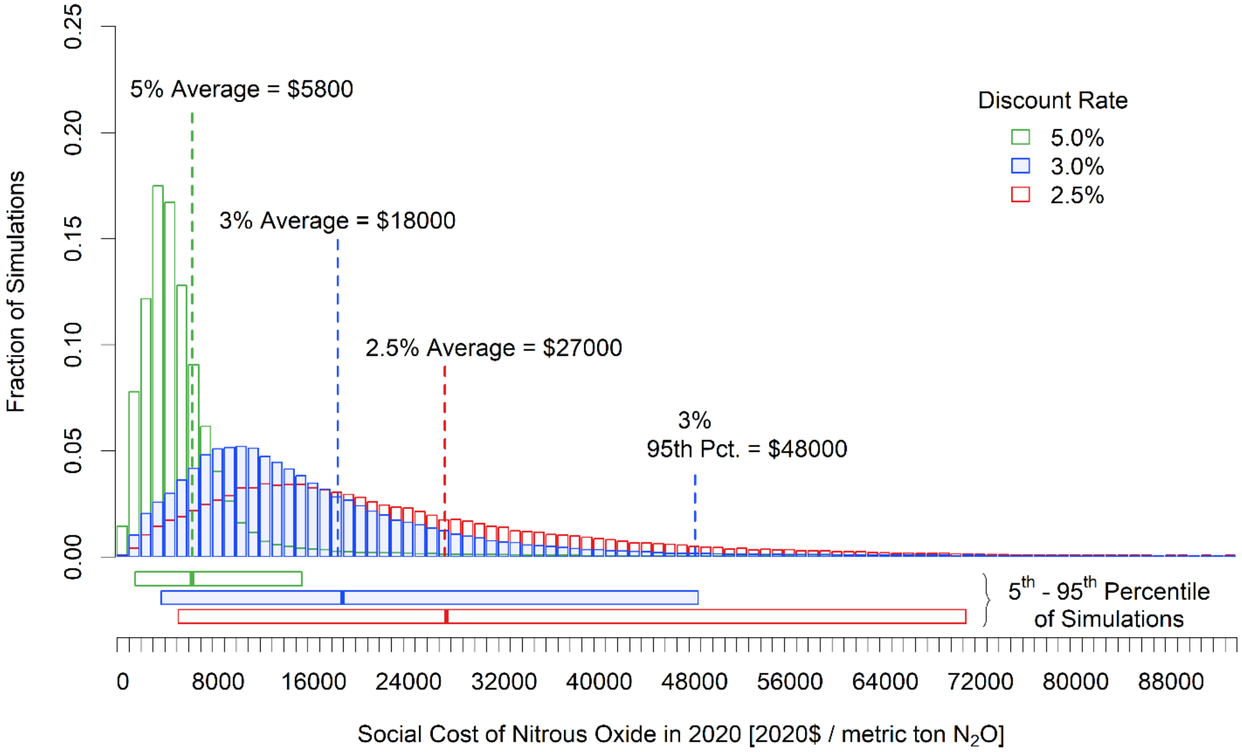
[35] Although the distributions and numbers in Figure 3 are based on the full set of model results (150,000 estimates for each discount rate), for display purposes the horizontal axis is truncated with 0.12 percent of the estimates falling below the lowest bin displayed and 2.84 percent of the estimates falling above the highest bin displayed.

Figure 4: Frequency Distribution of SC-N_2O Estimates for 2020[36]



While the figures above reflect the uncertainties that are explicitly considered in a quantitative manner, there are other areas of uncertainty that are not quantitatively reflected in the interim SC-GHG estimates. The scientific and economics literature has further explored known sources of uncertainty related to estimates of the SC-GHG. For example, published studies explore the sensitivity of IAMs and the resulting SC-GHG estimates to different assumptions embedded in the models (see, e.g., Hope 2013, Anthoff and Tol 2013a, and Nordhaus 2014). However, there remain additional sources of uncertainty that have not been fully characterized and explored due to data limitations and lack of consensus in the scientific or economic literature about how to represent them. Additional research is needed to expand the quantification of various sources of uncertainty in estimates of the SC-GHG (e.g., developing explicit probability distributions for more inputs pertaining to climate impacts and their valuation).

4.2 Other Modeling Limitations

The interim SC-GHG estimates presented in this TSD have a number of limitations, as would be expected for any modeling exercise that covers such a broad scope of scientific and economic issues across the complex global landscape. These include the incomplete treatment of catastrophic and non-catastrophic impacts in the IAMs, their incomplete treatment of adaptation and technological change, the incomplete way in which inter-regional and intersectoral linkages are modeled, uncertainty in the extrapolation of

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[36] Although the distributions and numbers in Figure 4 are based on the full set of model results (150,000 estimates for each discount rate), for display purposes the horizontal axis is truncated with 0.1 percent of the estimates falling below the lowest bin displayed and 2.85 percent of the estimates falling above the highest bin displayed.

damages to high temperatures, and inadequate representation of the relationship between the discount rate and uncertainty in economic growth over long time horizons.

There are newer versions available of each of the IAMs used to calculate the interim SC-GHG estimates in this TSD that offer improvements in some of these areas beyond the version of the models used for the interim estimates. For example, the latest version of the PAGE model, PAGE-ICE (Yumashev et al. 2019, Yumashev 2020), extends PAGE09 (Hope 2013) with representation of two nonlinear Arctic feedbacks (permafrost carbon feedback and surface albedo feedback) on the global climate system and economy, among other changes. The newest version of the DICE model, DICE2016-R3 (Nordhaus 2017), includes numerous updates, including changes to the carbon cycle (to better simulate the long-run behavior of larger models with full ocean chemistry) and updated methods for estimating economic activity.[37] At comparable discount rates, DICE2016-R3 would result in SC-CO_2 estimates roughly twice that of the interim estimates presented in this TSD. For example, using a 3% constant discount rate and other IWG modeling assumptions, DICE2016-R3 yields an average SC-CO_2 of $104 (2018 international dollars) for 2020 emissions (Nordhaus 2019a). However, even DICE2016 and PAGE-ICE do not include all of the important physical, ecological, and economic impacts of climate change recognized in the climate change literature and the science underlying their damage functions lags behind the most recent research. Likewise, the socioeconomic and emissions scenarios used as inputs to the models in this TSD do not reflect new information from the last decade of scenario generation or the full range of projections.

The modeling limitations discussed above do not all work in the same direction in terms of their influence on the SC-GHG estimates. However, it is the IWG's judgment that, taken together, the limitations suggest that the interim SC-GHG estimates presented in this TSD likely underestimate the damages from GHG emissions. In particular, the IPCC's Fourth Assessment Report (IPCC 2007), which was the most current IPCC assessment available at the time when the IWG decision over the ECS input was made, concluded that SC-CO_2 estimates "very likely…underestimate the damage costs" due to omitted impacts. Since then, the peer-reviewed literature has continued to support this conclusion, as noted in the IPCC's Fifth Assessment report (IPCC 2014) and other recent scientific assessments (e.g., IPCC 2018, 2019a, 2019b; U.S. Global Change Research Program (USGCRP) 2016, 2018; and National Academies 2016b, 2019). These assessments confirm and strengthen the science, updating projections of future climate change and documenting and attributing ongoing changes. For example, sea level rise projections from the IPCC's Fourth Assessment report ranged from 18 to 59 centimeters by the 2090s relative to 1980-1999, while excluding any dynamic changes in ice sheets due to the limited understanding of those processes at the time (IPCC 2007). A decade later, the Fourth National Climate Assessment projected a substantially larger sea level rise of 30 to 130 centimeters by the end of the century relative to 2000, while not ruling out even more extreme outcomes (USGCRP 2018). Section 5 briefly previews some of the recent advances in the

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[37] Relative to the previous version of DICE, DICE2013, the DICE2016 updates to the carbon cycle and the methods for estimating economic activity had the greatest impact on the SC-CO_2. Based on Archer et al. (2009), DICE2016's three-box carbon cycle model aims to better simulate the long-run behavior of larger models with full ocean chemistry. In measuring economic activity, one of the important changes in DICE2016 was to move from market exchange rates to measures adjusted for purchasing power parity when comparing monetary values across countries. See Nordhaus (2017, 2019a) for more discussion of these and other updates included in DICE2016-R3. Nordhaus has also recently explored side extensions of DICE2016. For example, DICE-GIS extends DICE2016 to include representation of sea level rise from melting of the Greenland Ice Sheet (Nordhaus 2019b, Pizer 2019).

scientific and economic literature that the IWG is actively following and that could provide guidance on, or methodologies for, addressing some of the limitations with the interim SC-GHG estimates.

5 Scientific and Economic Advances

The research community has made considerable progress in developing new data and methods that will provide a path forward for bringing the USG SC-GHG estimates closer to the current frontier of climate science and economics and could address many of the National Academies' (2017) recommendations. This research since 2010/2013 has advanced knowledge regarding each key component in the process of estimating the SC-GHG. This TSD does not intend to provide a detailed review of all these advancements, but this section does highlight some of the key research and new information that the IWG will be reviewing as it works to improve the SC-GHG estimates. As part of the process for updating the SC-GHG estimates by January 2022, the IWG will survey the scientific literature, including the economic literature, to identify advances to address the National Academies (2017) recommendations.

Climate system representation. There have been advancements in climate science since the publication of the IPCC's Fourth Assessment Synthesis report (IPCC 2007), which was the basis for the IWG decision on what equilibrium climate sensitivity (ECS) input to use in the IAM model runs. The conclusions of recent scientific assessments, e.g., from the IPCC (2014, 2018, 2019a, 2019b), the USGCRP (2016, 2018), and the National Academies (2016b, 2019), confirm and strengthen the science, updating projections of future climate change and documenting and attributing ongoing changes. In addition, there are reduced complexity climate models that could offer meaningful improvement over current representation of climate dynamics in existing IAMs (Nicholls et al. 2020). For example, the National Academies (2017) stated that the FAIR model (Smith et al., 2018) satisfies all of the criteria set by National Academies (2017) recommendations related to the representation of climate system dynamics, generates projections of future warming consistent with more complex, state of the art models, can be used to accurately characterize current best understanding of uncertainty, and can be easily implemented and transparently documented. Reduced complexity sea level rise models are also being developed that can provide projections for damage functions that require sea level estimates, including the contributions of thermal expansion and glacial and ice sheet melting based on recent scientific research (e.g., Wong et al. 2017).

***Damage functions*.** At the core of IAMs are "damage functions" that map global mean temperature changes and other physical impacts of climate change into economic (both market[38] and nonmarket[39]) damages. Relative to how much progress has been made in modeling and improving our understanding of climate system dynamics and the physical impacts resulting from temperature change, efforts involved in, and the public resources targeted at, understanding how these physical changes translate into economic impacts have been significantly smaller (Auffhammer 2018). Even so, as illustrated in Figure 5, in the time since the versions of the IAMs used in this TSD were published, there has been an explosion of research on climate impacts and damages.

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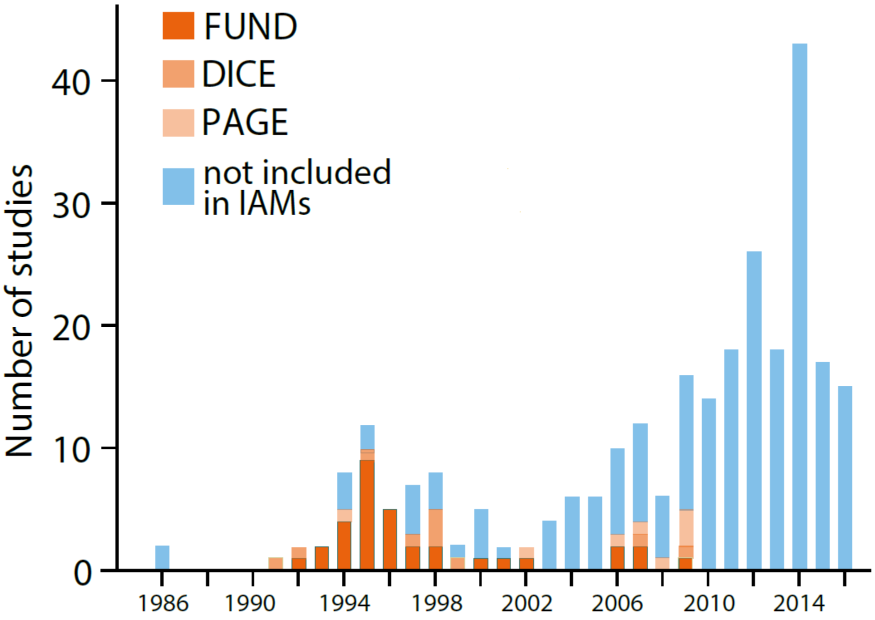
[38] Examples of market damages include changes in net agricultural productivity, energy use, and property damage from increased flood risk.

[39] Examples of nonmarket damages include services that natural ecosystems provide to society.

Figure 5. New Research on Climate Impacts[40]



Source: Greenstone (2016).

Several efforts are underway to draw on recent literature for improving damage functions and to generate new damage estimates. In particular, the Climate Impact Lab is undertaking an effort to quantify and monetize damages at a fine spatial scale, relying on rigorous empirical methods to develop plausibly causal estimates for several sectors, including health (Carleton et al. 2020), energy (Rode et al. 2021), labor productivity (Rode et al. 2020), agriculture, conflict, and sea level rise.[41] Other research efforts have sought to update the damage function for one sector in an existing IAM based on an updated review of the empirical literature on climate impacts pertaining to that sector (e.g., Moore et al. (2017) for agriculture damages in the FUND model). Damage functions specific to impacts within the U.S. have also been developed and improved for a number of sectors, such as impacts on coastal property, mortality due to extreme temperatures, transportation infrastructure, electricity supply and demand, water quality, recreation, and allergies (Neumann et al. 2020) and impacts of climate change on air quality and human health (Fann et al. 2021). There is also an emerging literature focused on incorporating interactions among

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[40] In many cases, the three IAMs used different studies for calibration. This is particularly true of FUND, which used studies relating to different subsectors of the model, whereas DICE and PAGE did not have as detailed a sectoral breakdown. That means that summing across these different models is likely valid in all but a few isolated cases. The blue bars include studies uncovered from a comprehensive literature review in the economics literature (and a few others in public health or relevant disciplines) by the Climate Impact Lab (CIL) through early 2016. Each of the studies counted in blue was determined by CIL to have employed a research design that allowed for the causal interpretation of results (Greenstone 2016).

[41] The Climate Impact Lab is a multidisciplinary collaboration of climate scientists, economists, computational experts, researchers, analysts, and students working to build empirically derived, local-level estimates of climate change damages and an empirically based SC-CO_2. More information on the Climate Impact Lab can be found at: http://www.impactlab.org/.

regions and impacts. For example, biodiversity loss (e.g., animal pollinators) as a result of climate-driven ecosystem stress could amplify impacts of climate change on agriculture. See National Academies (2017) for more discussion of recent research addressing these and other types of interactions.

Related to the development of damage functions, damages from climate change are uncertain and hence pose additional risks. Reductions in GHG emissions reduce not only expected damages, but also reduce the uncertainty and risks of catastrophic events. Evaluating the damages using the mean outcome does not account for the benefits of reducing uncertainty. Some researchers have raised the need to include this consideration in the SC-GHG (e.g., Carleton and Greenstone 2021) consistent with the observation that individuals are regularly willing to pay for insurance against bad outcomes.

Furthermore, E.O. 13990 instructs the IWG to consider how best to reflect environmental justice and intergenerational equity concerns in assessing climate damages. In the context of climate policy, equity considerations are discussed by economists, ethicists, and others in several ways: distributional effects within a specific country, effects across countries, and intergenerational equity impacts. Economists, ethicists, and others have proposed potential ways to incorporate equity into the SC-GHG. For example, IAM developers have introduced the use of equity weights potentially incorporate these concerns (e.g., Hope 2008; Anthoff and Emmerling 2019).

Socioeconomic and Emissions Projections. The socioeconomic and emissions projections underlying current USG SC-GHG estimates were developed around 2007. Since that time, there have been efforts to develop updated baseline scenarios. Several researchers have started using deterministic scenarios available as part of the IPCC's Fifth Assessment Report Working Group 3 database and the Shared Socioeconomic Pathways (SSPs) linked with the Representative Concentration Pathway (RCP) emissions scenarios (Riahi et al. 2017 and Moss et al. 2010) as benchmark scenarios. Resources for the Future (RFF) has engaged in a research effort to implement each of the National Academies' (2017) recommendations, in collaboration with research partners.[42] One part of this effort is focused on developing probability distributions for future paths of population, GDP, and emissions via using econometrics and expert elicitation techniques. For example, economic growth projections are being built off the results of a formal expert elicitation of leading growth economists together with recent research by Muller, Stock and Watson (2020), who have refined a foundational statistical methodology for generating long-run projections of economic growth at the country level. RFF plans to make these probabilistic scenarios easily usable on Mimi.jl, an open-source modular computing platform used for creating, running, and performing analyses on IAMs.[43]

Discounting. Another area of active research relates to discounting, including the best available evidence on the consumption rate of interest and the application of discount rates to regulations in which some costs and benefits accrue intra-generationally while others accrue inter-generationally. As described in Section 3.2, new empirical evidence suggests that consumption interest rates are now below the previous estimate of 3 percent presented in OMB's Circular A-4. This empirical evidence is also consistent with long-term forecasts by the Congressional Budget Office, suggesting these lower rates will persist (U.S. CBO

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[42] For more information on RFF's Social Cost of Carbon Initiative, see: https://www.rff.org/topics/scc/.

[43] Mimi.jl was developed by a team of researchers at UC Berkeley led by David Anthoff in response to a core recommendation from the National Academies (2017) to create an integrated modular approach to draw more readily on expertise from the wide range of scientific disciplines relevant to SC-CO_2 estimation. Mimi.jl provides an interface for defining components and building models in a modularized, transparent way (mimiframework.org).

2020). Future updates to the SC-GHGs estimates will need to reflect the best available evidence from the time series of risk-free rate data and expectations of these rates into the future.

As described in Section 3.3 uncertainty in the discount rate over time yields a declining certainty-equivalent discount rate schedule and can have a dramatic effect on the size of the SC-GHG. While this is not a new theoretical result, new literature has proposed methods for how to incorporate discount rate uncertainty (e.g., Arrow et al., 2013; Cropper et al., 2014) and other nations have implemented declining discount rate schedules for policy analysis (e.g., United Kingdom, France, and Germany). Recent recommendations by the National Academies (2017) and EPA's Science Advisory Board (2021) have encouraged the development and use of a declining certainty-equivalent discount rate schedule as theoretically appropriate and as a method of introducing consistency into analyses that have both near-term and long-term impacts.

In light of new science and evidence, including many of those highlighted in the paragraphs above, other jurisdictions are already considering or have implemented some of the scientific and economic advances discussed above. For example, some states that use SC-GHG estimates in policy analysis have recently updated their approach to discounting based on the increasing evidence that a 3% discount rate is too high for intergenerational analysis. In December 2020, New York issued guidance recommending state agencies use SC-GHG estimates based the same IWG modeling and input decisions as presented in this TSD but with lower discount rates: 2 percent in central scenarios (\$125/mtCO_2 for 2020 emissions (2020 dollars), along with sensitivity analysis at 1 percent and 3 percent (New York Department of Environmental Conservation 2020). Similarly, in Washington state an April 2019 law required utilities to use estimates based on the IWG methodology with a 2.5% discount rate when developing “lowest-cost analyses” for its integrated resource planning and clean energy plans.[44]

Canada is also in the process of updating the SC-GHG estimates used in their regulatory analyses. While the update is underway, they are continuing to use the estimates they adopted in 2016 (which are an adaptation of the IWG global SC-GHG estimates presented in this TSD) as well as a side analysis based on more recent estimates from the academic literature. Based on their review of the literature and latest climatological and economic evidence, they present their current estimates as a “likely underestimate [of] climate-related damages to society” and the side analysis as a way “to illustrate a range of plausible values if the Department were to update its [social cost of carbon] estimate based on new versions of the models currently used.”[45] Specifically, the side analysis includes SC-CO_2 estimates based on DICE2016 and PAGE-ICE (\$135 and \$440/mtCO_2 for 2020 emissions (2019 Canadian dollars)).[46]

The IWG will consider the new science and evidence as it works towards a more comprehensive update, including the new research and information described in this section.

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[44] Wash. Sen. Bill. 5116 (signed by Gov. Inslee on May 7, 2019). More information on Washington and other states' use of SC-GHG estimates is compiled by the Institute for Policy Integrity at NYU School of Law (see http://www.costofcarbon.org/states) and discussed in U.S. GAO (2020).

[45] Proposed Clean Fuel Regulations (published for public comment on 12/20/20) http://www.gazette.gc.ca/rp-pr/p1/2020/2020-12-19/pdf/g1-15451.pdf.

[46] Proposed Clean Fuel Regulations (published for public comment on 12/20/20) http://www.gazette.gc.ca/rp-pr/p1/2020/2020-12-19/pdf/g1-15451.pdf.

6 Path Forward

E.O. 13990 reaffirms that “[a]n accurate social cost is essential for agencies to accurately determine the social benefits of reducing greenhouse gas emissions when conducting cost-benefit analyses of regulatory and other actions” (E.O. 13990 2021). The E.O. instructs the IWG to publish interim SC-CO_2, SC-CH_4, and SC-N_2O estimates (collectively, SC-GHG estimates) within 30 days and to publish a set of final estimates by no later than January 2022.[47] In doing so, the E.O. instructs the IWG to consider the recommendations of the National Academies of Science, Engineering, and Medicine as reported in Valuing Climate Damages: Updating Estimation of the Social Cost of Carbon Dioxide (2017) and other pertinent scientific literature; solicit public comment; engage with the public and stakeholders; seek the advice of ethics experts; and ensure that the SC-GHG estimates reflect the interests of future generations in avoiding threats posed by climate change.

In developing the SC-GHG estimates in 2010, 2013, and 2016 the IWG used consensus-based decision making, relied on peer-reviewed literature and models, and took steps to disclose limitations and incorporate new information by considering public comments and revising the estimates as updated research became available (U.S. GAO 2014). Going forward the IWG commits to maintaining a consensus driven process for making evidence-based decisions that are guided by the best available science and input from the public, stakeholders, and peer reviewers.

While the IWG assesses the current state of the science in each component of the SC-GHG modeling exercise, the IWG is beginning by asking for public comment on how best to incorporate the latest, peer reviewed science to develop an updated set of SC-GHG estimates. The IWG will soon issue a Federal Register notice with a detailed set of requests for public comments on the new information presented in this TSD, as well as other topics and issues the IWG will address as we develop the next set of updates. Among other things, the IWG will ask for public comment on how to incorporate the best available science in the updated SC-GHG estimates, due to be published by January 2022, and how to incorporate the recommendations of the National Academies (2017).

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[47] The Executive Order also requests that the IWG assess the application of the SC-GHG to inform government decision making beyond regulations, in addition to recommending a robust long-term structure for ensuring the SC-GHGs continue to reflect the best available science and economic and that long-term research needs are met.

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Appendix – Annual SC-CO_2, SC-CH_4, and SC-N_2O Values, 2020-2050

The values in Tables A-1 through A-3 are the same as those reported in the 2016 TSD and Addendum adjusted for inflation to 2020 dollars using the annual GDP Implicit Price Deflator values in U.S. Bureau of Economic Analysis (BEA) NIPA Table 1.1.9: 113.626 (2020)/ 92.486 (2007) = 1.228575 (U.S. BEA 2021). Values of SC-CO_2 presented in this TSD are rounded to the nearest dollar; SC-CH_4 and SC-N_2O are rounded to two significant figures. The annual unrounded estimates are available on OMB's website for use in regulatory and other analyses: https://www.whitehouse.gov/omb/information-regulatory-affairs/regulatory-matters/#scghgs.

Table A-1: Annual SC-CO_2, 2020 – 2050 (in 2020 dollars per metric ton of CO_2)

	Discount Rate and Statistic			
Emissions Year	5% Average	3% Average	2.5% Average	3% 95th Percentile
2020	14	51	76	152
2021	15	52	78	155
2022	15	53	79	159
2023	16	54	80	162
2024	16	55	82	166
2025	17	56	83	169
2026	17	57	84	173
2027	18	59	86	176
2028	18	60	87	180
2029	19	61	88	183
2030	19	62	89	187
2031	20	63	91	191
2032	21	64	92	194
2033	21	65	94	198
2034	22	66	95	202
2035	22	67	96	206
2036	23	69	98	210
2037	23	70	99	213
2038	24	71	100	217
2039	25	72	102	221
2040	25	73	103	225
2041	26	74	104	228
2042	26	75	106	232
2043	27	77	107	235
2044	28	78	108	239
2045	28	79	110	242
2046	29	80	111	246
2047	30	81	112	249
2048	30	82	114	253
2049	31	84	115	256
2050	32	85	116	260

Table A-2: Annual SC-CH_4, 2020 – 2050 (in 2020 dollars per metric ton of CH_4)

	Discount Rate and Statistic			
Emissions Year	5% Average	3% Average	2.5% Average	3% 95th Percentile
2020	670	1500	2000	3900
2021	690	1500	2000	4000
2022	720	1600	2100	4200
2023	750	1600	2100	4300
2024	770	1700	2200	4400
2025	800	1700	2200	4500
2026	830	1800	2300	4700
2027	860	1800	2300	4800
2028	880	1900	2400	4900
2029	910	1900	2500	5100
2030	940	2000	2500	5200
2031	970	2000	2600	5300
2032	1000	2100	2600	5500
2033	1000	2100	2700	5700
2034	1100	2200	2800	5800
2035	1100	2200	2800	6000
2036	1100	2300	2900	6100
2037	1200	2300	3000	6300
2038	1200	2400	3000	6400
2039	1200	2500	3100	6600
2040	1300	2500	3100	6700
2041	1300	2600	3200	6900
2042	1400	2600	3300	7000
2043	1400	2700	3300	7200
2044	1400	2700	3400	7300
2045	1500	2800	3500	7500
2046	1500	2800	3500	7600
2047	1500	2900	3600	7700
2048	1600	3000	3700	7900
2049	1600	3000	3700	8000
2050	1700	3100	3800	8200

Table A-3: Annual SC-N_2O, 2020 – 2050 (in 2020 dollars per metric ton of N_2O)

	Discount Rate and Statistic			
Emissions Year	5% Average	3% Average	2.5% Average	3% 95th Percentile
2020	5800	18000	27000	48000
2021	6000	19000	28000	49000
2022	6200	19000	28000	51000
2023	6400	20000	29000	52000
2024	6600	20000	29000	53000
2025	6800	21000	30000	54000
2026	7000	21000	30000	56000
2027	7200	21000	31000	57000
2028	7400	22000	32000	58000
2029	7600	22000	32000	59000
2030	7800	23000	33000	60000
2031	8000	23000	33000	62000
2032	8300	24000	34000	63000
2033	8500	24000	35000	64000
2034	8800	25000	35000	66000
2035	9000	25000	36000	67000
2036	9300	26000	36000	68000
2037	9500	26000	37000	70000
2038	9800	27000	38000	71000
2039	10000	27000	38000	73000
2040	10000	28000	39000	74000
2041	11000	28000	39000	75000
2042	11000	29000	40000	77000
2043	11000	29000	41000	78000
2044	11000	30000	41000	80000
2045	12000	30000	42000	81000
2046	12000	31000	43000	82000
2047	12000	31000	43000	84000
2048	13000	32000	44000	85000
2049	13000	32000	45000	87000
2050	13000	33000	45000	88000

Attachment 31

Excerpts from Buck Project EA

United States
Department of
Agriculture

Forest
Service

May 2020

Environmental Assessment

Buck Project

Tusquitee Ranger District, Nantahala National Forest
Clay County, North Carolina

For information Contact: Steverson Moffat
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1.2 Description of the Project Area

The analysis area (AA) encompasses approximately 20,638 acres of National Forest System lands in Compartments 100–104 and 106–116 located in southeastern Clay County, North Carolina. National Forest System acres in the compartments are in management area (MA) 3B, 4C, 4D, 5, 13, and 14. Embedded within the compartments is management area 18 (riparian areas around perennial water bodies). MAs 3B and 4D are suitable for vegetation management through commercial timber sales. MAs 5 (Backcountry), 7 (Wilderness), 13 (Special Interest Areas), and 14 (Appalachian Trail Management Corridor) are not suitable for timber sales. No treatments are proposed in the unsuitable management areas. These are included in the analysis area to better project the wildlife effects of proposed treatments.

Elevations in the AA range from approximately 2,400 feet to over 4,600 feet. The AA includes the Boteler Peak, Chunky Gal, and Sharptop Ridge Inventoried Roadless Areas, the Buck Creek Serpentine Barrens Special Interest and Botanical Areas, North Carolina Natural Heritage Natural Areas and sites near Deep Gap and the Southern Nantahala Wilderness. There are no developed Forest Service recreation facilities in the AA. The Appalachian Trail is at the eastern end of the AA and the Chunky Gal Trail cuts across the AA along the main ridge of Chunky Gal Mountain. The majority of recreational use occurs at dispersed camp sites along Buck Creek and Deep Gap Road, in and adjacent to the rockhounding area at Glade Gap, with hunting and fishing occurring throughout the AA near open and closed National Forest System (NFS) roads.

The geology in the AA includes pockets of the Blue Ridge Belt, which is made up of sedimentary and metamorphic rock that contains mafic minerals with a basic pH. As most soils in the Southern Appalachians are acidic, these basic soils support unique plant assemblages. The Buck Creek Serpentine Barrens occur on soils derived from Blue Ridge Belt parent materials and provide habitat for locally rare insects. True to their name, the Barrens had a fewer number of widely spaced trees more typical of woodland rather than forest condition, but fire suppression during the 20th Century has allowed forests in the Barrens to become more fully stocked with woody vegetation, which is a departure from the range of historic variation in the area.

Access to compartments within the AA are provided by a combination of state roads and closed and open NFS roads. U.S. Highway 64 bisects the analysis area in and adjacent to compartments 104, 107, 108, 111, and 113. Portions of compartments 104, 108, 109, 110 and 114 are included in the inventory for potential additions to wilderness that was completed as part of the ongoing forest plan revision process. These areas have varying degrees of wilderness characteristics with the highest wilderness characteristics within the inventoried roadless areas which are unencumbered by NFS roads. Inclusion in the inventory is not a designation that conveys or requires a particular kind of management, and all treatments proposed in the Buck Project are consistent with the Land and Resource Management Plan for the Nantahala and Pisgah National Forests.

There is a long history of land use in the AA. Ritter Lumber Company, Gennett Lumber Company, and Champion Lumber Company owned large tracts of land in Clay County and Macon County in the late 19th and early 20th centuries. For a number of years Ritter Lumber Company operated a sawmill in the Rainbow Springs area just east of the Buck AA. Most easily accessible portions of the AA not included in a 20 year diameter limit contract between the

Pregitzer and Euskirchen (2004) synthesized results from 120 separate studies of carbon stocks and carbon fluxes for boreal, temperate, and tropical biomes. They found that in temperate forests NEP is lowest, and most variable, in young stands (0-30 years), highest in stands 31-70 years, and declines thereafter as stands age. These studies also reveal a general pattern of total carbon stocks declining after disturbance and then increasing rapidly during intermediate years and then at a declining rate over time until another significant disturbance (timber harvest or tree mortality resulting from drought, fire, insects, disease or other causes) kills large numbers of trees and again converts the stands to a carbon source where carbon emissions from decay of dead biomass exceed that amount of carbon removed from the atmosphere by photosynthesis within the stand.

The impacts of the action alternatives on global carbon sequestration and atmospheric concentrations of CO_2 are miniscule. However, the forests of the United States significantly reduce atmospheric concentrations of CO_2 resulting from fossil fuel emissions. The forest and wood products of the United States currently sequester approximately 200 teragrams (200 teragrams, or Tg, equals 196,841,306 U.S. tons) of carbon per year (Heath and Smith, 2004). This rate of carbon sequestration offsets approximately 10% of CO_2 emissions from burning fossil fuels (Birdsey et al., 2006). U.S. Forests currently contain 66,600 teragrams of carbon.

The short-term reduction in carbon stocks and sequestration rates resulting from the proposed project are imperceptibly small on global and national scales, as are the potential long-term benefits in terms of carbon storage. The currently large carbon sink in US forests is a result of past land use changes, including the re-growth of forests on large areas of the eastern U.S. harvest in the 19th and 20th centuries, and 20th century fire suppression in the western U.S. (Birdsey et al. 2006). The continuation of this large carbon sink is uncertain because some of the processes promoting the current sink are likely to decline and projected increases in disturbance rates such as fire and large-scale insect mortality may release a significant fraction of existing carbon stocks (Pacala et al. 2008; Canadell et al. 2007).

Management actions - - such as those proposed - - that improve the resilience of forest to climate-induced increases in frequency, and utilize harvested trees for long-lived forest products and renewable energy sources may help sustain the current strength of the carbon sink in U.S. forests (Birdsey et al. 2007).

Cumulative Effects to Climate Change: The contribution of the proposed project activities to the carbon cycle is extremely small under Alternatives B, B Modified, C, D, and G. Conducting regeneration harvesting on approximately 845 acres (Alternative B), 770 acres (Alternative B Modified), 953 acres (Alternative C), 497 acres (Alternative D), or 795 acres (Alternative G) would result in new ESH on approximately 4.1%, 3.7%, 4.5%, 2.4%, or 3.85%, respectively, of the 20,638 analysis area acres.

The long-term ability of forests to sequester carbon depends in part on their resilience to multiple stresses, including increasing probability of drought stress, high-severity fires, and large-scale insect outbreaks associated with projected climate change. Thus, even though some management actions may in the near-term reduce total carbon stored below current levels, in the long term they may improve the overall capacity of the forest to sequester carbon. Sustainable forestry practices can increase the ability of forests to sequester atmospheric carbon while enhancing other ecosystem services. Planting new trees and improving forest health through thinning and prescribed burning, for example, are some of the ways to increase forest carbon in the long run. Harvesting and regenerating forests can also result in net carbon sequestration in wood products and new forest growth.

When combined, the carbon from this and past projects in the analysis area has a minimal cumulative effect not only at the local level, but at the larger level. When implemented, the rate of carbon release through timber regeneration would be minimal for the reasonably foreseeable future. There are no ongoing projects within the analysis area that would appreciably contribute to climate change, although the 129 acres of existing ESH in the 0 □20 year age classes on National Forest System lands in the AA are still net carbon producers as described in the direct effect section previously, and no reasonably foreseeable future Forest Service actions that would affect climate change in the Buck AA.

3.16 Effects to Wilderness Characteristics

Treatments proposed by Buck Project action alternatives in units 104/13, 104/18, 104/19, 104/23 108/20, 108/23, 109/7, 110/7, 110/18, 110/22, and 114/7 are within either the Chunky Gal or Boteler Peak areas inventoried for potential additions to wilderness. Treatments would affect the naturalness of the areas in their immediate vicinity but would not affect consideration of the overall areas for recommended wilderness. Timber operations would cause short-term immediate impacts to solitude in the activity areas during implementation of silvicultural treatments. Post-harvest impacts to naturalness and solitude would be of limited duration and would not preclude future consideration of these areas for wilderness recommendation as impacts from past management treatments alone do not preclude an area from being included in the inventory.

Stands 104/13, 104/18, 104/19, and 104/23 proposed in Alternatives B, B-Modified, C, and/or G lie within the 10,524 acre Boteler Peak inventory area, adjacent to but outside of the Boteler Peak Inventoried Roadless Area. They are all either directly adjacent to or within a half mile of an open Forest Service system road. Open roads, an existing communication tower, and a cleared utility corridor all detract from naturalness and solitude in the vicinity of the proposed treatments. Proposed silvicultural treatments on up to 100 acres across four units would not further impact wilderness characteristics in the eastern portion of the Boteler Peak inventory area. Other treatments in this area include temporary road construction to provide access for silvicultural treatments and prescribed burning. Both prescribed fire and temporary road construction would be short-term impacts on the landscape that would not further detract from wilderness characteristics in the vicinity. None of the proposed treatments in Compartment 104 would result in an irretrievable commitment of resources nor would they result in any permanent improvements. Therefore, proposed activities would not preclude future potential recommendation for wilderness.

Stands 108/20, 108/23, 109/7, 110/7, 110/18, 110/22, and 114/7 proposed in Alternatives B, B-Modified, C, and/or G lie within the 7,785 acre Chunky Gal inventory area, adjacent to but outside of the Chunky Gal and Sharptop Ridge Inventoried Roadless Areas. All units are either directly adjacent to or within a half mile of an open Forest Service system road. Proposed silvicultural treatments in compartment 108 are less than a quarter mile from U.S. Highway 64

<u>Attachment 32</u>

Excerpts from Southside Project EA

United States
Department of
Agriculture

Forest
Service

February 2019

Final Environmental Assessment

Southside Project

Nantahala Ranger District, Nantahala National Forest
Jackson and Macon Counties, North Carolina

For Information Contact: Steverson Moffat
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Existing Condition and How Climate Change Affects Analysis Area Resources: The existing condition is an analysis area that is typical of the southern Appalachians, with a range of elevations from approximately 1,950 feet to 4,900 feet. Climate change models are continuing to be developed and refined, but the two principal models found to best simulate future climate-changed conditions for the various regions across the country are the Hadley Centre model and the Canadian Climate Centre model (Climate Change Impacts on the United States 2001). Both models indicate warming in the southern region of the United States. However, the models differ considerably. One predicts little change in precipitation until 2030, followed by much drier conditions over the next 70 years. The other predicts a slight decrease in precipitation during the next 30 years, followed by increased precipitation.

Either of these climate scenarios with their attendant changes could affect forest productivity, forest pest activity, vegetation types, major weather disturbances (droughts, hurricanes), and streamflow. These effects would likely be seen across the entire national forest system in the United States. In the Southern Appalachians, it is possible that in the long run, a warmer climate will result in certain species□(cold-adapted ones such as northern hardwoods) ranges moving northward. In turn, species that currently have a more southerly range might start appearing here. In general, concerning both vegetation and wildlife, species that are generalists and can tolerate a wider range of habitat conditions will probably fare better than those with a set of narrow habitat requirements and conditions.

Scope of Analysis: The scope of this analysis for direct, indirect, and cumulative effects on climate change includes the 18,944 acres of National Forest System lands in the compartments that comprise the analysis area. The time frame used in this analysis is up to ten years after completion of the project activities.

Direct and Indirect Effects: **Alternative A** - Alternative A (No Action) would result in no change to the current trend for carbon storage or release. Forested stands are expected to be less resilient to possible climate change impacts, such as changes in productivity or insect and disease.

Alternatives B and C - It is not expected that Alternatives B and C would substantially alter the effects of climate change in the project area. The regeneration in the areas to be harvested would provide more structural diversity to the area and establish young, vigorous stands that may be more resilient to the changes in climate than those ages 70 and older. In addition, it is anticipated that the existing forest types in the stands to be regenerated would regenerate naturally to these same forest types; climate change will not make any difference.

Both Alternatives B and C would remove biomass as a result of timber harvest. This would reduce the amount of carbon stored in the treated stands. A portion of the carbon removed would remain stored for a period of time in wood products. Regeneration harvests would reduce existing carbon stocks at the harvest sites. The harvest of live trees, combined with the increase in down dead wood, would temporarily convert stands from a carbon sink that removes more carbon from the atmosphere than it emits, to a carbon source that emits more carbon through respiration than it absorbs. These stands would remain a source of carbon to the atmosphere until carbon uptake by new trees and other vegetation exceeds the emissions from decomposing dead organic material. The stands would likely remain a carbon source for several years depending on the amount of dead biomass left onsite and new trees' growth rates once reestablished. As the stands continue to develop, the carbon source would change to a carbon sink. The strength of the carbon sink would increase until peaking at approximately 85 years of age (Vose 2009) and then would gradually decline but remain positive.

Recent scientific literature confirms this general pattern of changes in net ecosystem productivity (NEP) and carbon stocks over the period of forest stand development. (The Net ecosystem productivity, or NEP, is defined as gross primary productivity (GPP) minus ecosystem respiration (ER) (Chapin et al. 2006). It reflects the balance between (1) absorbing CO_2 from the atmosphere through photosynthesis (GPP) and (2) the release of carbon into the atmosphere through respiration by live plants, decomposition of dead organic matter, and burning of biomass (ER). When NEP is positive, carbon accumulates in biomass. Ecosystems with a positive NEP are referred to as a carbon sink. When NEP is negative, ecosystems emit more carbon than they absorb. Ecosystems with a negative NEP are referred to as a carbon source.) Most mature and old stands remain a net sink of carbon. Pregitzer and Euskirchen (2004) synthesized results from 120 separate studies of carbon stocks and carbon fluxes for boreal, temperate, and tropical biomes. They found that in temperate forests NEP is lowest, and most variable, in young stands (0-30 years), highest in stands 31-70 years, and declines thereafter as stands age. These studies also reveal a general pattern of total carbon stocks declining after disturbance and then increasing, rapidly during intermediate years and then at a declining rate, over time until another major disturbance (timber harvest or tree mortality resulting from drought, fire, insects, disease or other causes) kills large numbers of trees and again converts the stands to a carbon source where carbon emissions from decay of dead biomass exceed that amount of carbon removed from the atmosphere by photosynthesis within the stand.

The impacts of the action alternatives on global carbon sequestration and atmospheric concentrations of CO_2 are miniscule. However, the forests of the United States significantly reduce atmospheric concentrations of CO_2 resulting from fossil fuel emissions. The forest and wood products of the United States currently sequester approximately 200 teragrams (200 teragrams, or Tg, equals 196,841,306 U.S. tons) of carbon per year (Heath and Smith, 2004). This rate of carbon sequestration offsets approximately 10% of CO_2 emissions from burning fossil fuels (Birdsey et al., 2006). U.S. Forests currently contain 66,600 teragrams of carbon. The short-term reduction in carbon stocks and sequestration rates resulting from the proposed project are imperceptibly small on global and national scales, as are the potential long-term benefits in terms of carbon storage. The currently large carbon sink in U.S. forests is a result of past land use changes, including the re-growth of forests on large areas of the eastern U.S. harvest in the 19-20th century, and 20th century fire suppression in the western U.S. (Birdsey et al. 2006). The continuation of this large carbon sink is uncertain because some of the processes promoting the current sink are likely to decline and projected increases in disturbance rates such as fire and large-scale insect mortality may release a significant fraction of existing carbon stocks (Pacala et al. 2008; Canadell et al. 2007). Management actions - - such as those proposed - - that improve the resilience of forests to climate-induced increases in frequency, and utilize harvested trees for long-lived forest products and renewable energy sources may help sustain the current strength of the carbon sink in U.S. forests (Birdsey et al. 2007).

Cumulative Effects to Climate Change: For the action alternatives, the contribution of the proposed project activities to the carbon cycle is extremely small. Conducting regeneration harvesting on approximately 317 acres (Alternative B) or 259 acres (Alternative C) would result in new early successional habitat on approximately 2% of the total 18,944 acre AA.

The long-term ability of forests to sequester carbon depends in part on their resilience to multiple stresses, including increasing probability of drought stress, high-severity fires, and large-scale insect outbreaks associated with projected climate change. Thus, even though some management actions may in the near-term reduce total carbon stored below current levels, in the long term they may improve the overall capacity of the forest to sequester carbon. Sustainable forestry practices can increase the ability of forests to sequester atmospheric carbon while enhancing other ecosystem services. Planting new trees and improving forest health through thinning and prescribed burning, for example, are some of the ways to increase forest carbon in the long run. Harvesting and regenerating forests can also result in net carbon sequestration in wood products and new forest growth.

When combined, the carbon from this and past projects in the analysis area has a minimal cumulative effect not only at the local level, but at the larger level. When implemented, the rate of carbon release through timber regeneration would be minimal for the reasonably foreseeable future.

Attachment 33

Excerpts from Mossy Oak Project EA

United States
Department of
Agriculture

Forest
Service

January 2018

Environmental Assessment

Mossy Oak Project

Nantahala Ranger District, Nantahala National Forest
Jackson and Macon Counties, North Carolina

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project alternative. Because no changes are proposed to the Forest Service road system, a separate Transportation Analysis Plan (TAP) is not required for this project.

3.15 Climate Change

Climate change is discussed here in two ways: (1) how climate change affects analysis area resources, and (2) how the project area alternatives affect climate change. Note: Some of the material in this section is adapted from (quoted) and referenced by the Cherokee National Forest Big Creek EA, July 2009.

Existing Condition and How Climate Change Affects Analysis Area Resources: The existing condition is an analysis area that is typical of the southern Appalachians, with a range of elevations from approximately 2,100 feet to 4,800 feet. Climate change models are continuing to be developed and refined, but the two principal models found to best simulate future climate-changed conditions for the various regions across the country are the Hadley Centre model and the Canadian Climate Centre model (Climate Change Impacts on the United States 2001). Both models indicate warming in the southern region of the United States. However, the models differ considerably. One predicts little change in precipitation until 2030, followed by much drier conditions over the next 70 years. The other predicts a slight decrease in precipitation during the next 30 years, followed by increased precipitation.

Either of these climate scenarios with their attendant changes could affect forest productivity, forest pest activity, vegetation types, major weather disturbances (droughts, hurricanes), and streamflow. These effects would likely be seen across the entire national forest system in the United States. In the Southern Appalachians, it is possible that in the long run, a warmer climate will result in certain species’ (cold-adapted ones such as northern hardwoods) ranges moving northward. In turn, species that currently have a more southerly range might start appearing here. In general, concerning both vegetation and wildlife, species that are generalists and can tolerate a wider range of habitat conditions will probably fare better than those with a set of narrow habitat requirements and conditions.

Mossy Oak Project Effects on Climate Change

Scope of Analysis

The scope of this analysis for direct, indirect, and cumulative effects on climate change includes the 9,543 acres of national forest lands in the compartments that comprise the analysis area. The time frame used in this analysis is up to ten years after completion of the project activities.

Direct and Indirect Effects

Alternative A - Alternative A (No Action) would result in no change to the current trend for carbon storage or release. Forested stands are expected to be less resilient to possible climate change impacts, such as changes in productivity or insect and disease.

Alternatives B and C - It is not expected that Alternatives B and C would substantially alter the effects of climate change in the project area. The regeneration in the areas to be harvested would provide more structural diversity to the area and establish young, vigorous stands that may be more resilient to the changes in climate than those ages 70 and older. In addition, it is anticipated that the existing forest types in the stands to be regenerated would regenerate naturally to these same forest types; climate change will not make any difference.

Both Alternatives B and C would remove biomass as a result of timber harvest. This would reduce the amount of carbon stored in the treated stands. A portion of the carbon removed would remain stored for a period of time in wood products. Regeneration harvests would reduce existing carbon stocks at the harvest sites. The harvest of live trees, combined with the increase in down dead wood, would temporarily convert stands from a carbon sink that removes more carbon from the atmosphere than it emits, to a carbon source that emits more carbon through respiration than it absorbs. These stands would remain a source of carbon to the atmosphere until carbon uptake by new trees and other vegetation exceeds the emissions from decomposing dead organic material. The stands would likely remain a carbon source for several years depending on the amount of dead biomass left onsite and new trees growth rates once reestablished. As the stands continue to develop, the carbon source would change to a carbon sink. The strength of the carbon sink would increase until peaking at approximately 85 years of age (Vose 2009) and then would gradually decline but remain positive.

Recent scientific literature confirms this general pattern of changes in net ecosystem productivity (NEP) and carbon stocks over the period of forest stand development. (The Net ecosystem productivity, or NEP, is defined as gross primary productivity (GPP) minus ecosystem respiration (ER) (Chapin et al. 2006). It reflects the balance between (1) absorbing CO_2 from the atmosphere through photosynthesis (GPP) and (2) the release of carbon into the atmosphere through respiration by live plants, decomposition of dead organic matter, and burning of biomass (ER). When NEP is positive, carbon accumulates in biomass. Ecosystems with a positive NEP are referred to as a carbon sink. When NEP is negative, ecosystems emit more carbon than they absorb. Ecosystems with a negative NEP are referred to as a carbon source.) Most mature and old stands remain a net sink of carbon. Pregitzer and Euskirchen (2004) synthesized results from 120 separate studies of carbon stocks and carbon fluxes for boreal, temperate, and tropical biomes. They found that in temperate forests NEP is lowest, and most variable, in young stands (0-30 years), highest in stands 31-70 years, and declines thereafter as stands age. These studies also reveal a general pattern of total carbon stocks declining after disturbance and then increasing, rapidly during intermediate years and then at a declining rate, over time until another major disturbance (timber harvest or tree mortality resulting from drought, fire, insects, disease or other causes) kills large numbers of trees and again converts the stands to a carbon source where carbon emissions from decay of dead biomass exceed that amount of carbon removed from the atmosphere by photosynthesis within the stand.

The impacts of the action alternatives on global carbon sequestration and atmospheric concentrations of CO_2 are miniscule. However, the forests of the United States significantly reduce atmospheric concentrations of CO_2 resulting from fossil fuel emissions. The forest and wood products of the United States currently sequester approximately 200 teragrams (200 teragrams, or Tg, equals 196,841,306 U.S. tons) of carbon per year (Heath and Smith, 2004). This rate of carbon sequestration offsets approximately 10% of CO_2 emissions from burning fossil fuels (Birdsey et al., 2006). U.S. Forests currently contain 66,600 teragrams of carbon. The short-term reduction in carbon stocks and sequestration rates resulting from the proposed project are imperceptibly small on global and national scales, as are the potential long-term benefits in terms of carbon storage. The currently large carbon sink in U.S. forests is a result of past land use changes, including the re-growth of forests on large areas of the eastern U.S. harvest in the 19-20th century, and 20th century fire suppression in the western U.S. (Birdsey et al. 2006). The continuation of this large carbon sink is uncertain because some of the processes promoting the current sink are likely to decline and projected increases in disturbance rates such as fire and large-scale insect mortality may release a significant fraction of existing carbon stocks (Pacala et al. 2008; Canadell et al. 2007). Management actions - - such as those proposed - - that improve the resilience of forests to climate-induced increases in frequency, and utilize harvested trees for long-lived forest products and renewable energy sources may help sustain the current strength of the carbon sink in U.S. forests (Birdsey et al. 2007).

Cumulative Effects to Climate Change

For the action alternatives, the contribution of the proposed project activities to the carbon cycle is extremely small. Conducting regeneration harvesting on approximately 220 acres (Alternative B) or 299 acres (Alternative C) would result in new early successional habitat on approximately 2.5% or 3.1% of the total 9,543 acre AA.

The long-term ability of forests to sequester carbon depends in part on their resilience to multiple stresses, including increasing probability of drought stress, high-severity fires, and large-scale insect outbreaks associated with projected climate change. Thus, even though some management actions may in the near-term reduce total carbon stored below current levels, in the long term they may improve the overall capacity of the forest to sequester carbon. Sustainable forestry practices can increase the ability of forests to sequester atmospheric carbon while enhancing other ecosystem services. Planting new trees and improving forest health through thinning and prescribed burning, for example, are some of the ways to increase forest carbon in the long run. Harvesting and regenerating forests can also result in net carbon sequestration in wood products and new forest growth.

When combined, the carbon from this and past projects in the analysis area has a minimal cumulative effect not only at the local level, but at the larger level. When implemented, the rate of carbon release through timber regeneration would be minimal for the reasonably foreseeable future.

Attachment 34

Excerpts from Andrew Pickens White Pine Management EA

United States Department of Agriculture

AP White Pine Management Project

Environmental Assessment

Forest Service
Sumter National Forest

June 2021

Eligible Wild and Scenic Rivers

The project includes portions of six stands that overlap with the ¼-mile corridor of eligible rivers (five stands in the Chauga River corridor, and one stand in the Cedar Creek corridor). Four of these six stands are proposed for uneven-aged treatments, which are selection cutting treatments that would remove only a portion of trees from the site. The overlap is approximately 4% of the Chauga and 1% of the Cedar Creek corridor area. Cumulatively, when factoring in the loblolly removal project, the overlap for timber harvest treatments is approximately 10% of the Chauga and 1% of the Cedar Creek corridor area. The treatments would change how the forest looks in these overlapping areas in terms of the type and age of forest vegetation, but the sites would remain forested. These vegetation changes would not affect the eligible wild and scenic rivers' outstandingly remarkable values or classification, nor would they affect the rivers' wild and scenic eligibility status.

Climate Change

This proposed project would affect a relatively small amount of forest land and carbon on the Sumter National Forest and might temporarily contribute an extremely small quantity of greenhouse gas (GHG) emissions relative to national and global emissions. This proposed action would not convert forest land to other non-forest uses, thus allowing any carbon initially emitted from the proposed action to have a temporary influence on atmospheric GHG concentrations, because carbon would be removed from the atmosphere over time as the forest regrows. Furthermore, the proposed project would transfer carbon in the harvested wood to the product sector, where it may be stored for up to several decades and substitute for more emission intensive materials or fuels. This proposed action is consistent with internationally recognized climate change adaptation and mitigation practices. A more detailed analysis of carbon stocks on the Sumter and factors that influence carbon trends is available in the project record (McKinley et al. 2019).

Herbicides

Public comments were received expressing concerns about the use of herbicides. This summary from the risk assessment is included in the EA to help clarify the effects from herbicides. The herbicides proposed for use are registered with the EPA and the State of South Carolina. None of the proposed herbicides are restricted use pesticides (RUPs) (EPA 2021, South Carolina Pesticide Regulation 2021). The methods proposed for applying herbicide are very focused and would limit the chance for chemicals coming into contact with anything other than the target vegetation. Backpack foliar spray is applied manually, directed at individual small trees. Cut surface treatments (hack and squirt) are also applied manually to larger trees with a hatchet and spray bottle by making cuts in the bole of the tree with the hatchet and spraying a small amount of herbicide into the cut.

An herbicide risk assessment documenting the risk to human health and the environment from the proposed action is in the project record. Design criteria related to herbicide use are listed in Appendix C and Forest Plan standards are listed in the assessment. These design criteria and standards would mitigate potential effects, especially those related to accidental exposure. The assessment results show that the exposure of humans, plants, and animals to herbicides would be at least 10 times less than the maximum exposure for no observable adverse effects (NOAEL), and depending on the scenario analyzed, is typically over 100 times less than the NOAEL.

Cumulatively, herbicide exposure would not increase from other projects, because the herbicide treatments do not occur on the same sites or at the same time. The chemicals are broken down via plant or

Attachment 35

Excerpts from Eastern Divide Phase II Project

Project scale Carbon Effects – Phase II Project
George Washington and Jefferson National Forests, Southern Region

1.1 Carbon and Greenhouse gas emissions

Forests play an important role in the global carbon cycle by taking up and storing carbon in plants and soil. Forestry has gained attention in recent decades because of its potential to influence the exchange of carbon with the atmosphere, either by increasing storage or releasing carbon emissions. Forests have a carbon "boom and bust" cycle. They take up and store atmospheric carbon as they grow through photosynthesis and release carbon through mortality due to aging or disturbances. Following mortality events, forests regrow and the cycle continues. Forests can store carbon in soils and plant material as well as in harvested wood products outside of the forest ecosystem. In addition, wood fiber can be used to substitute for products that are more energy-intensive to produce, such as concrete and steel, creating a substitution effect which can result in lower overall greenhouse gas emissions.

A complete and quantitative assessment of forest carbon stocks and the factors that influence carbon trends (management activities, disturbances, and environmental factors) for the George Washington and Jefferson National Forests (NF) is available in the project record (Dugan *et al.,* 2019). This carbon assessment contains additional supporting information and references.

1.1.1 Direct and Indirect Effects Analysis Boundaries

The effects analysis area for carbon includes forested lands within the George Washington and Jefferson NF because this is where timber harvest activities are proposed and where carbon stocks may be affected. The effects analysis for greenhouse gas emissions is the global atmosphere given the mix of atmospheric gases can have no bounds.

1.1.2 Affected Environment

The carbon legacy of the George Washington and Jefferson NF's is tied to the history of Euro-American settlement, land management, and disturbances. As the first region to be widely settled in the United States, eastern forests have had a long history of intensive harvesting and conversion of forest to agriculture. Historical disturbance dynamics, forest regrowth and recovery, and forest aging have been most responsible in driving carbon accumulation trends since 1950. The George Washington and Jefferson NF's are maintaining a carbon sink and forest carbon stocks have increased by about 20 percent between 1990 and 2013 (USDA Forest Service, 2015). The negative impacts on carbon stocks caused by disturbances and climate conditions have been modest and exceeded by forest growth. Over half of the stands in the George Washington and Jefferson NF are middle-aged and older (greater than 80 years) and there has been a sharp decline in new stand establishment in recent decades (Birdsey *et al.,* in press). If the Forest continues on this aging trajectory, more stands will reach a slower growth stage in coming years, potentially causing the rate carbon accumulation to decline.

According to satellite imagery, timber harvest has been the dominant disturbance type on the George Washington and Jefferson NF from 1990 to 2011, although harvesting has typically affected less than 0.1 percent of the forested area annually. During this period, about two percent of the forested area experienced some level of harvest treatments. Carbon losses from the forest ecosystem associated with harvests have been relatively small compared to the total amount of carbon stored in the forest, with losses from 1990 to 2011 equivalent to about 1 percent of non-soil carbon stocks (Birdsey *et al.,* in press).

However, these estimates represent an upper bound, because they do not account for continued storage of harvested carbon in wood products or the effect of substitution.

According to satellite imagery, fire has been the second largest disturbance type on the George Washington and Jefferson NF from 1990 to 2011, affecting approximately 0.02 percent of the forested area annually (Birdsey *et al.,* in press). During this period, about 0.6 percent of the forested area experienced some level of fires including prescribed fires and wildfires. However, some prescribed fires that burned only along the forest floor may have gone undetected because they did not cause a change in canopy cover. Carbon losses from the forest ecosystem associated with fires have been relatively small compared to the total amount of carbon stored in the forest, with losses from 1990 to 2011 equivalent to about 0.3 percent of non-soil carbon stocks.

1.1.3 Direct, Indirect and Cumulative Effects

The proposed Phase II project includes timber harvesting that would be conducted on approximately 1,172 acres of the George Washington and Jefferson NF. This scope and degree of change would be minor, affecting less than 1 percent of the 1.1 million acre*s* of forested land in the George Washington and Jefferson.

In addition, the effect of the proposed action focuses on the aboveground carbon pool that is stored in live woody vegetation, which comprise about 44 percent of the total ecosystem carbon stocks of the Jefferson (USDA Forest Service 2015). The effect of the proposed mechanical harvest focuses on the understory and forest floor pools, which together comprise about eight percent of the Forest-wide ecosystem carbon stocks. About 33 percent or more of the ecosystem carbon is in mineral soils, a very stable and long-lived carbon pool (McKinley *et al.,* 2011; USDA Forest Service 2015; Domke *et al.* 2017). Timber harvesting generally results in a negligible amount of carbon loss from the mineral soils typically found in the United States, particularly when operations are designed in a way that minimizes soil disturbance (Nave *et al.,* 2010; McKinley *et al.,* 2011). Although timber harvest can also affect the carbon stored in the understory and forest floor organic layer consisting of debris in various stages of decomposition, the carbon loss would be negligible given it is not stable or long-lived and would be replaced with months to a few years.

In the absence of timber harvests, the forests where this proposed action would take place will thin naturally from mortality-inducing natural disturbances and other processes resulting in dead trees that will decay over time, emitting carbon to the atmosphere. The wood and fiber removed from the forest in this proposed project will be transferred to the wood products sector for a variety of uses, each of which has different effects on carbon (Skog *et al.,* 2014). Carbon can be stored in wood products for a variable length of time, depending on the commodity produced. Wood can be used in place of other materials that emit more GHGs, such as concrete, steel, and plastic (Gustavasson *et al.,* 2006; Lippke *et al.,* 2011; McKinley *et al.,* 2011). Likewise, biomass can also be burned to produce heat or electrical energy, or converted to liquid transportation fuels that would otherwise come from fossil fuels. In fact, removing carbon from forests for human use can result in a lower net contribution of GHGs to the atmosphere than if the forest were not managed (McKinley *et al.*, 2011; Bergman *et al.,* 2014; Skog *et al.,* 2014). The IPCC recognizes wood and fiber as a renewable resource that can provide lasting climate-related mitigation benefits that can increase over time with active management (IPCC 2000). Furthermore, by reducing stand density, the proposed action may also reduce the risk of more severe disturbances, such as insect and disease outbreak and severe wildfires, which may result in lower forest carbon stocks and greater GHG emissions.

Some tree species and forest communities within eastern U.S. are well adapted to fire and in some cases may depend on it for survival and regeneration. Historical fire suppression has allowed some fire-dependent forests in the eastern U.S. to become unnaturally dense and altered species composition and structure (Olson 1996; Nowacki and Abrams 2008). By reducing vegetative competition through mechanical harvest in the understory, the proposed harvest would help establish oak habitat and increase the ability of harvested areas to regenerate more quickly. This would help to support forest health in a changing climate and reducing GHG emissions over the long-term. Any initial carbon emissions from this proposed action will be balanced and possibly eliminated as the stand recovers and regenerates, because the remaining trees and newly established trees typically have higher rates of growth and carbon storage (Hurteau and North 2009; Dwyer *et al.,* 2010; McKinley *et al.,* 2011).

Climate change is a global phenomenon, because major GHGs[1] mix well throughout the planet's lower atmosphere (IPCC 2013). Considering emissions of GHGs in 2010 were estimated at 13,336 ± 1,227 teragrams[2] carbon globally (IPCC 2014) and 1,881 teragrams carbon nationally (US EPA, 2015), the Phase II project makes an extremely small direct contribution to overall emissions. Because local GHG emissions mix readily into the global pool of GHGs, it is difficult and highly uncertain to ascertain the indirect effects of emissions from single or multiple projects of this size on global climate. Any initial carbon emissions during the implementation of the proposed project would have a temporary influence on atmospheric carbon concentrations, because carbon will be removed from the atmosphere as forests regrow, minimizing or mitigating any potential cumulative effects.

From 2000 to 2009, forestry and other land uses contributed 12 percent of the human-caused global CO_2 emissions[3] (IPCC 2014). The forestry sector's contribution to GHG emissions has declined over the last decade (IPCC 2014; Smith *et al.* 2014; FAOSTAT 2013). The largest source of GHG emissions in the forestry sector globally is deforestation (e.g., conversion of forest land to agricultural or developed landscapes) (Pan *et al.,* 2011; Houghton *et al.,* 2012; IPCC 2014). However, forest land in the United States has had a net increase since the year 2000, and this trend is expected to continue for at least another decade (Wear *et al.,* 2013; USDA Forest Service 2016). The proposed activities in the Phase II project will not result in the loss of forest land from the George Washington and Jefferson NF. In fact, forest stands are being retained and harvested to maintain a vigorous condition that supports enhanced tree growth and productivity, reduces the risk of insect and disease, and supports sustainable ecosystems thus contributing to long-term carbon uptake and storage.

Some assessments suggest that the effects of climate change in some United States forests may cause shifts in forest composition and productivity or prevent forests from fully recovering after severe disturbance (Anderson-Teixeira *et al.,* 2013), thus impeding their ability to take up and store carbon[4] and retain other ecosystem functions and services. Climate change is likely already increasing the frequency and extent of droughts, fires, and insect outbreaks, which can influence forest carbon cycling (Kurz *et al.,* 2009; Allen *et al.,* 2010; Joyce *et al.,* 2014). In fact, reducing stand density, one of the outcomes of this proposed action, is consistent with adaptation practices to increase resilience of forests to climate-related

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[1] Major greenhouse gases released as a result of human activity include carbon dioxide (CO_2), methane, nitrous oxide, hydrofluorocarbons, and perfluorocarbons.

[2] This report uses carbon mass, not carbon dioxide (CO2) mass, because carbon is a standard unit and can easily be converted to any other unit. To convert carbon mass to CO2 mass, multiply by 3.67 to account for the mass of the oxygen (O2).

[3] Fluxes from forestry and other land use (FOLU) activities are dominated by CO_2 emissions. Non-CO_2 greenhouse gas emissions from FOLU are small and mostly due to peat degradation releasing methane and were not included in this estimate.

[4] The term "carbon" is used in this context to refer to carbon dioxide.

environmental changes (Joyce *et al.,* 2014). This proposed action is consistent with options proposed by the IPCC for minimizing the impacts of climate change on forests, thus meeting objectives for both adapting to climate change and mitigating GHG emissions (McKinley *et al.,* 2011). The relatively small quantity of carbon released to the atmosphere and the short-term nature of the effect of the proposed action on the forest ecosystem are justified, given the overall change in condition increases the resistance to wildfire, drought, insects and disease, or a combination of disturbance types that can reduce carbon storage and alter ecosystem functions (Millar *et al.,* 2007; D'Amato *et al.,* 2011).

In summary, this proposed project affects a relatively small amount of forest land and carbon on the George Washington and Jefferson and might temporarily contribute an extremely small quantity of GHG emissions relative to national and global emissions. This proposed action will not convert forest land to other non-forest uses, thus allowing any carbon initially emitted from the proposed action to have a temporary influence on atmospheric GHG concentrations, because carbon will be removed from the atmosphere over time as the forest regrows. Furthermore, the proposed project will transfer carbon in the harvested wood to the product sector, where it may be stored for up to several decades and substitute for more emission intensive materials or fuels. This proposed action is consistent with internationally recognized climate change adaptation and mitigation practices.

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Attachment 36

Excerpts from Fightingtown Creek Project EA

United States
Department of
Agriculture

Forest
Service

September 2017

Environmental Assessment

FIGHTINGTOWN CREEK EARLY-SUCCESSIONAL HABITAT PROJECT

Conasauga Ranger District, Chattahoochee-Oconee National Forests
Fannin and Gilmer Counties, Georgia

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Effects from Alternative 2. Either Alternative would maintain good water quality and healthy aquatic habitats.

Cumulative Effects: Riparian corridors within the Fightingtown Creek project area are almost completely forested with mid-late successional forest. Past, present, and reasonably foreseeable future actions in the project area are displayed in Table 10. Activities with the potential to affect or have affected aquatic habitats in the past 10 years are limited to ongoing system road maintenance, consisting of road grading, the periodic addition of gravel, and cleaning out ditches and culverts, which can result in localized sedimentation when near stream crossings. As a result of site-specific analysis, planning, and implementation of these other actions, aquatic habitats and associated biotic communities are intact and healthy, with no impairments to important functions or designated uses.

Effects to Climate

Current Conditions: Atmospheric levels of greenhouse gases (GHG), including carbon dioxide (CO_2) have increased over the last century due to increased burning of fossil fuels and land-use conversions (Ryan et al. 2010). Elevated levels of CO_2 in the atmosphere have increased global surface temperatures and are expected to alter climatic patterns in the future. In the southeast, climate change models indicate significant increases in air temperatures from historic and current levels. Precipitation patterns are predicted to be relatively stable, averaging slightly less to slightly above current conditions (TACCIMO 2012). Although the magnitude and temporal and spatial distribution of climate change are uncertain, all indications suggest that some change is certain.

Predicted changes in regional climate could affect forest productivity (both positively and negatively) and intensify disturbance events, including weather disturbances (droughts, storm intensities), insect and disease outbreaks, and wildfires. Forest management actions that condition forest communities for climate change by improving their resilience and resistance to climate-driven disturbances and that emphasize structural and age-class diversity have been recommended as strategies for adapting to predicted climate change patterns (Joyce et al. 2009).

Effects of Alternative 1 (No Action)

In general, no changes to current trends in carbon storage and release in the analysis area would occur. Current forest conditions would be unchanged and less resilient to climate change impacts, including more severe disturbances (drought, insect and disease outbreaks, and wildfires).

Cumulative Effects □Because no activities are proposed under this alternative, there would be no effects that could be combined with past, present, or reasonably foreseeable future actions that could cause adverse cumulative effects to climate change or its impacts on vegetation in the analysis area.

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<u>Effects of Alternatives 2 and 3:</u> The effects of treatments proposed under the action alternatives on altering the impacts of climate change on the forest communities in the analysis area are uncertain; however, proposed management actions are compatible with adaptation strategies recommended for responding to potential impacts associated with climate change.

The forest community in the Fightingtown Creek project area is dominated by late-successional forest. Even-aged regeneration treatments would create young, vigorous stands more resilient to disturbances associated with climate change and improve age-class and structural diversity within the forest community.

Forests play a major role in the carbon cycle. The carbon stored in live biomass, dead plant material, and forest soils can offset concentrations of CO_2 emitted into the atmosphere. Additionally, forest and wood products contribute to carbon storage. In the U.S., forests and forest products have sequestered the equivalent of 10 to 19 percent of the nation's CO_2 emissions from burning fossil fuels during the last decade (Birdsey et al. 2006, Ryan et al. 2010, U.S. EPA 2012).

The proposed action and other action alternatives include timber harvesting to meet multiple resource objectives. These actions would temporarily reduce carbon storage in the analysis area. However, forest land-use and forestry practices continue to be a net carbon "sink," with carbon storage gains exceeding carbon losses (U.S. EPA 2012).

In stands proposed for regeneration harvest, most trees would be removed, temporarily decreasing carbon storage on these sites. Increased dead plant material (slash) resulting from the harvest would release carbon as this material decomposes, temporarily converting these areas to a "source" of carbon emission. Carbon storage losses would be somewhat offset by the amount of carbon stored in wood products removed from the site. The regenerated sites would recover carbon lost from removals and decomposition if the recovery period is long enough (Ryan et al. 2010). Time periods for recovery would depend on the rate at which vegetation re-establishes, growth rates of the vegetation, and frequency/severity of future disturbances. Predicted increases in disturbances related to climate change could interrupt recovery periods. Maintaining healthy forests by improving age-class structure could minimize impacts of climate change-driven disturbances predicted in the future.

The impacts of Alternatives 2 and 3 on global carbon sequestration and atmospheric concentrations of CO_2 are miniscule. Forest and forest products currently serve as a major carbon sink, offsetting 10 percent or more of the nation's CO_2 emissions. Predicted changes in climate patterns and associated increases in frequency and intensity of disturbances have the potential to reduce the carbon sequestration capacity of our forests. Forests that are more resilient to climate change impacts could help sustain carbon storage potential. Proposed activities included in the action alternatives would make the forest more resilient and resistant to predicted climate change impacts.

Cumulative Effects – Action alternatives include timber harvesting to improve structural and age-class diversity. These actions would also reduce existing carbon stocks in the analysis area, but could improve resilience and resistance characteristics in response to predicted climate change patterns/disturbances. These effects represent the trade-offs associated with mitigation strategies designed to increase carbon storage and adaptation strategies designed to condition forests for changing environmental conditions (D'Amato 2011).

There are no past, present, or reasonably foreseeable activities potentially affecting climate which may be combined with those of the action alternatives.

Wildlife Habitat

Effects to Demand Species - Ruffed Grouse

Current conditions: Ruffed grouse reach the southernmost extent of their breeding range in the mountains of north Georgia (Schneider et al. 2010). They are uncommonly found in the Blue Ridge Mountains of Georgia, typically above elevations of 1800 feet. Although grouse populations in the southern Appalachians have historically occurred at lower densities than those in aspen-dominated landscapes, grouse numbers in the southern Appalachians have been declining for several decades as a result of a decrease in young forest cover (Dessecker and McCauley 2001). Hunting pressure in Georgia is low due to limited grouse distribution, low grouse numbers, and the type of rough cover and terrain that grouse inhabit (Georgia DNR 2016). Hunting mortality is not considered a major factor influencing population decline in Georgia.

Across their range, ruffed grouse prefer ESH, specifically early-successional deciduous forest habitats with high stem densities and dense herbaceous cover (Devers et al 2007). This type of habitat is valuable for nesting, drumming, and escape cover, which protects both chicks and adult grouse from their primary predators: raptors. It also provides protection from the elements and an abundant food source (buds, fruit, leaves, seeds). Survival is higher for grouse with home ranges containing more ESH (Clark 2000). This type of habitat is extremely rare on the Chattahoochee National Forest due mainly to the lack of widespread even-aged timber harvest over the last few decades. The gradual aging of the Forest has led to serious declines in ruffed grouse populations. This has been documented in Georgia and other southern Appalachian states by data collected during breeding bird surveys (Schneider et al. 2010), drumming surveys, and grouse hunter harvest surveys (Georgia DNR unpublished information).

The Fightingtown Creek project area's current mix of successional stages is detailed in the section of this document titled Successional Stage Habitats. Timber harvest prior to National Forest ownership, then a series of timber cuts since the 1970s provided a well-distributed mosaic of ESH which formerly supported a healthy ruffed grouse population. Most timber cuts are optimal for grouse from 6 to 20 years after regeneration (Jones and Harper 2004). The youngest patches of forest in the project area have gradually become less suitable as grouse habitat as stem densities have decreased and canopies have closed. Drumming surveys conducted on FSR 792 (Williamson Cove) during 2012 detected one drumming male, surveys in 2013 and 2014 failed to detect any drumming males, and no surveys were

Attachment 37

Additional Literature Cited

Age Structure and Disturbance History of a Southern Appalachian Virgin Forest
Author(s): Craig G. Lorimer
Source: *Ecology*, Vol. 61, No. 5 (Oct., 1980), pp. 1169-1184
Published by: Ecological Society of America
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Accessed: 15/04/2009 13:58

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Ecology, 61(5), 1980, pp. 1169–1184

AGE STRUCTURE AND DISTURBANCE HISTORY OF A SOUTHERN APPALACHIAN VIRGIN FOREST[1]

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Abstract. The frequency and intensity of natural disturbances in a montane old-growth forest of coniferous (*Tsuga*) and broad-leaved deciduous trees (*Liriodendron, Betula, Castanea, Fagus, Acer*) were investigated by means of diameter frequency distributions, diameter-age regressions, and analysis of radial growth patterns. The seven species investigated are represented by all-aged populations regardless of the degree of shade tolerance. The age distributions of most species, however, are irregular with several prominent peaks. Apparent coincidence of these peaks among some species suggests a common underlying influence. Radial growth patterns reveal higher than average numbers of abrupt and sustained increases in growth rate in most decades of peak recruitment which, along with other evidence, suggest that disturbance was the principal causal factor. There is evidence for eight partial disturbances in the past 250 yr, each of which generally removed <10% of the overstory trees. These fairly light disturbances appear to be sufficient to cause major deviations from the theoretical equilibrium diameter distribution of the forest.

Key words: age distribution; climax; disturbance; forest ecology; forest structure; southern Appalachian Mountains; windthrow.

INTRODUCTION

The frequency and magnitude of natural disturbances in temperate forests are important considerations in shaping management policies of both wilderness areas and commercial forest land. In the realm of forest management, uncertainty persists as to whether forests unaffected by man would be predominantly even aged or uneven aged. In some temperate regions the abundance of even-aged virgin stands and direct evidence on the frequency of large-scale natural disturbances have suggested that most forest stands would be in constant successional flux under natural conditions (Maissurow 1935, Graham 1941, Cline and Spurr 1942, Bloomberg 1950, Heinselman 1973, Spurr and Barnes 1973). Under such conditions all-aged forests would seldom develop even in the absence of man (Raup 1964, Sprugel 1976). In regions or physiographic sites with this type of disturbance regime, clearcutting of timber would more closely mimic the predominant process of tree mortality and replacement than would more conservative selection cutting.

The type of disturbance regime has significant bearing on wilderness area management as well since relatively few management options exist if an area is to retain its "primeval character and influence" and is to be "protected and managed so as to preserve its natural conditions" (Wilderness Act of 1964, United States Public Law 88-577). The mounting evidence that many conifer forests in western North America are adapted to frequent lightning fires has already led to a change in the traditional policy of total fire suppression is some wilderness areas (Kilgore and Briggs 1972, Chapman 1977). Difficult management problems arise when it is discovered, as in northern Minnesota, that the natural fire regime is probably one of frequent and extensive crown fires (Heinselman 1973). Thus it is clear that the natural disturbance regime of each region should be known in as much detail as possible.

The investigation of stand history in western conifer forests has concentrated primarily on determining fire frequency by cross-dating fire scars on old trees and by stand age determination (Fritz 1929, Wagener 1961, Arno 1976). Comparable evidence on the stand histories of eastern deciduous forests has been sparse because virgin stands in the region are rare and stand historical data are not easily obtained by nondestructive means. Most of the pertinent evidence in second-growth stands was removed by early logging or land clearing.

If climax forest types are defined as those capable of self-perpetuation in the absence of severe disturbance (such as beech-maple-hemlock), then it appears that climax types occupied a large proportion of eastern North America in presettlement times (e.g., Reed 1905, Lutz 1930, McIntosh 1962, Siccama 1971, Finley 1976). Such evidence, however, does not necessarily indicate that natural disturbances were rare or that these forests were predominantly uneven aged. Many types of disturbances such as windthrow, insect attack, drought, clearcutting, and repeated selection cutting cause mortality primarily among the overstory trees and leave many of the understory trees intact. In seral forest types, disturbance often hastens succession to the climax type by releasing the understory from suppression by the overstory (Lutz and Cline 1947, Brender and Nelson 1954, Spurr 1956, Trimble 1965, 1970, Henry and Swan 1974). Once the climax

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type is dominant, further disturbance will often perpetuate the existing climax type by releasing understory trees of the same species (Hough 1936, Leak and Wilson 1958, Ohmann and Ream 1971, Tubbs 1977). All-aged climax stands, however, can probably develop only if the recurrence interval between any type of major disturbance is longer than the life-span of the dominant tree species. Thus age structure is a more sensitive indicator of disturbance history than is species composition.

The recent study by Henry and Swan (1974) on the Pisgah Forest of New Hampshire appears to have been the first study specifically on stand history in a virgin forest of eastern North America. Detailed analysis of a 400-m^2 plot suggested that this particular stand, dominated by eastern white pine (*Pinus strobus*), experienced two severe disturbances over a 300-yr period, both of which killed nearly all the overstory trees in the plot. It is probable that many or most of such pine stands had a catastrophic origin, since those checked for age structure were usually even aged (Cary 1894, Spalding and Fernow 1899, Hough and Forbes 1943). An estimate of regional disturbance frequency from stand data, however, will require stand historical analysis for a variety of forest types and physiographic sites, with greater weight given to results from the predominant forest types. Although detailed stand historical analyses are lacking for the predominant cover types of the eastern deciduous forest region, the majority of remnant virgin stands in mesophytic types has been found, from examinations of cut stumps or increment cores, to be broadly uneven aged or all aged (Zon and Scholz 1929, Gates and Nichols 1930, Morey 1936, Maissurow 1941, Hough and Forbes 1943, Leak 1975, Hett and Loucks 1976, Tubbs 1977). Irregular age distributions in these stands, however, appear to have been common (Gates and Nichols 1930, Maissurow 1941, Hough and Forbes 1943), which raises the possibility that reasonably severe partial disturbance could still have a major influence on forest population dynamics. In the forest of Tionesta Creek, Pennsylvania, the age structure of hemlock (*Tsuga canadensis*) was nearly discontinuous so that well-defined age groups could easily be recognized, even though they were spatially intermixed (Hough and Forbes 1943). Since wind, fire, drought, climatic variation, and irregularities in seed production could all conceivably cause such peaks and gaps in the age structure, examination of additional evidence is necessary to establish the cause in such cases.

The purpose of this paper is to report evidence on the frequency and severity of natural disturbances in a virgin deciduous forest having a species composition typical of one of the principal cover types in the eastern deciduous forest region. The forest selected for study is one of the largest virgin forest remnants in the southern United States and in terms of species composition is fairly representative of the mixed mesophytic forest type of the southeastern and central United States (Braun 1950, Küchler 1964). This type generally contains a mixture of shade-tolerant and shade-intolerant species. An investigation of the relationship of size structure to age structure and natural disturbances should provide not only a case study of disturbance history in a specific forest, but also some general principles that can be used to interpret disturbance history for virgin tracts from which only tree diameter data are available.

THE STUDY AREAS

Two sites at low and middle to upper elevations were selected for detailed study in the Joyce Kilmer Memorial Forest, a wilderness preserve which encompasses the watershed of Little Santeetlah Creek in the mountains of southwestern North Carolina. This watershed has long been considered to have virgin timber because of its remoteness, the lack of road access for logging, and the large size of its trees (Anonymous 1910). Although most of the forest has never been cut, some logging did take place near the mouth of the creek around the turn of the century, and the early local residents started frequent surface fires in the area (Ayres and Ashe 1905). However, no evidence of fire, such as fire scars on trees or charcoal in the soil, was found on the two north-facing sites selected for study.

Braun (1950) lists the common dominants of the Mixed Mesophytic association as beech, tulip tree, white basswood, sugar maple, chestnut, yellow buckeye, red oak, white oak (*Quercus alba*), and hemlock, with local concentrations of sweet birch, yellow birch, and mountain silverbell (see Table 1 for scientific names). All of these species are important components of the study areas except for white oak. The two study areas comprise the principal tracts of mixed mesophytic forest within the preserve. Much of the remainder of the watershed was originally oak-chestnut forest.

The smaller study area is a 12-ha tract along the main branch of Little Santeetlah Creek and adjacent lower slopes (elevation range 715–790 m). The larger area is a 19-ha tract on the middle and upper slopes of Poplar Cove, 1.1 km from the Little Santeetlah tract and separated from it by two large spur ridges. The Poplar Cove tract has a greater variety of community types, ranging from hemlock forest of ravines or "coves" to oak forest of the ridge and upper slopes, in which chestnut was once a codominant species. The study area ranges from 757 to 930 m in elevation and spans 30–70% of the slope distance from valley bottom to ridgetop.

The soil in the two study areas is a brown stony loam with a brown loam or clay-loam subsoil (Goldston and Gettys 1953). Based on soils data of Oosting and Bourdeau (1955) and the regression of tulip tree site index on depth of the A_1 soil horizon by Auten (1945), the predicted site index of the study areas for tulip tree (in metres at age 50 yr) is 25–30 at Little

Santeetlah Creek and 30 in the hemlock forest at Poplar Cove. Auten (1945) found that the average site index for 77 stands in which tulip tree occurred was 25. The site quality of the lower slopes would therefore probably be rated as good to excellent.

Based on an analysis of weather data from stations in the North Carolina-Tennessee mountains (Dickson 1959), the expected mean annual temperature for an elevation of 760 m is 13°C, with a January mean of 4°C and a July mean of 22°C. The mean annual precipitation at the nearest weather station (Andrews, North Carolina) is 156 cm, but the rainfall at the higher elevation Kilmer Forest is probably greater (Shanks 1954).

METHODS

Instead of a complete census of a small plot, sampling of tree diameters and ages was conducted on moderate-sized tracts. This makes it possible to identify disturbances that occurred over a fairly extensive area and eliminates the chance of confining the study area to a portion of the forest with an unrepresentative age structure. However, such data do not provide a detailed view of the spatial arrangement of age classes in a particular spot. This limitation was partly overcome by more intensive sampling in some areas.

The forest at Little Santeetlah Creek was sampled by 30 circular 405-m^2 plots, amounting to 10% of the study area. The first coordinate for each pair of plots was determined systematically at 85-m intervals along a sample line traversing the center of the tract. The second coordinate was determined by random distances on both sides of the line. All trees taller than 0.3 m were identified on each plot and diameters were measured at breast height (dbh or 1.4 m) for all trees $>$ 2.5 cm dbh.

The Poplar Cove tract was mapped and sampled by the point-centered quarter method (Cottam and Curtis 1956) to provide a more detailed view of species and size distributions. The study area was divided into 92 rectangular blocks (64 × 32 m each) and the corners located on the ground by staff compass and pacing. A stratified random sample was obtained by locating two sample points by random coordinates in each block. At each sample point, distances to the nearest overstory tree ($>$15 cm dbh) and understory tree ($>$1 m tall and $\leq$15 cm dbh) and tree diameters were measured in each of four quadrants. The sampling intensity of overstory trees was 40 trees/ha, or $\approx$15% of the trees on the tract. A topographic map of the Poplar Cove study area (Fig. 1) was constructed from data on aspect and percent slope at each of the sample points. Slope distance was converted to horizontal distance and the elevation of each sample point computed trigonometrically, using the memorial plaque as a reference point.

In Poplar Cove several community types were recognized to facilitate an analysis of spatial variation in the data. The distribution of canopy trees of the principal species was the main criterion for recognition of community types. Some species such as tulip tree and white basswood are well distributed over the tract, whereas others are common only in certain habitats or elevation ranges. The boundary of the hemlock type (Fig. 1) corresponds to the approximate upper limit of overstory hemlock, excluding scattered outliers. The boundary between the lower and upper cove hardwoods corresponds to the upper limit of overstory beech and the lower limit of red maple, sweet birch, and pignut hickory. The transition between the upper cove hardwoods and the oak type was determined by the points at which silverbell ceases to be the most common canopy tree and is superceded by chestnut oak. Maps for each species are available elsewhere (Lorimer 1976). At Little Santeetlah Creek two types of hemlock forest were recognized (with and without a dense woody understory of *Rhododendron maximum*) because of spatial segregation of yellow birch, silverbell, and beech in the two types.

Since chestnut was once a major component of the Poplar Cove forest, quantitative data were obtained on its density, diameter distribution, and age struc-

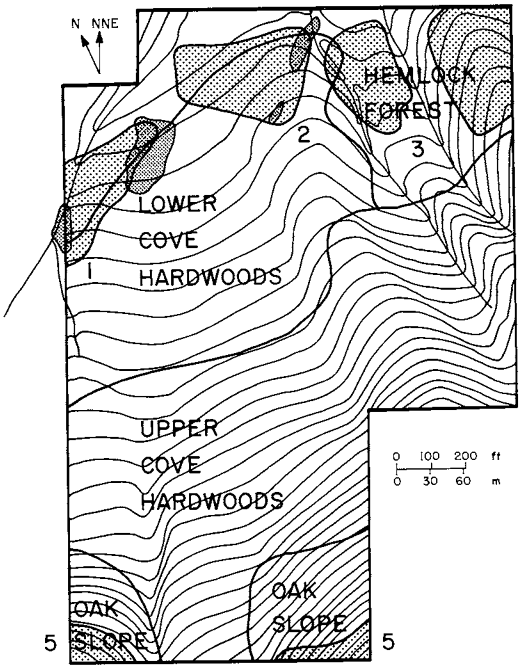


FIG. 1. Poplar Cove study area showing community types classified according to spatial distributions of dominant tree species. Lowest contour line (top) is 757 m (2485 ft). Contour interval is 4.6 m (15 ft). Numbered areas with light stippling indicate location of forest stands with higher than average number of age determinations. Areas 4 and 5 extend beyond study area boundaries (total of 2.5 and 4 ha, respectively) due to distribution of sectioned chestnut logs. Areas of dark stippling indicate location and size of windfalls that occurred in 1973–1975.

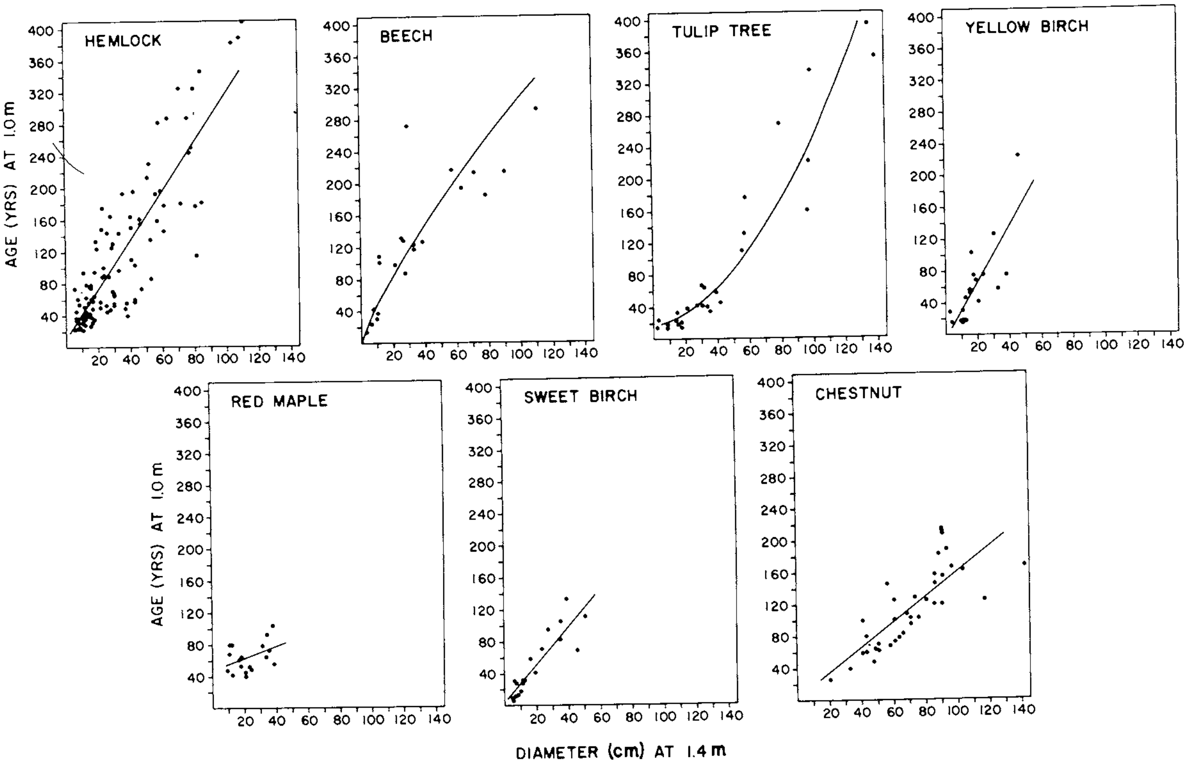


FIG. 2. Diameter-age relationships of the species selected for age structure analysis. Equations are: hemlock ($y = 9.1 + 3.09\,x$; $r^2 = .75$), beech ($y = 7.4x^{0.81}$; $r^2 = .81$), tulip tree ($y = 14.2 + 0.50x + 0.0188x^2$; $r^2 = .71$), yellow birch ($y = -4.1 + 3.53x$ $r^2 = .64$); red maple ($y = 47.8 + 0.74x$; $r^2 = .14$), sweet birch ($y = 3.7 + 2.36x$; $r^2 = .80$), chestnut ($y = 3.6 + 1.55x$; $r^2 = .61$).

ture. The chestnut logs are still well preserved due to their high tannin content, and can be readily distinguished in the field from those of associated species by their ring-porous wood and lack of visible rays (Panshin and de Zeeuw 1970). The diameter at breast height of a chestnut log and distance from sample point to root crown were recorded whenever the root crown was closer than the nearest standing tree in the quadrant.

Age determinations of 202 trees were made for seven of the principal species in order to plot diameter-age relationships. The 59 age determinations of hemlock were supplemented by an additional 42 determinations from the same site by Oosting and Bourdeau (1955 and original field notes). Because of restrictions on the number of cores, sampling among diameter classes was basically uniform to increase reliability over the range of tree size. Increment cores were taken at a height of 1.0 m. The center of the tree was reached in most coring attempts, but for some trees the center was not reached because of heartrot or because the tree radius was somewhat longer than the 46-cm bore length. Some of the incomplete cores (15) were retained if <10 cm of the radius (calculated as one-half the diameter) were missing. The remainder were rejected. The number of missing rings on incomplete cores was estimated from the diameter-age regression curves (Fig. 2). By this procedure, if the inner 2.5 cm of a core were lacking, the number of missing rings was estimated from the average age of a 5.0 cm diameter tree of that species. Since the standard error of the age of a small tree is relatively small, such a procedure should yield good estimates. Three incomplete cores from very large tulip trees, in which 20–28 cm were missing, were retained because of the low variability and rapidity of growth in young trees of this species.

Age determinations of chestnut logs were made from cross-sections of logs cut along trails. Correction factors to express age at the 1.0-m level were added or subtracted by assuming an average growth rate in height of 60 cm/yr (Zon 1904, Ayres and Ashe 1905). The average correction factor was 7 yr.

Yearly ring width measurements for 130 increment cores were made to the nearest 0.025 mm under a binocular microscope equipped with a movable stage and dial micrometer. It was necessary to sand or shave all increment cores of diffuse porous deciduous species in order to reveal the growth rings. In the laboratory, increment cores were moistened with water

TABLE 1. Relative density (%) of tree species in community types of the two study areas, Joyce Kilmer Memorial Forest (trees > 15 cm dbh).

	Little Santeetlah Creek		Poplar Cove			
Species	Hemlock/ rhodo- dendron	Hemlock/ herb	Hemlock/ herb	Lower cove hardwoods	Upper cove* hardwoods	Oak slope†
Hemlock (*Tsuga canadensis*)‡	32.2	47.4	55.0	7.4	6.8	0.9
Yellow birch (*Betula lutea*)	32.2	5.0	1.9	0.5	0.0	0.0
Sycamore (*Platanus occidentalis*)	0.8	0.0	0.0	0.0	0.0	0.0
Blackgum (*Nyssa sylvatica*)	0.8	0.0	0.0	0.0	0.0	0.0
Tulip tree (*Liriodendron tulipifera*)	8.1	8.8	6.3	9.3	5.8	8.6
White basswood (*Tilia heterophylla*)	7.2	9.6	8.1	7.9	8.2	6.0
Silverbell (*Halesia carolina*)	0.8	13.2	9.4	29.6	23.2	6.0
Beech (*Fagus grandifolia*)	0.8	7.9	6.9	17.1	1.7	1.7
Sugar maple (*Acer saccharum*)	0.8	2.6	8.1	12.5	10.6	0.0
Fraser magnolia (*Magnolia fraseri*)	2.4	0.0	0.0	0.9	0.3	0.0
Cucumber tree (*Magnolia acuminata*)	0.0	2.6	0.0	2.3	0.3	1.7
Buckeye (*Aesculus octandra*)	1.6	0.9	0.0	2.8	1.4	0.0
White ash (*Fraxinus americana*)	0.8	0.9	0.0	3.7	3.8	0.0
Sweet birch (*Betula lenta*)	8.1	0.0	0.6	2.3	13.7	14.6
Red maple (*Acer rubrum*)	1.6	0.0	2.5	0.9	12.0	18.1
Pignut hickory (*Carya glabra*)	0.0	0.0	0.0	2.3	6.5	8.6
Sourwood (*Oxydendrum arboreum*)	1.6	0.9	1.3	0.0	1.4	4.3
Black cherry (*Prunus serotina*)	0.0	0.0	0.0	0.0	0.3	0.0
Sassafras (*Sassafras albidum*)	0.0	0.0	0.0	0.0	0.0	0.9
Black locust (*Robinia pseudoacacia*)	0.0	0.0	0.0	0.0	0.0	1.7
Black oak (*Quercus velutina*)	0.0	0.0	0.0	0.0	0.3	0.9
Northern red oak (*Quercus rubra*)	0.0	0.0	0.0	0.5	2.7	7.8
Chestnut oak (*Quercus prinus*)	0.0	0.0	0.0	0.0	0.3	18.1
Number of sample trees	124	114	160	216	292	116
Number of trees per hectare	191.0	234.6	261.3	235.1	265.0	239.2

* Chestnut (*Castanea dentata*) comprised ≈20% of the trees >15 cm dbh in this community prior to the blight.
† Chestnut comprised ≈23% of the trees >15 cm dbh in this community prior to the blight.
‡ Nomenclature follows that of Fernald (1950).

after having been previously in an air-dry condition, thereby providing data on relative growth rates within a tree rather than absolute rates.

RESULTS AND DISCUSSION

Species composition and successional status

Species composition of each community is shown in Table 1. Based on species composition alone, the mesic communities would generally be considered climax because of the heavy dominance of shade-tolerant species. Climax status is also suggested by the fact that seedlings of all species are reasonably abundant, either beneath the forest canopy or in small gaps.

Species intermediate or intolerant of shade are well represented, however (cf. Baker 1949). The intermediate yellow birch is a codominant species in the hemlock forest along Little Santeetlah Creek. The intolerant tulip tree comprises 6–9% of the overstory density and 17–37% of the basal area in the various communities. Compared to 93 virgin mixed mesophytic stands sampled by Braun (1950), the Kilmer stands rank in the 74th percentile with respect to the relative density of overstory tulip tree. The oak-dominated forest has relatively few oak saplings >1 m tall (13 saplings/ha) but does have a dense understory of sugar maple, beech, basswood, hemlock, silverbell, and red maple (561 saplings/ha). Thus its successional status is uncertain. Since the observed type of species mixtures could be produced by a wide range of disturbance intensities, an analysis of age structure and growth rates is necessary to clarify the actual stand history and predominant mode of tree replacement.

Age structure

Scatter diagrams of age vs. diameter indicate that the forest is all aged for those species investigated, including the intolerant tulip tree and the mid-tolerant chestnut, yellow birch, and sweet birch (Fig. 2). There are sample trees in most 10-yr age classes up to ≈400 yr. These results support the principle that in old-growth forests, diameter distributions having a general form similar to the negative exponential distribution for individual species usually indicate all-aged structures (Hough 1932, Meyer and Stevenson 1943, Hett and Loucks 1976), even if the curves are highly irregular with prominent peaks. However, since each age class tends to have an approximately normal diameter distribution, and since the variance of diameter can be quite large in old age classes (Meyer 1930, Hough

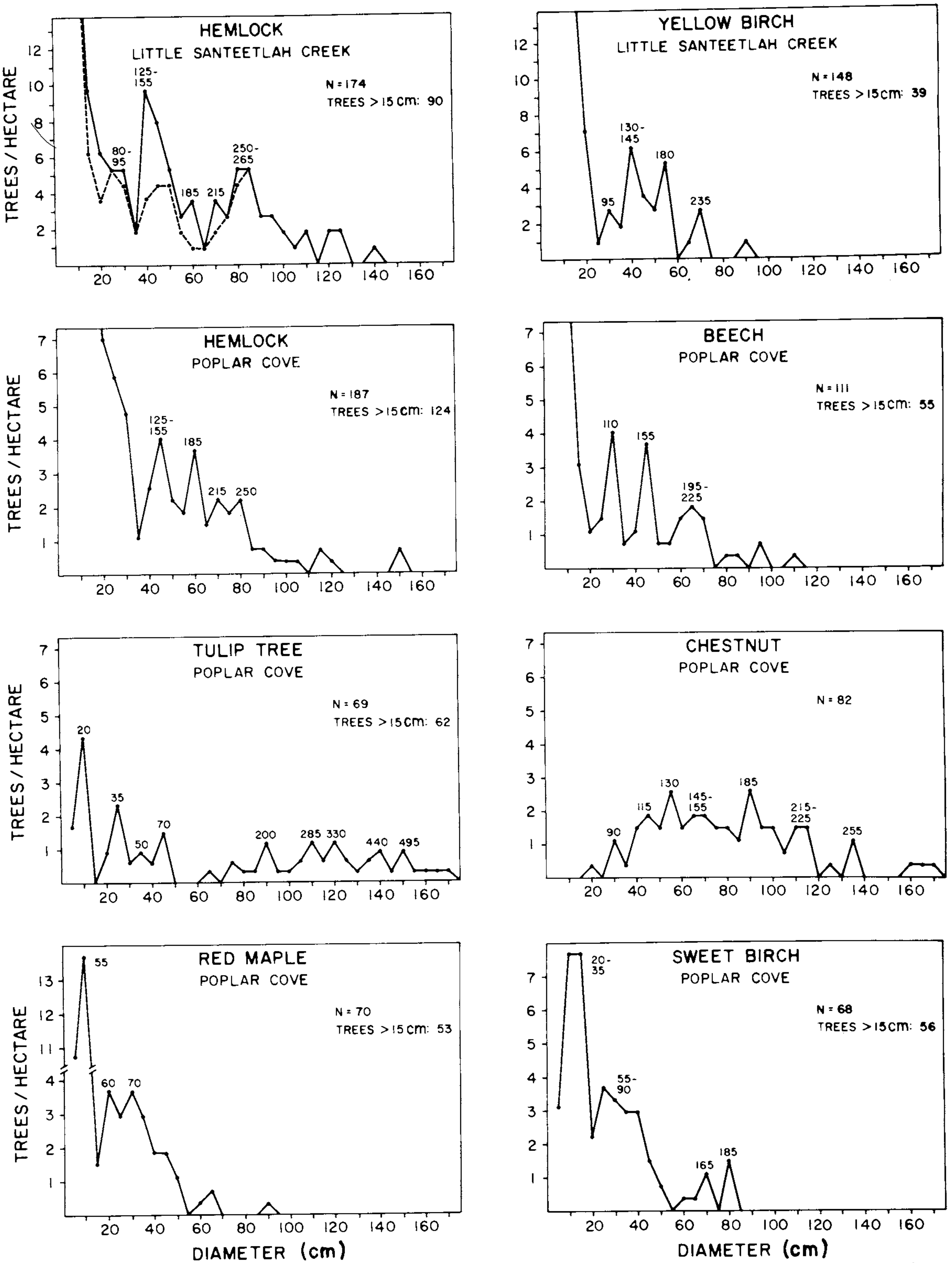


FIG. 3. Diameter frequency distributions for species having a sample size of >60 trees and for which age data are available. Numbers along abscissa are end points of 5-cm classes, and numbers above the peaks indicate predicted average age in years for class midpoints. Where more than one diameter class is incorporated into a peak, a range of age is indicated. For chestnut, 45 yr have been added to the predicted ages to account for the time elapsed since the chestnut blight. The dotted line for hemlock at Little Santeetlah Creek shows the distribution if four adjacent plots in the eastern part of the study area are omitted, revealing spatial clumping of trees in the 40- and 60-cm peaks.

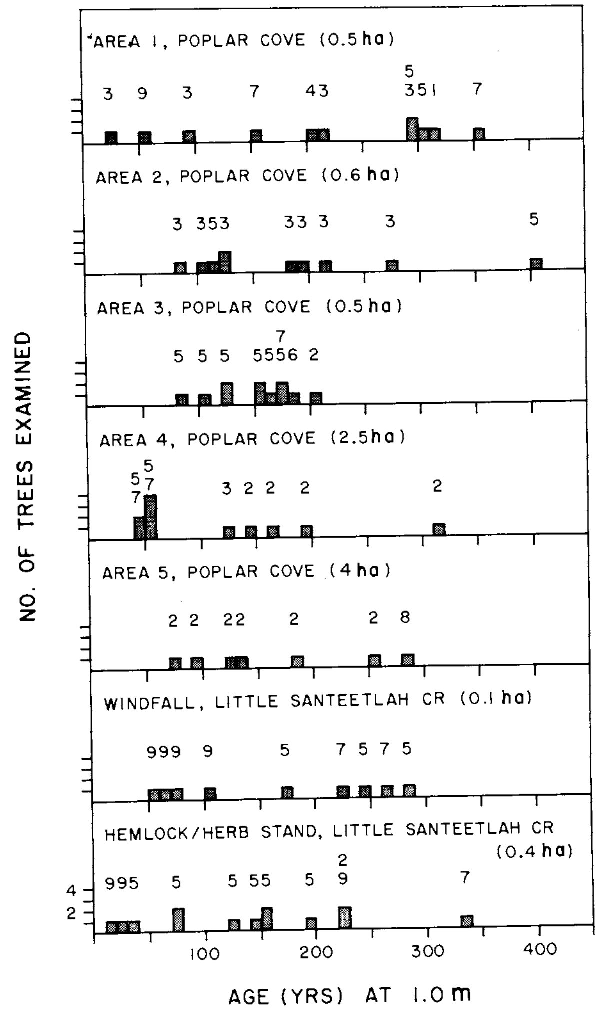


FIG. 4. Age distributions for stands having a higher than average number of age determinations (cf. Fig. 1). Numbers above the bars are species codes: 1 = white ash, 2 = chestnut, 3 = beech, 4 = sugar maple, 5 = hemlock, 6 = basswood, 7 = tulip tree, 8 = red oak, 9 = yellow birch.

1932), some exceptions can be anticipated. The most common case would be an old, even-aged overstory of a shade-tolerant species with a developing uneven-aged understory of the same species. Such a stand would have a curve similar to that of yellow birch in Fig. 3 and would be difficult to distinguish from a truly uneven-aged stand. It is also known that even-aged stands can have a negative exponential diameter distribution if many stands of dissimilar ages are grouped together or if several species in a stand are grouped (Hough 1932, Roach 1974, Oliver 1978).

Although both study areas are clearly all aged when considered as a whole, it is possible from the above considerations that each study area could be made up of several sizable even-aged stands of different ages. Therefore the data from five more intensively cored stands in Poplar Cove (Fig. 1) and two stands from Little Santeetlah Creek were analyzed separately. Each was found to have a wide range in tree age typical of uneven-aged stands, even for tracts as small as 0.1–0.5 ha (Fig. 4). Evidence of uneven-aged stand structure was found not only for mesic sites dominated by shade-tolerant species but also for a drier ridge site originally dominated by chestnut (Area 4, Fig. 1) and for the chestnut component of a high-elevation oak-chestnut site (Area 5, Fig. 1). The less intensively cored parts of the study areas also revealed trees of various ages scattered throughout. Thus if the study areas are comprised of even-aged groups, the average group size would have to be fairly small.

The diameter distributions of nearly all species have two or more well-defined peaks (Fig. 3). Since diameter is highly correlated with age (Fig. 2) it is probable that the peaks correspond to decades in which there was unusually high survival or recruitment of young trees. The estimated average age of trees in each peak was obtained from regression curves fitted by standard least squares techniques. This method provides unbiased parameters, which makes it preferable to the logarithmic transformations used in a previous analysis of the data (Lorimer 1976). For most species, linear equations were used. The data for tulip tree and beech showed strong indications of curvilinearity and were fit by the equations

$$y = a + bx + cx^2$$
$$\text{and } y = ax^b,$$

respectively, where y = tree age, x = diameter, and a, b, c are estimated parameters. The types of equations chosen seem biologically reasonable in view of the observed radial growth patterns.

A comparison of the predicted ages for peak diameter classes reveals apparent coincidence of peak age classes among a number of species (Fig. 3). For example, peaks in the 145–155 yr and 180–190 yr age classes are present in five out of seven cases in Fig. 3 (omitting red maple due to lack of old trees). Other peaks common to two or more curves are the 85–95, 110–115, 215–225, and 250–260 yr age classes. The major peak for hemlock (predicted age 125–155 yr) is included in the 145–155 yr age class because none of the trees sampled from this peak was between 125 and 145 yr old.

Independent random subsamples of various sizes (25, 50, and 75% of total sample) were used to test for variability in the form of the diameter distributions. Although there is some fluctuation in the height of the peaks among subsamples, the location of the peaks on the abscissa was consistent throughout.

It is evident that each age class peak generally covers a broad span of years. Even if the causal factor occurred within a narrow time interval (as with windthrow), it would not be possible to identify the exact year because a peak diameter class will inevitably include trees of other age classes due to differential

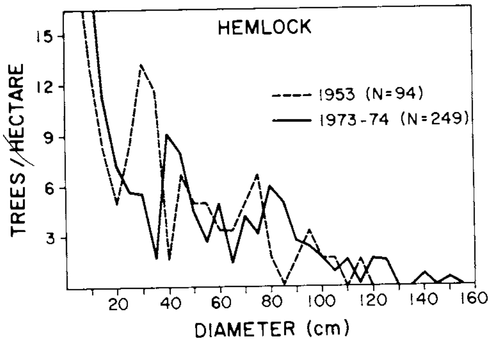


FIG. 5. Combined hemlock diameter distribution of both study areas compared to the distribution in 1953, plotted from original field notes of Oosting and Bourdeau (1955).

growth rates. Equally important is the fact that disturbances that kill primarily overstory trees (e.g., windthrow, insects, disease) typically "release" many understory saplings that not only pre-date the disturbance but may also have a wide range in age among individual saplings (Henry and Swan 1974). Even colonization of the disturbed site by new individuals will generally take place over a period of a decade or more. A diameter class peak can also be caused by the cumulative effect of several small disturbances over a period of one or more decades (Oliver and Stephens 1977).

The apparent coincidence of any particular peak age class among species is affected by this age span, and also by the standard error of the slope of each regression curve. The latter cannot be computed due to the marked inequality of variance (heteroscedasticity) in the diameter-age data. However, it is possible to test whether the overall pattern of coincidence among species could have occurred simply by chance as a result of random, unrelated events. If each age class has an equal chance of having peak numbers of trees of a species, then the number of age classes having 0, 1, 2 . . . *n* peaks should follow a Poisson distribution (Snedecor and Cochran 1967). A χ^2 test showed that the differences between the observed and expected (Poisson) distributions are significant at the .004 level ($\chi^2 = 13.21$, df = 3). There are more age classes with four or five peaks than expected by chance (observed = 4, expected = 1.2) and more age classes with zero peaks (observed = 8, expected = 4.1). The χ^2 test indicates that the probability of the 29 peaks $\geq$70 yr and $\leq$260 yr being distributed among 10-year age classes in such an aggregated manner by chance is <1%.

Development of the all-aged structure

The behavior of these peak diameter classes over time is suggested by a comparison of the combined 1973–1974 hemlock data from both study areas with a survey of the same sites by Oosting and Bourdeau in 1953. The three major peaks in 1953 are still recognizable in the 1973–1974 data, allowing for growth during the 20-yr period and reduction in height of the peaks due to mortality (Fig. 5). This evidence supports the interpretation of the peaks as well-defined size classes that retain their identity over a substantial period of time.

The apparent average mortality rate for trees in the three peaks during this time (20%) is close to what would be predicted from the slope of a smooth negative exponential curve (16%) or negative power function curve (16%) fitted by iterative nonlinear regression. The reduction in height of the peaks and extension of the smooth "tail" of the curve into the 20–30 cm classes have resulted in a better fit of the more recent data to theoretical distributions. For the negative exponential model ($y = 11.44e^{-0.0175x}$, $15 \leq x \leq 150$) the r^2 value has increased from .52 in 1953 to .66 in 1973–1974. For the negative power function model ($y = 102.29x^{-0.833}$, $15 \leq x \leq 150$) the r^2 has increased from .41 to .62. These changes suggest a trend toward a more balanced age structure over time in line with the classic all-aged model of stand structure (Meyer and Stevenson 1943). A poorer fit to the negative exponential distribution after 23 yr was found by Christensen (1977) in second-growth oak forest, and this was largely due to high mortality in the smaller size classes and low recruitment. It is probable that such species as oak and hickory are not sufficiently tolerant of shade to maintain the large numbers of suppressed trees needed for a good fit to the negative exponential distribution, and thus this distribution should probably be expected only for tolerant species.

Cause of age class peaks

Evidence on the cause(s) of the age class peaks can be obtained from the radial growth patterns of trees in the same decades. A period of favorable climate would be expected to produce a synchronous increase in growth rate among most overstory trees that would persist for the duration of the favorable period. Fritts (1962), for example, obtained multiple correlation coefficients of .83 to .90 for the relationship between radial growth and evapotranspiration data for beech and white oak. Disturbance would be expected to cause abrupt and sustained increases in growth of understory trees "released" by the death of nearby overstory trees, and also increased growth of some of the adjacent surviving trees in the upper crown classes. The number of trees responding to release would depend on the severity of the disturbance. A period of high seed production, on the other hand, would not cause an increase in radial growth unless it also was followed by a period of favorable climate.

Examination of the increment cores revealed no consistent trends in growth during the decades in question. The radial growth patterns are typically irregular, with major overlapping cycles of growth lasting from 5 to >60 yr, upon which are superimposed smaller yearly fluctuations. It is clear, however, that some trees display the types of growth patterns expected following disturbance. Several criteria were established for distinguishing trees showing a clear response to disturbance from other trees in which no pattern is evident or in which the pattern could have been caused by either climate or disturbance. The period of rapid growth had to occur abruptly and be sustained for at least 15 yr, and also had to be preceded by a period of relatively slow growth lasting at least 15 yr. The average rate for the 15-yr period following release had to be at least 100% greater than the previous average rate. For trees already showing moderate growth before release (>1.0 mm yearly radial increment, lab conditions), a 50% increase in average rate was required.

Examples of trees meeting these criteria are shown in Fig. 6. Periods of rapid growth following the year of abrupt increase, as well as the preceding periods of slow growth, are frequently sustained for 30 consecutive yr, and in some cases as many as 90 consecutive yr (Fig. 6, trees 2, 5, 6, 9, 10, 12). Due to the prolonged nature of these changes in growth rate, it is unlikely that climatic variation could be the principal causal factor. Climatic records for the nearest weather station in Andrews, North Carolina indicate short-term fluctuations with no evidence of longer cycles for the period of record (1909–1975). For example, the mean time interval between precipitation maxima and minima (years in which the deviation from the mean exceeds 50% of the maximum observed deviation) is 4.2 yr. Oscillations in temperature during the growing season have been less pronounced, and temperature alone or in combination with precipitation shows no consistent relationship to the frequency of abrupt increases in growth. Also, since sustained increases in growth can be found in nearly every decade (Figs. 6 and 7) it is unlikely that they are responses to favorable changes in climate. Even for trees showing abrupt increases in the same decade, the duration of rapid growth is highly variable. Disturbance is a reasonable explanation for the longer cycles because for many trees, growth rates prior to release are comparable to those of trees currently in the understory, while the rates following release are comparable to those of trees in the dominant or codominant crown class (Fig. 6, trees 1, 4, 5, 6, 7, 9, 11, 12, 13, 14). It is probable, however, that the smaller yearly fluctuations are climatically related.

Although releases, and hence overstory tree mortality, have occurred in nearly every decade, a summary compilation indicates higher than average frequency for the decades 1780, 1820, 1870, 1890–1900, 1920, and 1930 (Fig. 7). These correspond to age class midpoints 195, 155, 105, 75–85, 55, and 45, respectively. The percentages of trees showing release in these decades range from 73–250% higher than the average of the intervening decades. Four of these six decades are also represented by peak age classes. The two exceptions (1780 and 1930) may be related to age class peaks 185 and 35, since the span of years in these age classes (at least 10 yr) is such that the actual period of establishment would have overlapped with the time of releases.

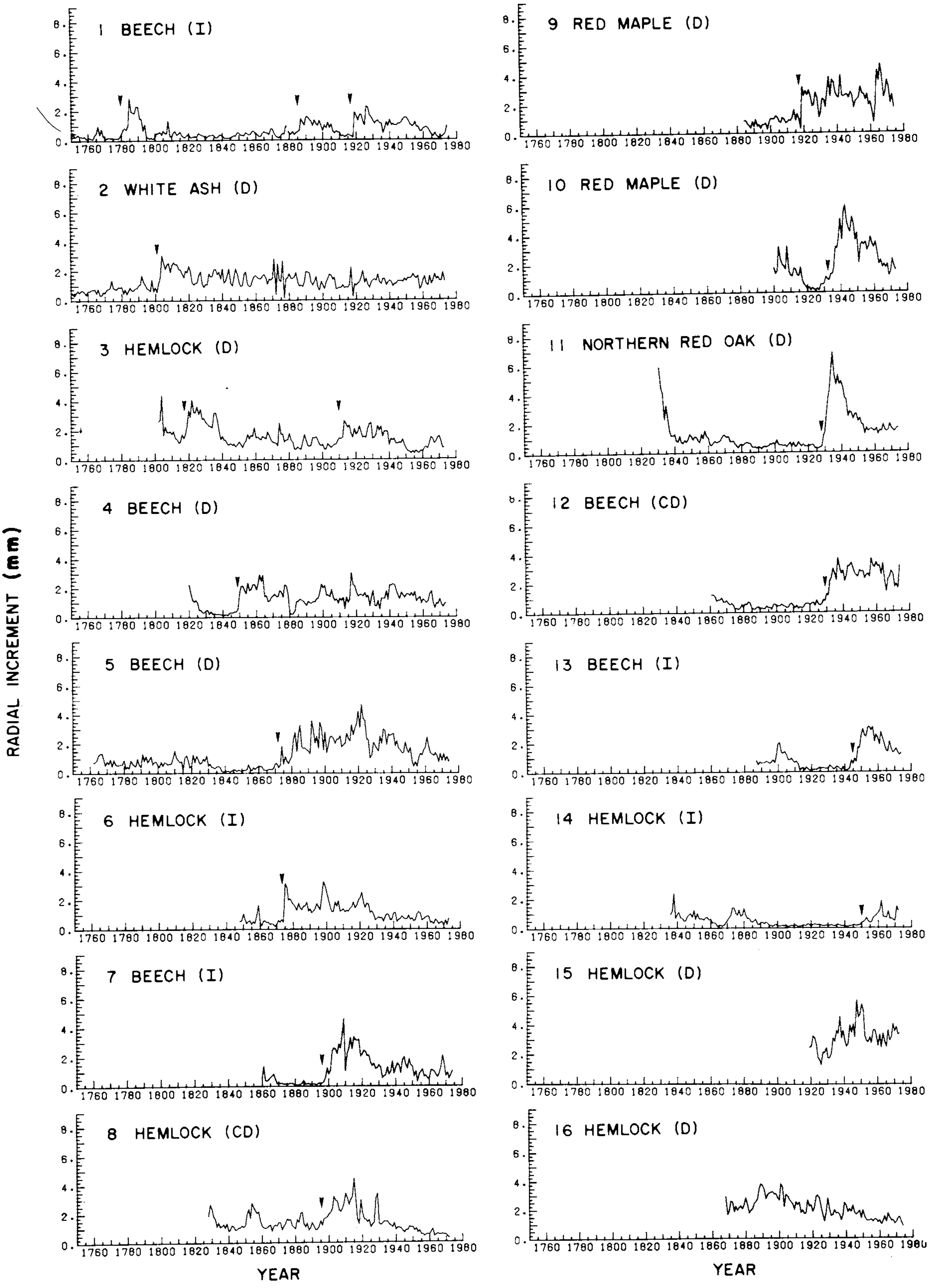
Further evidence on the growing conditions of trees during the decades of peak age classes pertains to the early growth of saplings. Since the standard core height was 1.0 m, the first 10 rings on each increment core reflect the growing conditions of each tree at the time that it was a small sapling. For trees in which the first 10 rings were formed in one of these six decades, 78.4% were growing at a moderate or rapid rate (>1 mm/yr radial increment). This is contrasted with 50.0% of the trees showing moderate or rapid initial growth in the intervening decades. Many of these trees were growing at rates comparable to those of canopy trees, suggesting that they were not growing under dense shade when in the sapling stage (e.g., Fig. 6, trees 4, 8, 11, 15, 16). This evidence is more direct in suggesting that many of the trees in peak age classes grew under postdisturbance conditions. Thus the possibility that the trees in peak age classes are simply a result of years of high seed production seems less plausible.

An illustration of the effects of a known disturbance is provided by the chestnut blight, an introduced disease which caused complete dieback of chestnut in the region from 1928–1938 (Braun 1942, Woods and Shanks 1959). The percentage of trees showing release in this decade is 153% higher than the average for nondisturbance decades. The percentage of releases in this decade is 717% higher if trees from the mesic communities that had little chestnut are eliminated from the sample (Fig. 7). The blight is probably responsible for major peaks in the age structure of sweet birch and tulip tree on the higher slopes (Fig. 3), although establishment of new seedlings appears to have lagged behind the immediate response of understory trees to the dieback of the crowns. Tulip trees of this age are largely restricted to the vicinity of chestnut logs in the upper cove hardwoods and oak slope communities of Poplar Cove.

It therefore seems that while radial growth patterns and population structure probably reflect the interaction of several factors, disturbance by itself is sufficient to account for abrupt and sustained increases in radial growth and peaks in the age structure. The other most likely factors (variation in climate and seed production) do not appear to be adequate causes in this case.

The principal disturbances and their extent

In addition to the disturbances for which both age class and radial growth evidence are available (1820,


1 BEECH (I)
2 WHITE ASH (D)
3 HEMLOCK (D)
4 BEECH (D)
5 BEECH (D)
6 HEMLOCK (I)
7 BEECH (I)
8 HEMLOCK (CD)
9 RED MAPLE (D)
10 RED MAPLE (D)
11 NORTHERN RED OAK (D)
12 BEECH (CD)
13 BEECH (I)
14 HEMLOCK (I)
15 HEMLOCK (D)
16 HEMLOCK (D)
RADIAL INCREMENT (mm)
0.
2.
4.
6.
8.
1760 1780 1800 1820 1840 1860 1880 1900 1920 1940 1960 1980
YEAR

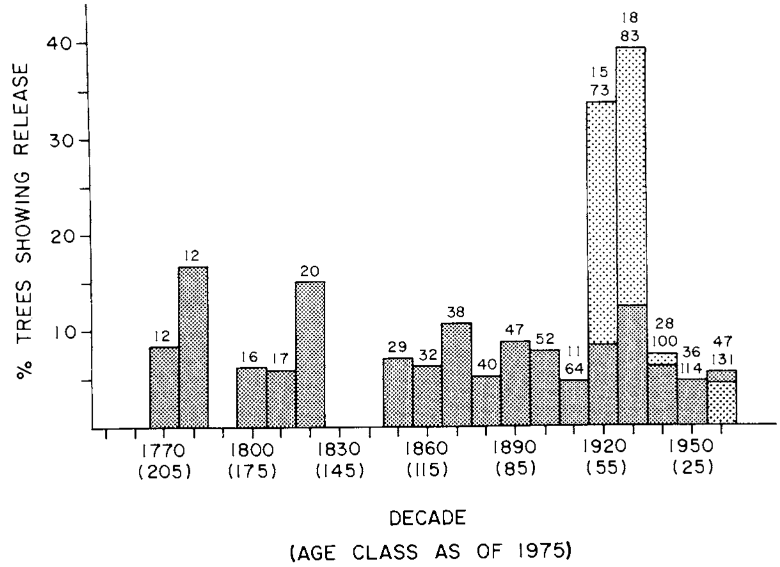


FIG. 7. Percentage of age-sampled trees showing release from suppression in each decade since 1750. Data from chestnut and tulip tree not included because these species rarely respond to release. Light stippled bars indicate percentage of trees showing release since 1905 in areas of >20% chestnut. No releases in chestnut areas were observed for 1910 and 1950; the percent release in 1960 in the chestnut areas is indicated by the lower horizontal bar. Sample size (number of trees) for each decade is shown above the bar; after 1905 the upper number indicates the sample size in the chestnut areas.

1870, 1890, 1920, 1930, and possibly 1785), there is evidence for several disturbances which either occurred so long ago that few radial growth data are available, or so recently that peaks have not yet developed in the diameter distributions. A disturbance or series of disturbances about 1755 is suggested by 215–235 yr peaks for hemlock in both study areas, yellow birch at Little Santeetlah Creek, and beech and chestnut at Poplar Cove. An earlier one about 1720 is suggested by 255-yr peaks for hemlock in both study areas and chestnut in Poplar Cove. Even earlier disturbances as far back as 1550 are almost certainly indicated by substantial numbers of the intolerant tulip tree in corresponding age classes.

There have been several cases of obvious disturbance in recent decades. Four plots in the hemlock-rhododendron forest of Little Santeetlah Creek were situated in windfalls which, based on the ages of yellow birch that had germinated on the fallen logs, all occurred about 1953. These windfalls were the final result of the death of several large trees and ranged from 200 to 400 m^2 in ground surface area for a total of 1330 m^2. From 1973–1975, while field work for this study was in progress, three sizable windfalls occurred in the lower portion of the Poplar Cove study area, amounting to 2200 m^2 or 1.2% of the tract (Fig. 1).

Disturbances of higher than average intensity have therefore occurred at ≈30-yr intervals in the past 250 yr for the study areas as a whole, and ≈40-yr intervals for a given community. The areal extent of each disturbance is indicated primarily by the location of communities in which corresponding age class peaks are found. The curves in Fig. 3 suggest that the disturbances of 1720, 1750, 1785, and 1820 were well distributed in both study areas whereas those of 1870, 1890, and 1920 were possibly more localized.

Severity of disturbances

The severity of disturbances is an important topic in relation to the management issues raised in the introduction. Estimates of severity are possible from two lines of evidence. Data on the percentage of trees showing release in each decade (Fig. 7) can be converted to estimates of overstory mortality in each decade if a ratio of the proportion of trees killed to the proportion released can be calculated. The number of

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FIG. 6. Yearly radial increments for a subsample of trees showing abrupt and sustained increases in growth rate (trees 1–14). Arrowheads identify periods that qualify as releases from suppression by the criteria outlined in text. Trees 15 and 16 are examples of trees with no history of suppression. Letters in parentheses indicate crown class in 1975. Dominant (D) and codominant (CD) trees are in the canopy layer. Intermediate trees (I) are taller than 12 m and partly overtopped by other trees.

trees sampled in each age class forms a roughly constant proportion of the trees in all older classes capable of responding to release. This permits a direct comparison of the intensity of different disturbances.

In principle, if overstory and understory trees are more or less uniformly distributed on an area, then mortality of overstory trees on 10% of the total area should remove 10% of the overstory trees and release 10% of the understory trees, giving a ratio of the percent mortality to percent release of 1:1. In reality, however, several complications occur. It is not likely that all understory trees released by disturbance would meet the necessary criteria for release in this study, yet lesser criteria would make it difficult to separate climatic effects. Large trees bordering a patch disturbance sometimes respond with increased growth. Also, a substantial proportion of understory trees is usually killed by the falling canopy trees, although this may be somewhat offset by a higher chance of subsequent survival for those not killed. These factors make it desirable to derive empirical estimates of the mortality:release ratio.

Long-range estimates of overstory mortality are probably best provided by the slopes of the relatively stable diameter distributions of shade-tolerant species. The slopes of the fitted negative exponential curves in conjunction with growth rate data from the diameter-age relationships yield a predicted average 10-yr mortality rate of 5.6% for hemlock and 6.5% for beech (among overstory trees >30 cm dbh). Since the mesic communities are heavily dominated by shade-tolerant species, an overall average of 6% for the forest is probably a reasonable estimate. From Fig. 7, the average 10-yr rate of release in the last 200 yr has been 6.9%. This suggests an approximate 0.9:1 ratio of mortality to release.

The chestnut blight, a disturbance of known severity, provides a second estimate of the mortality:release ratio. There is currently an average of 51.9 chestnut root crowns/ha in the two upper communities in Poplar Cove, the only areas in which chestnut was common. Assuming an average total stand density of 250 trees/ha (Table 1), chestnut would have comprised 20% of the stand in this area. The proportion of releases during the decade of the blight (1928–1938) was 33%. If we assume that 6% release was due to normal mortality unrelated to the blight, as calculated above, the incidence of release caused by the blight alone would have been 27%. This yields a mortality:release ratio of 0.7:1.

From the average 0.8:1 ratio we can estimate from Fig. 7 that in the disturbance decades of 1780, 1820, 1870, 1890, 1920, and 1930, the amount of overstory mortality was ≈14%, 12%, 8%, 6%, 6%, and 10%, respectively. Since the average or normal 10-yr mortality rate for the 200-yr period is 5.5% (6.9% × 0.8) and even nondisturbance decades have an expected mortality of 3.8%, we can conclude that the premature mortality of trees that could be accounted for by exogenous disturbance alone was in the range of 2–10%, and perhaps closer to a range of 1–9%. The latter estimate is probably more accurate since disturbance in a particular decade would generally remove old trees that would have died anyway during that time, and because premature removal of trees in disturbance decades would cause the mortality in subsequent decades to be below average. Thus the long-range average mortality rate of 6% probably most closely approximates expected mortality under steady state conditions. The indicated disturbance intensity of <1% in 1890 and 1920 might seem rather low, but the disturbances of 1973–1975 in Poplar Cove that blew down several large patches of trees were of this same magnitude (Fig. 1).

The relative numbers of the intolerant tulip tree in each decade provide independent evidence on the severity of disturbances. Fewer tulip trees appear to have been established between 1785 and 1920 than during the chestnut blight, which removed 10% of the overstory on the Poplar Cove tract and has resulted in ≈4 tulip trees/ha surviving in age classes 35–50. In the upper communities, where chestnut comprised 20% of the overstory, 11 tulip trees/ha are present in these age classes, and in the lower cove hardwoods, where chestnut comprised 6% of the overstory, 3 tulip trees/ha are present. By comparison, 2.5 tulip trees/ha are present in age classes 75–200 yr. If tulip tree mortality rates are similar to those of even-aged oaks and tulip trees on comparable sites (30% mortality from age 40–60, 25% from age 60–80, 15% from age 80–100; McCarthy 1933, Schnur 1937), then 2.5 trees/ha projected backward from age 180 to age 40 would give a predicted maximum density of 10.7 trees/ha. The disturbance level necessary for this level of density would therefore be ≈20% (exogenous) overstory mortality, distributed over a period of 115 yr. Three principal disturbances occurred during this period, which suggests that each would have removed <10% of the overstory. The relatively high numbers of tulip trees in age classes 250 and older may indicate that disturbances between 1550 and 1725 removed >10% of the stand. The presence of large 30–50 yr peaks for sweet birch and red maple in areas affected by the chestnut blight (Fig. 3) supports the viewpoint that relatively light disturbance (i.e., 20% mortality) can cause major deviations from the negative exponential form.

Spatial variations in disturbance intensity

Although the trees killed in most of the disturbances seem to have been distributed over a large area, there is evidence that at least some of the disturbances were spatially variable in their intensity. The location of trees in peak diameter classes on maps revealed a few cases of pronounced clumping. Twelve of the 24 trees in the peak 40–50 cm class of hemlock at Little San-

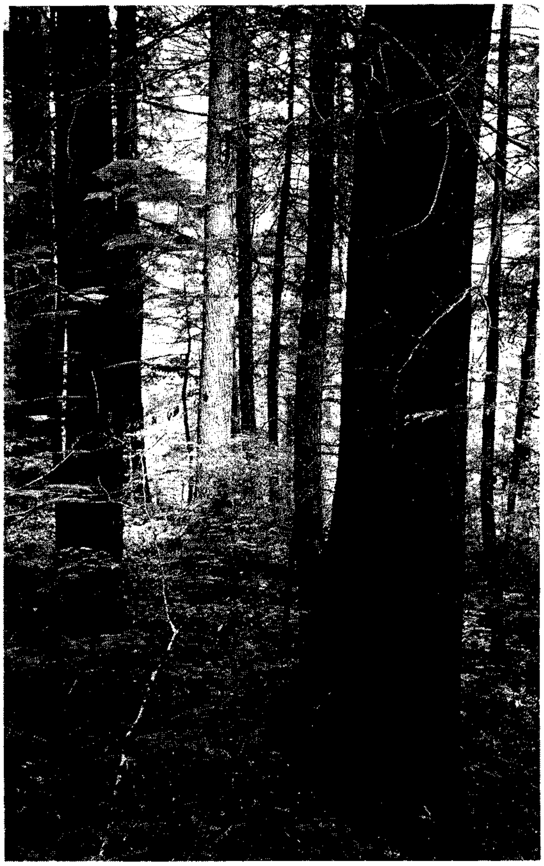

FIG. 8. Hemlock-dominated forest, Area 3 of Poplar Cove (cf. Figs. 1 and 4), heavily influenced by disturbances in 1785 and 1820. Most canopy trees range from 32 to 96 cm in diameter and 125–185 yr in age.

teetlah Creek occurred in four adjacent plots in the eastern part of the study area. Coring of trees in this area revealed that trees of this size were 150–160 yr old with a history of moderate to rapid growth and no suppression, and therefore probably date from the disturbance of 1820. Hemlock trees in the 60–70 cm peak were entirely restricted to this area. Five of the 12 yellow birch trees in the 35–45 cm peak were located in one plot in the northwest part of the study area. Trees in other peaks were fairly well distributed.

Direct age analysis also revealed some spatial clumping of age classes. Area 3 of Poplar Cove has a relatively restricted age distribution, with most overstory trees between 125 and 185 yr old (Figs. 4 and 8). Area 4 in Poplar Cove was heavily influenced by the disturbance of 1920, and subsequently by the chestnut blight. Most of the overstory hemlocks in this area were 55 yr old in 1975, showing rapid growth dating from 1920 and a further increase about 1930 (e.g., Fig. 6, tree 15). In some cases, areas heavily influenced by disturbance may have had complete mortality in patches as was the case for the disturbances of 1953 and 1973–1975. In other cases it was apparently partial mortality within a group. The eastern portion of the Little Santeetlah tract, for example, still has survivors from age classes much older than 1820 (Fig. 4, graph 7).

Causes of disturbance

Relatively little direct evidence is available on the causes of these disturbances. All of the patch disturbances of 1973–1975 were clearly windfalls. Instances of scattered standing dead trees, alone or in groups of two, were also common at this time. Such mortality can be due to a number of causes, including physiological senescence, insects, disease, lightning strikes (Taylor 1973), ice storms (Downs 1938), and drought (Hough 1936, McIntyre and Schnur 1936). Epidemics of insects and disease are rare among species of mixed mesophytic forests (Fowells 1965, Hepting 1971), but minor fluctuations in mortality from these causes may partly account for the past disturbances.

Hurricanes have been known to move inland over North Carolina (Tannehill 1956) but such storms ordinarily lose much of their force due to friction with the land surface and loss of sustaining energy from the warm ocean surface (Battan 1961, Critchfield 1974). Records from the nearest weather station with windspeed data (Asheville, North Carolina) indicate that the two highest windspeeds recorded for the period 1911–1972 were 84 and 97 km/h, generally below the windspeeds of hurricanes and tornadoes. Two periods in which windfalls were known to occur (1951–1953 and 1973–1974) each had maximum observed windspeeds of 64 km/h. This might indicate that patch windfalls can be caused by moderate windspeeds, although localized occurrence of brief, intense winds could easily escape documentation.

Fire is a possible but less likely cause for the disturbances after 1800, because the fires would have had to be intense enough to cause significant overstory mortality, but not intense enough to kill the fire-sensitive saplings of hemlock and beech, many of which clearly survived the disturbances. There is insufficient evidence for disturbances prior to 1800 to assess the possibility of fire in those centuries.

GENERAL DISCUSSION

Irregular diameter or age distributions of the negative exponential type, such as those from the Kilmer Forest, appear to have been common in mixed-species virgin forests of eastern North America. Curves for single species reported in the literature (Hough and Forbes 1943, Leak 1975, Willis and Coffman 1975, Hett and Loucks 1976) and curves for single species plotted from stand table data (Cary 1896, Hosmer 1902, Chittenden 1905, Gates and Nichols 1930, Oosting and Billings 1939-original field notes, Maissurow 1941, Whittaker 1948, Eyre and Zillgitt 1953) reveal that of 48 stands, 56% were of the irregular negative exponential type, 25% were of the smooth negative exponential type, and 19% approached a normal distribution. An additional seven virgin deciduous forest stands in northern Wisconsin were reported to be all aged by Zon and Scholz (1929), making a total of 32 remnant virgin stands known to be uneven aged or all aged by direct analysis. Thus the evidence available to date suggests that the uneven-aged condition was probably common in mesic forest types, especially those dominated by shade-tolerant species. Even at the northern limits of the eastern deciduous forest, where natural catastrophes are fairly common, the recurrence interval for such disturbances appears longer than the time interval needed to develop an uneven-aged structure (Lorimer 1977, Canham 1978). The even-aged condition has been more frequently observed in pine stands than in forests of shade-tolerant species (Cary 1894, Spalding and Fernow 1899, Hough 1932, Hough and Forbes 1943, Henry and Swan 1974), but it is probable that a significant proportion of mesophytic stands did develop following tornadoes or other violent windstorms.

Age structures in oak-chestnut and oak-pine forests are not well known. This study and that of Morey (1936) provide preliminary evidence that indicates the existence of uneven-aged structures in stands in which oak and chestnut were intermixed with more mesic species. Clearly, more work needs to be done in the more xeric types, but due to the geographical distribution of virgin stands, such analyses will be largely limited to montane oak stands in the Great Smoky Mountains National Park and the Kilmer Forest. The oak-chestnut forests, however, had a high frequency of fire disturbance of human origin during the early settler period. Ayres and Ashe (1905), for example, found evidence of frequent light fires on 80% of a 2.3×10^6 ha area examined.

It appears from the results of this study that relatively light disturbances are capable of causing peaks commonly found in irregular diameter distributions. Allowing for the normal 10-yr mortality rate of $\approx$6% due to such factors as competition and old age, an additional 6–8% premature mortality in a particular decade caused by disturbance appears capable of generating a major peak in the distribution. Such peaks appear to form when many gaps occur in the overstory over a short period of time, leading to higher than normal survival of the understory trees (Hough and Forbes 1943, Eyre and Zillgitt 1953) and new recruitment of seedlings (Trimble 1965). After the gaps close, new recruitment of seedlings would probably drop off for several decades as usually occurs in young even-aged stands. Although exogenous disturbance (including drought) is probably the major cause of irregular patterns of mortality, it is probable that even random fluctuations in mortality unrelated to disturbance could also cause irregular diameter distributions due to their effect on temporal patterns of seedling establishment. Because of the apparent sensitivity of diameter distributions to light disturbance, irregular distributions with major peaks may not necessarily indicate two- or three-aged stands that developed in response to moderately severe disturbances.

Light disturbances do not always appear to produce peaks in the size structure, especially in stands of young trees that have the capacity to fill gaps by crown expansion (Oliver and Stephens 1977). And since diameter distributions of even-aged stands may show some irregularities with several peaks (Hough 1932), it is possible that a partial disturbance can also cause more than one peak. For these reasons, the analysis of radial growth pattern for release from suppression in conjunction with diameter distributions is preferable to analysis of size or age distributions alone.

With respect to wilderness preservation, much concern over possible aesthetic deterioration of the Kilmer Forest and similar areas has been expressed in relation to the possible loss of the famed tulip trees or yellow-poplars, which attain diameters up to 2 m and for which Poplar Cove was originally named. Under the assumption that these stands were even aged and that tulip trees would be succeeded by more tolerant species, silvicultural manipulation was recommended or discussed by some individuals and groups (North Carolina Forestry Council 1971, Miller 1972, Simmons 1972). The fairly stable age distribution and the frequency of disturbance, however, indicate that major change in tulip tree abundance would be unlikely in the next two centuries. The recent windfalls in Fig. 1 have been colonized by many vigorous seedlings of this species. There also has not been much apparent change in the population in recent decades. The percentage of tulip tree in the dominant/codominant crown class was 16% in 1940 as sampled by Braun (1950), compared to 14% in 1973–1974. A moderate decline can be anticipated as a result of the small number of trees in the 90–190 yr age classes, but the young trees from recent disturbances will help counteract this trend.

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Natural disturbance regimes in northeastern North America—evaluating silvicultural systems using natural scales and frequencies

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Abstract

Many scientists and foresters have begun to embrace an ecological, natural disturbance paradigm for management, but lack specific guidance on how to design systems in ways that are in harmony with natural patterns. To provide such guidance, we conducted a comprehensive literature survey of northeastern disturbances, emphasizing papers that studied late-successional, undisturbed, or presettlement forests. Evidence demonstrates convincingly that such forests were dominated by relatively frequent, partial disturbances that produced a finely patterned, diverse mosaic dominated by late-successional species and structures. In contrast, large-scale, catastrophic stand-replacing disturbances were rare, returning at intervals of at least one order of magnitude longer than gap-producing events. Graphing the contiguous areas disturbed against their corresponding return intervals shows that these important disturbance parameters are positively related; area disturbed increases exponentially as the return interval lengthens. This graph provides a convenient metric, termed the natural disturbance comparability index, against which to evaluate both single and multi-cohort silvicultural systems based on their rotations or cutting-cycles and stand or gap sizes. We review implications of these findings for silvicultural practice in the region, and offer recommendations for emulating natural disturbance regimes. © 2002 Elsevier Science B.V. All rights reserved.

Keywords: Biodiversity; Northern hardwoods; Spruce; Hemlock; Acadian forest; Gap dynamics; Fire; Wind; Benchmarking

1. Introduction

During the past decade, many scientists and foresters in North America have begun to embrace an ecological, natural disturbance paradigm for management (Seymour and Hunter, 1999). The degree to which on-the-ground management actually conforms to natural patterns varies widely, however, due in part to a lack of specific, quantitative guidelines for emulating natural patterns and processes. Indeed, one can encounter forest managers purporting to embrace a natural disturbance model, with statements such as "All forests are wiped out periodically; our clearcuts are no different". As we illustrate below, this superficial statement ignores the growing body of evidence about what the natural disturbance regimes were really like in presettlement forests before people dramatically altered them. Here, we adopt Hunter's (1996) definition of "natural" as meaning "without human influence", and accept that the condition of the forest before European colonization is the best modern surrogate for this condition.

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Practicing foresters lack ready access to the contemporary disturbance literature, and hence have little specific guidance on how to design and manipulate disturbance parameters in ways that emulate natural patterns. To provide such guidelines, we conducted a comprehensive literature survey of all northeastern disturbances, emphasizing papers that studied late-successional, undisturbed, or presettlement forests. We focused on the northern hardwood and mixed-conifer forest types common in Acadian region of northern New England and New York, the upper Midwest, and the Maritime Provinces of Canada. We summarize results graphically in a manner that allows forest managers to evaluate how closely any silvicultural system approximates natural conditions. We conclude by suggesting some implications of these findings relative to silvicultural practice in the northeast.

We are not necessarily advocating that all forest lands be managed under the ecological principles discussed herein; this is obviously a larger societal decision that balances biodiversity with economics. Our purpose is to advance the practice of ecological forestry beyond the application of simple principles, to a more rigorous approach that is benchmarked against what we know about the dynamics of natural forests.

2. Methods used to study disturbances

2.1. Sources of information

Sources of information about disturbance regimes are varied but few have used the combination of historical, paleoecological, dendrochronological, and other approaches recommended by Foster et al. (1996). Old-growth stands are a common source of disturbance regime information. Often current disturbances, especially canopy gaps, are measured and converted to frequencies and size ranges (Runkle, 1985). Alternatively, dendrochronology is used to determine age structures and growth patterns, both of which can be used to make inferences about past disturbances (Lorimer, 1985). The major drawback of using old-growth stands is that they are rare and not necessarily representative of the landscape. They can give a biased view of the landscape because stands severely disturbed by natural disturbance in the past historically have not been set aside. Nonetheless, old-growth stands are directly observable and contain a wealth of information.

Land survey records are another common source of information (Bourdo, 1956). They essentially provide a coarse-scale, low-resolution sample of forests as they existed just prior to extensive settlement. Deriving quantitative information about disturbance frequency requires treating the survey lines as transects and converting length of line disturbed by a particular agent (e.g. fire) into a rate, making assumptions about the length of time such evidence remains discernible (Lorimer, 1977). Most surveyors recorded only major disturbances, such as fires and windthrows, thus limiting the types of disturbances about which inferences can be drawn from the survey notes.

Palynology is another source of information about disturbances. The occurrence of charcoal and sharp changes in pollen composition signify disturbances. However, disturbances like windthrow can leave little palynological evidence (David Foster, personal communication). Although palynology offers the longest time perspective of any of the techniques, high temporal resolution is costly to obtain. Most studies use cores extracted from lakes, ponds, or wetlands that sample pollen that was deposited from a fairly large area. These records are best for landscape-scale interpretations. In contrast, the less common approach of extracting cores from small forest hollows or vernal pools can give stand-level information (e.g. Schauffler, 1998; Foster et al., 1992).

2.2. Literature search

We searched the literature for studies of disturbance regimes in the northeastern quarter of North America. The region, which we will call the northeast, extends from Nova Scotia through New Brunswick, southern Quebec and southeastern Ontario, and from northern New England westward through New York, northeastern Pennsylvania, and the upper Lake States. The forest types emphasized lie within the temperate forest zone or transitional to the boreal zone, and include northern hardwoods and mixed-species forests in the Acadian region (Seymour, 1995). Studies from the boreal zone, and fire-dependent communities such as *Populus* spp. and various *Pinus* spp. common in the Lake States, were excluded. Of the many parameters that can be used to describe disturbance regimes, size

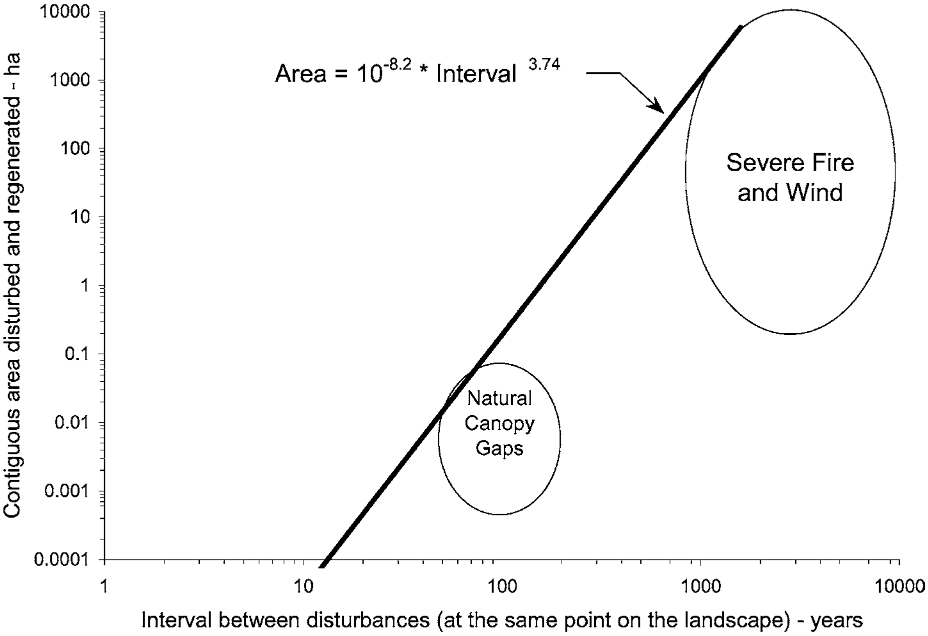

Fig. 1. Boundaries of natural variation in studies of disturbance in northeastern North American forests. The hand-fitted diagonal boundary line defines the upper limits on these disturbance parameters in combination, all of which fall in the lower right of the diagram. Upper limits of the area and return interval of severe fires and windstorms were truncated at 10^4 ha and 10^4 years, respectively.

and frequency are most directly analogous to choices made in formulating silvicultural systems (Seymour and Hunter, 1999); consequently, they are the focus of this paper. The cause or disturbance agent was also considered.

We tried to include only studies of natural disturbances, i.e. those minimally influenced by people, but some human influence was unavoidable (Cronon, 1983; Whitney, 1994). For example, although Lorimer (1977) was studying disturbances noted in early land survey records (ca. late 1700s and early 1800s) from northern Maine, he recognized that some of the recorded fires were associated with land clearing by settlers. Even old-growth forests may not be free of human influence. For example, Chokkalingam (1998) found that gap dynamics in old-growth Maine hardwood and mixed wood stands were partially related to beech bark disease, a disease complex introduced to North America from Europe around 1900 (Houston, 1975).

By including studies from a variety of locations and that used various research techniques, we have bracketed the range of frequencies and sizes associated with several disturbance types. We examined patterns in size and frequency by graphing the contiguous area disturbed over its corresponding return interval, following the model of Alverson et al. (1994). To examine natural limits of these parameters, we plotted ellipses that encompass both means and ranges from individual studies. In general, all data were included, except anomalous events thought to have human causes such as the very large (80,000 ha) fire reported by Lorimer (1977). For simplicity in illustration, we truncated the upper limits of area disturbed and return interval for stand-replacing disturbances at 10^4 ha and 10^4 years, respectively, slightly less than the reported maxima (Fig. 1).

3. Results and discussion

3.1. Patterns in frequencies and sizes of disturbances

Small canopy gaps are a common form of disturbance in several forest types, ranging from subalpine

Table 1
Summary of natural disturbance regimes in northeastern North America[a]

Type of disturbance	Disturbance agents	Range in patch size		Return interval (years)	Number of References
		Individual patches	Study means		
Natural canopy gaps	Senescence; wind; pathogens; insect herbivory	4–1135 m^2	24–126 m^2	50–200	12
Stand-replacing	Wind	0.2–3785 ha	14–93 ha	855–14300	4
Stand-replacing	Fire	2 to >80000 ha	2–200 ha	806–9000	8

[a] Includes forests in the Acadian region and Lake States dominated by northern hardwoods, red spruce, or eastern hemlock. Excludes boreal forests, forests dominated by balsam fir, and forests dominated by aspen, jack pine, white pine, or red pine in the upper Great Lakes region.

spruce-fir (Worrall and Harrington, 1988; Perkins et al., 1992; Battles and Fahey, 1996) to hardwood and hemlock-hardwood (Payette et al., 1990; Runkle, 1982, 1990; Cho and Boerner, 1991; Krasny and Whitmore, 1992; Tyrrell and Crow, 1994; Chokkalingam, 1998). Mean gap size is small, ranging from 24 to 126 m^2 (Table 1), with an overall mean of 53 m^2. Even the range of individual gap sizes peaks at a small size, 1135 m^2 (Battles and Fahey, 1996). The return interval of these gap disturbances is usually in the 50–200 year range (Table 1) in accordance with the estimate by Runkle (1982).

At the other end of the spectrum are catastrophic fires and windstorms (Table 1). Although often considered stand-replacing disturbances, they can create patches 2 ha or less (Seischab and Orwig, 1991; Whitney, 1986; Canham and Loucks, 1984; Marks et al., 1992; MacLean and Wein, 1977) or as large as 80,000 ha (Lorimer, 1977). The larger recorded sizes might not have included total destruction of the canopy throughout the burn, however. Many of the fires recorded in the land survey notes may have been influenced by early settlers (Lorimer, 1977), whereas fires recorded in the era prior to effective suppression (MacLean and Wein, 1977; Wein and Moore, 1977, 1979; Fahey and Reiners, 1981; Abrams and Nowacki, 1992) typically occurred in landscapes already significantly altered by human settlement. In contrast, sizes of patches derived from land survey records of catastrophic windstorms (Seischab and Orwig, 1991; Whitney, 1986; Canham and Loucks, 1984; Marks et al., 1992) probably were not significantly influenced by people.

3.2. Temporal and spatial boundaries of northeastern disturbances

When disturbed areas are plotted over their corresponding return intervals on a log–log scale, we found that all studies could be accurately depicted by ellipses that encompass their ranges in both space and time (Fig. 1). Data fell into two distinct clusters, corresponding to gap-phase and stand-replacing agents. Clearly, return intervals and areas disturbed are not independent, as is sometimes assumed. Gaps were small and frequent, as expected, whereas catastrophic fires and blowdowns were rare and highly variable in size. This pattern is distinctly different from the disturbance regime of the nearby boreal region where forests cycle more frequently at largescales (Cogbill, 1985).

All data appear to be bounded by a line tangent to the two ellipses (Fig. 1); the equation hand-fitted to this line indicates that the area disturbed increases exponentially as return intervals lengthen. Natural disturbances in the northeastern forest types included here fall below and to the right of the line. Combinations of space and time above and to the left of the line resulting from natural events evidently are undocumented, and thus, we conclude, outside the boundary of natural variation in this region.

Notably absent are moderate disturbance events with several-century return intervals at a medium (1–100 ha) scale. This could be an artifact of the methodology used in the studies cited herein, or a real void. The argument for a methodological explanation centers on two points. First, land surveyors may have recorded only larger, more dramatic disturbances; unfortunately, the minimum size recorded is unknown and may have varied by survey. Second, gap dynamics studies usually focus on small gaps (<1.0 ha) and, because of the techniques used, may involve small plots. Furthermore, areas that have had moderate to large disturbances may be avoided for gap dynamics studies.

The argument in support of the hypothesis that moderate disturbances were truly absent from the presettlement forest depends on assumptions about disturbance agents and how they operate in these types of forests. All known biotic disturbance agents have fairly narrow host ranges, and are typically species-specific, often attacking only old individuals in the population. Given the high levels of species and age diversity in the presettlement forest, it is easy to see how such agents would almost always produce gap dynamics at small scales. The only other possible sources of mid-size disturbances are abiotic agents (fire, wind). Interestingly, such disturbances are common in the boreal forest of eastern Canada (just to the north of the region discussed in this paper) where fire and spruce budworm (*Choristoneura fumiferana* Clem.) outbreaks cause stand-replacement over large areas every 100–250 years (Cogbill, 1985). Species which dominate this boreal region (e.g. *Betula papyrifera* Marsh., *Populus tremuloides* Michx., *Pinus banksiana* Lamb., *Picea mariana* (Mill.) B.S.P., *P. glauca* (Moench) Voss, and *Abies balsamea* (L.) Mill.) are also present and locally abundant in the northeast. But these boreal species rarely form extensive monocultures in the northeast, except after rare large-scale, stand-replacing disturbances to which they are well adapted. Where these species dominate stand composition in the northeast (e.g. extensive *Abies balsamea* "flats" in northern Maine; *Pinus banksiana* sand plains in New Brunswick), the resulting disturbance dynamics are more akin to their northerly counterparts than to the generally stable matrix of northern hardwoods (dominated by *Acer* spp., *Fagus grandifolia* Ehrh., and *Betula alleghaniensis* Britton) and long-lived conifers (*Picea rubens* Sarg., *Thuja occidentalis* L., and *Tsuga canadensis* (L.) Carr.) that typifies the Acadian region.

For the sake of simplicity in application, we treat the apparent void as a methodological artifact, thus assuming that disturbances of intermediate size and frequency are part of the natural disturbance regime but have simply gone unrecorded or have not been studied. This assumption is consistent with our hypothesis that a line tangential to the two ellipses establishes the boundary of natural variation in disturbance regimes in this region (Fig. 1).

3.3. Silvicultural implications

Silviculturists in the northeast seeking to emulate natural disturbance regimes have historically relied on general ecological principles and intuition, without really knowing how closely their management resembled natural processes. The existence of the boundary condition in Fig. 1 suggests a more rigorous approach to the process of formulating ecologically based silvicultural systems. Here, we can use the fact that return intervals and contiguous areas disturbed (i.e. the axes in Fig. 1) both have direct silvicultural analogues (Seymour and Hunter, 1999). In ecosystems where stand-replacing events dominate, the range of return intervals is directly comparable to the rotations of single-cohort stands, and their spatial extent would essentially define stand sizes. Where partial disturbances are the rule, return intervals are related to cutting-cycles for managed multi-cohort stands, and gap sizes would be similar to the small, within-stand patches where regeneration is recruited under single-tree or group selection silviculture. Any silvicultural treatment or system can thus be displayed identically to the disturbance data as in Fig. 1, and compared to the boundary condition as a natural benchmark.

We can best illustrate this approach with a simple example. During the past two decades, some industrial landowners in northern New England and the Maritimes have begun to manage modest areas under production forestry, typically by growing plantations of various *Picea* spp. on rotations of ca. 50 years (Seymour, 1995). Due to "green-up" adjacency requirements and government regulations, clearcuts (and thus plantations) average about 20 ha. When plotted on the disturbance spectrum, we see immediately that such a plantation falls well outside the boundary of natural disturbances (Fig. 2). One way to quantify its departure from the natural forest is to calculate the lower limit on the natural return interval for the same-sized area on the landscape. Substituting 20 ha into the boundary equation and solving for the interval yields a value of 347 years. The ratio of the managed

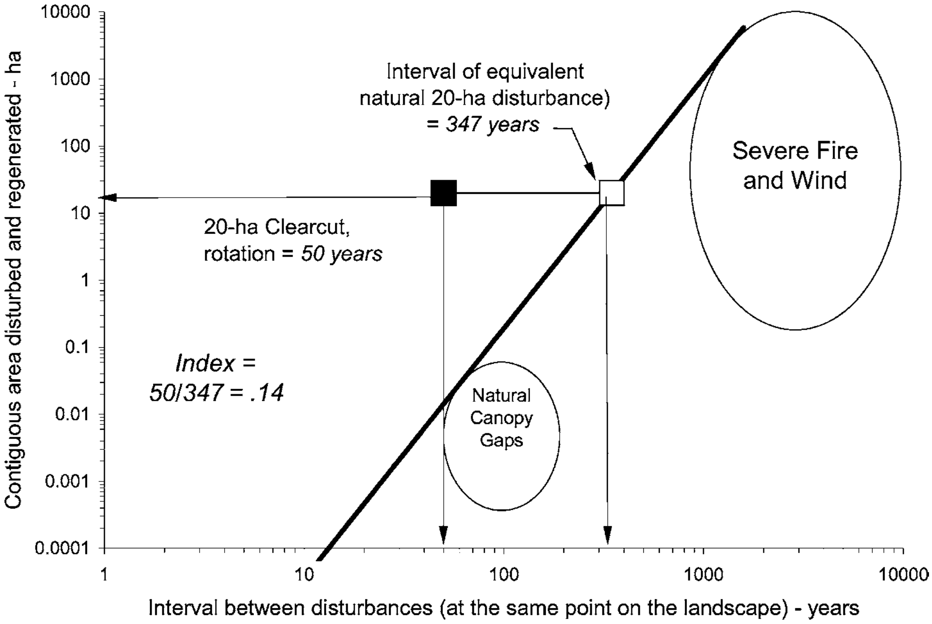

Fig. 2. Natural disturbance comparability index illustrated for a 20 ha forest plantation managed on a 50-year rotation. The index is defined as the actual rotation of the managed stand expressed as a proportion of the natural return interval of the same size patch.

rotation and its natural disturbance-frequency counterpart ($50/347 = 0.14$) can be thought of as a metric that quantifies how closely the system emulates a natural disturbance regime of comparable scale or time, where a value of 1.0 represents exact replication of the boundary condition. We term this the natural disturbance comparability index. In this example, an industrial landowner who plans to convert an entire landscape to production forestry would, in effect, be harvesting and regenerating a given place about seven times more frequently than would natural events.

This approach for benchmarking forest practices can readily be extended to other silvicultural systems; typical examples are shown in Fig. 3. Here, we have shortened the axes relative to the complete range of natural variation on the grounds that feasible silvicultural systems would fall well below the limits of 100 ha and 1000 years. Reference lines are also added corresponding to natural disturbance comparability indices of 0.05, 0.10, 0.25, and 0.50. To evaluate multi-aged (selection) systems that regenerate different patches within the stand at each harvest, we must first convert the cutting cycle (time between harvest entries) into an effective "rotation", or the time it would take to cycle through and regenerate the entire stand. Here, we must estimate the proportion of the total stand area that is regenerated at each entry, and divide this value into the cutting cycle (Nyland, 1996, p. 230). For example, a selection system that opened 10% of the stand for regeneration at each entry on a 15-year cutting cycle would effectively equate to a 150-year rotation for a given spot in this stand.

Typical multi-aged systems fall within natural limits. For example, a group selection system designed to perpetuate some species with low shade tolerance using openings ranging between 0.04 and 0.10 ha, repeated on effective rotations of 80–120 years, lies near the upper limit of natural processes. Single-tree selection systems with return intervals of 100–150 years with opening sizes of 0.001–0.01 ha, are comparable to natural tree-fall gaps.

Single-cohort systems based on natural regeneration without reserve trees (i.e. trees left for structural enhancement after the overwood removal) all tend to fall somewhat outside the natural boundary unless the rotation is very long. The natural return interval for a 2 ha patch, the minimum size area that would be

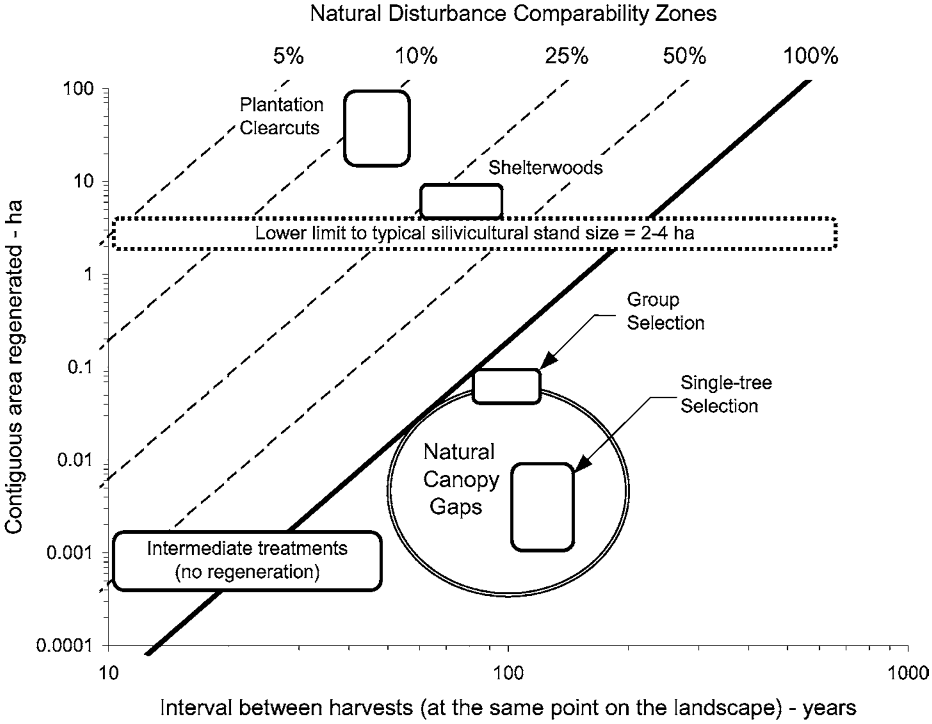

Fig. 3. Natural disturbance comparability zones (defined as in Fig. 2) displayed against typical northeastern silvicultural systems. Note that the upper limit of natural canopy gaps (ca. 0.1 ha) is at least one order of magnitude smaller than the smallest stand size (2 ha).

considered a stand in this region, is 188 years (from Fig. 1)—considerably longer than what most foresters would consider to be a long rotation in the region (Seymour, 1995). Natural disturbances recurring at a 100-year interval did not exceed 0.19 ha, at least one order of magnitude smaller than the minimum size stand. Thus, the common practice of making small patch clearcuttings of 1–3 ha has surprisingly little ecological justification when judged against this benchmark, unless some diversity in age or vertical structure is left within the patch.

The degree to which single-cohort systems can be mitigated or "softened" by varying degrees of structural retention at the final harvest (Franklin et al., 1997) is in the forefront of silvicultural research and debate in the northeast, as it is elsewhere (Carey et al., 1999). If the goal is to emulate most northeastern natural disturbance regimes faithfully, then the majority of the landscape must be under some type of continuous-canopy, multi-aged silviculture that maintains ecologically mature structures at a finely patterned scale. Two-cohort stand structures resulting from variable retention (Franklin et al., 1997) practices represent a fairly wide band on the continuum between simplified single-cohort and complex multi-cohort structures. Thus, ecological robustness of two-cohort systems appears to be directly related to the magnitude of retention of both living and dead trees as biological legacies. Leaving a few scattered reserve trees (under 2–3 m^2 of basal area per ha) could offer only limited benefits, whereas 10–15 m^2 per ha of reserves might be impossible to distinguish from a true multi-cohort structure.

Intermediate treatments such as thinnings, which by definition do not lead to regeneration, form small canopy openings that are quickly reoccupied by vigorous residual trees. Any gaps below ca. 0.002 ha (equal to the crown area of an average tree in a stand with a density of 500 trees per ha) usually do not result in the initiation of new cohorts and thus should not be evaluated using this framework. By ignoring such small gaps, we remain consistent with some studies cited above that attempted to eliminate such events from the disturbance chronology by adjusting the threshold growth responses accordingly (e.g. Frelich and Lorimer, 1991; Nowacki and Abrams, 1997).

What if our assumptions when drawing the line bounding the ellipses are wrong? Two types of possible errors illustrate the consequences. First, what if the ellipses themselves changed, perhaps due to new studies or application to a new region or different set of forest types? In these situations the solution is to construct a new boundary line and use it to calculate new comparability index values. This is a simple solution but obviously would affect the estimation of where a particular practice fell relative to natural conditions. Second, what if the void to the right of the boundary line is real, rather than an artifact of research bias as the tangential line assumes? In such a situation, the boundary line is no longer appropriate because some areas to the right of the line or on the line itself, such as the point used in the illustration of how to calculate the index, would not be within the range of natural conditions. In this case, the index could still be used, but the natural point of comparison in the graph would be the nearest point on the ellipse that corresponded to that size opening. Thus, both types of errors result in changes to absolute values, but the approach of comparing a particular practice to the natural disturbance regime remains valid.

3.4. Landscape considerations

Evidence reviewed above supports the conventional wisdom that disturbances were frequent throughout the presettlement landscape of the northeast. There is much less consensus, however, regarding the finding that the effects of common disturbances were quite dispersed, and occurred at scales at least one order of magnitude below that of the smallest stands that are presently delineated by foresters for silvicultural purposes. Extensive, single-cohort stands were uncommon in the presettlement forest of the northeast; for example, Lorimer (1977) estimated that stands less than 75 years old occupied 16% of the landscape in northern Maine ca. 1820. Widespread application of single-cohort silviculture on rotations of under 100 years thus creates a landscape that has no natural pre-cedent for the types of forests we reviewed. Management that deliberately produces such stands thus cannot claim to be emulating natural disturbances, as in the common industrial situation where multiple, short rotations are planned, or where such stands dominate the landscape.

Furthermore, basing regeneration rates on natural disturbance frequencies alone (e.g. 1% per year), without accounting for the scale of the disturbance, greatly oversimplifies the natural pattern where landscape-level, stand-replacing disturbances are much rarer than small, within-stand patches. If we ignore this relationship between space and time, then management activities might have negative consequences on landscape structure. Consider the example of a landowner who limits stand sizes between 4 and 20 ha and manages everything in single-cohort stands on 100-year rotations. Although this system seems benign relative to the more aggressive industrial plantation example in Fig. 2, it would effectively eliminate the small-scale, within-stand gap processes that dominated the natural forests in this region. The long-term consequence is an unnatural landscape that becomes homogenized in both time and space. This example raises questions about strategies that are designed to address biodiversity issues strictly at the landscape-scale using a continually shifting mosaic of variable-size, single-cohort stands managed on ecologically short rotations. Such a landscape will not contain a natural diversity of conditions unless silvicultural systems make substantial provisions for retaining within-stand structure during the regeneration harvest. Once single-cohort stands occupy over ca. 15–25% of the landscape, every stand that is converted or maintained in a single-cohort structure contributes toward an increasingly artificial landscape pattern.

The stand-level benchmarking approach (Fig. 3) can readily be extended to evaluate a forest structure at the landscape level if the age structures of stands are known and the management plan is site specific. Disturbance comparability indices could be calculated for each stand, and a weighted average could be determined for various sized landscape units. Using a triad model for landscape allocation (Seymour and Hunter, 1999), a network of ecological reserves (Norton, 1999) could then be designed to counterbalance limited areas allotted to production forestry. To enhance ecological robustness, the production forests and reserves would be embedded within a diverse matrix of stands managed according to the principles outlined here.

4. Recommendations

The discussion above should help to clarify for northeastern practitioners what sorts of silvicultural systems to favor in order to emulate natural disturbance regimes, and by implication, which to avoid. The following specific recommendations should help to clarify our working hypotheses.

1. For the purpose of silvicultural prescription, think of northeastern landscapes in terms of relatively large stands with substantial within-stand diversity in age, not as many small, uniform single-cohort stands. Match stand boundaries with large-scale, enduring physiographic and edaphic features; do not designate stands simply on the basis of the present age structure and species composition if these are substantially altered from the presettlement condition, as is common. Doing the latter could reinforce a landscape pattern that could become increasingly artificial and self-perpetuating. Use a within-stand, gap-based paradigm to manage the regeneration process (Coates and Burton, 1997).
2. Regenerate new cohorts at rates $= 0.7-1.3\%$ per year. This will produce average canopy residence times of 75–150 years (not including any early suppression period), which were apparently typical of presettlement forests (e.g. Frelich and Lorimer, 1991; Dahir and Lorimer, 1996). Set an operational cutting cycle, then multiply by the chosen regeneration rate to determine how much total gap area to create in each harvest.
3. When starting with stands exhibiting mid- to late-successional structures and reasonable species diversity, create a range of gap sizes ranging from the crown area of a single large tree up to a maximum of ca. 0.2 ha. Above 0.1 ha, gaps will admit enough light to ensure some representation of commercially important, shade-intolerant species, if that is an objective. Avoid harvesting all the largest or most valuable trees in a single entry, as this tends to create a network of large, interconnected patches (Lorimer, 1989; Lorimer and Frelich, 1994). Rather, try to form distinct gaps around senescent individuals or clumps using vigor and risk classifications (Seydack, 1995).
4. In more uniform stands that lack much structural or species diversity, larger patches created by more intense harvests may be necessary to avoid short-term financial loss. In these cases, long-term restoration, rather than maintenance, is the goal. Take advantage of every opportunity to conserve legacy trees and advance regeneration of longer-lived species. This will mimic the natural successional pattern that would eventually restore the later-successional condition if no harvesting occurred, and will expand future treatment options.
5. Finally, the practice of multi-aged silviculture does not risk loss of early-successional communities that depend on infrequent catastrophic disturbances. Most such disturbances will occur regardless of human activity, so there is no justification for emulating them. In addition, landscapes in regions such as the Lake States have substantial areas of forest communities that naturally depend on stand-replacing disturbances (e.g. pine forests) that are intermingled among the more stable communities (e.g. northern hardwoods-hemlock) discussed in this paper. Using a natural disturbance paradigm to manage both types of communities would result in a naturally diverse landscape comparable to the presettlement era. Given that people cannot prevent most abiotic disturbances, management should strive to complement the natural background levels, not duplicate them; otherwise, the overall disturbance rate will be unnaturally high. The balanced response to catastrophic events is to salvage economic losses when they occur, with due attention to biological legacies such as surviving living trees, standing snags, and coarse woody material.

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Scale and frequency of natural disturbances in the northeastern US: implications for early successional forest habitats and regional age distributions

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Abstract

Recent declines in the amount of habitat suitable for early successional wildlife in the northeastern US have prompted public land managers to consider establishing minimum levels of young forest, based on the natural range of variation, in order to maintain viable populations of these species. In this paper, we review evidence on the frequency, severity, and scale of natural disturbances in four major forest regions of the northeastern US. Using six independent lines of evidence, we examined the influence of natural disturbances in presettlement and modern times. In situations where estimates of annual disturbance rates were available, we estimated the regional age distribution of forest stands based on the assumption of random spatial pattern of disturbance. Available evidence suggests a gradient of generally decreasing disturbance frequency from coastal regions to the interior uplands and mountains. The proportion of the presettlement landscape in seedling–sapling forest habitat (1–15 years old) ranged from 1 to 3% in northern hardwood forests (*Fagus–Betula–Acer–Tsuga*) of the interior uplands to possibly >10% in coastal pine–oak (*Pinus–Quercus*) barrens. Within a region, variability in the amount of young forest is not well known, but upper slopes and ridges generally had the highest disturbance frequency and severity. Comparison of line transect data of the presettlement land surveys with modern plot surveys suggests that present-day amounts of young forests in northern hardwood and spruce–hardwood forests in some regions may be several times higher than in presettlement times. In coastal oak forests and pine–oak barrens, the amount of young forests and open woodlands may be less because of reduced fire frequency.

Keywords: Blowdowns; Early successional habitat; Fire regimes; Insect epidemics; Natural disturbances; Old-growth forests; Presettlement forests; Windthrow

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1. Introduction

Historical land-use changes in the eastern US, as in many other parts of the world, have caused extreme swings in forest habitat conditions. In the late 19th and early 20th centuries, much of the northeast was dominated by young forest stands as a legacy of extensive logging, land clearing, fuelwood utilization, repeated fires on cutover land, and widespread farm abandonment. For example, >75% of the forest in central Massachusetts was less than 30 years old in 1885 (Foster et al., 1998). Government surveys in 1908 revealed that 56% of the forest land in 12 eastern states was classified as "cutover land" (Whitney,

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1994, p. 192). Subsequent declines in the rate of forest utilization allowed many forests in eastern US to recover (McKibben, 1995). As these regenerating forests have matured, a number of animal populations also recovered that were uncommon a hundred years ago (Kendeigh, 1946; Whitney, 1994).

Concurrent with the maturation of second-growth stands, the abundance of early successional habitats has declined in much of the northeast. Seedling–sapling stands currently represent 4–18% of forests in the region (Trani et al., 2001). Yet the effects of the reduction in habitat on wildlife species that require young forests have received relatively little attention. As Askins (2001) has pointed out, shrublands, clearcuts, and thickets are ''unpopular habitats'' among the general public. There is also a widespread notion, even among some conservation groups, that wildlife species dependent on early successional habitats are ''weedy generalists'' that thrive in human-dominated habitats and therefore require no special conservation measures. However, substantial population declines have occurred among early successional obligates. For example, of the 126 neotropical migrant bird species in the northeast, 74 require disturbance-generated habitats or young forests, and these species are scarce or absent in mature and old-growth stands (Smith et al., 1993). Thompson and DeGraaf (2001) pointed out that no breeding bird species are dependent upon uneven-aged stands, whereas many species require even-aged habitats. Among the disturbance-dependent bird species throughout eastern North America, Hunter et al. (2001) predicted that if early successional habitats continue to decline, many species will be extirpated from portions of the eastern US and others risk extinction. Fourteen of these species are federally listed as endangered or threatened, and 18 others are on a national watch list (Hunter et al., 2001).

Broader-level landscape considerations also are important in maintaining suitable conditions. Because early successional species utilize habitats that persist only for a short time, continual turnover of stands somewhere on the landscape is necessary. In addition to the total amount of young-forest habitat, the scale of disturbance also is an important consideration. For example, prairie warblers (*Dendroica discolor*) and yellow-breasted chats (*Icteria virens*) will utilize moderate or large patches of habitat, but apparently avoid small openings (Annand and Thompson, 1997).

The wide historical swings in habitat conditions have prompted ecologists to consider appropriate historical benchmark conditions in the amount of early and late-successional habitats. Although natural disturbance regimes in the post-glacial era have fluctuated (e.g., Anderson et al., 1986), managing landscapes within a natural range of variability may help sustain population viability by maintaining landscape characteristics to which the regional plants and animals have become adapted (Seymour and Hunter, 1999; Thompson and DeGraaf, 2001). An understanding of natural disturbance regimes is also needed to fulfill the broader goals of ecosystem management on public lands (Thomas, 1996). Therefore, the purpose of our paper is to review evidence on the frequency, severity, and scale of natural disturbances in the major forest regions of the northeastern US. In regions with sufficient evidence, we provide estimates of the relative amount of young and old forest in presettlement times, but the temporal scope of the paper includes evidence spanning a broader period of several thousand years to characterize the range of variability.

A distinction is made in this paper between natural disturbances (caused by lightning, windstorms, insect outbreaks, etc.) and anthropogenic disturbances caused by Native Americans and European settlers, although these causes are often not distinguishable using historical or scientific evidence. Some disturbances in modern times also are discussed in situations where these events clarify the potential scope and impacts of natural disturbance, even though they may be operating on landscapes considerably affected by humans. However, analysis of recent anthropogenic disturbances such as logging, land clearing, and invasion of exotic species lies beyond the scope of this paper. Some types of anthropogenic disturbance (e.g., logging, grazing, prescribed burning) may be useful in conserving species at risk and are covered in other papers in this issue (cf. Foster and Motzkin, 2003; DeGraaf and Yamasaki, 2003; Litvaitis, 2003).

2. Investigative methods and limitations

2.1. Methods of analyzing disturbance frequency

There are six principal methods or sources that can be used to investigate natural disturbance regimes.

These include analysis of sedimentary pollen and charcoal (e.g., Patterson and Backman, 1988; Clark and Royall, 1995a), presettlement land survey records (Siccama, 1971; Cogbill, 2000), early descriptions by travelers, naturalists, and foresters (Day, 1953; Whitney, 1994), reconstructions of disturbance history in old-growth stands (Lorimer and Frelich, 1989; Chokkalingam, 1998), modern records and aerial photos (Fahey and Reiners, 1981; Jenkins, 1995), and computer modeling (Frelich and Lorimer, 1991a; Boose et al., 2001). Each method contributes unique information, but also has limitations. For example, paleoecological evidence from lakes and small forest hollows provides the only long-term record of fire activity, often extending over thousands of years, but it is often difficult to distinguish individual fires and usually not feasible to determine size or spatial extent. On the other hand, it is possible to map the boundaries of large fires in reasonable detail using presettlement land survey records, but the sampling period for this method spans only a few decades. There also are site and geographic limitations for most methods, as well as interpretive ambiguities and possible methodological biases. For example, remnant old-growth stands are rare and often restricted to steep or inaccessible topography that may not be representative of the larger landscape. Fortunately, the strengths of one method can often be used to compensate for the limitations of another, so that a synthesis of multiple lines of evidence often provides an excellent overview of disturbance regimes in a region. Nevertheless, there are still some large gaps in the record. In the best cases, there may be sufficient systematic evidence to estimate the proportion of the landscape in young forest habitat, but usually only for a short period prior to settlement or for a specific type of disturbance.

2.2. Estimating forest age structure on the presettlement landscape

For each forest type, we harmonized the evidence obtained from various methods to provide an integrated summary on disturbance regimes. In situations where quantitative information was available, we have provided preliminary pooled estimates of mean annual disturbance rates for the major categories of disturbance (primarily wind and fire) within a forest type. The composite figure was then used to estimate the proportion of the presettlement landscape in each of several forest age classes. These estimates should be considered region-wide averages, as the evidence in most cases is not sufficient to quantify local variation.

Estimates of regional forest age distribution based on annual disturbance rates are influenced by assumptions about the spatial pattern of disturbances. If stands are heavily disturbed only after they reach the rotation age (as, for example, in even-aged forest management), the resulting landscape age distribution is uniform, with an equal proportion of the landscape in each age class. However, if the pattern of disturbance is random (all stands have the same probability of disturbance in a given year regardless of age), even young stands can be disturbed again, and some old stands can escape disturbance for long periods by chance. This leads to a landscape with fewer young stands and more old stands than under the uniform assumption. The rotation period in these situations does not correspond to the maximum stand age, but rather the mean age. With the random distribution of disturbances, the age distribution of a forest landscape approaches a negative exponential curve, and 37% of the stands are actually older than the rotation period (Van Wagner, 1978).

Empirical age distributions from the boreal forest and conifer forests of western North America—landscapes dominated by stand-replacing fires—generally do follow a descending curve (Yarie, 1981; Johnson et al., 1995; Reed et al., 1998). This evidence suggests at least a quasi-random pattern of burned areas, and overlapping burns and reburns have been reported (Brown and Davis, 1973). Landscape age models have not been extensively investigated in temperate hardwood or mixed conifer–hardwood forests of the eastern US. But it is clear from the fire history of Maine, for example, that overlapping burns and reburns have been very common (Coolidge, 1963). Therefore, it seems reasonable to apply the negative exponential model as a first approximation to spruce–hardwood forests of northern New England.

Young forest stands are much less susceptible to windthrow than mature and old stands (Foster, 1988b). But there is little reason to think that wind will only blow down stands near the end of the rotation period, especially when the rotation period is 1000 years or more. In our initial empirical trials, an assumption of zero probability of windthrow for stands younger than 45 years had negligible effects on the estimated amount of young forest. In the final analyses (Tables 1–3), we used a simple negative exponential model in which the annual disturbance rate p is the pooled disturbance rate for both catastrophic windthrows and stand-replacing fires. Pooling these rates in the negative exponential model allows random (and overlapping) spatial patterns

Table 1

Expected percentage of regional landscape occupied by different age classes of northern hardwood forest under various rotation periods and assumptions about disturbance spatial patterns[a]

Age class	500-year rotation (fire 1000 years, wind 1000 years)		1364-year rotation (fire 3000 years, wind 2500 years)	
	Uniform	Random	Uniform	Random
Seedling–sapling (1–15 years)	3.0	3.0	1.1	1.1
Small pole (15–30 years)	3.0	2.9	1.1	1.1
Large pole (30–60 years)	6.0	5.5	2.2	2.1
Mature even-aged (60–100 years)	8.0	6.8	2.9	2.8
Old even-aged (100–150 years)	10.0	7.8	3.7	3.4
Transitional uneven (150–300 years)	30.0	19.2	11.0	9.3
Old uneven-aged (300+ years)	40.0	54.9	78.0	80.2

[a] Uniform refers to spatially non-overlapping disturbances that occur only when the stand has reached the rotation age. Random signifies a random spatial pattern in which all age classes have an equal probability of disturbance (see text).

Table 2

Expected percentage of regional landscape occupied by different age classes of spruce–northern hardwood forest under various rotation periods and assumptions about disturbance spatial patterns (see text)

Age class	230-year rotation (fire 385 years, wind 575 years)		335-year rotation (fire 800 years, wind 575 years)		388-year rotation (fire 1200 years, wind 575 years)	
	Uniform	Random	Uniform	Random	Uniform	Random
Seedling–sapling (1–15 years)	6.5	6.3	4.5	4.4	3.9	3.8
Small pole (15–30 years)	6.5	5.9	4.5	4.2	3.9	3.6
Large pole (30–60 years)	13.0	10.7	9.0	7.8	7.8	6.9
Mature even-aged (60–100 years)	17.4	12.3	12.0	9.4	10.3	8.4
Old even-aged (100–150 years)	21.8	12.6	14.9	10.3	12.9	9.3
Transitional uneven (150–200 years)	21.8	10.2	14.9	8.9	12.9	8.2
Old uneven-aged (200+ years)	13.0	42.0	40.2	55.0	48.3	59.8

Table 3

Expected percentage of regional landscape occupied by different age classes of spruce–northern hardwood forest under various rotation periods and assumptions about disturbance spatial patterns (continued)[a]

Age class	545-year rotation (fire 1200 years, wind 1000 years)		606-year rotation, mixed uplands (fire 800 years, wind 2500 years)		210-year rotation, swamps and flats (fire 800 years, wind 285 years)	
	Uniform	Random	Uniform	Random	Uniform	Random
Seedling–sapling (1–15 years)	2.8	2.7	2.5	2.4	7.1	6.9
Small pole (15–30 years)	2.8	2.6	2.5	2.4	7.1	6.4
Large pole (30–60 years)	5.6	5.1	5.0	4.6	14.2	11.6
Mature even-aged (60–100 years)	7.3	6.3	6.6	5.8	19.0	13.0
Old even-aged (100–150 years)	9.2	7.3	8.2	6.7	23.8	13.2
Transitional uneven (150–200 years)	9.2	6.7	8.2	6.2	23.8	10.4
Old uneven-aged (200+ years)	63.1	69.3	67.0	71.9	4.8	38.5

[a] The last two columns provide preliminary estimates for two broad habitat types: mixed spruce–hardwood forest on uplands and conifer-dominated sites in swamps and stony flats.

for both types of disturbance; stands blown down by wind can be burned at any subsequent time during the rotation period, and areas burned can later be windthrown. Following the approach of Van Wagner (1978), the cumulative proportion of the landscape in age classes up to age x, given an annual disturbance rate p, is calculated as

$$\sum f(x) = 1 - e^{-px}$$

In our analyses, the transition period from even-aged to uneven-aged stands is assumed to begin about 150 years after catastrophic disturbance, based on average life expectancy data for major tree species in the region (Moesswilde, 1995; Chokkalingam, 1998; Tyrrell et al., 1998).

3. Pine–oak barrens

Pine–oak barrens in the northeast are commonly dominated by pitch pine (*Pinus rigida*) with varying amounts of canopy oaks (e.g., *Quercus alba*), scrub oaks (e.g., *Q. ilicifolia*), and ericaceous shrubs (e.g., *Gaylussacia* and *Vaccinium* spp.). They are commonly associated with the sandy soils of the Coastal Plain (Lull, 1968) or similar soils inland; they may also be found on rocky ridges and hilltops. Although pine barrens typically occupy droughty, nutrient-poor sites, average annual precipitation (102–122 cm) is similar to that of other community types in the region (Lull, 1968). Temperatures are moderate with a typical frost-free growing season of 180–210 days (Lull, 1968).

Proximity of pine–oak barrens to the coast facilitated early exploitation by European settlers, so detailed presettlement records of species composition and disturbance patterns are lacking. However, there are some general descriptions, such as that of a surveyor in 1687, who described the pine barrens of southeastern New Jersey as ‘‘a great tract of barren lands consisting of pine land and sand’’ (Wacker, 1979). Paleoecological data confirm that pines and oaks dominated for many centuries prior to European settlement (Florer, 1972; Watts, 1979; Fuller et al., 1998; Parshall et al., 2003), although in some areas pitch pine replaced white pine (*Pinus strobus*) and oaks as a result of logging, burning, and plowing by European settlers (Patterson and Backman, 1988).

Of the tree-dominated communities in the northeast, the pine–oak barrens are often considered the most fire prone. Sediment cores often contain abundant presettlement charcoal, sometimes in quantities comparable to post-settlement times (Patterson and Backman, 1988; Fuller et al., 1998; Copenheaver et al., 2000; Parshall et al., 2003). Recorded fires in recent centuries have been frequent and often large. In the New Jersey pine barrens, Forman and Boerner (1981) reported that a few fires had burned >40,000 ha each during the last 150 years, with large fires more commonly ranging from 8000–16,000 ha. Individual fire years in which >50,000 ha burned (~10% of the total area) occurred about once every 20 years. Local variation in fire regimes was noted due to the distribution of natural fire breaks (e.g., swamps), community composition, and weather patterns. Pine–oak barrens in other portions of the northeast may have experienced smaller fires, partly because of their smaller total contiguous area. Even so, individual fires of more than 1000 ha have been reported (Schweitzer and Rawinski, 1988). Humans, both Native Americans and European settlers, were the primary ignition sources, as lightning fires are uncommon in the northeastern barrens (Lutz, 1934; Forman and Boerner, 1981).

Quantitative estimates of natural fire frequencies for northeastern pine–oak barrens are not available, and the only estimates are based on relatively recent historical records. Using different approaches, both Lutz (1934) and Forman and Boerner (1981) estimated an average return interval (for mostly intense fires that open the canopy) of approximately 20 years for the late 1800s and early 1900s. The average return interval had increased to about 65 years by the mid-1970s (Forman and Boerner, 1981). The return interval was estimated to only be 8 years for the ‘‘Plains’’, an area within the New Jersey pine barrens characterized by short (<4 m), sprout-origin pines and oaks (Lutz, 1934). Although pine–oak barrens often succeed to a mature forest of oaks in the absence of fire (Little, 1979), harsh site conditions can sometimes keep barrens in an early successional state for prolonged periods. Winne (1997) reported that an area of barrens in eastern Maine has remained open since created by a major fire 1700 years ago.

Of the four major forest types and geographic regions in the northeast, the pine–oak barrens probably had the highest incidence of severe disturbance, mostly from fires and secondarily from periodic hurricanes. As a result, pine–oak barrens may have had the highest proportion of land area in recently disturbed habitat and young forest suitable for early successional wildlife. The lack of quantitative evidence on stand-replacing fires prior to European settlement precludes an estimate of the expected proportion of the landscape in young forest habitat under natural conditions. Therefore, we simply note that return intervals spanning a broad range of 40–150 years for severe fires would yield estimates of 10–31% of the landscape in the seedling–sapling stage (stand ages of 1–15 years) under the assumption of a random spatial pattern of ignitions.

4. Eastern oak forests

Forests dominated by oaks and hickories (*Carya* spp.) are dispersed across eight distinct physiographic and soil regions in the northeast, from unconsolidated sandy soils of the coastal plain to the predominantly stony loam and sandy loam soils on steep slopes of the Allegheny Mountains. However, the boundaries of the region correspond well with climatic variables, especially a mean frost-free period of 150–180 days (Lull, 1968).

Oaks of all species made up 35–75% of the witness trees in presettlement land surveys, usually accompanied by hickories, chestnut (*Castanea dentata*), and pines (e.g., Russell, 1981; Abrams and Ruffner, 1995; Black and Abrams, 2001). Paleoecological studies indicate oak dominance has been remarkably stable over the past 9000 years in most areas (Watts, 1979; Maenza-Gmelch, 1997). In recent decades, however, oak forests have developed dense understories of shade-tolerant species, including maples (*Acer* spp.) and beech (*Fagus grandifolia*) (Lorimer, 1984; Abrams, 1992). These shade-tolerant species may be capable of displacing oaks on some mesic and dry-mesic sites (Abrams and Downs, 1990; Abrams and Nowacki, 1992).

4.1. Windstorms

Atlantic hurricanes are one of the principal disturbance agents in parts of the oak region. Although hurricanes begin to lose energy as they move inland or move across cold ocean currents (e.g., Gulf of Maine), some hurricanes move northward across Long Island and into interior New England, causing severe damage even 100 km from the coast. Damage can range from light crown damage and scattered treefalls to nearly complete blowdown of forest stands. Boose et al. (2001) documented 67 New England hurricanes that occurred from 1620 to 1997, averaging one storm every 6 years. At the Harvard Forest in central Massachusetts, three hurricanes reached or exceeded F2 intensity, sufficient to cause blowdown of entire stands, with a mean recurrence of 150 years. Using historical reports on damage to calibrate a hurricane simulation model, Boose et al. (2001) provided evidence of a geographical gradient in mean recurrence of hurricane damage, ranging from 85 years for F2 damage in southern coastal New England to more than 380 years for F2 damage in northern Maine (Fig. 1).

Studies at a local scale have revealed a number of factors that may explain variation in the degree of disturbance caused by hurricanes. In central Massachusetts, stands on level ground or on windward slopes (S, SE, E) were exposed to the brunt of the storm and had the highest levels of damage after the hurricane of

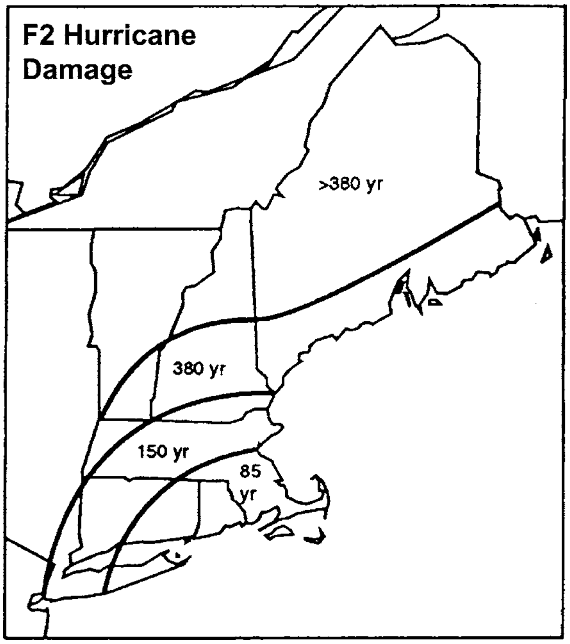

Fig. 1. Zones of hurricane frequency in New England, showing mean recurrence intervals between consecutive F2 hurricanes capable of causing extensive blowdown of forest stands (from Boose et al., 2001; reproduced with permission of the Ecological Society of America).

1938 (F2+ intensity). The storm had less effect on leeward slopes (W, NW, N). Damage varied by species composition and tree size. Uprooting and trunk breakage were greater in conifer stands than in hardwoods, but were generally high in mature stands >15 m tall. Mature conifer stands averaged >75% mortality of canopy trees on exposed sites, and often had >50% tree mortality on protected sites. Mature hardwood stands, in contrast, had 50–75% tree mortality on exposed sites, but <25% tree mortality on protected sites (Fig. 2). Young stands (only 10 m tall) had 25–75% damage depending on species composition and slope exposure (Foster and Boose, 1992).

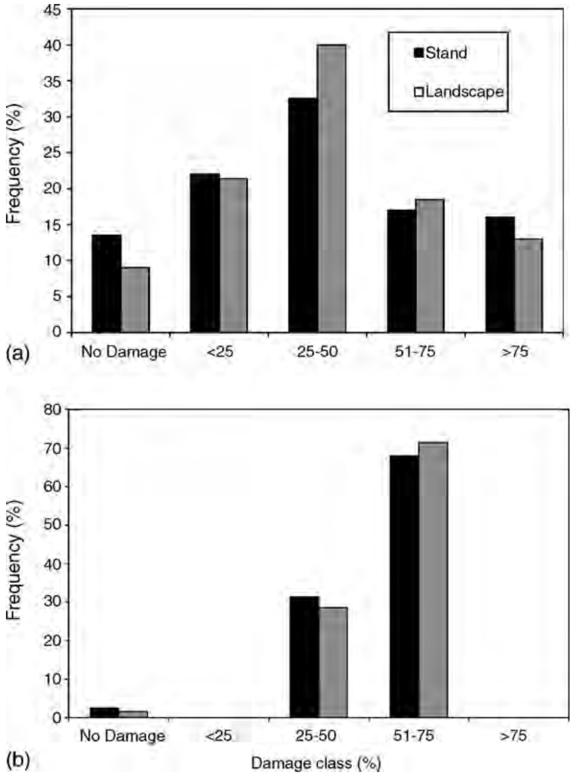

Fig. 2. Proportions of the landscape in central Massachusetts in different forest damage classes after the 1938 hurricane. The graphs show the predicted outcome of various scenarios based on a landscape model calibrated with actual stand damage data. (a) Percentage of stands (dark bars) and percentage of landscape area (stippled bars) in various damage classes for hardwood stands based on observed 1938 stand heights. (b) Distribution of hardwood stand damage classes expected if all stands were mature with a uniform height of 20 m (from Foster and Boose, 1992; reproduced with permission from the British Ecological Society).

Topographic variation, therefore, leads to landscape heterogeneity in forest age class distribution because of variations in slope exposure, physiographically influenced differences in forest species composition, and pre-existing variation in stand ages. In the gently rolling portions of southern and central New England, most of the landscape would be rated as exposed or vulnerable to the effects of intense hurricanes. For example, 82% of the landscape of the town of Petersham in central Massachusetts was classified as exposed to the storm tracks of the three F2 hurricanes that had occurred there since 1620 (Boose et al., 2001). If old-field pine stands are included, the 1938 hurricane caused >75% canopy-tree mortality on nearly 40% of the landscape and >50% mortality on 60% of the landscape. About 25% of the landscape had light damage (<25% mortality), and only 15% had no damage (Foster and Boose, 1992).

Despite the battered condition of the landscape after exposure to an F2 hurricane, there is considerable small-scale variability in the condition of stands. Few storms are powerful enough to blow down all the trees over a large area (Boose et al., 2001). The resulting mosaic of stands with light, moderate, and heavy damage creates conditions suitable for a wide variety of organisms that depend on a range of seral stages. Most of the lightly damaged stands can restore full crown cover within a decade or two, and hence become refugia for late-successional species in the battered, post-hurricane landscape. However, the long intervals (e.g., 150 years) between these powerful, stand-replacing hurricanes probably results in erratic population cycles for some early successional species. In the intervening decades, the more mobile species must seek out patches of disturbed habitat caused by thunderstorm winds, floods, ice storms, and forest fires.

For oak stands distant from the coast (e.g., central Pennsylvania), other types of windstorms such as thunderstorm downbursts are probably an important feature of the natural disturbance regime. Tornadoes are rare, with an estimated mean point recurrence interval of 10,000–20,000 years (Whitney, 1994).

4.2. Fire regimes

Most fires in oak forests occur in early spring and late fall after the leaves have been shed. At these times herbaceous vegetation is largely in a cured stage and the bare tree crowns allow direct sunlight to dry out the ground fuels on dry, windy days. Fire intensity is enhanced by the loose, porous leaf litter of curled oak leaves, the retention of some dead oak leaves on trees overwinter, and the presence in some areas of flammable shrubs such as *Kalmia latifolia*. Gusty winds can blow the burning leaves ahead of the main fire front, igniting spot fires that help accelerate fire spread and area burned.

The earliest explorers and colonists of the oak–hickory region described rather open, park-like woods free of dense undergrowth and sometimes with abundant grass (see reviews by Day, 1953; Russell, 1983; Whitney, 1994; Lorimer, 1993, 2001). The surveyor Peter Lindestrom wrote in 1656 that in eastern Delaware, "there indeed grows a great deal of high grass, which reaches above the knees of a man . . . there is also no thickly grown forest but the trees stand far apart, as if they were planted" (Lindestrom, 1656). In 1684, Lawrie noted that in eastern New Jersey "the trees grow generally not thick, but in some places ten, in some fifteen, and in some twentyfive or thirty upon an acre" (Whitney, 1994, p. 118). Very similar descriptions of park-like oak woodlands were made in coastal Massachusetts (Morton, 1632; Wood, 1634; Whitney, 1994), portions of the southern Maine coast (Rosier, 1605; Grant, 1946; Day, 1953), the Connecticut Valley of Massachusetts, and the Lake Ontario lowlands of western New York (Maude, 1826; Day, 1953).

The openness of the woodlands was often attributed by early observers to intentional burning by native tribes (e.g., Morton, 1632; Van der Donck, 1656). Because mature oak forests normally develop closed canopies and very dense understories of woody shrubs and saplings (e.g., Nowacki and Abrams, 1992), and lightning fires are infrequent (Patterson and Sassaman, 1988; Schroeder and Buck, 1970), anthropogenic burning is probably necessary to maintain open woodlands on moderately productive sites. Early descriptions suggest that uncontrolled fires in the 17th century were often moderately intense (e.g., Morton, 1632; Van der Donck, 1656). Regardless of their cause, these wildfires were more likely to create open habitats suitable for early-successional animals than would a series of modern, low-intensity prescribed burns.

Because of their anecdotal nature, these early descriptions do not permit quantitative estimates of the proportion of the landscape dominated by open woodlands. Geographical and temporal biases are also likely to influence these observations. Most early descriptions were obtained from the first-settled regions along the seacoast and in the major river valleys. Because Native American villages were heavily concentrated in these same areas (Fig. 3), open woodlands and savanna-like vegetation could have been localized and largely restricted to areas of relatively high human population density (Russell, 1983; Patterson and Sassaman, 1988; Whitney, 1994). Villages, however, were less than 20–40 km apart in most areas (Fig. 3). Annual use of fire to drive game on hunting grounds between the villages could easily have had a major impact on the general landscape, depending on average fire size.

There are unfortunately few if any good early descriptions of the interior upland vegetation. By the time the interior zones were settled, the novelty in describing North American vegetation for a European audience had worn off, and few people continued to write detailed descriptions. Furthermore, the initiation of widespread trade and warfare between Native Americans and Europeans altered the aboriginal way of life, including village size and location, patterns of hunting, and possibly fire use (Russell, 1983; Foster and Motzkin, 2003). Conceivably, the late 17th and 18th century reports of open woodlands may have reflected a relatively late development in response to increased trade and warfare. Other lines of evidence are therefore needed to resolve these ambiguities.

Presettlement land survey records in the oak region are early "metes and bounds" surveys and contain little or no direct evidence on disturbance or forest developmental stage. However, the witness trees used to mark survey boundaries do preserve some important indirect evidence of fire frequency over a vast portion of the otherwise little-known interior uplands. Surveys in the eastern half of Pennsylvania and northern New Jersey show that shade-tolerant, late-successional species were a remarkably minor component of the presettlement forest, generally making up <7% of the witness trees except in the high plateaus and mountains (Russell, 1981; Abrams and Downs, 1990; Abrams and Nowacki, 1992; Abrams and Ruffner, 1995; Black and Abrams, 2001). Yet the modern forests on similar sites face strong successional pressures toward canopy dominance by shade-tolerant species (Abrams and Downs,

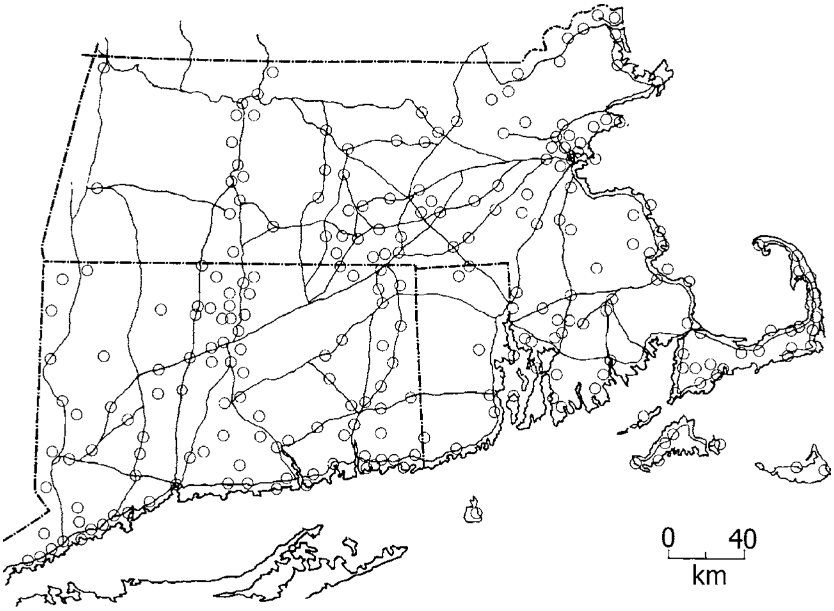

Fig. 3. Map of recorded Native American village sites (circles) and trails in southern New England at the beginning of the 17th century. The highest concentration of village sites, and probably the greatest amount of fire-influenced oak woodland, occurred in coastal areas and major river valleys (from Russell, 1980; reproduced with permission of the University Press of New England).

1990; Abrams and Nowacki, 1992). The low abundance of tolerant species in presettlement times is readily explainable if fires in the interior uplands were widespread and at least moderately frequent (cf. Grimm, 1984; Leitner et al., 1991 for clear analogs in the midwestern US), but is less easily explained if presettlement fires are presumed to have been infrequent or patchy in their occurrence. Climate change may also have been a factor, but oak forests of much drier climates such as southwest Wisconsin and central Missouri have also not been self-perpetuating on mesic sites and likewise have been succeeding toward shade-tolerant species (Pallardy et al., 1988; Hix and Lorimer, 1991).

A more direct approach to investigating presettlement fire frequency is the analysis of fire scars in old-growth stands or on remnant old (legacy) trees scattered in a matrix of second-growth forest. Detailed and systematic work of this type has been done only recently by Shumway et al. (2001) for an old-growth oak forest on steep slopes above the Savage River in western Maryland and by Dey and Guyette (2000) in nine oak–pine stands in southern Ontario. Both studies revealed a high presettlement frequency of fires, with mean surface-fire intervals of 6–20 years for the period 1650–1850. These are comparable to estimates from oak forests in Ohio (Sutherland, 1997; McCarthy et al., 2001) and limited evidence from New Jersey (Buell et al., 1954). More work on local fire-scar evidence is urgently needed while living or recently dead trees that germinated between 1600 and 1800 are still available.

Paleoecological evidence has the potential for clarifying presettlement fire frequencies, but currently the standards for relating fire frequency to charcoal abundance are being debated (Patterson and Backman, 1988; Clark and Royall, 1995b; Campbell and McAndrews, 1995). Studies in coastal Massachusetts have reported high charcoal–pollen ratios of 300–1000 on some sites (Patterson and Backman, 1988; Fuller et al., 1998; Parshall et al., 2003). As might be expected, charcoal influx is sometimes higher after the onset of European contact and settlement, reflecting the widespread use of fire in land-clearing activities (Perley, 1891). Yet charcoal influx at many oak–pine sites showed relatively little change after the point of European colonization (Watts, 1979; Maenza-Gmelch, 1997; Patterson and Backman, 1988; Fuller et al., 1998; Parshall et al., 2003), a trend also seen in fire scar studies (Dey and Guyette, 2000; Shumway et al., 2001).

A notable feature of the pollen record at many sites in southern coastal New England is the low percentage (1–4%) of the total pollen count contributed by grass, shrub, and herbaceous pollen. This suggests that open grasslands and heathlands were probably at most local occurrences in presettlement times (Motzkin and Foster, 2002; Foster and Motzkin, 2003). The implications of pollen data for the extent of open woodlands is currently less clear. Janowiak (1987) reported only 7% non-arboreal pollen from a varved lake in a region of southern Wisconsin that was heavily dominated by oak savanna in presettlement times (see also Winkler, 1985). Detailed land surveys taken at about the same time suggest tree densities of <50 trees/ha (Tans, 1976; Bollinger et al., 2003). Further studies may be needed to determine if non-arboreal pollen counts can provide reliable distinctions between open oak woodlands and closed-canopy forests.

4.3. *Forest age structure on the presettlement landscape*

Oak forests of the mid-Atlantic and southern New England uplands probably ranked second to pine–oak barrens in frequency of severe disturbances and the proportion of habitat suitable for early successional species. Recurrence intervals of 85–380 years for catastrophic wind damage in the southern half of New England imply that most stands would have been heavily dominated by the post-hurricane age cohort. Relatively few stands on exposed sites would have been old-growth stands except toward the end of the time interval between consecutive hurricanes (cf. Henry and Swan, 1974). Multi-cohort stands with a component of mature and old trees would have been common on protected sites and on some exposed sites, occupying roughly 25–40% of the landscape (cf. Foster, 1988b; Foster and Boose, 1992; Boose et al., 2001; Orwig et al., 2001). Calculating the mean proportion of the landscape in young forest habitat in this region, however, would not be very meaningful because of the erratic temporal fluctuations. The proportion of landscape in seedling–sapling habitat could vary from 40 to 50% immediately after a severe hurricane to probably <3% once the post-disturbance cohort had moved into the pole and mature age classes.

For oak forests in coastal sites and major river valleys, a regime of low- to moderate-intensity fires set mostly by Native Americans may have been superimposed upon this regime of severe wind disturbance. There is, however, currently no direct quantitative evidence that would permit numerical estimates of how much of the region may have been fire-influenced and dominated by open oak woodlands.

Oak forests distant from the influences of F2 hurricanes may have had a much different disturbance regime. Limited fire-scar evidence (Buell et al., 1954; Shumway et al., 2001) and the low frequency of shade-tolerant species cited in many presettlement land surveys suggest that fire frequency may also have been high in interior locations. Catastrophic wind disturbance was probably infrequent. Modern ecological studies suggest a natural disturbance rate of about 0.6–1% per year, mostly as small gaps within a matrix of old-growth forest (Runkle, 1990). So the proportion of the landscape in young forest stands (1–15 years) created by windthrow in the interior sections was probably low, perhaps similar to the prevailing rates in northern hardwood forests (1–3%; cf. Table 1).

5. Northern hardwood forests

Northern hardwood forests, dominated by American beech, sugar maple (*Acer saccharum*), and yellow birch (*Betula alleghaniensis*), occur in a climatic zone with a frost-free period of 120–150 days (Lull, 1968). In New England and northern New York, northern hardwoods are found on spodosols of the New England Upland physiographic province and at moderate elevations along the Appalachian mountain chain. Northern hardwoods also dominate the Allegeheny Plateau of southern New York and northern Pennsylvania, found primarily on inceptisols. Extensive tracts of beech–maple forest also occurred in presettlement times on alfisols of the Lake Ontario plain in western New York (Seischab, 1990).

Northern hardwoods are usually considered to be late-successional forests, but the dominant species persist well after catastrophic windthrow because of the resilience of advance regeneration. In presettlement times, beech was often the major dominant, usually averaging 30% or more of the witness trees (McIntosh, 1962; Siccama, 1971; Cogbill, 2000). Eastern hemlock (*Tsuga canadensis*), a shade-tolerant conifer, was often intermixed with hardwoods but was somewhat patchy in occurrence, averaging 12–20% of the witness trees. White pine, an early to mid-successional species, locally dominated sites after catastrophic disturbance, especially fire. Across the region, white pine was a relatively minor species, averaging 1–5% of the witness trees, but made up to 20% of the forest in some of the major river valleys. It was most commonly dominant on sandy river terraces (Siccama, 1971; Whitney, 1994; Cogbill, 2000).

5.1. Windstorms and ice storms

Windstorms are probably the major natural cause of catastrophic disturbance in the northern hardwood region. These include hurricanes and other cyclonic storms, thunderstorms, derechos, and tornadoes. Hurricanes are probably the dominant type of destructive windstorm in much of southern and central New England. In the Pisgah State Park of southern New Hampshire, about 25% of the landscape was severely disturbed and 50% moderately disturbed in the 1938 storm (Foster, 1988a). In 1950, a great cyclonic storm originating over the Appalachian Mountains passed across the Adirondack Mountains of New York and caused moderate to severe disturbance on a district of about 160,000 ha (Jenkins, 1995).

Derechos are fast-moving convective storms that potentially may have more impact in inland regions. Derechos can cover a wide area and produce several episodes of violent downdrafts with speeds exceeding 160 km h^{-1}. Individual downbursts can range from 4 to 40 km in length (Jenkins, 1995). Derechos and downbursts are common in the midwestern US but have been less frequently reported in the northeast. However, a derecho crossed the Adirondack Mountains in 1995, causing a mosaic of lightly and heavily disturbed patches across a region of approximately 364,000 ha. The total area on which >30% of trees were killed was about 36,000 ha (Jenkins, 1995). A great windfall in 1845 that extended from western New York to western Vermont may also have been caused by a derecho (Jenkins, 1995). Similar thunderstorm downbursts may be responsible for many of the large windfalls reported in 18th and 19th century land survey records (Canham and Loucks, 1984).

Little evidence is available on the size range of blowdowns in northeastern northern hardwoods. Probably the most detailed map was prepared by Jenkins (1995) for the Adirondack blowdown based on low-altitude aerial photos of one of the most heavily impacted areas (Fig. 4). We digitized this map to produce estimated patch size distributions (Fig. 5). Individual patches ranged from <1 to 700 ha. As in the analysis of the 1938 hurricane by Foster and Boose (1992), most of the patches in the Adirondack blowdown are small. Several large patches account for much of the blowdown area; more than 40% of the total blowdown area is in patches >100 ha (Fig. 5). Such large patches may provide important habitat for early successional wildlife species that are area-sensitive.

Of the 36,000 ha of moderate and severe damage in the 1995 Adirondack blowdown, 29,000 ha were classified as having moderate damage (30–60% of trees toppled) and 7000 ha had severe damage (>60% of trees down; Jenkins, 1995). As in the 1938 hurricane, the pattern of damage was heavily influenced by topographic exposure. In the portion of the blowdown shown in Fig. 4, the storm hit windward west and northwest slopes and ridges most heavily. This study, along with those by Hough and Forbes (1943), Foster (1988a), Engstrom and Mann (1991), and Mann et al. (1994) demonstrate that ridgetops and upper slopes generally have higher than average disturbance rates and are potentially more dependable sources of suitable habitat for early successional species.

Ice storms can also cause damage over extensive areas. The ice storm of 1998 affected >6.9 million ha of northern hardwood and spruce–hardwood forest in northern New England and New York. Effects at the stand level were highly variable, reflecting complex interactions of meteorological conditions, stand composition and structure, and topography. Surveys of the affected areas revealed that >50% of the sampled trees had no crown loss whereas more than 21% had heavy (50–79%) or severe (80–100%) crown loss. Only the 12% with severe loss were categorized as unlikely to survive (Miller-Weeks et al., 1999). This suggests that although regionwide volume and financial losses were considerable, the 1998 ice storm was not typically a stand-replacing event. Given that this was one of the most intense and severe ice storms on record for the region, it seems reasonable to assume that ice storms

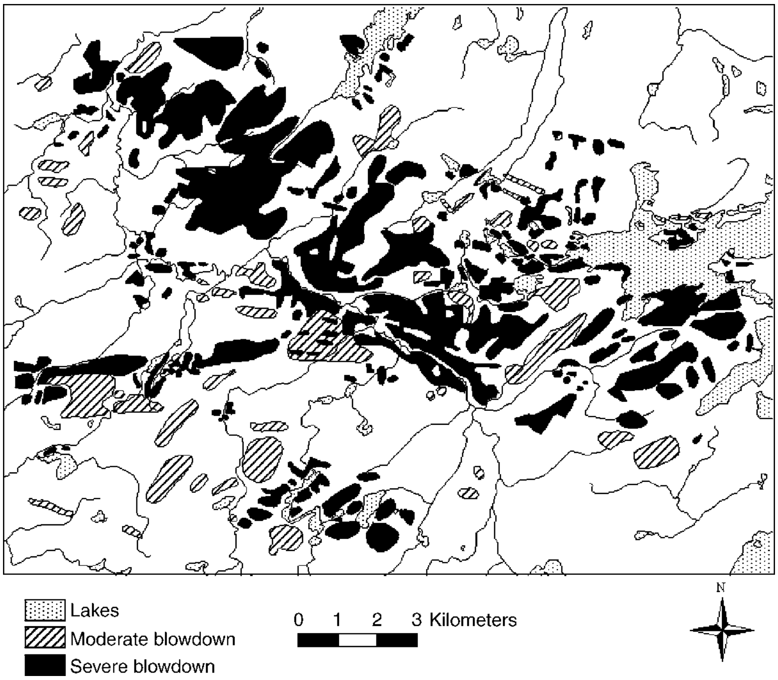

Fig. 4. Map of blowdown patches caused by the 1995 derecho in the Adirondack Mountains, New York, based on low-altitude aerial photos and reconnaissance in the region of heaviest damage (after Jenkins, 1995; reproduced with permission of the Wildlife Conservation Society).

are generally an agent of patch dynamics within forest stands (see also Seischab et al., 1993).

Aside from these isolated case studies, there has been little effort to systematically quantify annual disturbance rates by recent windstorms and ice storms. The best estimates of natural disturbance rates at the landscape scale are therefore still the presettlement land survey records of the 18th and 19th centuries (Cogbill, 2000). These records suggest that in spite of some of the spectacular storms in the historical record, the average rotation period for catastrophic storm disturbance is rather long for areas not normally subjected to F2 Atlantic hurricanes. Studies of land survey records of 1763–1810 in New Hampshire, Vermont, New York, and Pennsylvania indicate that large patches of standing dead and fallen timber covered only 0.2–1.5% of the landscape (Seischab, 1990; Whitney, 1990; Seischab and Orwig, 1991; Marks et al., 1992; Cogbill, 2000). If surveyors recorded storm-damaged areas up to 15 years old, this translates into mean rotation periods of 1000–7500 years. These 15-year storm disturbance rates are comparable to rates of 0.7–3.5% reported in northern hardwood forests of the Great Lakes region based on mid-19th century survey records (Canham and Loucks, 1984; Whitney, 1986; Zhang et al., 1999), and from reconstructive field studies of old-growth stands for the period of 1850–1980 (Frelich and Lorimer, 1991b; Frelich and Graumlich, 1994; Ziegler, 1999). This concurrence of evidence from different methods over a vast region and over a long period of time suggests that these are probably reasonable estimates of 18th and 19th century storm-disturbance rates. In northern hardwood regions under the influence of Atlantic hurricanes, the average interval between stand-replacement events was much shorter, ranging from 150 to 380 years on exposed sites (Boose et al., 2001).

5.2. Fire regimes

Lightning fires are uncommon in the northern hardwood region, usually making up less than 3% of all fires (Jenkins, 1995). Northern hardwood forests are generally less susceptible to intense fires than most other forest types in the region. In contrast to the loose and porous fuel beds of oak and pine, leaves of maple

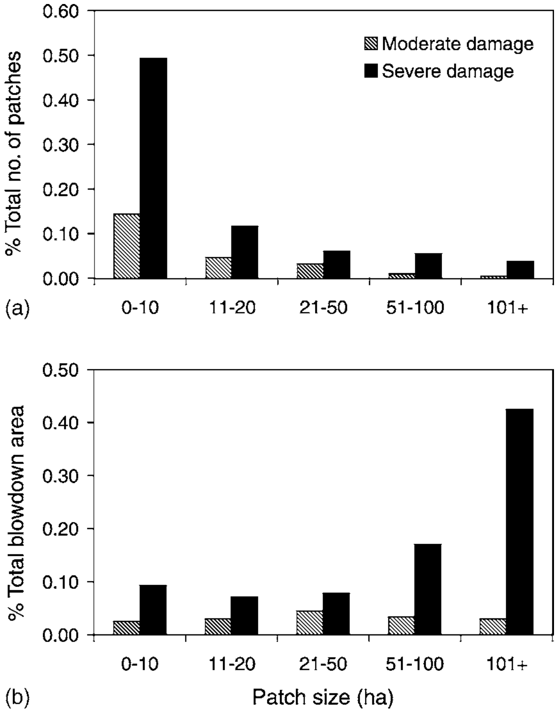

Fig. 5. Patch size frequency (a) and proportion of total blowdown area in different patch sizes (b) based on the map of the 1995 Adirondack blowdown in Fig. 4.

and birch are thin and easily matted down by rain and snow. The mesic soil conditions of many northern hardwood sites also hinder fire spread. However, fire scars and modern fire records show that surface fires can spread over large areas under certain conditions. For example, in October 2000, a lightning-ignited surface fire, fanned by strong northwest winds, spread over more than 560 ha of mature sugar maple forest in the Porcupine Mountains, Michigan in three days, despite prompt suppression attempts (Anon., 2000). Early surveyors in northern Maine recorded several consecutive miles of "hardwood killed by fire" during the dry year of 1825 (Lorimer, 1977). But generally, intense fires are common only in areas of heavy down fuel accumulations such as logging slash and blowdowns. Ayres (1909) noted that nearly all fires in the White Mountains of New Hampshire originated on cutover land, and the boundaries of the fires closely corresponded to the recently logged areas, as did fires in the Adirondack Mountains and northern Maine at about the same time (Spring, 1904; Jenkins, 1995).

Studies of old-growth stands confirm on-site evidence of both surface fires and stand-replacing fires, although most reported cases have been in pine–hemlock stands (Lutz, 1930; Hough and Forbes, 1943; Foster, 1988a; Mann et al., 1994; Abrams and Orwig, 1996). In a pine–hemlock forest on exposed knobs and escarpments in the Green Mountains of Vermont, Mann et al. (1994) documented an 18-year mean fire interval between 1504 and 1851, including three fires that led to some new canopy recruitment. Lutz (1930) also reported abundant charcoal fragments in an old-growth hemlock–beech forest about 8 km from the Allegheny River in Pennsylvania. A field examination of stumps adjacent to this stand suggested as many as 40 fire scar dates (not cross-dated) spanning the period from 1725 to 1925. At least five major surface fires occurred over the 200-year period, two of them during the presettlement era.

Quantitative evidence on the frequency of stand-replacing fires is available from three independent methods, all suggesting rather long return intervals. Presettlement land survey records indicate that the proportion of survey lines intercepted by burned lands was very low, probably 0.5% or less over much of northern New England, New York, and the western Allegheny Plateau of Pennsylvania (Siccama, 1971; Seischab, 1990; Seischab and Orwig, 1991; Marks et al., 1992; Cogbill, 2000). This would result in rotation periods of >3000 years. Burned windfalls, however, were probably one of the major successional pathways that led to dominance of stands by white pine (Hough and Forbes, 1943; Foster, 1988a; Abrams and Orwig, 1996) and birch–aspen. Burned lands and white pine stands were most common along the major river corridors (Cogbill, 2000), probably because of more intense human activity in those areas.

Fahey and Reiners (1981) used 20th century historical records to derive independent estimates of natural fire frequency in Maine and New Hampshire. Reasoning that the most accurate records of lightning fire frequency are probably the more recent ones (because of improved detection) but that the best estimates of mean fire size are those in the pre-fire suppression era, they multiplied mean fire size from 1903 to 1910 by recent estimates of mean annual frequency of lightning fires. This method yielded an estimate of 1070 years for a natural fire rotation in northern hardwoods.

Paleoecological evidence also suggests long fire intervals between severe fires in northern hardwoods. Using a new thin section technique on sediment cores, Clark and Royall (1996) reported that the sedimentary record showed low rates of charcoal accumulation but no clear evidence of local fires from two northern hardwood sites in Maine. In a forest hollow dominated by hemlock in central Massachusetts, Foster and Zebryk (1993) identified six charcoal horizons from the past 8000 years, but only one of these dated to the past 3000 years. Other paleoecological studies of hemlock–hardwood stands in central New England have found either little evidence of fire (Patterson and Backman, 1988; McLachlan et al., 2000) or relatively long fire intervals of about 600 years (Schoonmaker, 1992).

5.3. *Forest age structure on the presettlement landscape*

Northern hardwoods on the interior uplands probably had the lowest frequency of stand-replacing disturbances among the major forest types in the northeast. In the interior Allegheny Plateau and Appalachian Mountain regions, most lines of evidence suggest average rotation periods of 1000–3000 years for catastrophic windthrow and similar rotation periods for stand-replacing fires. The pooled disturbance rates for wind and fire result in an overall rotation period for catastrophic disturbance of 500–1500 years. The proportion of seedling–sapling habitat (1–15 years old) under these disturbance regimes ranges from 1 to 3%. Stands 15–30 years old would make up an additional 1–3% of the landscape (Table 1).

Because of the long rotation periods, the slope of the negative exponential curve is very shallow, and so estimates of young and mature forest based on the uniform and negative exponential models are quite similar. For example, with an overall rotation period of 500 years, the proportion of the landscape in stands up to 100 years of age is 20% with the uniform model and 18% with the negative exponential model. The effects of underlying model assumptions on the total amount of old-growth forest are also slight for overall rotation periods longer than 500 years. The estimated proportion of the landscape in old-growth forest (>150 years old) is 70–89% depending upon rotation period and model form.

The presettlement age distributions in Table 1 suggest ratios of uneven-aged to even-aged stands ranging from about 3:1 with a 500-year rotation period to 8:1 for a rotation period of 1365 years. Among mature and old stands only, the ratio of uneven-aged to even-aged ranged from 5:1 to 14:1. Field verification of these ratios is difficult because of the rarity of old-growth stands, but a predominance of uneven-aged stands is evident from stand age data on inland sites obtained by Hough and Forbes (1943), Leak (1975), Chokkalingam (1998), and Ziegler (1999). The 1365-year rotation period (2500 years for wind, 3000 years for fire) results in an overall landscape age distribution that is quite similar to an age distribution reconstructed from field data and simulations for large natural landscapes of northern hardwood forest in Michigan (Frelich and Lorimer, 1991a).

In spite of the low rates of stand-replacing disturbance, disturbances of moderate intensity were much more common and probably provided suitable habitat for some early successional species (cf. Greenberg and Lanham, 2001). In the 1995 Adirondack windstorm, the zone of moderate disturbance was four times as large as the zone of catastrophic disturbance. Likewise, a notable feature of many old-growth hemlock–hardwood stands is the dominance by 2 or 3 age cohorts, suggesting episodes of partial canopy removal at intervals of 60–400 years (Hough and Forbes, 1943; Frelich and Lorimer, 1991b; Chokkalingam, 1998; Ziegler, 1999). These gaps are often rather small ($<300\ m^2$), but given the point recurrence intervals suggested by the old-growth stands, moderate disturbances could potentially affect an additional 3–25% of the landscape over a 15-year period.

6. Spruce–northern hardwood forests

Northern hardwood forests with a substantial admixture of spruce (*Picea* spp.) and other conifers are common on a variety of habitats in northern New England and New York. Spruce–hardwood forests are best discussed separately from other northern hardwood forests because of differences in both physical environment and disturbance regime. Spruce and fir (*Abies balsamea*) in particular are much more vulnerable to windthrow, insect epidemics, and crown fires than most of the associated species.

Spruce–northern hardwood forests occur mostly on spodosols and in a climatic zone with a frost-free period of 90–120 days (Lull, 1968). Four distinct habitats are often recognized, each of which probably has a different disturbance regime: (1) spruce swamps, (2) "spruce flats" on relatively level and often stony soils near lakes and streams; (3) mixed forests of spruce and northern hardwood species on the better soils of lower slopes and low ridges, and (4) the "spruce slope" type on high mountain slopes or on ridges with thin, rocky soil. The proportion of spruce in the forest averaged about 15–20% across the region prior to settlement, but often reached 60–80% on the high slopes. Other conifers such as balsam fir, hemlock, northern white cedar (*Thuja occidentalis*), larch (*Larix laricina*), and eastern white pine collectively averaged 12–40% of the trees in different habitats and in different geographic regions (Siccama, 1971; Lorimer, 1977; Cogbill, 2000). Paleoecological studies indicate that for much of the mid-Holocene, northern hardwood species largely dominated the region. The invasion of spruce is a relatively recent phenomenon of the last 2000 years, probably in response to climatic cooling (Anderson et al., 1986; Schauffler and Jacobson, 2002).

6.1. Windstorms

Spruce and fir are susceptible to windthrow because of their shallow root systems and the tendency to predominate in swamps, on upland sites with thin and stony soils, and on mountain slopes exposed to severe winds. Graves (1899) noted that on these extreme habitats, windfall was common and that trees were usually blown down before reaching (biological) maturity. As a consequence, pure stands of spruce in the Adirondacks were usually composed of comparatively young trees. Graves (1899) also indicated a typical average diameter of 33 cm for canopy trees in spruce swamps (equivalent to an average age of 180 years), and noted that spruce forests in the swamps, flats, and high mountain slopes facing the prevailing winds were often relatively even aged. In contrast, extensive tracts of mixed spruce–northern hardwoods were seldom destroyed by natural forces, but individual trees were continually dying and being replaced, resulting in a forest that included trees of all sizes and ages (Graves, 1899). This overall assessment was shared by Hawley and Hawes (1912) and Murphy (1917).

As in the northern hardwood region to the south, the spruce–hardwood region is subjected occasionally to large storms capable of causing windthrow over extensive areas. Outside of the Adirondacks, the largest known windstorm documented in reasonable detail occurred about 1795 north of the Piscataquis River in northern Maine (Morse, 1819; Lorimer, 1977). Descriptions by land surveyors working in the area in 1801 suggested a typically heterogeneous patchwork of heavily and lightly disturbed stands. The full extent of the storm damage was difficult to ascertain because portions of the region were surveyed at different times, and subsequent fires burned much of the tract in 1803, 1811, and 1825. But references to extensive windfalls and burned windfalls that seemed to be connected with the 1795 storm occurred in a zone about 72 km from west to east and about 20 km from south to north, nearly all in the Piscataquis Valley. The overall boundaries of the region known to have had numerous windfalls encompassed 144,000 ha, about 40% as large as the zone impacted by the 1995 Adirondack derecho. The total area of blowdown within this zone was not known, but based on the available descriptions, severity seems comparable to the 1995 Adirondack storm. Another large storm occurred in northern Maine in November 1871. According to Hough (1882), a snowstorm in that month was followed by a severe gale that blew down many trees and extended hundreds of miles in Maine and New Brunswick. In recent times, one of the more notable storms, albeit of much smaller magnitude, was a blowdown of about 2500 ha in northcentral Maine in 1974 (Kolman, 1978).

Presettlement land surveys provide the best evidence of disturbance rates and rotation periods at the landscape level. Windfalls and burned areas recorded by surveyors in New Hampshire, Vermont, and northern New York occupied about 0.5% of the landscape, and suggest long rotation periods. In northeastern and northcentral Maine, surveyed between 1793 and 1827, recent windfalls occupied somewhat less than 2.6% of the landscape (Lorimer, 1977). This suggests rotation periods for severe windthrow on the entire landscape of 575–1150 years, depending upon whether a 15- or 30-year time span is assumed. There was clearly a difference in windthrow frequency, however, between spruce-dominated forests in lowlands and mixed spruce–hardwood forests on the uplands, with most of the blowdown area located in conifer-dominated forests in swamps and on stony flats. If these habitats are analyzed separately, the rotation period (15-year time window) for severe windthrow in the mixed spruce–hardwood forests and northern hardwood dominated forests on the better soils is 2585 years, while the rotation period for conifer forests on lowland sites and on poor, rocky soils is only 290 years.

6.2. Insect epidemics

Many insects are capable of killing trees in spruce–northern hardwood forests. Only a few native insects, however, are known to cause widespread mortality. These include the spruce budworm (*Choristoneura fumiferana*), spruce beetle (*Dendroctonus rufipennis*), and larch sawfly (*Pristiphora erichsonii*). Because these insects are reasonably host-specific, in mixed forests they normally kill scattered trees or small groups. However, in stands heavily dominated by spruce, fir, or larch, they can kill most of the stand. Epidemics can be triggered by an abundance of susceptible trees, weather patterns favorable for buildup of insect populations, and abiotic factors that weaken the host trees. Stands with a high percentage of mature balsam fir are particularly vulnerable to spruce budworm. Stands of mature spruce weakened by windthrow or logging are likewise susceptible to spruce beetle outbreaks (Schmid and Beckwith, 1975).

Two large-scale epidemics of spruce budworm have occurred in recent times in northern Maine, including outbreaks that caused very heavy mortality in 1913–1919 and 1972–1986. An earlier epidemic of 1876–1879 reportedly killed one billion board-feet of spruce, about half of the merchantable spruce volume, in the Allagash River region of northwestern Maine (Hough, 1882). The 20th century outbreaks are believed to have been exacerbated by decades of selective logging of the more valuable red spruce (*Picea rubens*), which led to unusual dominance by more susceptible balsam fir (Zon, 1914; Swaine and Craighead, 1924; Seymour, 1992). Nevertheless, there are at least two credible reports of major insect outbreaks in spruce forests that occurred too early to have been triggered by human activities. Packard (1881) cited two independent reports of an insect outbreak in 1818 that killed almost every spruce west of the Penobscot River. Hopkins (1901) cited widespread spruce mortality from 1844 to 1859 in two New York counties in the Adirondack Mountains. The geographic extent of these early outbreaks is not well known, however, and different writers attributed dieback to various causes including spruce beetle, spruce budworm, drought, and cold weather. Hough (1882) suggested that the dieback of 1876–1879 was caused by insects and triggered as a result of trees weakened by the extensive windstorm of November 1871.

Because insect-killed conifers are easily windthrown and are highly flammable, these various disturbances do not act independently. It is therefore possible that some areas of windthrow and burned land in the early land surveys might have been first subjected to insect outbreaks. At the time of the 1793–1827 land surveys in northeastern Maine, however, only four small areas were explicitly described as having “much dead and down spruce” (Lorimer, 1977).

6.3. Fire regimes in spruce–hardwood forests

Natural fire frequency in spruce–hardwood forests is a difficult and complex topic. There is no question that large and severe fires do occur in this forest type but evidence on the frequency and rotation periods of such fires is difficult to interpret. There also may be important regional differences. Siccama (1971) found no reports of burned land in the presettlement land surveys of 1783–1787 in the northern half of Vermont, and Cogbill’s (2000) analysis of a much larger portion of the northeast also showed burned lands to be uncommon. But Maine has had a long history of large and intense fires (Hawley and Hawes, 1912; Coolidge, 1963).

Mature and old-growth forests of spruce and northern hardwoods do not burn readily. Hawley and Hawes (1912) pointed out that most fires occur in areas where trees were harvested, and uncut stands were usually resistant to fires. However, some large fires in mature stands have occurred in the past two centuries (Lorimer, 1977, 1980).

Prior to European settlement, abundant windfalls provided similar fuel conditions to those of cutover lands and some of these did burn (e.g., the 1795 blowdown in northern Maine). However, ignition probabilities of blowdowns are not well known, and some large historic blowdowns did not catch fire (Jenkins, 1995). The overall frequency of lightning fires in Maine is low (0.45–0.81 lightning fires per 1000 km^2 per year; Fobes, 1944; Fahey and Reiners, 1981), similar to lightning fire frequency elsewhere in the Northeast (Schroeder and Buck, 1970). According to survey records (Siccama, 1971; Lorimer, 1977; Cogbill, 2000), early successional and post-fire species (e.g., *Betula papyrifera*, *Populus tremuloides*, *Prunus pensylvanica*, *P. strobus*) made up only a small proportion of the presettlement forest in northern New England. This evidence is consistent with comments by foresters familiar with areas of old-growth forest in the late 19th century. Dana (1930) stated that white birch and aspen were poorly represented in the original forest. Of the intolerant birches (*B. papyrifera* and *B. populifolia*), Cary (1896) remarked that ‘‘neither, of course, was a large element in the natural forest, but they have come in largely on burnt and cleared land.’’ These pioneer species usually dominate stand-replacement burns on upland sites, although shade-tolerant species such as spruce and beech are often present and are sometimes locally dominant (Dana, 1909; Ayres, 1909; Lorimer, 1980; Patterson et al., 1983).

Shortly before settlement, recently burned lands (1–15 years old) occupied 3.9% of the landscape in northeastern Maine, mostly from two very large fires of 10,000 ha and 32,000 ha that occurred in 1825. An additional 5.1% of the landscape was recorded as old burn, mostly from the 80,000 ha fire of 1803 (Lorimer, 1977). At the time of the 1825–1827 surveys, the 1803 burn was occupied by white birch and aspen. An additional 0.6% was occupied by pole-mature stands (probably up to 75 years old; cf. Cary, 1894b, 1896; Dana, 1909) in which white birch and aspen were listed among the dominant species. Interpretation of these data is complicated by the fact that the year 1825 was the worst fire year in the state’s recorded history (Coolidge, 1963), and by the use of land-clearing fires by the first settlers from 1799 to 1825 (Loring, 1880). Overall, the amount of forest burned over a 75-year period suggests a fire rotation period of about 800 years. The amount of land burned only during the more recent 15-year period from 1811 to 1826 suggests a shorter rotation of 385 years, and use of a 30-year period from 1796 to 1826 would imply a rotation of about 330 years.

Fahey and Reiners’ (1981) use of mean fire size in the early pre-suppression era (1903–1910), combined with modern data on lightning fire frequency, is the only other historical estimate of natural fire rotation periods. This approach suggested a fire rotation of 1240 years for spruce–fir forests in Maine, the longest for any of the major forest types. We examined the same data but made some different assumptions about fire potential in drought years to provide a sensitivity analysis. In most years, lightning fires account for only about 1.5% of the total burned area, but lightning fires contribute 17–21% of the burned area in some years (mean intervals of about 5 years; Fobes, 1944). We therefore applied these two different proportions to the early fire data from 1903 to 1915 in the Maine Forestry District, which at that time comprised about 3,846,150 ha of mostly spruce–hardwood forest (Coolidge, 1963). We assumed that lightning caused 20% of the total burned area in the three peak fire years of 1903, 1908, and 1911, and 1.5% in each of the other 10 years. This resulted in fire rotation periods (including slash fires and ground fires) of 1253 years for 1903–1910 and 1519 years for 1903–1915.

Analysis of sedimentary charcoal at three sites in northcentral and northeastern Maine also has suggested infrequent fires. Clark and Royall (1996) considered the size of charcoal particles and the concentration too low to indicate local fires in the past 1500 years near Conroy Lake, and Schauffler and Jacobson (2002) could find little evidence of fire over a period of 5000 years at the Big Reed Preserve. Anderson et al. (1986), however, found evidence of at least two major fires that occurred about 3500 and 1500 years ago at South Branch Pond. In the spruce–fir forests of coastal Maine, Schauffler and Jacobson (2002) found only small amounts and small-sized particles of charcoal, suggesting infrequent fire in those habitats for most of the Holocene.

As in the case of northern hardwoods, age-structure analysis of remnant old-growth spruce–hardwood stands provided limited verification for these estimates of disturbance frequency. Among 10 old-growth stands at the Big Reed Preserve in northcentral Maine and two stands in the White Mountains of New Hampshire, there was a strong predominance of uneven-aged or multi-aged stands and low importance values for pioneer species that commonly regenerate after fires (Leak, 1975; Moesswilde, 1995; Chokkalingam, 1998;

Chokkalingam and White, 2001). Data collected from 18 old-growth stands by early foresters also indicated a predominance of species and stands (about 60%) with relatively smooth, descending size distribution curves, even in the spruce slope type (Cary, 1894a, 1896; Hosmer, 1902; Chittenden, 1905). Size distributions are less conclusive evidence than direct examination of tree cores, but descending size distributions for individual species are usually characteristic of uneven-aged or multi-aged structures (cf. Leak, 1973, 1975; Moesswilde, 1995; Chokkalingam, 1998). These limited age and size distribution data appear consistent with broader generalizations about age structure in old-growth spruce–hardwood stands by early foresters. Hawley and Hawes (1912) stated that "the (spruce–hardwood) forest as a whole is composed of trees of all ages; it is thus an 'uneven-aged' or 'all-aged' forest. Exceptions to this character occur; for sometimes a part of the forest is found where the trees are all of one age over considerable area; i.e., an 'even-aged' forest. But such cases are in the minority."

6.4. *Forest age structure on the presettlement landscape*

Regional average rates of catastrophic disturbance in spruce–hardwood forests are rather difficult to verify because of the pronounced temporal and habitat-related variability. Tables 2 and 3 show estimated regional stand age distributions for rotation periods ranging from 575 to 1000 years for windthrow and 385–1200 years for fire. These correspond to overall rotation periods for stand-replacement events of 230–545 years. The proportion of seedling–sapling habitat (1–15 years) ranges from 2.7 to 6.5% depending upon the rotation period and model form. Stands 15–30 years old make up a similar proportion of the landscape. Stands 60–150 years old range from 13 to 39% and old-growth stands (>150 years old) from 35 to 76%.

The Maine land surveys enable a preliminary estimate of habitat-related variability, in this case the contrast between mixed spruce–hardwood forests on the better soils of lower slopes and low ridges and the conifer-dominated sites in swamps, stony flats, and steep, high-elevation sites (Lorimer, 1977 and Table 3). The pooled wind/fire rotation period of 606 years for mixed spruce–hardwood forests results in a landscape age distribution similar to northern hardwood forests elsewhere, with 2.4% in seedling–sapling habitat and 75–78% old growth. The conifer forests in swamps and on rocky sites, in contrast, had a pooled rotation period of only 210 years, giving a landscape with 7% seedling–sapling habitat and 29–39% old-growth forest.

Although the age distributions in Tables 2 and 3 vary widely, some of these may be more plausible than others based on other lines of evidence. The scenario in Table 2 with the shortest overall rotation period (375 years for fire, 575 years for wind, 230 years overall) implies that even-aged stands would occupy about 48% of the landscape, and that post-fire stands up to 75 years of age would have dominated about 18% of the landscape. Given the generally low representation of post-fire species in the presettlement forest, the predominance of uneven-aged stands described by early foresters, the predominance of uneven-aged structures among remnant old-growth stands, and the low levels of sedimentary charcoal, an average 230-year rotation may be too short for the region as a whole. However, it may lie within the historic range of variability and may even be a common disturbance frequency for some habitats and subregions. A 1000-year rotation period for severe windthrow and 1200 years for stand-replacing fire would result in about 24% of the landscape occupied by even-aged stands and 76% by uneven-aged stands, with 6% of the landscape in post-fire stands up to 75 years old. These figures seem more consistent with the currently available evidence. The best tentative estimates of the proportion of the landscape in seedling–sapling habitat (1–15 years old) thus lie between 2.5 and 4.5%, with a similar proportion between the ages of 15 and 30 years.

7. Conclusions

Natural disturbance regimes vary markedly across the northeastern US, being influenced primarily by geographic location, forest type, and local habitat conditions. In presettlement times, there seemed to be a gradient in the frequency of stand-replacing disturbances from coastal regions to interior zones. From Delaware to southern Maine, coastal regions often had the highest disturbance frequencies because of the large expanses of sandy pine–oak barrens near the coast, abundant populations of Native Americans, longer snow-free periods, and the influence of occasional destructive hurricanes. In situations where presettlement or early historic disturbance rates can be quantified, the average proportion of the landscape occupied by seedling–sapling habitat was probably only 1–3% in the northern hardwood forests of northern New England, New York, and Pennsylvania. However, seedling–sapling habitat may have occupied, on average, more than 10% of the landscape in some of the coastal pine–oak barrens. To this must be added an unknown proportion of the landscape in fire-influenced oak woodlands, some of which may have been rather ‘‘open’’ with less than full crown cover.

These regional averages do not reflect the high spatial variability that clearly occurred among specific habitats within a region or the high temporal variability that occurred within moderate-sized landscapes. In all forest regions, high ridgetops and sites with thin or rocky soil had a much higher than average frequency of blowdown and fire, and were probably the most dependable locations where early successional organisms might find suitable habitat. In the spruce–hardwood region, there is some evidence that seedling–sapling habitat may have occupied about 7% of the spruce swamp and spruce flat habitats, roughly twice the regional average. Also, within smaller subregions, the influence of occasional large-scale catastrophic events such as the 1938 hurricane or the 1995 Adirondack windstorm could disrupt the existing age distribution of forest over sizable areas, leading to major but temporary regional irruptions of early successional wildlife populations. Range of variability in time and space are probably more important than averages, yet are even more difficult to quantify given our current state of knowledge.

The regional averages also do not reflect the very significant impact of moderate-severity disturbances that typically removed 30–60% of the overstory. Stand reconstruction studies as well as contemporary observations suggest that these disturbances were much more frequent, and hence much more influential, than catastrophic disturbances in determining the overall structure of stands and landscapes. Evidence presented by Greenberg (2001) and Greenberg and Lanham (2001) for the southern Appalachian Mountains suggested that moderate-sized gaps 0.1–1.2 ha (created by a hurricane) can be important in creating suitable habitat for some early successional birds and reptiles.

Systematic data from the federal network of Forest Inventory and Analysis plots indicate that the current average proportion of seedling–sapling habitat in the northeast is about 16%, ranging from 4% in Massachusetts to 25% in Maine (Trani et al., 2001). If the modern seedling–sapling category (defined as young stands with predominant tree diameters <12.7 cm) is comparable in age range to those of recent blowdowns and burned areas cited by early surveyors, these data suggest that the current proportion of seedling–sapling habitat is probably similar to presettlement levels in some areas but higher or lower in others. For example, the current proportions of seedling–sapling habitat of 4–5% in Massachusetts and Connecticut are only moderately higher than presettlement estimates for the interior northern hardwood region. The current estimates of 9–25% for the northern New England states are probably several times higher than presettlement levels, even given the most generous estimates for the role of windthrow and fire in the spruce–hardwood region (see also Seymour et al., 2002). However, it is possible that the current estimates of 10–18% for the mid-Atlantic states may be comparable to (or possibly lower than) presettlement levels if fire was as widespread in the coastal regions as suggested by historical and fire-scar evidence, and if the pine barrens had presettlement fire rotations of less than 150 years. Because the ‘‘natural’’ proportion of the landscape in young forest habitat seems to be low in some regions, managers should be aware that restoring natural conditions could have unfavorable effects on many early successional species that may already be declining for various other reasons. Thus the ‘‘natural range of variability’’ concept may have some serious drawbacks in guiding conservation policy if it were to be implemented mechanically across the entire region (Litvaitis, 2003).

It is not yet clear if forest management practices that increase the amount of seedling–sapling habitat will be sufficient to reverse the declining trends of some early successional species. Many of the threatened and watch-list species cited by Hunter et al. (2001) are primarily grassland, heathland, and marshland specialists that make some use of young or open forest habitats. About 70% of grassland and shrubland birds have been undergoing long-term declines, for which habitat loss and human development patterns are clearly major causes (Hunter et al., 2001; Litvaitis, 2003).

Fewer species with distributions centered in the northern hardwood or spruce–hardwood regions have downward population trends, yet more than 40% of the species in those regions may be experiencing significant declines (Franzreb and Rosenberg, 1997; Hunter et al., 2001). Some of these cases may represent a readjustment of populations to habitat levels similar to those in presettlement times, whereas species with more serious declines (e.g., wood thrush (*Hylocichla mustelina*) and golden-winged warbler (*Vermivora chrysoptera*)) may be affected by other threats, including decline in quality of forest habitats, threats on the tropical wintering grounds, modern migration hazards, and loss of habitat to human development (DeGraaf and Miller, 1996; Franzreb and Rosenberg, 1997). To the extent possible, investigators should try to determine if increasing the amount of young forest habitat or modifying habitat structure is sufficient to reverse the population trends of species at risk.

Guidelines for restoring a more natural forest age structure in specific landscapes (e.g., national or state forests) may require more detailed investigations of historic disturbance regimes that are spatially linked to local habitat and site variation. Sampling, for example, could be stratified by habitats using existing ecological land classification systems (e.g., Smith, 1995; Gawler, 2000). All of the standard methods could be used in such investigations, including analysis of presettlement land surveys, stand history reconstructions of the larger old-growth remnants, paleoecological analysis of a network of small forest hollows, observations of contemporary natural disturbances, and computer simulation. Major progress toward interdisciplinary work of this type has been achieved at the Harvard Forest in Massachusetts (e.g., Fuller et al., 1998; Boose et al., 2001) and the Big Reed Forest Reserve in Maine (e.g., Moesswilde, 1995; Shauffler and Jacobson, 2002; Chokkalingam and White, 2001). Further interdisciplinary work of this type would be useful in those and other areas to provide some of the site-specific information needed by land managers as an aid in devising long-range management plans.

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Severe thunderstorms and climate change

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ABSTRACT

As the planet warms, it is important to consider possible impacts of climate change on severe thunderstorms and tornadoes. To further that discussion, the current distribution of severe thunderstorms as a function of large-scale environmental conditions is presented. Severe thunderstorms are much more likely to form in environments with large values of convective available potential energy (CAPE) and deep-tropospheric wind shear. Tornadoes and large hail are preferred in high-shear environments and non-tornadic wind events in low shear. Further, the intensity of tornadoes and hail, given that they occur, tends to be almost entirely a function of the shear and only weakly depends on the thermodynamics.

Climate model simulations suggest that CAPE will increase in the future and the wind shear will decrease. Detailed analysis has suggested that the CAPE change will lead to more frequent environments favorable for severe thunderstorms, but the strong dependence on shear for tornadoes, particularly the strongest ones, and hail means that the interpretation of how individual hazards will change is open to question. The recent development of techniques to use higher-resolution models to estimate the occurrence of storms of various kinds is discussed. Given the large interannual variability in environments and occurrence of events, caution is urged in interpreting the observational record as evidence of climate change.

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Contents

1. Introduction

The global average temperature has increased significantly in the last few decades, mostly as the result of increasing greenhouse gases from anthropogenic sources (IPCC, Intergovernmental Panel on Climate Change, 2007) This trend is predicted to continue over the next century, barring changes in fossil fuel consumption. Global average temperature is, at some level, of little practical importance to most of society. Questions about changes in local weather events, particularly extreme events, are of greater concern. Assessment reports, such as those from the Intergovernmental Panel on Climate Change (IPCC) and the United States Climate Change Science Program (CCSP) have focused on such issues in recent reports (IPCC 2007, 2012; CCSP, 2008). Temperature and hydrologic cycle (e.g., heavy precipitation

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and drought) have been particularly emphasized. In large part, this is because of the more direct connection between global temperature and local temperature and moisture processes via the Clausius–Clayperon relationship and land-surface processes such as evaporation.

Tropical cyclones have also received a great deal of attention in the last decade, fueled in part by the coincidence of some papers that suggested a strong relationship between global temperatures and tropical cyclone strength being published near the time of Hurricane Katrina (Emanuel, 2005; Webster et al., 2005). The controversy that ensued after those papers led to significant research resulting in a much more complex description of the relationship and an understanding that different basins may respond differently to changing global temperature (e.g., Kossin and Vimont, 2007). Some of that work involved making estimates of the magnitude of impacts of inconsistent data collection procedures (Vecchi and Knutson, 2011), as well as the use of improved global climate models to look at projected changes.

Research into severe thunderstorms (those that produce large hail, damaging straight winds and/or tornadoes) in relationship to global climate has been more limited. In large part, this is a result of the even greater problems associated with data collection than are found in the tropical cyclone arena. The small horizontal scale of the phenomena also makes it more difficult to deal with them with global models that have horizontal grid spacings on the order of tens of kilometers or larger. As a result, the description of trends within the IPCC Assessment Reports has tended to be a paragraph or two. The extremely deadly spring 2011 tornadoes in the United States have raised attention to the threat posed by severe thunderstorms. As a result, there has been a great deal of discussion in the popular media about the relationship of tornadoes and climate, with commentators from a wide variety of viewpoints using elements of the observed record to support a wide variety of hypothesized relationships. Unfortunately, very little of that discussion has taken place in the context of an understanding of the limitations of the observed data or of the current state of understanding within the relatively small severe thunderstorm/climate research community. In this paper, I will discuss those limitations and that understanding. I will point to promising opportunities for research that could lead to better understanding for the next round of assessment reports.

2. Reports

In general, a logical place to start when considering climate change is to start by considering the observational record. For instance, a time series of temperature records represents a simple place to look for changes. Severe thunderstorm reports, typically, are of a very different class. While regularly observed phenomena such as temperature, precipitation and winds provide time series that can be analyzed in time and space with straightforward procedures, severe thunderstorm events are typically “target of opportunity” observations. They require the presence of an observer and a system to collect the observations. Obviously, if there is no one to witness the event or the damage it causes, the event will not be recorded. Even if someone sees it, however, if there's no system in place to collect the reports, then no record will available in the future. Also, in places where there are organized forecasting activities, the reports are frequently collected primarily for the purposes of aiding the forecast process, be it for creating or evaluating the forecast.

The nature of target of opportunity reports leads to the potential for significant problems in interpretation. Inhomogeneities in time and space in procedures for collecting reports can lead to large non-meteorological impacts in the record. Brooks et al. (2003a), Verbout et al. (2006) and Doswell et al. (2009) illustrated historical changes in the reporting and damage rating of tornadoes in the USA. Tornadoes in the early part of the official National Weather Service record (1950-approximately 1975) are rated with higher ratings than the 1975–2000 period, which, in turn, had higher ratings than 2001–2007. Doswell et al. (2005) showed large differences in reporting practices for non-tornadic severe thunderstorms across regions in the USA. Taken together, these studies demonstrate the challenges associated with reporting databases. It is particularly important to note that the USA has made the most significant efforts over the years within its national meteorological service to collect reports of severe thunderstorms. Other countries have even greater problems with consistency in reporting. Also, there is little to no relationship between changes in reports and changes in population in time and space. This is consistent with King (1997), who looked at the impact of population on reporting of tornadoes in southwestern Ontario. This region is particularly interesting because the effects of lake breezes from three of the Great Lakes lead to large differences in thunderstorm and tornado occurrence over short distances. Some of these gradients in occurrence are, effectively, perpendicular to the gradient of population density. Because of this, there were regions of high population density and high tornado occurrence, low population density and high tornado occurrence, high population density and low tornado occurrence, and low population density and low tornado occurrence. King's primary result was that it took a relatively low threshold of population density to report tornadoes at a consistent level, given that a system to collect the reports exists.

The one phenomenon that has some systematic data collection that has been analyzed for trends is hail. In China, there are a large number of stations with human observers that report the occurrence (or non-occurrence) of hail on a daily basis and, at many of the sites, record the size of the largest hailstone. Xie et al. (2008) showed a large decrease in the number of days with hail reported since about 1990. Xie et al. (2010) showed no consistent pattern to changes in size of hail, but carried out analyses of radiosonde observations, finding both an increase in CAPE and a rise of the height of the freezing level. The former change should, other things being equal, be favorable for the production of more and larger hail, but the latter would be unfavorable as stones would melt over a larger distance of fall. Berthet et al. (2010) and Eccel et al. (2011) examined hailpad observations in southwest France and northern Italy, respectively, over a number of years. In both cases, there was no statistically significant change in the occurrence frequency over time, but measures that weighted hailfall by the size of the hailstones support the notion that the distribution of hail had shifted towards larger stones. This is conceptually consistent with the Xie et al. results, where larger hail might be formed in higher CAPE environments. The higher freezing levels would lead to greater melting than with lower freezing levels. This is particularly important for small stones, which fall more slowly, losing much more of their mass. Xie et al. indicate that stones that start at 2 cm lose 20% of their diameter as they fall, while 5 cm stones lose less than 5%. Increasing the freezing level would impact the smallest stones the most since they would spend more time in the melting region of the atmosphere, so that, relative to large stones, they lose even more mass. As a result, it is plausible that the distribution of hail on the ground might shift towards larger stones. Unfortunately, the hailpad studies only go back 25–35 years, so it is difficult to have confidence in the long-term trends. It is also important to note that most of the hail the Chinese, French, and Italian studies is relatively small, with almost all of it 2 cm in diameter or smaller. It is quite possible that the impacts of increased instability and higher freezing levels would be small for very large hail (~5 cm or larger). Such very large hail typically forms in supercell thunderstorms, in environments associated with large vertical wind shear. Storms that form in such environments can have their updrafts and, hence, hail size, increased by the interaction between the updraft and the environmental wind shear, meaning that the instability is a poorer estimate of the updraft speed in strongly-sheared environments than in weakly-sheared environments (Brooks and Wilhelmson, 1993). Intriguingly, for the relationship of severe thunderstorms to a warming climate, Berthet et al. (2010) found that the fraction of total precipitation that fell in the warm season as hail increased with the average summer temperature in their region of study, so that in the heat wave of 2003, there was a decrease in precipitation, but a much larger than normal fraction of it fell as hail.

3. Environmental estimates

Given the challenges associated with the use of reports, another approach must be employed in order to investigate the questions at hand. In particular, the use of meteorological covariates (Brown and Murphy, 1996) is particularly attractive. Covariates relate environmental conditions that may be well-observed to weather events that are of greater interest, but are not well-observed. Brown and Murphy (1996) applied the concept to icing and turbulence for aviation weather. For tornadoes, in particular, there is a long history of a related thread of research into so-called proximity soundings (Rasmussen and Blanchard, 1998 and references therein) where observations from radiosondes taken in the vicinity of tornadoes have been used to understand the environmental conditions associated with tornadoes. This understanding has led to many of our current forecasting approaches for severe thunderstorms.

Brooks et al. (2003b) used environmental conditions derived from reanalysis models (Kalnay et al., 1996) with horizontal grid spacing of about 1.9° × 1.9° to estimate the global distribution of severe thunderstorms and tornadoes. Frequency of occurrence of combinations of various atmospheric parameters was determined in phase space associated with two variables derived from the reanalysis and the analysis performed on smoothed fields in that phase space. The relationships between environments and events were derived from the US data and then applied to the rest of the globe, based on a simple discriminant analysis using convective available potential energy associated with a parcel mixed over the lowest 100 hPa (CAPE), the magnitude of the vector difference between the surface and 6 km winds (SHR6), the lifted condensation height of a parcel mixed over the lowest 100 hPa of the atmosphere (MLLCL), the magnitude of the vector difference between the surface and 1 km winds (SHR1), and the lapse rate between 2 and 4 km above ground level, which was used to represent convective inhibition. Severe vs. non-severe thunderstorm discrimination was based on the CAPE and SHR6 terms, while tornadic vs. non-tornadic was based on MLLCL and SHR1, given that conditions had been identified as severe. Recently, Cecil and Blankenship (2012) have used satellite observations to estimate the distribution of hail globally, showing a qualitatively similar pattern to the severe thunderstorm estimate from Brooks et al. (2003b). This provides some support for the reanalysis approach.

In follow-up work, Brooks (2009) used a slightly different technique for smoothing the results of the combining the parameters and found that the probability of occurrence of severe thunderstorms was a function of the distance from the discrimination line found in Brooks et al. (2003b) with increasing probability as CAPE and SHR6 increases. Significantly, they used data from the European Severe Weather Database (ESWD) (Dotzek et al., 2009) and found that the best discriminator line for ESWD data was parallel to the best discriminator from the US data and that, again, probabilities were roughly parallel to that line. Of interest is the fact that for a specific combination of CAPE and SHR6, the probabilities in Europe were higher than in the US data, although the combination of high CAPE and SHR6 was much less likely to occur in Europe than in the US, so that the overall frequency of severe thunderstorms would be expected to be higher in the US than in Europe. The higher probabilities in Europe are likely due to differences in convective initiation (e.g., differences in convective inhibition, sources of initiation such as topographic features). Allen et al. (2011) did a similar analysis for Australia using short-range NWP models as the input for the soundings and found a roughly parallel discrimination line for their dataset. The discrimination line in all cases was approximately

$$\mathrm{CAPE} * \mathrm{SHR6}^{1.6} = k \qquad (1)$$

where k is some constant. (For some datasets, the exponent is closer to 1.7, but the differences in practical application between 1.6 and 1.7 are small and are likely inside of reasonable uncertainty bounds based on sampling, although that has not been quantified.) This relationship indicates that the deep shear is more important than CAPE for discriminating between severe and non-severe thunderstorms, a significant result when future changes in parameters are discussed later. The consistency between the different regional datasets provides encouragement that the discriminant is based on real physical behavior in the atmosphere and not on details in the regions. The different values of k, however, could be determined regionally. Brooks (2009) speculated that the differences reflected issues involving convective initiation, a process that is not resolved in reanalysis scale data.

A larger dataset of reanalysis soundings for the US reveals more information on the different types of severe thunderstorms and their intensity. Following Doswell et al.(2009), indicating that the 1990s seemed to be a reasonably consistent period for rating tornado intensity in the US, every sounding from the period 1991–1999 from the US has been analyzed for its deep-layer shear (SHR6) and the vertical velocity derived from simple parcel theory (WMAX) associated with the CAPE, with

$$WMAX = \sqrt{2 * CAPE} \tag{2}$$

Physically, there is no difference between looking at WMAX and CAPE, but WMAX may offer some advantages in interpretation. For one, the units of WMAX and SHR6 are the same. The notion of smoothing in the two-dimensional phase space becomes somewhat more intuitive. Also, the display naturally compresses the effect of extremely large values of CAPE. Brooks et al. (2003b) attempted to do the latter by plotting the logarithm of CAPE, but that led to large parts of the graphs being relatively “uninteresting” (CAPE between 1 and 100 too up just as much space as CAPE between 100 and 10,000). Although the discrimination capabilities of the different thermodynamic descriptors are similar, WMAX and SHR6 are more similar in their importance for the discrimination with the exponent on SHR6 being ~0.8 (as expected taking the square root of (1)). There are probably advantages to looking at both CAPE and WMAX. Here, we will concentrate on WMAX-SHR6 space, noting that much of what is presented is similar to that shown in Brooks (2009), except for the much larger dataset.

Similar to the approach of Brooks (2009), all soundings from 1991 to 1999 with non-zero WMAX have been analyzed on a grid in two-dimensional phase space (WMAX and SHR6), using a kernel density estimation technique with a Gaussian smoother with $\sigma = 5\ \mathrm{m\ s^{-1}}$ in both directions. The zero WMAX soundings are not considered for two reasons. First, it is not clear what meaning there is in smoothing from 0 to a positive value. The smoother is intended to mimic uncertainty in the parameter values, among other things, but, obviously, some WMAX = 0 soundings are much farther away from having positive WMAX than others. Second, if free convection is the primary concern, it is not clear how to treat zero WMAX. The distribution of sounding values shows that most soundings occur for combinations of relatively small WMAX and SHR6 (Fig.1). This is not surprising and is a reflection of the fact that the atmosphere visits regions of extreme values of environmental parameters relatively rarely.

The distribution in WMAX-SHR6 of significant severe thunderstorm soundings (in US usage, this means a tornado of at least F/EF-2 intensity, hail of 5 cm diameter, and/or wind gusts of at least 120 km h^{-1}) shows that they tend to occur off of the WMAX or SHR6 = 0 axes (Fig. 2). As was done for overall distribution of soundings, these distributions have been smoothed with the same Gaussian smoother. The probability of the event of interest occurring, given the combination of WMAX and SHR6 can be calculated just by dividing the distribution of the weather event by the overall unconditional probability of the sounding being observed. For example, the probability of a significant severe thunderstorm shows much higher probabilities as the WMAX and SHR6 increase, with values at some combinations approaching 10%, 20 times the overall base rate of 0.5% from the dataset. (Fig. 3). Note that this illustrates one of the primary challenges of severe thunderstorm forecasting-low conditional probability events that occur in overall conditions that are relatively common compared to high conditional probability events that occur in rare combinations of WMAX and SHR6. Even though the probability of a significant severe thunderstorm is low for a combination of, say, 10 m s^{-1} of both WMAX and SHR6, those conditions are common enough that a non-negligible number of events occur in those environments, as seen in Fig. 2.

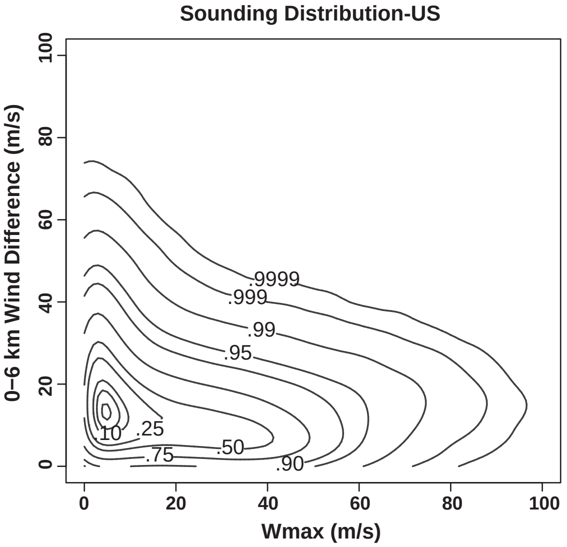


Fig. 1. Distribution of soundings in US from 1991 to 1999 as a function of magnitude of the difference between the near-surface and 6 km above ground level winds (SHR6) and parcel theory vertical velocity (WMAX). Contours represent fraction of observed soundings contained within contour.

As mentioned before, there are similarities in the distribution for the ESWD data (Fig. 4), but notably, the distribution does not extend to as high of values of WMAX, but the conditional probability for severe given that a combination of atmospheric conditions occurs is much higher in the ESWD data. The lack of high WMAX soundings is a result of the lower values of boundary moisture and steep mid-tropospheric lapse rates (Brooks et al., 2007). It is not obvious what the explanation for the higher conditional probabilities is, but it could be related to the frequency of convective initiation.

It is interesting to consider the probability of a particular severe weather event (significant tornado, hail, or wind) given the occurrence of a significant severe thunderstorm (Fig. 5). It is clear, from a cursory examination of these probabilities, that the probability of a significant severe thunderstorm producing tornado or hail increases with increasing values of SHR6, and the probability of wind increases with decreasing values of SHR6. The gradient of probability for tornadoes as a function is SHR6 is particularly

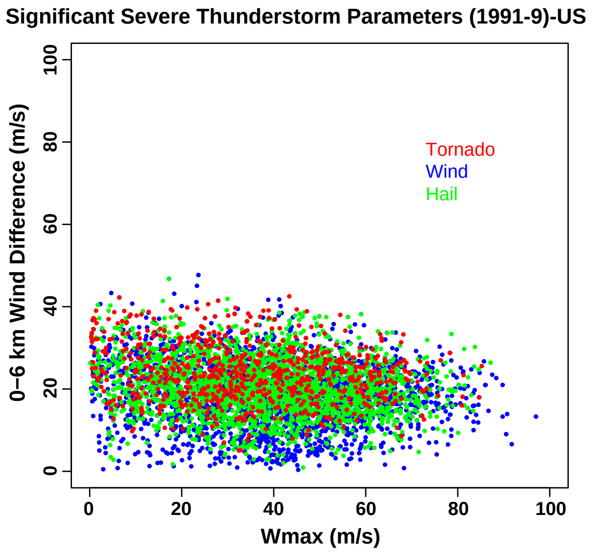


Fig. 2. Location of points of significant severe thunderstorms in WMAX-SHR6 space for US from 1991 to 1999. Tornado in red, wind in blue, hail in green.

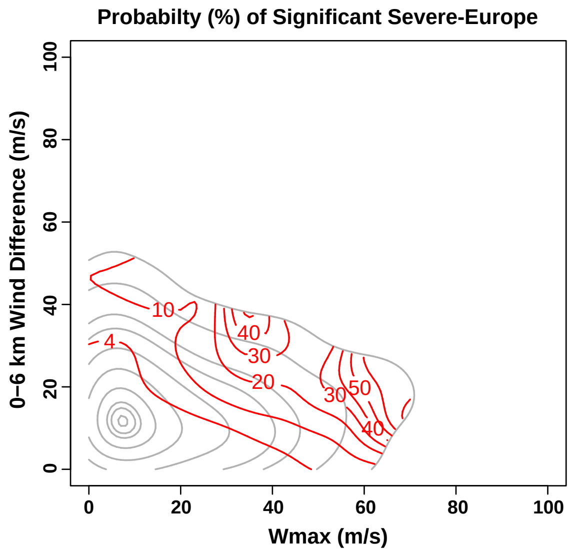


Fig. 4. As in Fig. 3, except for ESWD soundings (1958–1999).

strong. Note that, to first order, there is little to no relationship with WMAX. It appears that once the atmosphere is capable of producing a significant severe storm (for which WMAX is an important component), the results of what comes out of the storm is not a strong function of WMAX. If the shear is large, the storm is much more likely to be tornadic and/or produce hail; if the shear is small, it is much more likely to produce winds. This can be summarized by considering the regions of the phase space where the conditional probability of a particular threat exceeds the base rate probability (Fig. 6). Remarkably, the dividing line between greater and lesser probability compared to the base rates is almost the same line for all three threats over a broad range of WMAX. The influence of shear is consistent with the annual cycle of different kinds of severe thunderstorms in the US. Tornadoes and hail tend to peak in occurrence in the spring and wind peaks in the summer (Brooks et al., 2003a; Doswell et al., 2005).

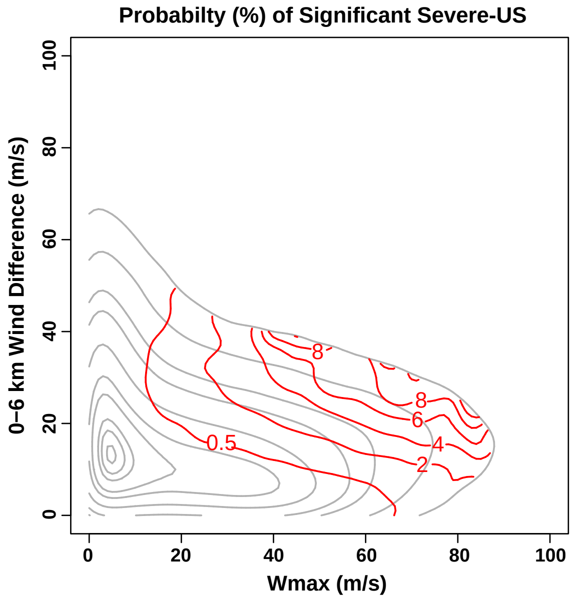
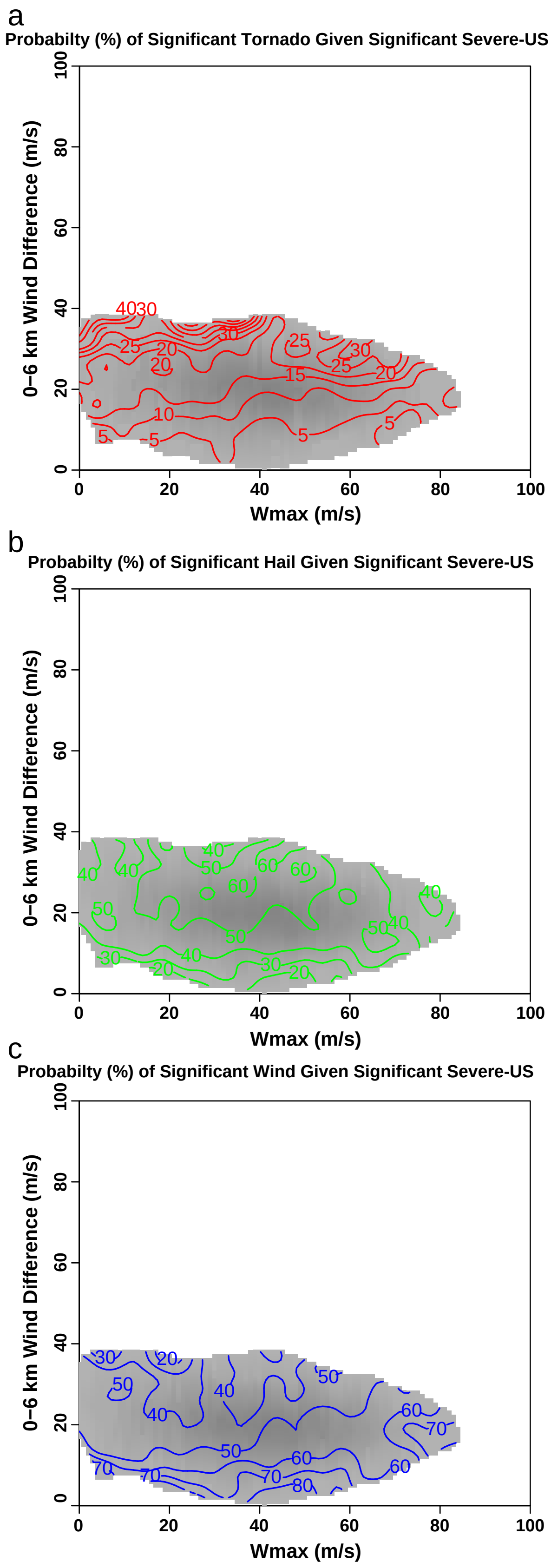


Fig. 3. Probability (%) of significant severe thunderstorm given combination of environmental conditions (red). Overall distribution of soundings from Fig. 1 shown in gray.

The importance of shear on what comes out of the storm is further illustrated if the probability of a very severe event, given a significant severe event, is considered. The probability of a storm producing an F3 tornado, given that an F2 tornado occurs, is, again, almost entirely a function of SHR6 (Fig. 7a). There is a slight dependence on WMAX in very large hail (Fig. 7b), but it is still mostly SHR6. The exception is in the distribution of very strong winds (approximately 140 km h^{-1}) (Fig. 7c). Winds do not seem to have as systematic dependence on these parameters. This could be a result of issues associated with the collection of the data or it may reflect the complex dynamical processes that lead to very strong winds. In any event, SHR6 is significantly more important in determining the intensity of tornadoes and hail than WMAX is. Grunwald and Brooks (2010) examined tornado data from the ESWD and also found that the probability of strong tornadoes (F2 given the occurrence of any tornado) was primarily a function of SHR6.

4. Modelling studies

The results in Figs.3–5 provide us with important guidance as to how future climate change might impact the occurrence of severe thunderstorms, but that depends upon the ability of models to depict the current and future distributions of environments. In recent years, a number of such studies have been carried out. Marsh et al. (2007, 2009) used the discriminator from Brooks et al. (2003b) and showed that global climate models (GCMs) are capable of producing reasonable spatial distributions of severe thunderstorms in the US and Europe, although there were



concerns about interpreting the magnitudes. Niall and Walsh (2005) and Leslie et al. (2008) attempted to estimate future changes in hailstorm occurrence in southeastern Australia and came to different conclusions, primarily because of differences in how the environments were estimated to change in their models.

More work that has yielded relatively consistent results has been done in the US. Del Genio et al. (2007), Trapp et al. (2007), and Van Klooster and Roebber (2009) all looked at late 21st century climate compared to late 20th century climate. Their results were similar, with CAPE (or WMAX) increasing over most of the US east of the Rockies, driven by increases in boundary layer moisture as surface temperatures warm, consistent with the observations of Peterson et al. (2011). SHR6 decreased over much of the US, driven at zeroth-order by a reduction of the equator-to-pole temperature differences and the associated thermal wind changes. The Del Genio and Trapp studies both showed that the increase in the thermodynamic parameter would be greater than the decrease in the SHR6 term, leading to more frequent favorable environments for severe thunderstorms. A question of interest, however, is if and how convective initiation might change. Just because a storm that formed in an environment would be severe does not mean that a storm would form at al. In general, for example, for a similar bulk value of CAPE and SHR6 in the southeastern US, convective initiation in much more likely in cool season than in the warm season. In part, this is a result of greater convective inhibition in the warm season and the lack of synoptic-scale forcing and significant fronts in the warm season. The difference in synoptic settings between the US and Europe is likely to be particularly important in determining frequency of initiation, given that the bulk of Europe is north of the bulk of the US and, as a result, frontal passages are more common even in the warm season. In addition, the presence of the Alps across a broad area of central Europe increases the influence of complex terrain in comparison to the US. This could have influences on how we can look at the ESWD, given that the largest number of reports in it come from Germany. It is not clear how to deal with this issue.

Trapp et al. (2009) broke the US down into regions and looked at a long transient climate simulation (1950–2100) and attempted to include convective initiation by considering the response of the convective parameterization of the model. They found, in general, an increase in CAPE, and a decrease in SHR6 in all regions, with an increase in favorable environments, particularly in summer. There was no clear signal in changes of relative frequency of convective inhibition for a particular combination of parameters during a given season of the year through the length of the run, but that could depend on the nature of the convective parameterization scheme used. There was very large interannual variability in the number of favorable environments, however, so that statistical significance was limited, although the overall picture supported that seen in Del Genio et al. (2007) and Trapp et al. (2007). Note that Trapp et al. (2009) looked

Fig. 5. Probability of individual threats given significant severe thunderstorm. Gray background represents smoothed field of significant severe thunderstorm soundings from Fig. 2. a) tornado (red, contour 5%), b) hail (green, contour 5%), and c) wind (blue, contour 5%).

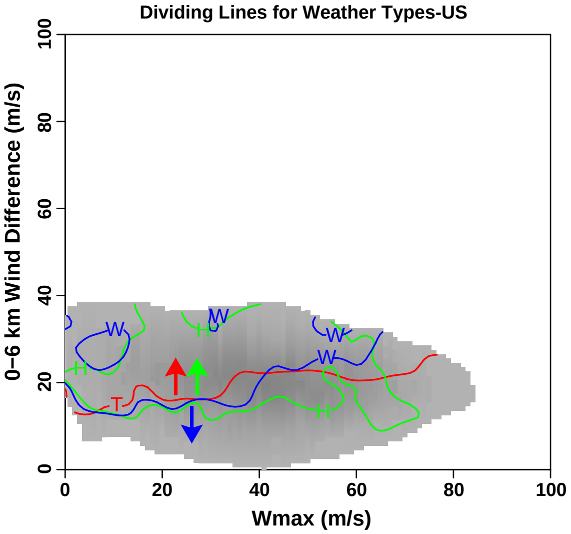


Fig. 6. Dividing lines between greater and lesser probability than overall base rate for occurrence of significant tornado (red), hail (green), and wind (blue). Arrows point towards general region of higher probability.

at the combination of individual favorable parameters (CAPE, SHR6) occurring together at individual points and times. It appears that the changes in the count of how often the favorable parameters occur together changes in concert with the overall mean changes in CAPE and SHR6. Obviously, it is conceptually possible that, even if the mean SHR6 goes down, there could be more days with large values of SHR6 or that the only days with high values of SHR6 would be days with large CAPE. However, it appears that, in the model simulations, this is not the case. The changes in the means of the parameters reflect the changes in the distributions of the parameters and the covariance doesn't appear to change significantly. As a result, the mean behavior of CAPE and SHR6 provide a large fraction of the information on the changes in the joint distribution of the two parameters.

Changes in the deep shear in the central US from the 20th century reanalysis dataset (Compo et al., 2011) are of a similar magnitude (0.5 m s^{-1} difference) from 1950 to 2008 as seen in Trapp et al. (2009), but there is no statistical significance yet, because of the large interannual variability (Fig. 8). This is of interest because the Trapp et al. (2009) results suggest that the SHR6 decrease will become statistically significant, but hadn't in the first half century of their simulation. The consistency of these two results means that there could have been physical changes in the distribution of shear in the past half century as a result of the planet warming, but they would not have been statistically significant. Complicating the interpretation of future changes, not all models agree on how shear and instability will change. Vecchi and Soden (2007) showed that roughly 2/3 of the simulations for the IPCC 4th Assessment Report indicated that shear would decrease over the US in summer and fall

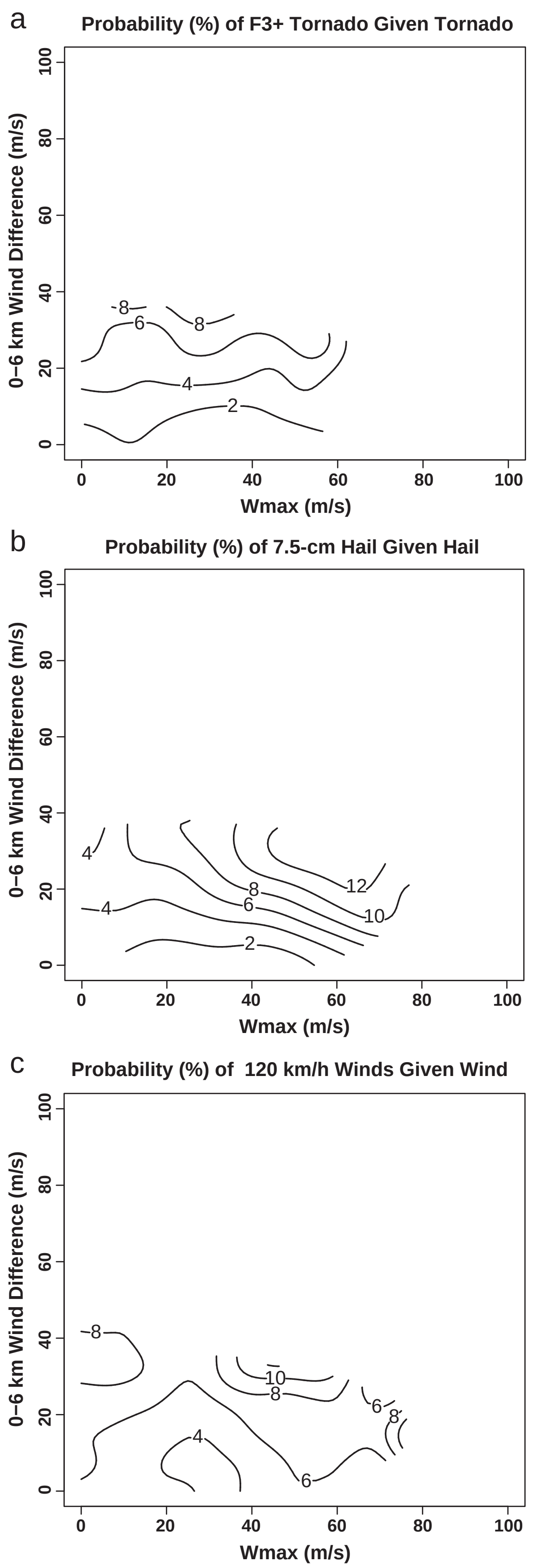


Fig. 7. Probabilities (%) of extreme events given significant severe weather event. a) F3 tornado given significant tornado, b) 7.5 cm diameter hail given significant hail, c) 120 km h^{-1} wind gusts given significant wind gust.

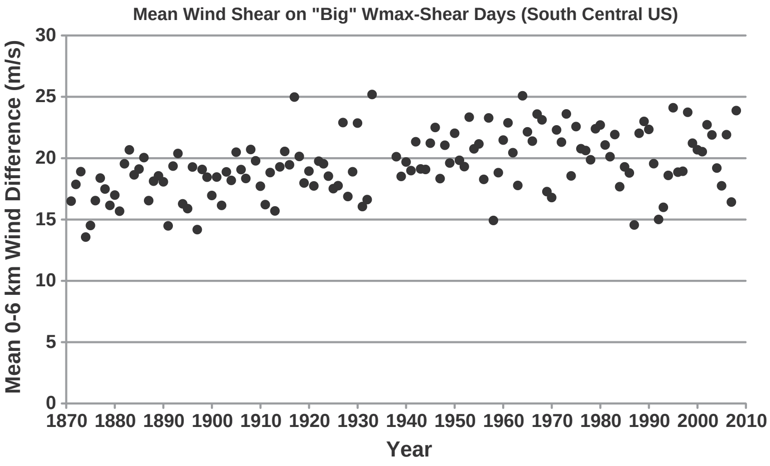


Fig. 8. Mean value of SHR6 for soundings in upper 10% of CAPE*SHR6 for May for south-central US by year. Data from 1934 to 1937 missing.

(their emphasis was on tropical cyclones). Soden (personal communication, 2011) reported that, for the IPCC AR5 simulations, 2/3 showed an increase in CAPE and a decrease in SHR6. This is similar to the Del Genio and Trapp simulations, but clearly shows that we cannot have complete confidence in the result.

5. Discussion and future directions

Observations suggest that some measure of atmospheric thermodynamic energy available for storms (CAPE or WMAX) and deep tropospheric shear (e.g., SHR6) provide reasonable discrimination between significant severe and non-severe thunderstorms. In general, SHR6 seems to be more important for the development and intensity of severe thunderstorms than CAPE is. High SHR6 environments favor tornadoes and hail, while low SHR6 environments favor non-tornadic wind events. Also, the intensity of tornadoes and hail is almost entirely a function of SHR6.

Our naïve expectations of how environments will change as the planet warms is that, on average, CAPE will increase as the surface temperature and boundary layer moisture increases and SHR6 will decrease as the equator-to-pole temperature gradient decreases. These expectations are supported by a majority of the climate model simulations that have looked at the variables. Those simulations, in general, suggest that the increase in CAPE will more than offset the decrease in SHR6 over the US, leading to more frequent environments favorable for severe thunderstorms. The change in SHR6, however, could impact the relative fraction of events by different weather types. We can look at this by carrying out a simple gedanken experiment where the distribution shown in Fig. 1 is simply shifted around and the probabilities seen in Figs. 3 and 5 are assumed to be constant as the overall distributions of environments change (Fig. 9). Assuming an increase in WMAX, tornadoes are much less likely to increase and, in fact, are more likely to decrease, for a particular decrease in the mean shear. A change in WMAX of 2 m s^{-1} is equivalent to a change in CAPE from 1000 to 1100 J kg^{-1}, on the order of what is seen in the model simulations over the US. For a plausible decrease in SHR6 (again based on the model simulations) of 1 m s^{-1}, that would lead to little change or, perhaps, a slight decrease in relative tornado and hail occurrence. Winds are more likely to increase. If true, a shift towards relatively more non-tornadic wind events will be seen in the future and the distribution tornadoes and hail will shift towards slightly less intense events. However, if the number of overall favorable environments increases, there may be little change, if any, in the number of tornadoes or hailstorms in the US, even if the relative fraction decreases. The signals in the climate models and our physical understanding of the details of storm-scale processes are sufficiently limited to make it extremely hazardous to make predictions of large changes or to focus

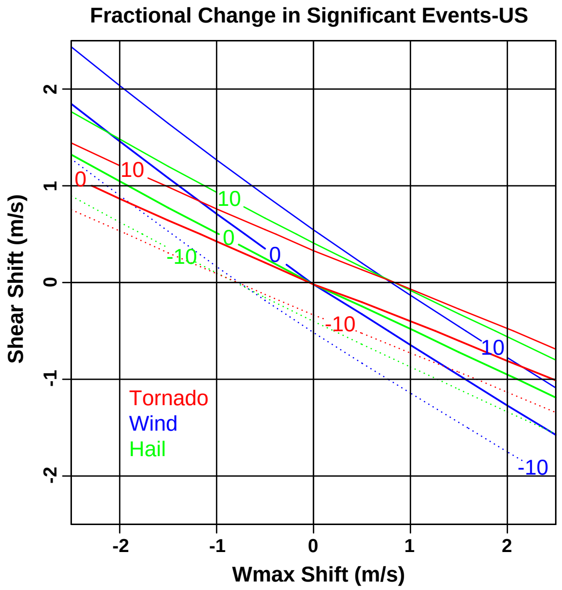


Fig. 9. Fractional change in severe weather occurrence given change in distribution of large-scale environments in US. Axes are shifts in distribution of soundings from Fig. 1. Heavy line represents line of no change, light line 10% increase in event, dotted line 10% decrease. Tornado in red, hail in green, wind in blue.

on small regions. Projected changes would be well within error estimates.

It is important to note that most of the work done has focused on the US and it is unclear how well the lessons learned there apply to the rest of the world. As shown in Brooks (2009) and Grunwald and Brooks (2010), even though CAPE and SHR6 appear to be important in both the US and Europe, the quantitative relationships are different. Also, physically, in the US, the presence of the Rocky Mountains as a source of steep mid-tropospheric lapse rates and the Gulf of Mexico as a source of warm air is critical for the environmental conditions. Signals in Europe are not as strong (e.g., Brooks et al., 2007) and it is not clear changes in boundary-layer moisture in Europe will be driven as directly by global temperatures as in the US. The role of the Mediterranean as a moisture source with the dry Sahara to the south complicates the problem. Trajectories carrying moist air into North America from the Gulf of Mexico are unlikely to have passed over a significant land mass for a considerable period of time, in contrast to the situation in Europe where the Mediterranean is relatively small from south to north and is enclosed by landmasses. On the other hand, because it is enclosed, it is not clear that the Mediterranean will respond to warming in the same way as the open Gulf of Mexico and Caribbean might. Thus, even though development of better relationships between the environments and storms is needed everywhere, it is particularly true outside of the US. This necessitates the development of more complete databases of observed events.

Trapp et al. (2011) have started investigating the use of high-resolution models (grid spacing of 4 km) embedded within global models to look at more direct simulations of a variety of hazards. The approach outlined in Brooks et al. (2003b) requires the development of relationships between large-scale environmental conditions and small-scale weather events. This could be thought of as equivalent to forecasting a high probability of severe thunderstorms with a lead-time of 24 h over large regions. The Trapp et al. approach is similar to modeling the warning process for a particular storm on time scales of 10s of minutes and space scales of 10s of km. It amounts to letting the numerical model develop the relationship between the environment and event with variables that are used in the interpretation of numerical models for operational weather forecasting. This has a great deal of promise for optimizing that process of building the relationships, but has the drawback of being computationally expensive.

The question of how severe thunderstorms will change in the future is still an open one. The quality of observational databases of severe thunderstorm reports is low enough that we cannot say if changes have already occurred with any degree of confidence. By their very nature, severe thunderstorms are rare events at any location so that analysis of time series of reports is a problem with a low signal-to-noise ratio. The presence of singular, or near-singular events, in the record cannot be used as evidence of trends.

Acknowledgments

Conversations over the years with a number of colleagues have been enlightening. Of particular note are Jeff Trapp, Nikolai Dotzek, David Karoly, Tony Del Genio, Patrick Marsh, Chuck Doswell, Greg Carbin, and John Allen. I would not claim that they all agree with everything written here, but they have influenced my perception of the issues. Brian Soden provided information on the AR5 simulations. Several undergraduate students participated in research relevant to this work in a variety of settings—as participants in the National Science Foundation Research Experiences for Undergraduates (Heather Flachs), as Ernest Holings NOAA Scholars (Jace Bauer, Alicia Klees), and as visiting students from Hamburg University (Kathrin Riemann-Campe, Irina Ebbers, Stefanie Grunwald). James W. Lee, Patrick Marsh, Aaron Anderson, and Nick Benson all helped in processing the global reanalysis data.

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GROWTH AND DEVELOPMENT OF YELLOW-POPLAR PLANTATIONS ON THREE SITES RANGING FROM 9 TO 18 YEARS

Wayne K. Clatterbuck[1]

Abstract—Planting pine for conversion of former agricultural land to managed forests is well-documented, but little information is available for hardwood plantings. This study references three yellow-poplar (*Liriodendron tulipifera* L.) plantations (ages 9, 12, and 18 years) on different sites totaling 95 acres. The sites were located in middle Tennessee along a site productivity gradient composed of a floodplain and terrace site near the Cumberland River and an upland footslope site. All three sites had been in row crops or pasture prior to planting. Tree survival at time of measurement was greater than 75 percent for all three plantings. Average annual diameter increment and height increment for the three plantations were 0.5 inch and 4.2 feet, respectively. Comparisons of age, height, diameter, basal area, density, site index, and volume are made between the planted yellow-poplar in this study and three other published studies: natural stands of yellow-poplar and two studies with planted loblolly pine (*Pinus taeda* L.). Although site productivity and stand parameters differ among the studies, the growth of yellow-poplar plantations compares favorably with the natural stands of yellow-poplar and planted loblolly pine.

INTRODUCTION

Tennessee is 55 percent forested (14.4 million acres), and 89 percent of the composition is in hardwoods (Schweitzer 2000). The state has gained nearly 1.4 million acres of new forest land during the last 10 years, with about 600,000 acres reverting to other uses (May 1991, Schweitzer 2000). Through the Conservation Reserve Program, many acres of former agricultural land have been planted to pine. However, most nonindustrial forest landowners in Tennessee prefer to plant hardwood. The incentives for planting pine are poor. Tennessee has a lack of competitive markets for pine resulting in the lowest prices for pine pulpwood and sawtimber in the Southeast (Timber Mart-South 2002). Additionally, the recent southern pine beetle epidemic in Tennessee has killed nearly 400,000 acres of both natural and planted pine, about 33 percent of the total acreage categorized in pine types (Kauffman 2001).

Little information is available on afforestation of hardwoods on the better, more productive sites. Loblolly pine is not as site specific as yellow-poplar which typically occurs on the better sites, with a site index of 80 feet or greater at base age 50 years (Beck and Della-Bianca 1981). Yellow-poplar, with its fast growth and straight form, is a likely candidate for planting, especially on the better sites. The objectives of this study are two-fold: (1) to determine height, diameter and survival of three yellow-poplar plantations (planted in 1985, 1991, and 1994) on similar sites in Tennessee and (2) to compare growth of these plantations with natural yellow-poplar stands and planted loblolly pine as reported in the literature.

STUDY AREA

The study was conducted on three yellow-poplar plantations in Smith and Trousdale Counties in middle Tennessee at the transition between the Eastern Highland Rim and Nashville Basin Physiographic Regions. Braun (1950) described the vegetation of this area as part of the Western Mesophytic Forest, a transition area between the mixed Mesophytic Forest Region of the mountains to the east and the Oak-Hickory Region to the west. The three sites are within 10 miles of each other. Characteristics of each plantation site are as follows:

Site 1 is on a terrace of the Cumberland River. Soils are Ultic Hapludalphs (Armour series) formed in old alluvium overlying or in residuum derived from limestone (USDA-NRCS 2002). It was planted in 1991 and consisted of 32 acres. Site 2 is on the floodplain of the Cumberland River. Soils are Cumulic Hapludolls (Arrington series) formed in silty alluvium on floodplains and along drainageways (USDA-NRCS 2002). It was planted in 1994 and consisted of 25 acres. Site 3 is on the well-drained, footslopes. Soils are Typic Paleudults (Dellrose series), typically deep and well drained, formed in cherty colluvium (USDA-NRCS 2002). It was planted in 1985 and consisted of 39 acres.

Annual precipitation is 51 inches, usually evenly distributed in all seasons. Average site index (base age 50) is from 80 feet for upland oaks to 100 feet for yellow-poplar (Smalley 1983). All three study sites were former agricultural fields that were either row-cropped or in pasture. Each area was site prepared during the fall for planting the following year. Sites were cross-directionally disked followed by subsoiling in one direction. Each site was hand-planted at a 10 by 10-foot spacing using 1-0 yellow-poplar seedlings produced at the Tennessee Department of Agriculture, Division of Forestry nursery. The study sites, though planted at different times (years), were supervised by the same forester using the same procedures.

The only competition control used on these plantations after planting was mowing once or twice a year for the first 2 or 3 years. No herbicide was used during site preparation or following planting.

METHODS

Each plantation was sampled during the fall of 2002. Three 0.1-acre rectangular plots were randomly located in each plantation. Although a larger sample size might be desirable

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with the size of these plantations, each plantation was fairly homogenous with little tree variation. Thus, larger sample sizes were deemed unnecessary. Diameter at 4.5 feet diameter at breast height (d.b.h.) was measured for each tree in the plot. Total height was recorded from a subsample of three dominant or codominant stems. Empty cells (missing trees) in each plot were recorded to determine survival percentages.

Data from this study on planted yellow-poplar stands is compared to three other studies: natural yellow-poplar stands (Beck and Della-Bianca 1970, 1981), loblolly pine plantations on abandoned fields in Tennessee, Alabama and Georgia (Smalley and Bailey 1974, Smalley and Bower 1968), and loblolly pine plantations on cutover sites in the West Gulf Coastal Plain (Feduccia and others 1979).

RESULTS

Growth and stand parameters of each of the three yellow-poplar plantations through the 2002 growing season are shown in table 1. Average annual diameter growth rate of all sampled trees ranged from 0.45 to 0.63 inch per year across the three sites. Average annual height growth rate of dominant trees ranged from 4.0 to 4.4 feet per year. If height growth continues at this rate, site index for yellow-poplar is estimated to be 115 to 120 feet at 50 years (Beck 1962), much greater than that estimated by the soil survey reports (USDA-NRCS 2002) and by Smalley's (1983) landtype classification system. Although Beck's yellow-poplar site index curves were formulated for the Southern Appalachian Mountains, they are the best available for the Eastern Highland Rim of Tennessee.

Table 1—Growth and stand parameters of three yellow-poplar plantations after the 2002-growing season in Smith and Trousdale Counties in Middle Tennessee

	Site 1	Site 2	Site 3
Growing season (yrs)	12	9	18
Mean dominant height (feet)	53	40	73
Mean diameter (inches)	7.6	5.3	8.5
Survival (percent)	76	87	78
Largest diameter tree (inches)	11.4	6.7	11.6
Basal area (ft2 per acre)	101	56	128
Number of measured trees (n)	96	110	98

Tree survival decreased as age of the stand increased (table 1). As trees became larger, the amount of available growing space became more limited causing mortality of lesser stems. Basal area per acre also increased with age.

Table 2 shows the stand, growth and yield parameters of the planted yellow-poplar at age 18 for site 3, natural yellow-poplar stands at age 20, and two growth and yield studies for planted loblolly pine at age 20. The planted yellow-poplar in this study had greater average diameters and dominant heights than the two loblolly pine studies and the natural yellow-poplar stands. The basal areas and volumes per acre fluctuated between studies presumably because of the number of trees per acre originally planted and the number of trees currently present. Site productivity (as measured in volume) in this study is similar to those in the other studies even though as measured by site index, the numerical value between species (yellow-poplar and loblolly pine) is expected to differ.

DISCUSSION

Each of these studies is based on different assumptions and parameters that make direct comparison of growth and yield difficult. The work of Feduccia and others (1979) was based on cutover sites without intensive site preparation before planting. The assumption was that intensively managed loblolly pine plantations, where pine density is carefully controlled throughout the rotation and utilization is more complete at final harvest, would need minimal site preparation for repeated crops of pine. The West Gulf Coastal Plain also has much different soils and geology than middle Tennessee.

Pine plantations measured to predict growth and yield in the Highland regions of Tennessee, Alabama, and Georgia

Table 2—Stand, growth and yield comparisons of planted yellow-poplar, natural yellow-poplar stands and two planted loblolly pine studies

	Planted YPP[a]	Natural YPP[b]	Loblolly pine[c]	Loblolly pine[d]
Age (years)	18	20	20	20
Mean height-overstory (feet)	73	66	61	53
Mean diameter (inches)	8.5	8.2	7.8	8.2
Basal area (ft^2 per acre)	128	112	144	112
Density (trees per acre)	325	310	436	303
Site index at 50 years (feet)	120 YPP	115 YPP	105 LP	105 LP
Average volume (ft^3 per acre)	3,116	2,939	3,625	2,920

YPP = yellow poplar.

[a] Eastern Highland Rim of TN (site 3 of this study) planted at 435 trees per acre.

[b] Southern Appalachian Mountains (Beck and Della-Bianca 1970).

[c] Abandoned fields of Highland areas in TN, AL and GA planted at 500 trees per acre (Smalley and Bowers 1968, Smalley and Bailey 1974).

[d] Cutover sites in the Gulf Coastal Plain planted at 400 trees per acre (Feduccia and others 1979).

(Smalley and Bailey 1974, Smalley and Bower 1968) were based on plantations established on agricultural fields with little evidence of damage from snow, ice, insects and disease; where stands have not been burned, thinned, or pruned; and where survival and tree distribution were judged reasonably good and no reinforcement planting had occurred. One would expect some degree of site preparation was used to establish loblolly pine on old-fields or pastures before planting. However, there is no mention in the text of site preparation or even if site preparation was considered in the selection of sample plantation sites. The sites in this study, although more upland, were more similar to the soils and geology found in middle Tennessee.

The natural yellow-poplar stands (Beck and Della-Bianca 1970) are from the mountains and coves of western North Carolina and north Georgia. Geology and soils are different from that of the study site in middle Tennessee. However, the differences between the planted stand and the natural stand are probably based on the patchiness that occurs with stems in natural stands and the more structured spacing in planted stands. Beck and Della-Bianca also chose to base their growth and yield predictions on the largest 25 yellow-poplar trees per acre in unthinned second-growth Southern Appalachian stands. These trees would be 3 to 5 inches larger than the average dominant at comparable ages (Beck and Della-Bianca 1970).

In this study, the fields were disked twice in perpendicular directions and subsoiled before planting. Disking and subsoiling are common practices to break plowpans or fragipans and soil compaction in old fields. This more intensive site preparation may have attributed to better growth and survival of planted seedlings when compared to the other studies. The fields were mowed once or twice a year for 2 or 3 years to control competition from grass and other broadleaf vegetation. Although mowing reduces the height of competing vegetation, the roots are still present, affecting the moisture uptake of planted seedlings. Most pine planting today uses a herbicide release to aid the early growth of pine seedlings. A herbicide release probably would have assisted the growth of planted yellow-poplar in this study.

SUMMARY

Although this study comparing growth, yield, and stand parameters of planted yellow-poplar to natural yellow-poplar and planted loblolly pine are not directly comparable because of the different site conditions, stand parameters, and assumptions made in collecting information for each study, it appears that the growth and yield of yellow-poplar and loblolly pine are similar. With height growth rates of greater than 4 feet per year and diameter growth rates of 0.5 inch per year after 18 years, yellow-poplar can achieve growth rates comparable to, if not greater than, loblolly pine especially on the more productive sites. Thus, hardwood planting of yellow-poplar is practicable on the studied areas as an alternative to pine planting. Site productivity for yellow-poplar is greater than hypothesized by the soil survey estimations (USDA-NRCS 2002) and Smalley's (1983) site classification system in the first 20 years after planting.

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An Ecoregion-Based Approach to Protecting Half the Terrestrial Realm

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We assess progress toward the protection of 50% of the terrestrial biosphere to address the species-extinction crisis and conserve a global ecological heritage for future generations. Using a map of Earth's 846 terrestrial ecoregions, we show that 98 ecoregions (12%) exceed Half Protected; 313 ecoregions (37%) fall short of Half Protected but have sufficient unaltered habitat remaining to reach the target; and 207 ecoregions (24%) are in peril, where an average of only 4% of natural habitat remains. We propose a Global Deal for Nature—a companion to the Paris Climate Deal—to promote increased habitat protection and restoration, national- and ecoregion-scale conservation strategies, and the empowerment of indigenous peoples to protect their sovereign lands. The goal of such an accord would be to protect half the terrestrial realm by 2050 to halt the extinction crisis while sustaining human livelihoods.

Keywords: Nature Needs Half, Aichi target 11, ecoregions, protected areas, global biodiversity conservation strategies

Protected areas are the cornerstone of biodiversity conservation (Coetzee et al. 2014, Wuerthner et al. 2015). Where networks of protected areas are large, connected, well managed, and distributed across diverse habitats, they sustain populations of threatened and functionally important species and ecosystems more effectively than other land uses (Noss and Cooperrider 1994, Gray et al. 2016). Protected areas also play an important role in climate-change mitigation (Baker et al. 2015, Melillo et al. 2015). Recognizing the importance of protected areas for conserving nature and its services, the Convention on Biological Diversity (CBD) established a goal to protect 17% of terrestrial land and inland water areas by 2020 through Aichi target 11. To date, approximately 15% of global land is protected (UNEP-WCMC and IUCN 2016).

Aichi target 11 is achievable but insufficient. Seventeen percent is not a science-based level of protection that will achieve representation of all species or ecosystems in protected areas and the conservation of global biodiversity, as are required by the CBD (Noss et al. 2012, Butchart et al. 2015, Wilson 2016). In contrast, reviews of conservation plans by Pressey and colleagues (2003) and Noss and colleagues (2012) demonstrated the scientific basis for a 50% protection target to achieve comprehensive biodiversity conservation. Authors of ecoregion-scale conservation plans from a variety of habitats who empirically evaluated what is required to represent and protect habitat and ecosystems (including marine) have agreed on the need to conserve about half of a given region (Noss and Cooperrider 1994, Pressey et al. 2003, Noss et al. 2012, O'Leary et al. 2016).

More recently, the scientific basis for protecting half the terrestrial realm was strengthened by Wilson's (2016) analysis of extinction in relation to area of natural habitat loss, of greatest concern in habitats rich in endemic species.

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Even before these biodiversity-based analyses of the land area required for conservation, Odum and Odum (1972) pointed to the need to conserve half of the land to maintain ecosystem function for the benefit of humans. On the question of how much to conserve, a species-conservation approach derived the same answer as an ecosystem-services paradigm—a striking example of convergence. Therefore, the aspirational goal of 50% protected has emerged and the science codified in several advocacy and policy papers under the name *Nature Needs Half* (NNH; e.g., Locke 2013).

Nature Needs Half addresses the spatial dimensions of conservation biology, which comprises four goals: (1) represent all native ecosystem types and successional stages across their natural range of variation, (2) maintain viable populations of all native species in natural patterns of abundance and distribution, (3) maintain ecological and evolutionary processes, and (4) address environmental change to maintain the evolutionary potential of lineages (Noss and Cooperrider 1994). Here, we evaluate progress toward Nature Needs Half within the framework of ecoregions, protected areas, and habitats. We answer two basic questions that must be addressed: (1) Is the aspirational goal of protecting half of nature in the terrestrial realm possible? (2) Which half should be protected, and how much of it has already been conserved?

To address these questions and enhance systematic planning for terrestrial biodiversity conservation, we revised the 2001 map of terrestrial ecoregions of the world (supplemental appendix S1; Olson et al. 2001). We then determined the extent of both protected areas and remaining natural habitat within each ecoregion. To designate the protected area network, we used the World Database of Protected Areas (UNEP-WCMC 2016), which is inclusive of International Union of Conservation of Nature (IUCN) categories I to VI (Dudley 2008), as well as many community conservancies, aboriginal ownership, and private lands without an IUCN category. To assess habitat, we used tree-cover maps in forested ecoregions (Hansen et al. 2013) and excluded globally significant patterns of human land use and populations (anthropogenic biomes, or “Anthromes”) in nonforested ecoregions (Ellis et al. 2010; detailed methods in supplemental appendix S2).

We conducted this analysis across all 846 terrestrial ecoregions distributed among the Earth’s 14 terrestrial biomes (supplemental appendix S1). We then sorted ecoregions into four categories defined by the extent of both remaining natural habitat and protected land:

(1) *Half Protected*: More than 50% of the total ecoregion area is protected. (2) *Nature Could Reach Half*: Less than 50% of the total ecoregion area is protected but the sum of total ecoregion protected and unprotected natural habitat remaining is more than 50%. Ecoregions in this category have enough remaining natural habitat to reach Half Protected if additional protected areas or other types of conservation areas are added to the system. (3) *Nature Could Recover*: The sum of the amount of natural habitat remaining and the amount of the total ecoregion that is protected is less than 50% but more than 20%. Ecoregions in this category would require restoration to reach Half Protected because the amount of available habitat outside protected areas plus the existing protected areas is below 50%. (4) *Nature Imperiled*: The sum of the amount of natural habitat remaining and the amount of the total ecoregion that is protected is less than or equal to 20%.

In many Nature Imperiled ecoregions, the remaining habitat exists as a mosaic of isolated fragments insufficient in size and orientation to adequately conserve biodiversity (Wilson 2016). We recognize that in the most heavily altered ecoregions, achieving Half Protected is inconceivable because of extreme rates of conversion. For example, in the tall grass prairie ecoregions of the United States and Canada, 99% of the land area is devoted to agriculture—an active land use that is unlikely to transition back to natural habitat.

To determine the shortcomings in conservation even where protected areas exist, we conducted a global survey of terrestrial ecoregions for which strategies to achieve long-term conservation goals have been developed. For each strategy, we assessed the extent to which all four goals of biodiversity conservation are addressed (appendix S3).

Evaluating protected area networks using ecoregions

The 2001 map of the terrestrial ecoregions of the world (Olson et al. 2001) facilitated the design of representative networks of protected areas. It has also been used to depict species distributions, to model the ecological impacts of climate change, to develop landscape-scale conservation plans, and to report on progress toward international targets. The revised map, named Ecoregions2017$^{\text{©Resolve}}$, that is the basis for this scheme is unchanged for large sections of the seven biogeographical realms but differs from the original map in four regions: the Arabian Peninsula, some of the desert and drier ecoregions of the African continent, Antarctica, and the southeastern United States (figure 1). Further details and justification for changes are presented in supplemental appendix S1.

Calculating the extent of protection by ecoregion and biome provides a scorecard to measure progress toward Half Protected (table 1, figure 2). Summing across all 14 biomes and their constituent 846 ecoregions, 98 ecoregions (12%) have already achieved Half Protected. The largest category is Nature Could Reach Half, with 313 ecoregions (37%), followed by the 228 ecoregions classified as Nature Could Recover (27%). Half Protected remains a reasonable goal in these regions. Within Nature Could Reach Half, 119 (38%) ecoregions have greater than 20% of their land area protected; the remaining 194 ecoregions (62%) have limited coverage of protected areas but retain considerable intact natural habitat. To achieve Half Protected, these 313 regions require only an expansion of their protected area network. The remaining 207 ecoregions (24%) classified as Nature Imperiled have little natural habitat and will require intensive efforts to achieve Half Protected or even to conserve the fraction that remains.

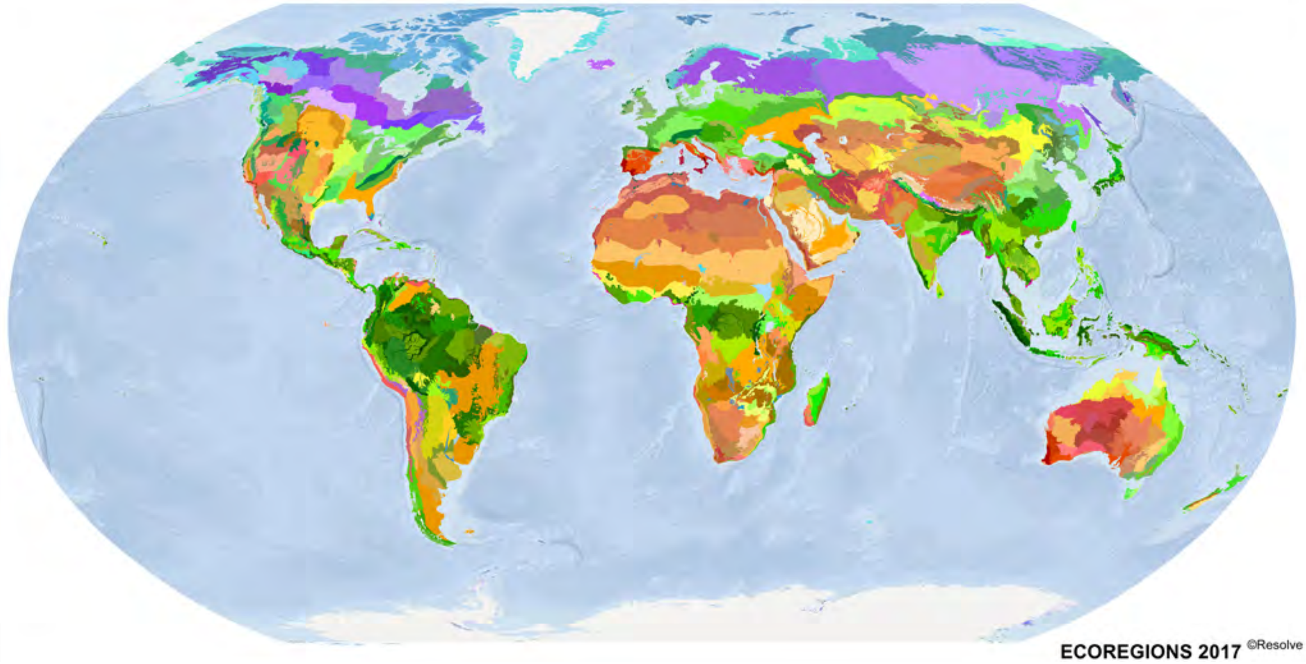

Figure 1. The 846 global ecoregions that comprise Ecoregions2017©Resolve nested within 14 terrestrial biomes. An interactive map is available at ecoregions2017.appspot.com. (A companion biome map is presented in supplemental appendix S1, supplemental figure S1).

Analyses conducted at a global scale inevitably involve error. Here, we were unable to differentiate "paper parks"—designated protected areas that remain unprotected because of lack of enforcement—from those that are well managed. Protected areas subjected to severe bushmeat-hunting pressures or overgrazing by domestic livestock are also ignored at this scale, although these are major threats. There are also protected areas where activities (e.g., industrial extraction) have been expressly allowed by governments even though these activities are plainly inconsistent with conservation objectives. We elucidate the major sources of error, including the assessments of tree-cover change and land-cover classes, in supplemental appendix S2.

Forested ecoregions and biomes. The 476 forested ecoregions are distributed unevenly among each of the four categories of protection: 40 (8%) achieve Half Protected; 198 and 130 fall into Nature Could Have Half and Nature Could Recover categories, respectively; and 108 are classified as Nature Imperiled.

The tropical and subtropical moist broadleaf forests biome has more species and ecoregions than any other on Earth. Covering only 14% of the Earth's land area, this biome supports at least 50% of the world's species (table 1), many of which have likely yet to be discovered (Mora et al. 2011). Fortunately, over half (61%; 140) of the ecoregions within this species-rich biome ($n = 230$) fall into the Half Protected or Nature Could Reach Half category: 24 (10%) ecoregions have achieved Half Protected (table 1, supplemental appendix S2), and 116 (50%) have achieved Nature Could Reach Half (many of which already exceed Aichi target 11). Of the best-protected ecoregions, the majority (15) occur in the Neotropics, followed by the Indomalayan realm (11; figure 2).

In contrast to the moist forests, the tropical and subtropical dry broadleaf forest is the most endangered biome on Earth; only 2 ecoregions (among 56) are Half Protected, 20 are Nature Could Recover, and 26 are Nature Imperiled. The temperate broadleaf and mixed forests biome has the second largest number of ecoregions (83) but shows a distribution of protection categories skewed toward those needing restoration: Nature Could Recover and Nature Imperiled. The boreal forest ecoregions are among the largest and have the greatest potential to reach Half Protected because of their vast remaining intact forest blocks.

The majority of mangrove ecoregions fall into the categories of Half Protected or Nature Could Reach Half. The remaining mangrove ecoregions are degraded but can recover through restoration (table 1, supplemental appendix S2).

The Nature Imperiled category includes 108 (23%) forest ecoregions ($n = 476$; table 1; supplemental appendix S2, supplemental table S1a, S1b). Assessing recent trends in tree cover, of the 16 forest ecoregions with the greatest extent of tree loss between 2000 and 2014 (ranging from 20% to 86%), 9 are in the Afrotropics, and 4 are in the Indo-Malayan realm of India. Deforestation was greatest in the Nigerian lowland forests and the Cross-Niger transition forests.

Nonforested ecoregions and biomes. The protected area network is far less extensive in nonforested biomes. The

Table 1. Progress toward Nature Needs Half by biome, showing the number of ecoregions in each category, based on habitat protected and habitat remaining.

Biome name and number	Percentage of Earth's terrestrial area	Mean percentage of protected within biome	(1) Half Protected	(2) Nature Could Reach Half	(3) Nature Could Recover	(4) Nature Imperiled	Total
Forested biomes							
1. Tropical and subtropical moist broadleaf forests	14.4	12	24	116	46	44	230
2. Tropical and subtropical dry broadleaf forests	2.9	8	2	8	20	26	56
3. Tropical and subtropical coniferous forests	0.5	12	1	6	7	1	15
4. Temperate broadleaf and mixed forests	9.3	10	7	21	30	25	83
5. Temperate conifer forests	2.8	17	2	16	19	10	47
6. Boreal forests or taiga	11.4	9	1	23	2	0	26
14. Mangroves	0.2	26	3	8	6	2	19
Forested biome subtotal	*41.5*	*13*	*40*	*198*	*130*	*108*	*476*
Nonforested biomes							
7. Tropical and subtropical grasslands, savannas, and shrublands	15.8	15	5	14	18	20	57
8. Temperate grasslands, savannas, and shrublands	7.8	4	0	11	13	24	48
9. Flooded Grasslands and Savannas	0.9	32	8	4	9	4	25
10. Montane grasslands and shrublands	3.6	25	9	11	14	12	46
11. Tundra	8.7	8	26	24	0	1	51
12. Mediterranean forests, woodlands, and scrub	2.4	18	2	5	25	8	40
13. Deserts and xeric shrublands	19.3	6	8	46	19	30	103
Nonforested biome subtotal	*58.5*	*10*	*58*	*115*	*98*	*99*	*370*
Total	100	12	98	313	228	207	846

Note: The ecoregion data can be found in supplemental tables S1 and S2. (1) Half Protected: 50% or more of the total ecoregion area is protected. (2) Nature Could Reach Half: Less than 50% of the total ecoregion area is protected, but the sum of the total ecoregion protected and unprotected natural habitat remaining is 50% or more. (3) Nature Could Recover: The sum of the amount of natural habitat remaining and the amount of the ecoregion that is protected is less than 50% but more than 20%. (4) Nature Imperiled: The sum of the amount of natural habitat remaining and the amount of the ecoregion that is protected is less than or equal to 20%.

tundra biome is best protected among the seven nonforested biomes: 26 of the 51 tundra ecoregions (51%) fall under Half Protected, and another 24 ecoregions (47%) are in Nature Could Reach Half. Desert and xeric shrubland ecoregions also have expansive networks of protected areas and large swaths of natural habitat remaining: over half fall into Half Protected or Nature Could Reach Half (figure 2). Ecoregions in the remaining nonforested biomes have been more heavily degraded: 99 (27%) nonforested ecoregions were categorized as Nature Imperiled.

Human impact and revisiting the most endangered biomes on Earth. Land-use change as a result of human activities is a dominant feature in the large majority of ecoregions, as has also been shown by Venter and colleagues (2016). In the 207 Nature Imperiled ecoregions, an average of 96% of natural habitat has been converted to an anthropogenic land use. Many of the fragments in these ecoregions are of disproportionately high biodiversity value. Here, protecting Key Biodiversity Areas (KBAs) will be crucial, and the goal of NNH remains aspirational and of secondary concern to protecting what remains (Eken et al. 2004).

Forested and nonforested biomes are evenly represented in the Nature Imperiled category (table 1). Hoekstra and colleagues (2005) described the temperate grasslands, savannas, and shrublands biome as the most endangered in the world. However, our results show that the most critically

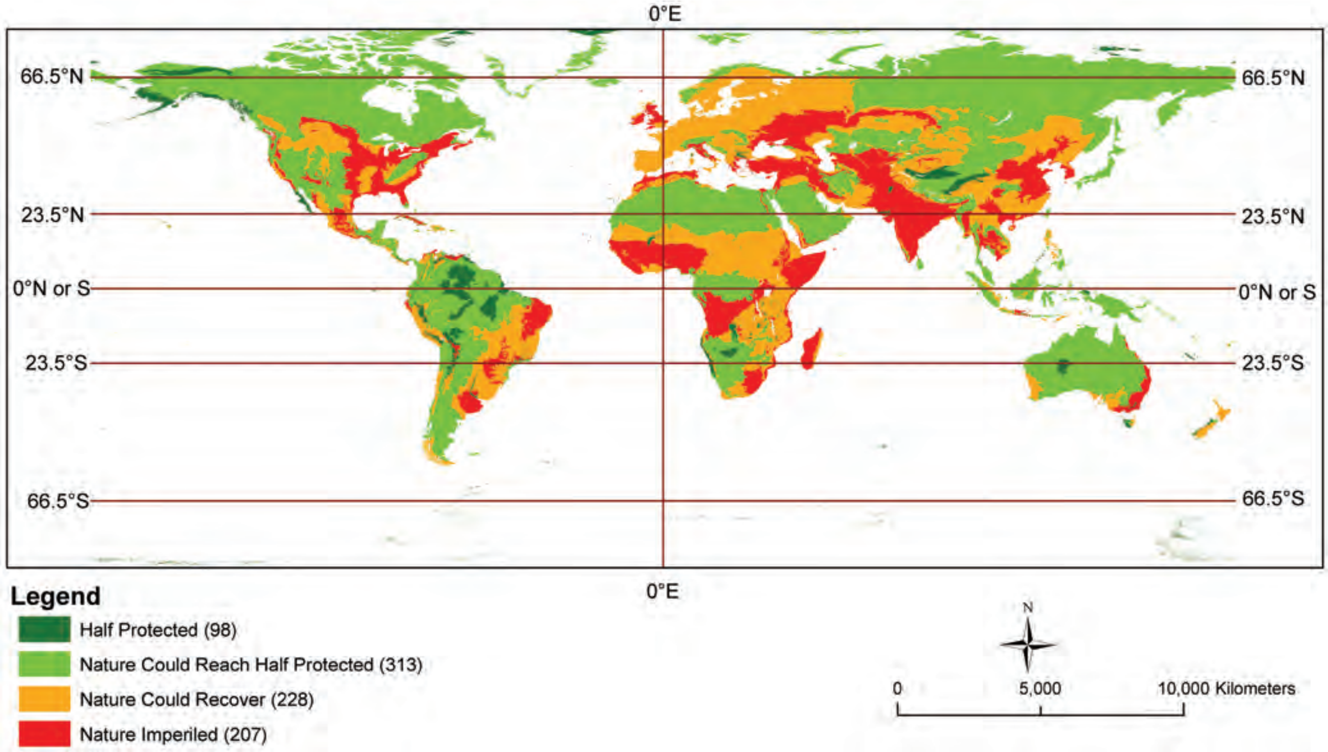

Figure 2. The protection statuses of ecoregions of the world. This map shows the high levels of habitat remaining in some of the most species-rich areas on Earth, including the Brazilian Amazon, the Congo basin, and the islands of Indonesia. Although enough habitat remains for nearly half of the ecoregions to exceed 50% protected in the coming decades, much of this forest is still unprotected, and just under 50% of ecoregions have adequate conservation plans in place to keep remaining forests intact (supplemental appendix S3). The numbers in parentheses for each category represent the entire number of ecoregions found in each category. The ecoregion protection categories are defined as the following: Half Protected, more than 50% protected; Nature Could Reach Half, less than 50% of the total ecoregion area is protected, but the sum of the total ecoregion protected and unprotected natural habitat remaining is more than 50%; Nature Could Recover, the sum of the amount of natural habitat remaining and the amount of the total ecoregion that is protected is less than 50% but more than 20%; Nature Imperiled, the sum of the amount of natural habitat remaining and the amount of the total ecoregion that is protected is less than or equal to 20%.

endangered biome—as is determined by the proportion of Nature Imperiled ecoregions that constitute each—is the tropical dry forests, whereas two nonforested biomes are nearly as endangered: (1) tropical and subtropical grasslands, savannas, and shrublands and (2) Mediterranean forests, woodlands, and scrub.

Without considering fine-scale endemism and beta-diversity (turnover of species with distance or along gradients), simple metrics of habitat loss and percent protection may underestimate the conservation crisis among biomes. Biodiversity loss would therefore be much greater and more sensitive to habitat conversion in tropical and subtropical grasslands, savannas, and shrublands; in Mediterranean forests, woodlands, and scrub; and in tropical moist and tropical dry forests. These four biomes support higher endemism and greater beta-diversity levels than those found in other biomes.

Beyond Aichi targets: Toward Half Protected

The need to go beyond Aichi protection targets was approved by delegates at the 2014 IUCN World Parks Congress. They further decided that the total area of protected areas and connectivity lands needs to be far higher than current conceptions and agreed on the importance of setting ambitious targets (IUCN 2014). Results from our global assessment suggest that the ambitious target of protecting half of terrestrial nature is attainable for many of the Earth's more intact ecoregions. Among the 846 ecoregions, 98 (12%) occupy the Half Protected category. Although these ecoregions are largely concentrated in two biomes—tropical and subtropical moist forest and tundra—there is at least one ecoregion achieving this status in 12 of the 14 biomes. Within Nature Could Reach Half ($n = 313$), 26 ecoregions (8%) are at least 40% protected and therefore require modest additional protection to reach Half Protected in each. These and the other 287 ecoregions constituting the Nature Could Reach Half category provide

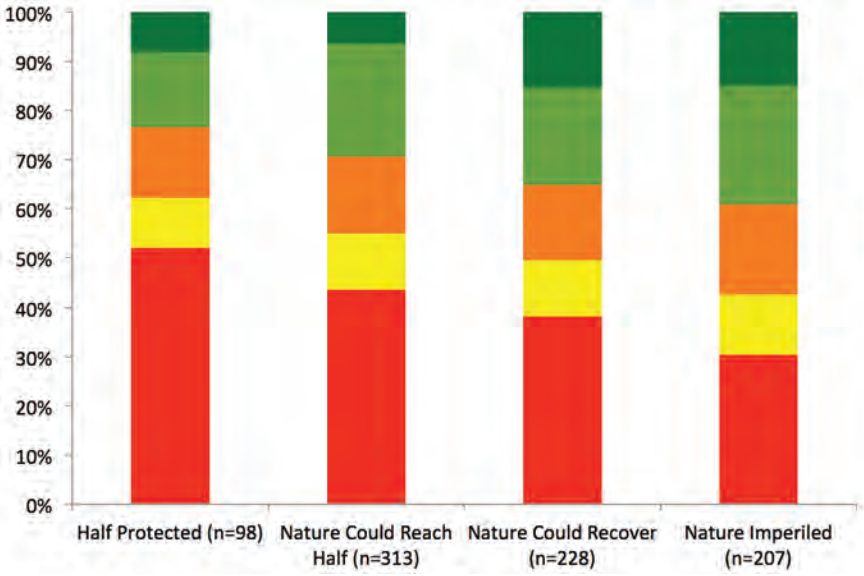

Figure 3. The proportion of biodiversity goals addressed within available conservation plans for all 846 ecoregions, distributed across the four protection-status categories. The colors represent the percentage of conservation strategies addressed within each protection-status category: 0 goals addressed, red; 1 goal addressed, yellow; 2 goals addressed, orange; 3 goals addressed, light green; 4 goals addressed, dark green. For a detailed list of conservation strategies and sources, see supplemental appendix S3.

the greatest conservation opportunity, because adequate habitat remains to reach Half Protected. These ecoregions are found within every biome and should rank high in the formulation of the next Aichi target 11 post-2020.

Because Aichi target 11 requires protected area networks to be ecologically representative, an ecoregion assessment provides an indispensable tool for meeting the new targets to be set in 2020. Greater effort is needed to complete these ecoregion strategies. For example, only 94 of the 846 terrestrial ecoregions (11%) have published plans that address all four biodiversity conservation goals (figure 3; see supplemental appendix S3 for methods). Formal conservation strategies that address three-fourths of the biodiversity conservation goals were published for 22% of ecoregions globally. Most of these strategies focus on identifying priority areas for protection and on conserving species of conservation concern (figure 3). Notably, a high percentage of ecoregions in the Nature Imperiled category have plans that address all four conservation goals. This is because biodiversity hotspots—biologically rich areas containing less than 30% of the original habitat—are explicitly targeted by Critical Ecosystem Partnership Fund (CEPF) profiles (Myers et al. 2000, Olson 2010). Of great concern are the 337 ecoregions that lack biodiversity plans (supplemental appendix S3).

Robust ecoregion strategies must be followed by effective implementation to realize biodiversity conservation goals formulated at a national scale. Three countries advancing to or already surpassing Half Protected—Namibia, Nepal, and Bhutan—are worth singling out as compelling examples of where effective implementation embodies key principles of biodiversity conservation. They also refute some of the criticisms raised over the NNH approach that (a) it could displace rather than empower indigenous communities, (b) it is a paradigm only suitable for wealthy countries, and (c) it can only succeed in sparsely populated, remote ecoregions.

Namibia's conservation strategy includes conservation areas managed by local communities alongside government-run strict nature reserves across all its ecoregions. These communities are awarded autonomy to manage vast tracts of land for wildlife conservation and income generation, in large part by allowing communities to own the wildlife. Now widely touted as a success story in global conservation, these lands were largely defaunated through poaching only 25 years ago. Community-managed lands, called *communal conservancies*, now contribute to Namibia's national protected area network, which covers 47% of the country. Communal conservancies range in size from 43 square kilometers (km^2) to 9120 km^2 (the mean being 1953 km^2). In fact, many conservancies function as vital corridors connecting other protected areas and allowing dispersal, movement, and range recovery of large mammals, including elephants, lions, and others that are in steep decline elsewhere in sub-Saharan Africa (figure 4a; Naidoo et al. 2016).

In Nepal, ecoregion conservation strategies that involve local communities are the rule and complement the country's strictly protected areas. In the lowlands and midlands, community forestry and agroforestry in designated landscapes yield economic returns while strategically extending habitat and connectivity among reserves (figure 4b, table 2; Wikramanayake et al. 2010). Community-managed forest parcels are small—some are as little as 20 hectares in size—but abundant and interspersed among larger protected areas, often facilitating population recovery of endangered large mammals (Wikramanayake et al. 2010). Community forests, linked together to form corridors, play a pivotal role in landscape conservation. Handing over forest management to communities, which then receive 50% of the revenue generated by wildlife parks in designated buffer zones, led to a 61% increase in tigers and a 31% increase in rhinos over a 5-year period (2008–2013). No rhinos, tigers, or elephants have been poached in Nepal in several years (Dhakal et al. 2014).

In the Himalayan and trans-Himalayan ecoregions overlapping Nepal, conservation areas managed by local

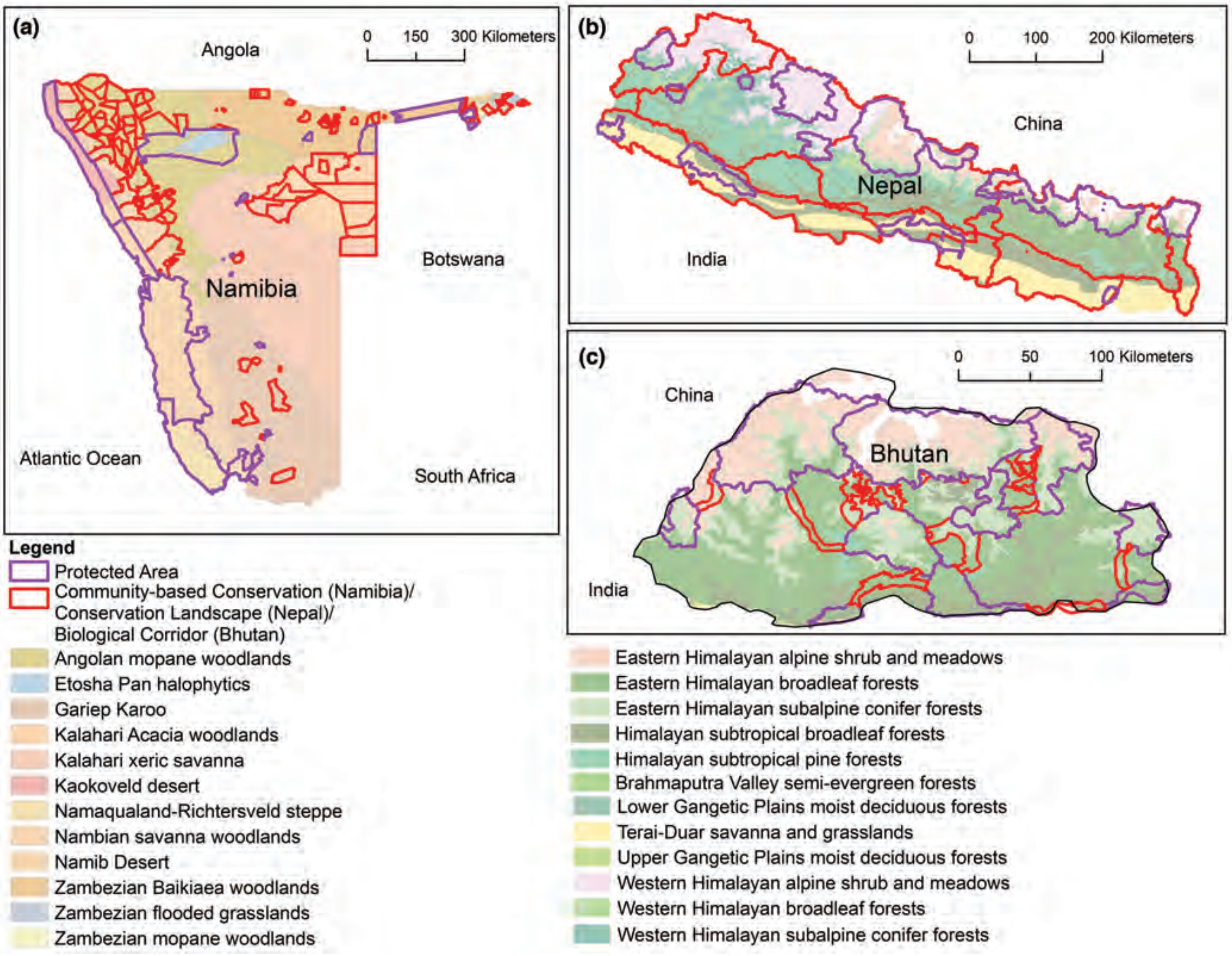

Figure 4a–c. Ecoregion conservation planning in three developing countries: (a) Namibia uses communal conservation areas to extend protection beyond protected areas and cover a diverse set of ecoregions, (b) Nepal uses a mixture of protected areas and conservation landscapes to protect along north–south and east–west gradients, and (c) Bhutan uses protected areas combined with biological corridors to provide connectivity between protected areas and across ecoregions.

communities exceed in area the land under national-park status and some, such as the Annapurna Conservation Area, return large sums of tourism-generated revenues annually to local funds. These are sparsely populated ecoregions. In contrast, the protected areas and community forests of the Terai-Duar savannas ecoregion in Nepal are intermingled with some of the highest rural population densities on Earth. In this densely settled, productive ecoregion situated on alluvial soils, there is room for intensive rice production and park protection (Dinerstein et al. 1999), the latter of which returns more than $1 million annually to local development funds in demarcated buffer zones.

Bhutan protects 51% of its land through national parks and corridors connecting reserves (figure 4c, table 2). In a novel policy framework, Bhutan's constitution requires that at least 60% of the country remains forested (currently, forest cover is estimated at 72%). Mid-elevation temperate broadleaf forests, which are so heavily converted elsewhere, are particularly well protected. Bhutan, as with Nepal, ranks among the nations with the lowest *per capita* GDP but protects enough habitat to conserve biodiversity (Dinerstein 2013).

All three examples stress core protected areas, buffer zones, and connectivity—all key components of ecoregion conservation strategies and securing biodiversity. The first two examples illustrate how extensive areas can be put under conservation management by engaging local communities. The example of Bhutan offers a different mechanism through constitutional decree. Both approaches work.

Strengths and weaknesses of the Nature Needs Half approach to conserving half the terrestrial realm

NNH, like any paradigm, has strengths and weaknesses. NNH offers a simple, inspirational, and science-based

Table 2. The conservation status of ecoregions within the countries of Namibia, Nepal, and Bhutan.

Ecoregion number	Ecoregion	Global ecoregion area (km^2)	Ecoregion area within country (km^2)	Percentage of global ecoregion area	Country's area protected in IUCN cateogry I-VI (km^2)	Percentage of country's ecoregions protected	Global ecoregion protection status
Country of Namibia							
34	Angolan mopane woodlands	191,639	151,443	79	66,620	44	2
47	Kalahari Acacia woodlands	106,411	68,004	64	46,214	68	1
64	Zambezian Baikiaea woodlands	358,546	86,277	24	20,469	24	2
65	Zambezian mopane woodlands	387,596	4,724	1	2,569	54	2
70	Etosha Pan halophytics	7,691	7,688	100	7,457	97	1
76	Zambezian flooded grasslands	201,936	4,239	2	2,137	50	1
94	Gariep Karoo	251,666	142,553	57	10,729	8	2
97	Kalahari xeric savanna	685,551	183,555	27	12,277	7	2
98	Kaokoveld desert	33,039	20,806	63	20,767	100	1
102	Namaqualand-Richtersveld steppe	52,727	20,044	38	18,065	90	2
103	Namib Desert	79,116	79,118	100	72,427	92	1
104	Nambian savanna woodlands	102,712	56,391	55	31,704	56	3
Namibia Total		*1,406,746*	*506,706*	*36*	*168,106*	*33*	
Country of Nepal							
233	Himalayan subtropical broadleaf forests	38,124	28,447	75	2,766	10	3
238	Lower Gangetic Plains moist deciduous forests	253,213	250	0	0	0	4
287	Upper Gangetic Plains moist deciduous forests	262,642	25	0	0	0	4
302	Himalayan subtropical pine forests	76,126	22,811	30	836	4	3
306	Eastern Himalayan broadleaf forests	82,915	15,418	19	2,180	14	2
308	Western Himalayan broadleaf forests	55,825	4,809	9	913	19	3
309	Eastern Himalayan subalpine conifer forests	27,436	4,928	18	2,778	56	2
310	Western Himalayan subalpine conifer forests	39,650	12,080	30	1,753	15	4
311	Terai-Duar savanna and grasslands	34,517	22,732	66	3,265	14	4
751	Eastern Himalayan alpine shrub and meadows	121,014	8,212	7	6,725	82	2
769	Western Himalayan alpine shrub and meadows	70,090	21,243	30	7,593	36	3
Nepal Total		*1,061,552*	*140,954*	*13*	*28,810*	*20*	
Country of Bhutan							
222	Brahmaputra Valley semi-evergreen forests	56,613	274	0	125	46	4
233	Himalayan subtropical broadleaf forests	38,124	4,143	11	1,090	26	3
302	Himalayan subtropical pine forests	76,126	671	1	244	36	3
306	Eastern Himalayan broadleaf forests	82,915	16,198	20	4,079	25	2
309	Eastern Himalayan subalpine conifer forests	27,436	9,232	34	6,031	65	2
311	Terai-Duar savanna and grasslands	34,517	139	0	33	24	4
751	Eastern Himalayan alpine shrub and meadows	121,014	7,463	6	6,102	82	2
Bhutan Total		*436,745*	*38,119*	*9*	*17,704*	*46*	

Note: The protected status of many of these ecoregions is ahead of the global average because of ecoregional planning and the use of communal reserves and corridors in addition to strict protected areas. A map of these three countries and their protected areas can be found in figure 4. Global ecoregion protection status' refers to 1 = Half Protected, 2 = Nature Could Reach Half, 3 = Nature Could Recover, 4 = Nature Imperiled.

Box 1. Protecting half in a policy context.

Nature Needs Half finds support in the United Nation's Sustainable Development Goals (SDGs). Among other items, the SDGs call on humanity to "take urgent and significant action to reduce degradation of natural habitats [and] … protect and prevent the extinction of threatened species" and to "halt deforestation" and "halt loss of biodiversity" by 2020. These internationally agreed-on conservation goals will be challenging to achieve without protecting in the realm of half. As such, we call on advocates and leaders around the world to set new global protected area targets accordingly: 50% of the terrestrial realm by 2050.

Calls to increase the global area under protection should be considered in the context of other political mechanisms, such as international development funding (e.g., G20) and The Bonn Challenge. The Bonn Challenge, a global effort to restore millions of hectares of deforested and degraded land by 2020 or 2030, can be a critical mechanism in ecoregions falling under Nature Could Recover and Nature Imperiled. There are other opportunities to weave the 50% goal into the global economic and development fabric. For example, the "G20," the world's 20 largest economies, have called for as much as $60 trillion–$70 trillion in investment for large infrastructure projects (Foundation Earth 2015). Holistic ecoregional planning must be included to ensure that future infrastructure and cities are built in harmony with a world where nature receives half.

A Paris-like deal that addresses biodiversity conservation at the highest political level—a Global Deal for Nature under the auspices of the CBD—is needed for nature conservation (for further details see www.resolv.org/blog/2017/global-deal-for-nature). An initiative of this scale would mobilize unprecedented financial resources to support countries to implement the goal of Half Protected. The estimated cost to add terrestrial protected areas, better protect existing reserves, and restore habitat varies by country, region, and ecoregion, ranging between $8 billion and $80 billion per year for the terrestrial realm (Balmford et al. 2003, McCarthy et al. 2012) and between $5 billion and $19 billion per year for the marine realm (Balmford et al. 2004). Implementing a Global Deal for Nature would employ a large number of currently unemployed or underemployed workers in rural communities.

At the current rate, the amount of land under formal protection increases by about 4% per decade. If the rate of increase doubled to 8% or achieved 10% per decade, the global goal, supported by a Global Deal for Nature, could be within reach.

message that can be easily understood by the general public. It also provides the conservation movement with a unifying goal. Incremental gains in global protection targets have proved insufficient in response to the magnitude of the biodiversity crisis. Conservation efforts have often been mired in process or targets that do not track onto an ultimate conservation goal or vision statement (Wilson 2016). NNH provides an endgame: Achieving Half Protected will help realize the outcomes and objectives of maintaining a living biosphere, avoiding mass extinction, and preserving ecological processes that benefit all human societies. NNH also provides a goal and a planning framework under which all conservation efforts can fit.

Importantly, 50% avoids setting targets too low and being surpassed by the synergistic effect of threats to nature from climate change and mass extinction. The recent Paris Agreement under the United Nations Framework Convention on Climate Change provides targets for stabilizing atmospheric greenhouse gas concentrations at a level that prevents "dangerous anthropogenic interference with the climate system." We contend that for the climate deal to succeed, we need a Global Deal for Nature (box 1). NNH provides a baseline from which we can monitor progress as the environmental data sets are increasingly dynamic, annually updated, and freely available and serve as a scorecard to underpin a Global Deal for Nature and assist the CBD in measuring progress. Finally, NNH could help provide government, lenders, citizens, and industry guidance about where to site extractive industries and develop large infrastructure projects.

Providing clear implementation guidelines can help address weaknesses associated with NNH. For example, insisting that NNH be empirically derived for each of the world's ecoregions is important. However, in trying to erect a simple, science-based target that nonscientists can understand—50% protected by 2050—the approach runs the risk of giving the misimpression that 50% is the "right" target for each ecoregion. In fact, the amount of habitat that needs to be conserved in each region will vary. This guideline will help avoid pitfalls, such as a case in which governments could assign large areas to be protected just to reach the 50% target (e.g., high-elevation rock and ice, barren desert, contaminated areas, unproductive soils, or lands of low economic value) without consideration of the design, through ecoregion strategies, of representative networks to capture unique patterns of biodiversity. One clear guideline is that site selection is as important as total area protected in achieving conservation objectives (Margules and Pressey 2000). Tools such as ecoregion conservation planning, CEPF hotspot profiles, Key Biodiversity Areas, and systematic conservation planning that focus on the quality or irreplaceability of areas considered for protection will be most useful to avoid this danger (Margules and Pressey 2000, Myers et al. 2000, Eken et al. 2004, IUCN 2016).

A potential pitfall is that policymakers not well versed in ecosystem function might view NNH as license to clear the other 50%. This would be a disaster in some ecoregions, such as those in the Amazon and Congo Basins, that perform vital ecological roles only if contiguous forest cover is maintained. Conservation planning will need to underpin the implementation of NNH to avoid these abuses.

Another concern is that the NNH approach risks overlooking, however unintentionally, those 207 ecoregions determined by our analysis to average only 4% of remaining natural habitat outside protected areas that fall into the Nature Imperiled category. Where ecoregions contain global centers of endemism but with only fragments of natural habitat remaining replete with irreplaceable sites, a concern is that the global importance of these sites of rarity could be downplayed. Donors and agencies might concentrate on those less biodiverse ecoregions but those likely to come closer to achieving the 50% target. In most of these ecoregions, Key Biodiversity Areas, if properly conserved will protect the biodiversity that remains (Eken et al. 2004). CEPF profiles should include all possible options for restoration (Butchart et al. 2015).

A possible concern expressed by critics of Wilson (2016) and of the NNH approach is that protecting half the terrestrial realm adversely affects humans in remote regions (e.g., Büscher et al. 2016). In contrast, implementing NNH is an opportunity to empower indigenous peoples and local communities. Many indigenous reserves in Latin America, Asia, Africa, and Australasia are an essential part of the formal protection network, but the decisionmaking is in the hands of those within the reserves. Several indigenous communities are also advocating for half their lands to be protected. The Dehcho Dene in northern Canada, for example, has articulated an explicit 50% protected goal for their own territory (Norwegian 2005). For many groups, such as the Dehcho Dene, protecting half is an approach derived from their traditional ecological knowledge. Conservation should be achieved through careful planning while respecting rights, improving livelihoods, and sharing decisionmaking.

Achieving Half Protected hinges on a reduction of human disturbance, sparing nature

Fortunately, two schools of thinking—how to save half for nature and how to feed and fuel advancing societies— are in growing concordance. As societies urbanize and develop, there is a well-documented trend toward “decoupling”: an increasingly efficient use of land and resources that reduces environmental degradation (Ausubel 2000, Fischer-Kowalski and Swilling 2011, Tilman et al. 2011, Ausubel et al. 2012). These trends have already produced major recoveries of woodland and other vegetation in many regions (Ellis et al. 2013, Blomqvist et al. 2015). The prospects for feeding growing human populations while recovering natural habitat are not only aspirational but also achievable as long as these aspirations are put to work guiding land-use policy and commodity-chain interventions (box 1; Lambin et al. 2014).

The global phenomenon of growing urbanization, accentuated in some ecoregions, sets the stage for reaching Half Protected. In remote areas in many parts of the world, depopulation due to socioeconomic changes such as increasing wages and career opportunities have resulted in rural populations moving to population centers; by 2050, 70% of people will live in cities. This phenomenon, driven by economics, could lead to expansion of the protected area network and restoration of disturbed or abandoned lands (Ellis et al. 2013).

Nature Needs Half is an ambitious goal that will allow humanity to maintain a world with space for all life and the continuance of critical ecosystem services. Our findings show that a large number of ecoregions are Half Protected and that NNH is achievable in the vast majority of remaining ecoregions. However, achieving NNH requires further research into the desirability, feasibility, and progress toward the goal at ecoregional and national scales. Here, we provide tools and information to chart progress toward NNH and call on advocates and leaders around the world to set new global protected area targets: 50% of the terrestrial realm by 2050. Doing so through carefully balanced ecoregion plans that promote economic development while sustaining nature will also make the planet more livable for humanity (Mulligan 2014, 2015).

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Supplemental material

Supplementary data are available at *BIOSCI* online.

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Planning for Biodiversity Conservation: Putting Conservation Science into Practice

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The growing recognition that the species extinction crisis has deepened and that there are limited conservation dollars to address this crisis has had a profound influence on the planning methods and conservation strategies of governmental and nongovernmental organizations. For example, World Wildlife Fund (WWF) and Conservation International have pinpointed priority ecoregions and biodiversity "hotspots," respectively, that represent some of the most significant remaining regions for conserving the world's biological diversity (Olson and Dinerstein 1998, Myers et al. 2000). Both The Nature Conservancy (TNC) (Master et al. 1998) and World Wildlife Fund (Abell et al. 2000) have set conservation priorities at the scale of large watersheds for freshwater ecosystems in the United States. The National Gap Analysis Program (GAP) of the US Geological Survey's Biological Resources Division is using biological survey data, remote sensing, and geographic information systems (GIS) technology at the state level to identify those native species and ecosystems that are not adequately represented in existing conservation lands—in other words, the aim of the program is to detect conservation "gaps" (Jennings 2000). Some state governments in the United States are also developing their own biodiversity conservation plans (e.g., Kautz and Cox 2001). Internationally, more than 175 countries are mandated, as signatories to the United Nation's Convention on Biological Diversity, to prepare National Biodiversity Strategy and Action Plans (Secretariat of the Convention on Biological Diversity 2000).

> A SEVEN-STEP FRAMEWORK FOR DEVELOPING REGIONAL PLANS TO CONSERVE BIOLOGICAL DIVERSITY, BASED UPON PRINCIPLES OF CONSERVATION BIOLOGY AND ECOLOGY, IS BEING USED EXTENSIVELY BY THE NATURE CONSERVANCY TO IDENTIFY PRIORITY AREAS FOR CONSERVATION

All of these assessments and priority-setting exercises have a common trait: They focus on relatively large spatial areas or regions inhabited by thousands of species and hundreds of identifiable natural communities. To implement conservation

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actions on priorities identified in these coarse-scale assessments requires a practical yet science-based planning framework for the conservation of biodiversity *within* these regions. Recognizing that most conservation efforts are reactive and that its own conservation investments needed to be more strategic, The Nature Conservancy has been developing such a framework for conservation planning in terrestrial, freshwater, and near-shore marine environments (Groves et al. 2000). This framework has been tested and revised through the preparation and implementation of over 45 ecoregional and regional conservation plans in the United States (figure 1), Latin America, the Caribbean, Micronesia, and Yunnan, China. The framework's methods are based on theories and principles from ecology and conservation biology and have been developed in consultations with scientists from research, natural resource management, and conservation institutions and organizations. It has been applied across many types of ecosystems by numerous scientists and practitioners under a variety of levels of funding and availability of information. In this article, we report the lessons learned from implementing TNC's planning framework as a model for the many agencies and institutions around the world that face similar challenges in conservation planning.

Four significant scientific advances in the last decade of the 20th century have shaped the development of this framework. First, the growing list of endangered species highlighted the need for approaches to conservation that are proactive and complement the reactive measures of most endangered species programs. Second, scientists increasingly recognized the importance of conserving the underlying ecological processes that support the patterns of biological diversity (e.g., Balmford et al. 1998). Third, we began to realize that biodiversity occurs at multiple spatial scales and levels of biological organization (Schwartz 1999) and that a greater emphasis to conserve this diversity must be placed at all appropriate levels and scales (Poiani et al. 2000). Finally, we learned that systematic conservation planning approaches are more effective at conserving biological diversity than are the ad hoc approaches of the past (Margules and Pressey 2000). These ad hoc approaches have resulted in a biased distribution of lands and waters set aside for conservation purposes, with the majority of these areas occurring at relatively higher elevations and on steeper slopes and poorer soils (Pressey et al. 1996, Scott et al. 2001).

TNC's seven-step, conservation planning framework incorporates all four of these scientific advances (see box 1). We have applied the framework to ecoregions—large areas of the earth's surface that have similarities in faunal and floral composition due to large-scale, predictable patterns of solar radiation and moisture (Bailey 1998). Most ecoregional classifications are based upon criteria such as climate, soils, geology, vegetation cover types, or in the case of marine systems, oceanographic factors (Bailey 1998), because these environmental variables are assumed to have a major influence on the evolutionary history and distribution of many species and communities. The US Forest Service and the US Environ-

Box 1. A Seven-Step Conservation Planning Framework

Step 1: Identify conservation targets

Communities and ecosystems

Abiotic (physically or environmentally derived targets)

Species: imperiled or endangered, endemic, focal, keystone

Step 2: Collect information and identify information gaps

Use a variety of sources

Rapid ecological assessments, rapid assessment programs

Biological inventories

Expert workshops

Step 3: Establish conservation goals

Two components of goal: representation and quality

Distribute targets across environmental gradients

Set a range of realistic goals

Step 4: Assess existing conservation areas

Gap analysis

Step 5: Evaluate ability of conservation targets to persist

Use criteria of size, condition, and landscape context

Use GIS-based "suitablity indices"

Step 6: Assemble a portfolio of conservation areas

Use site or area selection methods and algorithms as a tool

Design networks of conservation areas employing biogeographic principles

Step 7: Identify priority conservation areas

Use the criteria of existing protection, conservation value, threat, feasibility, and leverage

Figure 1. The Nature Conservancy (TNC) map of the ecoregions of the United States and adjacent regions of Mexico and Canada, as adapted from Bailey (1995). The different colors represent the boundaries of distinct ecoregions. TNC is also working on ecoregional plans in Latin America, the Caribbean, and the Asia–Pacific realms.

mental Protection Agency developed ecoregional classifications for the United States (Omernik 1987, Bailey 1995, 1998), and the World Wildlife Fund has done so for every continent (Olson et al. 2001). For this planning framework, we used a modified version of Bailey's (1995) ecoregions for the United States and relied on WWF's ecoregional classifications for other countries. Although intended for application at an ecoregional scale, this framework should be applicable to other types of planning regions (e.g., Conservation International's biodiversity hotspots) at similar spatial scales. Redford and colleagues (forthcoming) provide an overview of approaches that various organizations use to conserve biodiversity, including the spatial scale at which these different approaches are intended to operate.

The primary product of applying this framework is the identification of a portfolio or network of lands and waters for conserving the elements of biodiversity within an ecoregion. We refer to these lands and waters as *conservation areas.* We separate the identification of conservation areas from their design and management (Scott and Csuti 1997). We emphasize that the primary purpose of regional-scale conservation planning as articulated in this article is to identify a set of conservation areas that best represents the native species and ecosystems of the region and the underlying ecological processes that sustain them. Determining how those areas are best designed and managed requires a more detailed analysis, usually at finer spatial scales. Planning at the scale of conservation areas (e.g., Nature Conservancy preserve, national park, national or state wildlife refuge) aims to maintain or improve the ecological condition of targeted biological or environmental features of these areas and to abate threats to these features (Poiani et al. 1998). Noss and Cooperrider (1994) and Meffe and Carroll (1997) provide overviews of the design and management of conservation areas.

A seven-step framework for conservation planning

Although we describe the framework step by step, the actual planning process is less linear and more dynamic. For example, the collection of information (step 2) occurs throughout the planning process from its inception to the point of setting priorities among the portfolios of conservation areas. Furthermore, the planning process itself should be viewed as adaptive, with continual improvements being made in both the methods of the steps and the conceptualization of the entire seven-step framework. Finally, for each step, we cite relevant scientific literature that provides some substantiation for the importance of the step.

Step 1: Identify conservation targets

For the purpose of this planning framework, we define "biodiversity" as the variety of living organisms, the ecological complexes in which they occur, and the ways in which they interact with each other and the physical environment (Redford and Richter 1999). Although biodiversity is defined many ways, this definition is consistent with one previously advanced by Noss (1990). It characterizes biodiversity as having three primary components: composition, structure, and function. From a conservation perspective, it is necessary to consider each of these components.

To represent the biodiversity of a region or ecoregion in conservation areas, we focus on conservation targets, the entities or features for which a conservation plan or project is attempting to ensure long-term persistence (Redford et al. forthcoming). The word "target" has also been used in a different context by some conservation planners and scientists to imply a particular goal, such as conserving a specific percentage of an ecosystem type (Soulé and Sanjayan 1998). Because it is impractical to conduct planning efforts for each of the hundreds to thousands of species that inhabit any one region, scientists and planners seek to identify a set of conservation targets that presumably represent the biodiversity of a region. These targets may be defined based on their biological features (e.g., species and communities), physical features (e.g., soils, geology, climate), or a combination of both biotic and abiotic features. The assumption is that, by focusing planning efforts on these targets, there will be a high likelihood of conserving the vast majority of living organisms in a region, both those known to science and the many yet to be discovered.

Considerable debate has taken place over which levels of biological organization are most appropriate to serve as targets for conserving biodiversity (e.g., species vs. communities vs. landscapes; Franklin 1993). Some scientists have recommended a "coarse filter" and "fine filter" approach to target selection (e.g., Hunter 1991, Noss and Cooperrider 1994, Noss 1996). The principal idea behind the coarse filter approach is that by conserving representative examples of the different biological communities and ecosystems that occur within a region, the majority of species of that region will also be conserved. Some types of conservation targets, however, such as rare or endangered species, do not always co-occur in a predictable fashion with certain communities or ecosystems. For these targets, individual or fine filter approaches are necessary. Which particular conservation targets can be captured with a coarse filter approach has never been tested empirically (Noss and Cooperrider 1994).

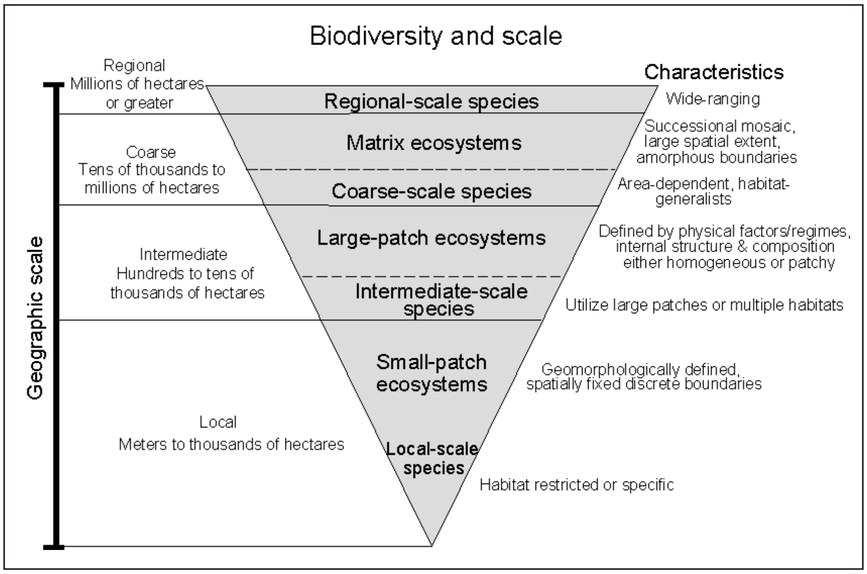

Figure 2. The spatial scales and levels of biological organization. Conservation targets can be viewed as occurring at four spatial scales from local to regional. The general range in size (hectares) for each spatial scale is indicated to the left of the pyramid and some general characteristics of two types of conservation targets (species and ecosystems) are shown on the right. Reprinted from Poiani et al. (2000), with permission.

Although the coarse–fine filter strategy is a practical approach to an otherwise complex problem, it can be confusing with regard to the spatial scale at which various coarse and fine filter targets occur. A more useful approach may be to recognize that conservation targets can be identified at a variety of levels of biological organization and spatial scales from local (fine) to regional (figure 2). Which targets are used in any particular planning exercise will depend to a great extent on what information is available (Margules and Pressey 2000). Some areas of the world, such as parts of the United States, Australia, and Europe, are relatively rich in information on individual species. However, many areas are not, particularly those in the tropical regions of the world; thus, some type of conservation target in addition to a species-specific one must be used. The only spatially consistent types of information available in most parts of the world are for physical variables (e.g., elevation, climate, soil type) and for communities or ecosystems classified according to vegetative composition. Based on these considerations, we suggest three general classes of conservation targets: (1) communities or ecosystems, (2) abiotic targets based on physical variables, and (3) species not likely to be subsumed under the other two classes of targets.

Communities and ecosystems. Like biodiversity, *community* and *ecosystem* have various definitions. For the purposes of this article "community" refers to an interacting assemblage of species that co-occur with some degree of predictability and consistency. "Ecosystem" includes the interactions of these communities with the abiotic or physical environment, such as through the transfer of energy and matter (Whittaker 1975).

Communities or ecosystems occur at a spectrum of spatial scales (figure 2) and can serve as practical surrogates for sampling finer levels of biological organization. Classifications of communities and ecosystems exist for many parts of the world at local, state, regional, and national scales (see Grossman et al. 1999, table 5, for a summary). Although data on the actual individual community and ecosystem units described in these classifications are often lacking, remote sensing imagery can contribute much information on communities and ecosystems described on the basis of dominant vegetation (Jennings 2000).

The Nature Conservancy and NatureServe (formerly known as the Association for Biodiversity Information), in collaboration with gap analysis programs, have developed an international classification of vegetation communities (Grossman et al. 1998). This classification system is a hierarchical taxonomic structure with physiognomic criteria used at the upper levels of the classification (coarsest spatial scale of resolution) and floristic criteria at the lower levels (finest spatial scale). Because these finer levels of the classification are difficult to detect and map with remote sensing technology, they are generally less useful for regional conservation planning in most parts of the world. Although The Nature Conservancy has used this classification in its ecoregional planning work, its use has largely been restricted to the United States (Groves et al. 2000). Scientists from TNC, gap analysis planners, and NatureServe are now modifying the classification to make it a more geographically robust tool by incorporating a classification level that identifies vegetation communities based on dominant species, that is detectable by remote sensing imagery, and that can be consistently applied across the spatial scale of ecoregions or similarly scaled planning units.

Abiotic targets. The increasing availability of regional, national, and global data sets on environmental variables such as elevation, soil, and geology makes them attractive targets for conservation planning, especially for parts of the world where there is a dearth of biological information. For example, Pressey and colleagues (2000) developed a classification of landscape targets covering all of New South Wales (NSW), Australia, that was derived mainly from abiotic features. The classification system was subsequently used as a surrogate for biodiversity to assess the extent to which conservation areas in NSW are representative of the state's biodiversity. Although environmental factors are known to influence the distribution of many species, other studies have demonstrated that combining abiotic targets with biotic targets results in a system of conservation areas that is more representative of a region's biodiversity (Kirkpatrick and Brown 1994). Several recent planning efforts in Australia (Smart et al. 2000), Papua New Guinea (Nix et al. 2000), the United States (Southern Rocky Mountains Ecoregional Team 2001), and South Africa (Cowling et al. 1999) have used approaches that combine abiotic and biotic targets.

Because of the paucity of biological information available for aquatic species and communities, TNC developed a classification framework for freshwater ecosystems that accomodates biological data, but is based on abiotic variables that have been shown to strongly influence biotic patterns at multiple scales (Lammert et al. 1997, Groves et al. 2000). This classification is used in conjunction with biotic data to inform the conservation planning process. Similar efforts are under way in the National Gap Analysis Program (Jennings 2000). The TNC classification loosely follows the hierarchical model of Tonn (1990); it includes regional-scale units (ecological drainage units) that take into account regional drainage (zoogeography), climatic, and physiographic patterns; mesoscale units (aquatic ecological systems) that are aggregations of local-scale units tied together by dominant ecological processes; and local-scale units (macrohabitats) that are small to medium-sized lakes and valley segments of streams defined by hydrology and map-based criteria (stream size, gradient, connectivity, catchment geology) to represent local environmental patterns and processes (figure 3).

In marine environments, most classification systems are based on a combination of biotic and abiotic units. Biotic units can be either vegetative (e.g., seagrass, saltwater marsh, kelp) or faunal (e.g., oyster, coral). Many marine classifications also include abiotic units (Dethier 1992), especially in offshore environments where there is less biological information (Day and Roff 2000). These classifications, whether described by biotic or abiotic factors, are generally known as "habitat" classifications, although they are often consistent with terrestrial ecosystem classifications. The most promising way to select conservation areas in marine environments is to focus on these habitats and the ecological processes that sustain them, an approach taken by TNC (Beck and Odaya 2001) and others (Ward et al. 1999).

Species. Several categories of species have been identified as being useful for management or conservation purposes (e.g., threatened or endangered, endemic, umbrella, flagship, indicator, landscape, focal, keystone). Because of their rarity, habitat specificity, or area needs, the majority of species in these categories are unlikely to be conserved by a focus on either community or ecosystem or abiotic targets. Most of these categories have received considerable attention in the scientific

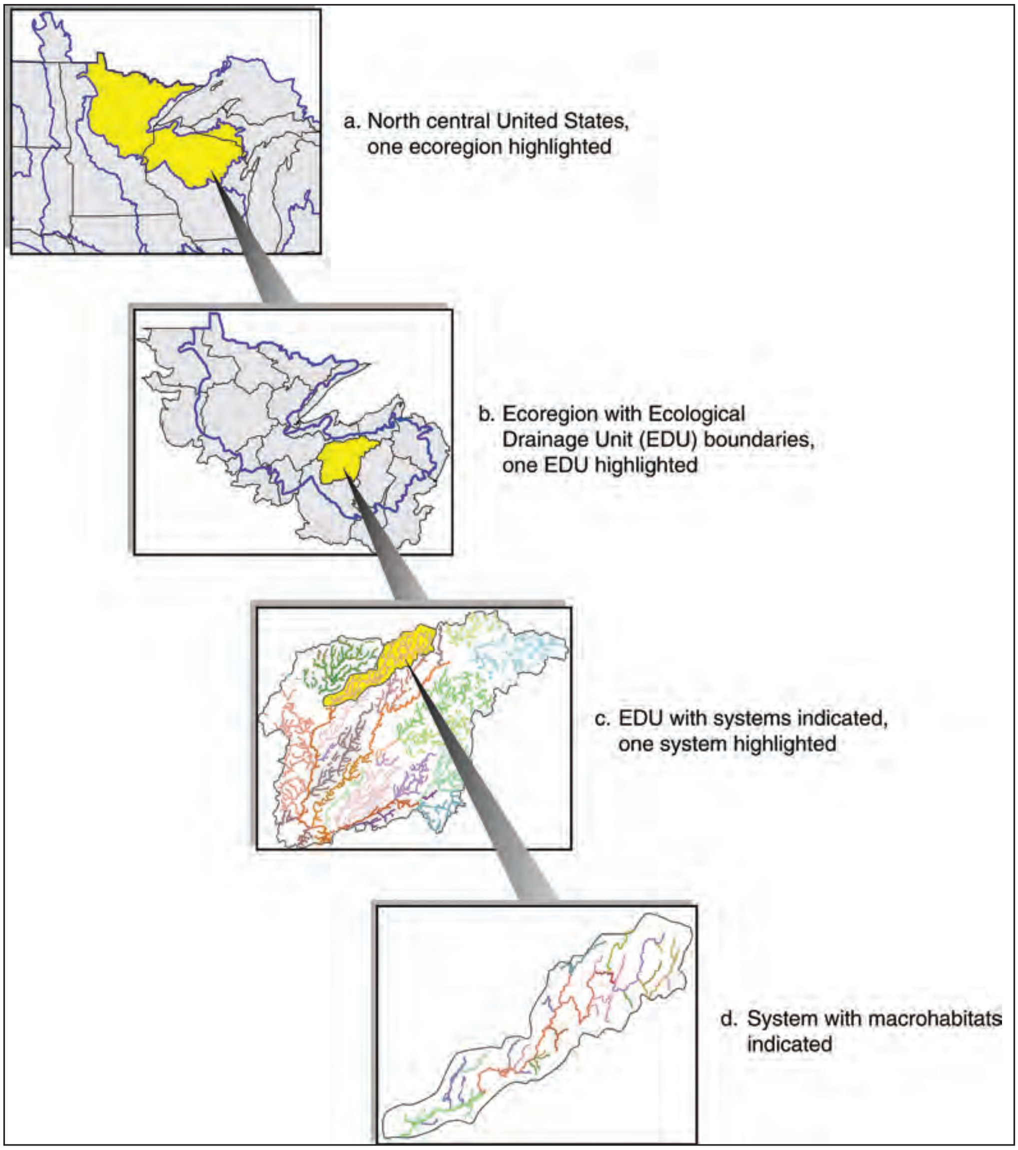

Figure 3. Aquatic classification framework of The Nature Conservancy showing the relationships among the different hierarchical levels of the classification, from ecoregions to macrohabitats. Ecological systems and rare macrohabitats are often selected as conservation targets, especially in the absence of biological information, which is commonly the case in freshwater ecosystems. Ecological drainage units are used to stratify the representation of freshwater conservation targets across environmental gradients.

literature, and several have been criticized on conceptual grounds. Because of questions concerning the utility and validity of flagship, umbrella, and indicator species (see, e.g., Simberloff 1997), this framework emphasizes imperiled, threatened or endangered, endemic, focal, and keystone species as conservation targets.

Imperiled and threatened or endangered species. This category of target species includes those ranked by NatureServe and the network of Natural Heritage programs as globally vulnerable, imperiled, or critically imperiled (for the current listing see *www.natureserve.org*; Master et al. 2000); species listed as threatened or endangered under the US Endangered Species Act (see *www.endangered.fws.gov/endspp.html*); and species listed on the World Conservation Union Red List as vulnerable, endangered, or critically endangered (see *www.redlist.org* for current listing; Hilton-Taylor 2000).

Endemic species. This category consists of species whose entire distribution is restricted to an ecoregion or a small geographic region within an ecoregion. These species make worthy conservation targets because of their limited distribution and associated vulnerability to extinction.

Focal species. Lambeck (1997) defined four types of focal species: area-limited, dispersal-limited, resource-limited, and limited by ecological process (e.g., natural flow regime). Others have defined focal species differently (Noss et al. 1999). For conservation planning purposes, populations of wide-ranging species whose home ranges often exceed that of individual ecoregions are among the most useful focal species (Carroll et al. 2001). Wide-ranging species can be both dispersal- and area-limited. Examples include brown bears, jaguars, sea turtles, and anadromous fishes.

Keystone species. Keystone species have an impact on a community or ecosystem that is disproportionately large relative to their abundance (Power et al. 1996). Although relatively few keystone species (e.g., starfish, beaver) have been identified, their importance to the conservation and function of ecosystems can be substantial (Kotliar 2000).

Step 2: Collect information and identify information gaps

A regional conservation plan for biodiversity requires a variety of data, ranging from human population trends and major land ownership patterns to environmental and biological information on conservation targets (table 1). Fortunately, a great deal of this information is available digitally, and much of it can be found on the Internet (see Groves et al. 2000,

Table 1. Useful categories of information for conservation planning.

Category	Type of information
Land use ownership	Transportation Administrative boundaries Land cover Locations of dams and diversions Water-quality monitoring stations Hydrological flow monitoring stations Point sources for pollution
Physical	Soils Geology Climate Terrain and elevation Wave exposure Wave depth Watersheds and hydrography
Biological	Vegetation cover Wetlands Species distribution Ecoregions and bioregions Shellfish distributions Fisheries data Coral reef distribution and status
Socioeconomic	Population density Population trends Economic trends

appendix A-10, for sources and descriptions). A special issue of *Science* (2000, vol. 289: 2308–2312) that focused on the emerging field of biodiversity informatics provides additional sources for accessing information on biodiversity, including links to a comprehensive list of global databases and Web sites.

The best regional conservation plans utilize information from all available sources, including conservation organizations, public natural resource agencies (local, state, provincial, federal), academia, research institutions, and individual experts. In many cases, critical information necessary for development of a conservation plan may be lacking. These gaps can be filled through use of a variety of techniques that utilize a combination of remotely sensed imagery, reconnaissance overflights, selective biological inventories, and visual display of information with a GIS to cost-effectively gather biological and ecological information about an area; among these techniques are TNC's Rapid Ecological Assessments (Sayre et al. 2000) and Conservation International's Rapid Assessment Programs *(www.biodiversityscience.org/xp/CABS/research/rap/aboutrap.xml)*. Taxon-specific biological inventories can be cost-effective (Balmford and Gaston 1999) and help fill data gaps, especially when the inventories are designed with the intent of providing more accurate estimates of the spatial distributions of species (Margules and Austin 1994). Finally, consultations with experts, often in a workshop setting, have proven extremely useful to both governmental and nongovernmental organizations involved in natural resource management or biodiversity conservation planning (Dinerstein et al. 2000). However, planners need to be aware of some of the assumptions, difficulties, and inherent biases of using expert-based information (Cleaves 1994).

Step 3: Establish conservation goals

Once conservation targets have been identified, planners need to establish explicit goals for them by answering these questions: How much or many of each target should be conserved, and how should these targets be distributed across the planning region? Determining goals is important for several reasons. First, with goals in place, planners can evaluate the effectiveness of a proposed system of conservation areas by asking whether those areas represent the targets at levels requisite for their conservation in the entire planning region (figure 4). Second, goals provide guidance to planners who may have to balance competing demands for lands and waters in the planning region (as happens, for example, when

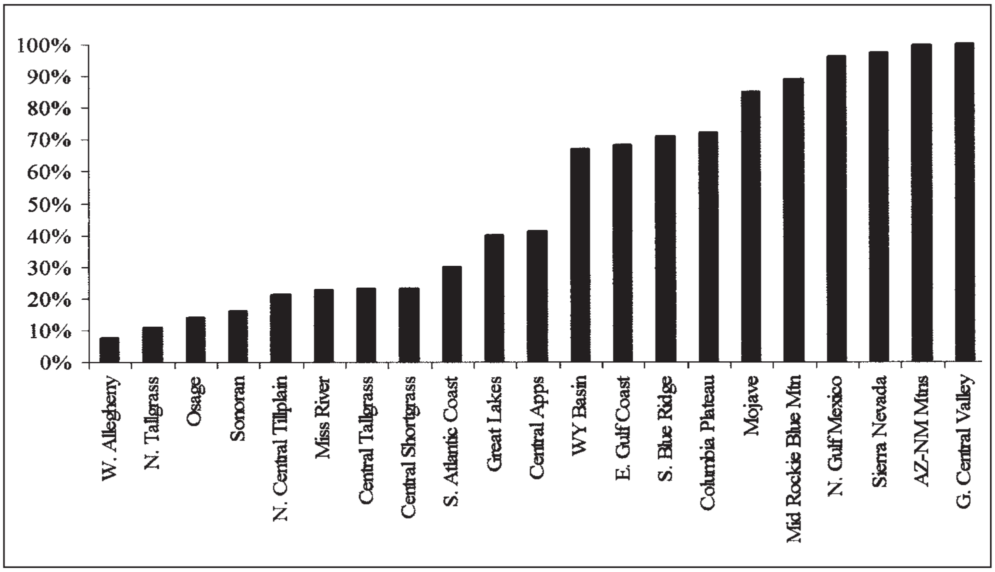

Figure 4. Percentage of conservation targets for which goals were met in several TNC ecoregional plans. "Meeting goals" refers to whether a conservation target is represented a specified number of times in a proposed conservation area across the range of the target within the ecoregion. This graph indicates a general pattern of lower percentages of goals met for ecoregions where natural vegetative cover has been extensively removed or converted. Where conservation goals are not met, it may be necessary to undertake additional biological inventories or restoration efforts. An assessment of conservation goals is one mechanism for measuring the effectiveness of a proposed system of conservation areas.

public agencies operate under multiple-use mandates). Third, goals for targets will ultimately have a strong influence on determining how many conservation areas are needed in a planning region and the extent of area within the region that they will occupy.

Setting meaningful and realistic conservation goals for targets is challenging. There is no scientific consensus on how many populations are needed or how large these populations need to be for conservation of target species (Beissinger and Westphal 1998), although most scientists suggest that a minimal level of redundancy is essential for long-term viability (Shaffer and Stein 2000). For communities and ecosystems, there is little empirical or theoretical research that addresses how best to represent these targets in a system of conservation areas. Finally, in many cases there will be tradeoffs in goals related to the need to conserve multiple examples of targets, on the one hand, while, on the other hand, conserving areas of sufficient quality (see step 5) to persist over the long term.

Conservation goals should have two components: a representation component that refers to the number of occurrences or percentage of each target that should be represented within conservation areas, along with some indication of how those targets should be distributed or stratified across a planning region; and a quality component that addresses the level of viability or ecological integrity thought necessary for these targets to persist over the long term. For example, most marine studies have suggested that ecologically functional reserves will need to cover at least 20% of a planning region if the biodiversity of that region is to be fully conserved. Broader goals have been suggested for marine reserves when an additional goal is to sustain fisheries (see Roberts and Hawkins 2000). Beyond these two components, additional criteria, such as the rangewide distribution of the target relative to the planning region, can be considered in goal setting. For example, if a particular target is endemic to or largely restricted to a planning region, then goals may be set correspondingly higher than for a target that is more widely distributed across several planning regions (Anderson et al. 1999).

Planners also need to ensure that conservation targets are, to the extent possible, distributed across the environmental gradients in which they occur. Doing so helps safeguard against natural catastrophes (storms, disease) that could eliminate targeted features occurring in relative proximity to each other and helps conserve the genetic and ecological variation that occurs in target species and communities across their range. Most ecoregional classifications are hierarchical and have already been divided into subunits based on differences in physical factors (Bailey 1998, Zacharias and Howes 1998). These subunits can be useful for stratifying the distribution of terrestrial conservation targets across the region or ecoregion. In freshwater ecosystems, the level of the classification identified as an ecological drainage unit (figure 3) can serve as a useful stratification unit for conserving aquatic conservation targets across their range of distribution.

Because of the scientific uncertainty involved in setting goals and the need for alternative solutions in most planning processes, biologists and planners should consider setting a range of numeric goals for targets (Jennings 2000). For example, in the Cape Floristic region of South Africa, planners established three goals—10%, 25%, and 50% of the original extent of each vegetation type within the planning area—and then examined alternative portfolios of conservation areas (see step 6) that corresponded to these different goals (Heijnis et al. 1999).

Step 4: Assess existing conservation areas for their biodiversity values

A logical early step in any planning process for conserving biodiversity is to determine what biological features are already under adequate management within existing conservation areas (Margules and Pressey 2000). The biota of many of the world's parks, refuges, wilderness areas, marine protected areas, and nature reserves have been poorly inventoried, in part because of the perception that these areas are already "protected" and that survey funds would be better spent on areas yet to be designated for conservation management. Nevertheless, interviews with resource experts for these protected areas often reveal considerable information on the status and distribution of biodiversity and the need to devote greater management attention to the conservation of this diversity. Remote-sensing imagery of vegetation cover for these areas can also be useful in assessing the status and distribution of community and ecosystem-level targets. Given the limited dollars available for new conservation areas, it is especially important to determine which conservation targets are already within existing conservation areas and the degree to which these areas are being appropriately managed for these targets. The final step in this framework, identifying priority conservation areas (step 7), will use this information as one of the criteria for setting priorities.

The Department of the Interior established the National Gap Analysis Program to undertake the assessment of the degree to which existing conservation areas adequately represent native vertebrate species, threatened and endangered species, and vegetation cover types (Jennings 2000). Irrespective of land ownership, gap programs typically assign a biodiversity management category ranging from 1 to 4 to each conservation area, with status 1 referring to those areas with permanent protection of natural land cover from conversion to status 4, where there is no legal mandate to prevent conversion of natural habitats. Those conservation targets found in status 1 and 2 lands are usually regarded as being under adequate conservation management (*Gap Analysis Handbook*, available at *www.gap.uidaho.edu/handbook)*. The World Conservation Union (1994) uses a somewhat similar though more restrictive approach to classify the world's legally declared protected areas, with six categories ranging from category I (strict nature reserve and wilderness areas) to category VI (areas managed primarily for the sustainable use of natural resources).

Step 5: Evaluate the ability of conservation targets to persist

Conservation planners have devoted considerable resources to representing the elements of biodiversity within a system of conservation areas, but traditionally have paid only scant attention to the factors responsible for the long-term persistence of conservation targets (Balmford et al. 1998, Margules and Pressey 2000). For species, this often means using population viability analyses to assess whether populations can persist over some specified time period (Beissinger and Westphal 1998), an approach largely restricted to a small group of species in the developed world for which data are relatively plentiful. For communities or ecosystems, it means assessing whether disturbance regimes are intact and areas are sufficient in size to ensure survival and recolonization from natural or human-caused disturbances (Poiani et al. 2000).

One practical approach for evaluating the ability of species, community, and ecosystem-level targets to persist is to use a qualitative ranking system that employs three criteria: size, condition, and landscape context (Anderson et al. 1999, Groves et al. 2000, Stein and Davis 2000).

Size is a measure of the area or abundance of a conservation target's occurrence. At the species level, size takes into account the area of occupancy and the number of individuals. For communities or ecosystems, size relates to the area needed to ensure survival from large-scale natural disturbances; it has been referred to as the minimum dynamic area (Pickett and Thompson 1978). Planning teams from TNC use both the concept of minimum dynamic area and the area requirements of wide-ranging species to assess the size criterion for community and ecosystem-level targets (figure 5).

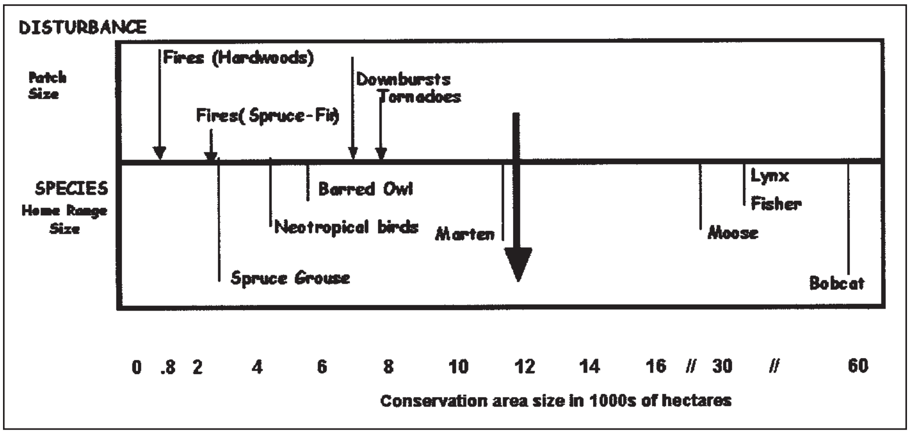

Figure 5. Factors used to assess the adequacy of size for proposed conservation areas of forested ecosystems in the Northern Appalachians Ecoregion. Two principal factors can be used to assess size: the home range of wide-ranging animal species or historical patch sizes from natural disturbances. In this figure, disturbance is defined as four times the patch size of the most severely disturbed patch, based on historic data suggesting that about 25% of any given forested area of New England is expected to be severely disturbed at any one time. The home range estimate is based on the area needed to accommodate a viable population of each species. In the Northern Appalachians Ecoregional Plan, the minimum size for forested conservation areas (large vertical down arrow) was set at approximately 12,000 hectares. From Anderson (1999).

Condition is an integrated measure of the composition, structure, and biotic interactions that characterize the occurrence of a conservation target. For example, this factor would include information on the reproduction and age structure of a population, the canopy or understory structure of a community, or any of several biotic interactions such as predation and disease. In assessing condition, it is often helpful to examine the extent of anthropogenic impacts (e.g., habitat fragmentation and degradation, introduction of exotic species) and the presence or absence of biological legacies—critical features of communities and ecosystems that take generations to develop (e.g., fallen logs and rotting wood in old-growth forests).

Landscape context is an integrated measure of two factors: intactness of dominant ecological processes that help maintain conservation targets (e.g., natural hydrological flow and fire regimes) and connectivity, which allows species to disperse, migrate, and otherwise move to adjacent habitats to meet life cycle needs.

In practice, planners have often found it adequate for their purposes to rate each occurrence of a conservation target, for each of these three criteria, as "very good," "good," "fair," or "poor." Occurrences of those targets that receive an overall fair or poor rating are generally excluded from further consideration in the planning process. Details on the use of this rating scheme and examples of its application are provided by Groves and colleagues (2000). Because of the paucity of information on minimum dynamic areas and disturbance regimes for many communities and ecosystems, much work remains to make these criteria more operational for conservation targets above the species level.

Time and funding, coupled with limited information, usually precludes an evaluation of each of these criteria for all occurrences of conservation targets. One shortcut is to combine various sorts of digitally available information to use as an index of the suitability of a site or area for conservation purposes. Davis and colleagues (1996) used GIS to combine information on road density, human population density, percentage of remaining natural land cover, distance to existing conservation lands, integrity of aquatic systems, and percentage of land in private ownership into a "suitability index" for a biodiversity assessment in the Sierra Nevada Ecoregion. This index, which has now been used in several TNC ecoregional conservation projects, effectively steers planners away from areas with high human use and conversion of natural land cover on the assumption that these areas will be more expensive to manage and that conservation targets in these areas will very likely have lower probabilities of persistence. In freshwater and marine ecosystems, TNC and other regional conservation planning projects have used similar GIS-based suitability indices that aggregate a number of physical and biological criteria (e.g., road density, number of dams, land use and land cover data, percentage of modified shoreline, and point sources of pollution) into an overall "integrity" value (Moyle and Randall 1998, Groves et al. 2000).

Step 6: Assemble a portfolio of conservation areas

Following the collection and mapping of data on conservation targets and assessment of the conditions for persistence, conservation planners can identify a set of potential conservation areas, including areas that do not have acceptable levels of viability and integrity but which may be restored in the future. In most situations, planning teams will have a substantial amount of information on conservation targets, ratings of persistence or suitability, land ownership and management, and other ancillary data sets. Because of the relative complexity of the task, there are a number of advantages to using computerized algorithms with GIS as a tool to aid the identification of conservation areas (figure 6). An algorithm is a step-by-step problem-solving procedure, usually a computational process defined by stipulations written into a computer program. In the case of biodiversity conservation, a common challenge is to select the set of conservation areas that best meets the target-based goals of the project within the smallest area. Fortunately for conservation planners, many such algorithms have been developed; several of them can be accessed for free on the Internet (see Williams 1998 for a review of algorithms for area selection).

The primary advantage of using algorithms is that they allow planners to delineate explicit "rules" to identify a set of conservation areas and to assess alternative portfolios of conservation areas by making changes in these rules. For example, a team might choose to examine a portfolio of conservation areas that is located mostly on public lands versus one that emphasizes private lands. Other teams may find it desirable to design a portfolio of conservation areas with a minimum size requirement for each area. A recent biodiversity plan for Papua New Guinea (Nix et al. 2000) demonstrated how algorithms can be used to integrate economic tradeoffs into the selection of conservation areas or to eliminate certain areas (e.g., highly altered lands) within the planning region from consideration.

Staff members or partner organizations that undertake conservation action or management for particular conservation areas need to be involved in the application of algorithms designed to select these areas. In Australia, interactive algorithms for area selection have been used to negotiate settlements between timber companies and conservationists regarding the use of public lands (Pressey 1998). Experiences in TNC's ecoregional planning efforts suggest that managers and conservation practitioners who do not understand the algorithms or why a particular place has been identified for conservation will be less supportive of a regional conservation plan than they otherwise might be (Groves et al. 2000).

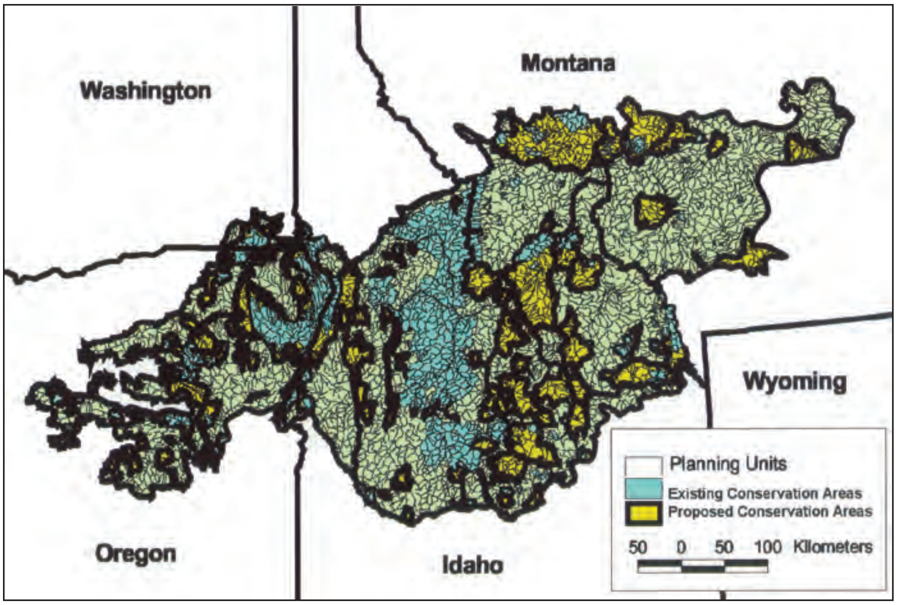

Figure 6. Portfolio of conservation areas for the Middle Rockies–Blue Mountains Ecoregion. Conservation areas are roughly delineated along the boundaries of watersheds referred to as HUCs (hydrological unit codes). HUCs make excellent base map units for organizing a variety of biological, socioeconomic, and environmental data and can serve as a generalized selection unit for conservation areas. HUCs are available digitally from the US Environmental Protection Agency at a variety of spatial scales. From Middle Rockies–Blue Mountains Planning Team (2000).

The final task in assembling a portfolio of conservation areas is consideration of the overall configuration or design of the portfolio. Several design principles for a network of conservation areas have emerged from biogeographic theory and landscape ecology (Noss et al. 1997). Collectively, these principles lead to an emphasis on selecting landscape-scale conservation areas. Typically, these areas contain larger, more viable occurrences of conservation targets and are more likely to be sustained by intact, functional ecological processes (Soulé and Terborgh 1999).

Decisions concerning the overall design or configuration of a network of conservation areas must balance the desirability of securing new conservation areas and enlarging existing ones with the need to consider proximity and connectivity among these areas. In practice, this has proven both difficult and contentious. It is difficult because there is often little biological information to guide the design of connectivity. It is contentious because there are convincing arguments in favor of establishing linkages among conservation areas (Beier and Noss 1998), but there is also compelling evidence that the configuration of conservation areas is not nearly as important to species survival as preventing overall habitat losses (Fahrig 2001).

Step 7: Identify priority conservation areas

Experience in TNC ecoregional planning projects indicates that most plans will identify over 100 potential conservation areas. Some of these areas are in urgent need of conservation action, while others are not. Therefore, a final step in this planning framework is to set priorities for action among the portfolio of potential conservation areas. Our planning framework uses five criteria for setting these priorities: degree of existing protection, conservation value, threat, feasibility, and leverage (Groves et al. 2000).

"Degree of protection" refers to how well or the extent to which conservation targets are already represented within the existing set of conservation areas in an ecoregion (step 4). Higher priority is given to areas with targets that are not already well represented. The conservation value of an area is based on the number of conservation targets, the diversity of these targets (e.g., terrestrial and aquatic), and their predicted ability to persist over the long term. Areas with more conservation targets (step 1) and higher persistence or suitability ratings (step 5) are assigned a higher priority. Conservation areas that face critical threats are assigned a higher priority than those that are not imperiled; the greater the degree of threat, the higher the priority. Feasibility refers to an organization's capacity to gain protection for an area (through land acquisition, for example) and to secure sufficient funding, staff, and strategies to abate critical threats. Finally, leverage is the ability to take conservation action at one area and thereby effect conservation action at other areas. In practice, a qualitative rank of high, medium, or low is assigned for each criterion (see Groves et al. 2000 for definitions of qualitative ranks) for each potential conservation area. These criteria rankings are summed for the conservation areas, each of which is assigned an overall priority rank. As with any qualitative ranking scheme, results should be used in setting priorities in conjunction with the sound judgment and personal knowledge of conservation areas by members of the planning team and other experts.

Approaches to regional conservation planning

Several scientists have advanced principles, characteristics, and criteria for the development of biodiversity conservation plans. For example, Shaffer and Stein (2000) outlined three principles for successful conservation of biodiversity that they termed representation, resilience, and redundancy. *Representation* in its simplest form means "saving some of everything"—ensuring that all species and communities native to a region can be found, to the greatest extent possible, within lands and waters that are primarily managed for conservation purposes (step 1). *Resilience* refers to ensuring that these species and communities can persist and evolve for long periods of time (step 5). *Redundancy* admonishes conservation practitioners to refrain from placing all of their eggs in one basket, thereby hedging bets of failure of any single population of a species or occurrence of a community to survive (step 3). Our framework is entirely consistent with these principles.

Margules and Pressey (2000) outlined a six-stage framework for systematic conservation planning. Shafer (1999) developed a similar set of steps for reserve planning in national parks. Their stages included identifying which biotic and abiotic features can serve as surrogates for biodiversity in the planning region and gathering information on these features (steps 1 and 2); setting explicit goals for these features, including goals for ecological processes (steps 3 and 5); assessing existing conservation areas for their representation of these features (step 4); selecting new conservation areas (step 6); implementing conservation action according to priority level (step 7); and effectively managing and monitoring conservation areas. With the exception of this final stage regarding the management of conservation areas, which we earlier suggested is best accomplished through a separate site or project planning process, the seven-step framework incorporates and is consistent with these stages.

Soulé and Terborgh (1999) outlined a scientific program for conserving nature in North America. The rationale for this program, the Wildlands Project, centers on the idea that networks of large and well-connected protected areas (referred to as core areas or wildlands) require keystone species, especially large carnivores, to stabilize prey populations and maintain ecological diversity. Core areas are selected on the basis of three criteria or types of conservation targets (Noss et al. 1999): representation, special elements, and focal species. Representation refers to conserving intact examples of each vegetation or habitat type (defined as target ecosystems in step 1) across the environmental gradients in which they occur. Special elements are rare species and communities, pristine sites (e.g., roadless areas), and other features unique to a region (e.g., artesian springs, mineral licks, indigenous sacred sites) that are thought to have high conservation value. Finally, focal species are conservation targets whose needs define answers to two questions: How large do conservation areas need to be, and what should their configuration be?

With the exception of some special elements, the three types of conservation targets used by Noss and colleagues (1999) are consistent with those identified in step 1. We elected to not include such features as mineral licks, springs, caves, and roadless areas as a type of conservation target, unless they had identifiable biotic targets associated with them or were part of an environmental or physically derived classification system.

In practice, the Wildlands Project has emphasized wide-ranging carnivores as targets and connectivity between core areas to a greater extent than TNC ecoregional projects, whereas TNC projects have placed greater emphasis on using a more comprehensive set of conservation targets at a variety of spatial scales to select conservation areas. Both steps are important aspects of conservation planning, and TNC's ecoregional projects are now moving to better incorporate wide-ranging species and network design, and the Wildlands Project is seeking to bring greater consistency to its conservation planning methods across projects (Barbara Dugelby, [The Wildlands Project, Blanco, Texas], personal communication, September 2000).

Conclusions

As the list of endangered species grows longer, it is clear that additional strategies and approaches are needed to conserve biological diversity. Because habitat loss and degradation are the leading causes of imperilment for most species (Wilcove et al. 1998, Hilton-Taylor 2000), it is equally clear that more lands and waters need to come under conservation management if future losses are to be prevented. We have outlined a framework for identifying the most important remaining areas for conservation and restoration. The seven-step framework is based upon scientific principles and theories that represent a synthesis of thinking from population biology, community ecology, and landscape ecology. Although the methodology for the framework differs from some other regional planning approaches, there are more similarities than differences. A consensus is emerging on the most important elements of planning for the express purpose of conserving biological diversity. Some of the underpinnings of the seven steps rest on assumptions that remain inadequately tested (e.g., surrogate measures for biodiversity) and methods that are not yet fully developed (e.g., assessing persistence of conservation targets). Nevertheless, the urgency of the conservation mission demands that conservation plans based on the best available scientific information and methods be implemented now, while explicitly acknowledging their limitations and working toward their improvement..

This seven-step approach to conservation planning, which has been applied to terrestrial, freshwater, and marine environments, offers numerous benefits. First, it allows conservation planners to set goals that are based on assessments of the biological needs of species, communities, and ecosystems, not on arbitrary, subjective estimates of how much land a society can set aside in protected areas (Soulé and Sanjayan 1998). Second, this framework complements single-species conservation approaches by incorporating a broad set of conservation targets at a variety of levels of biological organization and spatial scales. Third, at a median cost of $234,000 per plan ($n = 24$ plans, staff salary, and all operating costs included) and an average completion time of just less than 2 years, application of the framework strikes a reasonable balance between planning and action. Fourth, the framework provides an explicit means for conservation planners to measure whether the set of conservation areas that they have identified will sufficiently represent the biodiversity of the region and achieve the target-based goals of the plan. Fifth, the proposed framework pays due diligence to a long-overlooked aspect of conserving biodiversity: the underlying ecological processes and functions that support the long-term persistence of biodiversity. Finally, by using an approach that represents biodiversity in a set of conservation areas across environmental regimes in which targeted features are known to occur, the framework may help conserve biodiversity in the face of global climate change (Halpin 1998).

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13.3: Prioritization - What Should be Protected?

Historically, the boundaries of protected areas was often determined through pragmatic considerations, such as the availability of funds and land, and political influence, rather than ecological considerations. Many conservation areas were thus established on “lands that nobody wants”: marginal areas with little agriculture and development potential, or areas that were too remote to have high commercial value (a trend that continues even today: Venter et al., 2018). Other protected areas were established in locations with charismatic megafauna, so ecosystems without those species remained unprotected. Consequently, some of Africa’s most threatened species and ecosystems remain under-protected (Beresford et al., 2011).

> In a crowded world with finite natural resources and limited funding, it is increasingly important to be strategic about where protected areas are established.

In a crowded world with finite natural resources and limited funding, it is becoming increasingly important to be strategic about where protected areas are established. To do this, conservation biologists and policy makers must answer three key questions: (1) What is most important to protect? (2) Where would it be best protected? (3) How could it be most effectively protected? Three criteria can be used to answer the first two of these questions:

- Distinctiveness (or irreplaceability): Ecosystems with species that are distinct in their taxonomy (e.g. ecosystems that contain the only species in a taxonomic group) or geographic distribution (e.g. endemic species), or ecosystems with unique attributes (e.g. scenic landscapes, unusual geological features).
- Endangerment (or vulnerability): Areas that contain concentrations of species threatened with extinction, or ecosystems in danger of being destroyed.
- Utility: Species and ecosystems that people value, including culturally significant species, economically valuable species or ecosystems, or areas that can contribute to combating climate change.

Using these criteria, scientists have developed several broadly complementary methods to prioritise areas for protection. The approaches differ more in what traits they emphasise rather than in their fundamental principles. Thus, although some people may argue about which approach is better, each approach contributes to the protection of biodiversity.

Species approach

Many protected areas are created to protect (e.g. threatened, culturally significant, or keystone) species. Species that provide the motivation to establish a protected area are known as focal species. As a prominent example using the focal species concept, the Alliance for Zero Extinction (http://www.zeroextinction.org) identified 67 priority sites across Sub-Saharan Africa (853 sites globally) that contain the last remaining populations of one or more Endangered or Critically Endangered species. Flagship species, such as gorillas, are a special kind of focal species because they capture public attention, have symbolic value, and are important for ecotourism purposes. Many flagship species and focal species are also umbrella species, because their protection indirectly benefits other species and ecosystem components with which they share their landscape.

Protected areas are often established to protect threatened or charismatic species, unique ecosystems, and or wilderness areas.

Ecosystem approach

There is debate among conservation biologists over whether ecosystems rather than individual species should be the primary target of conservation efforts. Supporters of an ecosystem approach argue that protecting and managing ecosystems can preserve more species and provide more value to people than spending the same amount of money to protect individual species. Focusing on ecosystems also allows for greater flexibility in justifying conservation efforts, because it can be easier to demonstrate the economic value of ecosystems for helping to control floods, filtering water, and providing opportunities for recreation. To that end, the WWF has identified 238 ecoregions across the globe (the “Global 200”)—57 of them in Sub-Saharan Africa—that are most crucial to the biodiversity conservation (Olson et al., 2002). This Global 200 analysis formed the basis of a more recent global assessment that identified 41 at-risk ecoregions—areas of high conservation priorities because they are undergoing high levels of habitat conversion and have low protected areas coverage (Watson et al., 2016). Africa has several at-risk ecoregions, particularly in Angola, South Africa, the DRC, and West Africa’s Sahel region. The IUCN Red List of Ecosystems (RLE, Section 8.5.1) is another example of an ecosystem-focused prioritization for conservation. While the ecosystems approach overcomes several limitations of the species approach, some conservationists argue that focussing on distinct ecosystems may, in itself, be detrimental, and that the scope of conservation should be expanded, for example by also including biogeographic transition zones (van Rensburg et al., 2013).

Wilderness approach

Wilderness areas are large areas where people have had little influence on the environment (relative to other areas), they have few people living in them, and are unlikely places for human development in the short term. These areas are conservation priorities because they may be the only places where animals that require large home ranges can continue to survive in the wild. Further, wildernesses can serve as controls or benchmarks for researchers to measure the effect of human disturbance on nature. The most popular way to identify wilderness areas is to identify areas without roads. While very few roadless areas remain, many of the world’s most important roadless wildernesses, some larger than 10,000 km2, are in Africa (Ibisch et al., 2016). Of concern is that, second to South America, Africa also leads the world in wilderness losses over the past decade (Potapov et al., 2017). It is worth emphasising that even wilderness areas have had a long history of human activity (Roberts et al., 2017). It is not always necessary or even possible to eliminate all human activity from such areas, if those activities do not obstruct conservation goals.

Hotspot approach

Multiple prominent initiatives have prioritised conservation in areas where large concentrations of species can be protected in a relatively small area. Perhaps the most prominent example is the Global Biodiversity Hotspots initiative. Combining a species approach with an ecosystem approach, Global Biodiversity Hotspots are areas with exceptionally high levels of biological diversity and endemism—that is, irreplaceable biodiversity—that are threatened with imminent habitat destruction (Table 13.2). Norman Myers, a British biologist who launched his conservation career as a wildlife photographer in Kenya, originally proposed the Biodiversity Hotspot concept (Myers, 1988). Working with a team of prominent scientists, Myers identified 25 Hotspots (five of them in Sub-Saharan Africa), which contained 44% of all vascular plant species and 35% of all terrestrial vertebrate species on only 1.4% of the Earth’s land surface (Myers et al., 2000). More recently, Conservation International (CI) identified an expanded set of 36 Biodiversity Hotspots (Mittermeier et al., 2005), eight of which are in Sub-Saharan Africa (Figure 13.3). This expanded set of Biodiversity Hotspots covers only 2.3% of Earth’s surface yet contains over 50% of all plant species and over 40% of all terrestrial vertebrate species.

Table 13.2 A natural history comparison of Sub-Saharan Africa’s eight Global Biodiversity Hotspots.

Location	Original extent (× 1,000 km2)	Remaining undisturbed vegetation (%)	Number of species		
Plants	Birds	Mammals			
Guinean Forests of West Africa	620	15	9,000	917	390
Succulent Karoo	103	29	6,356	225	75
Cape Floristic Region	90	20	9,000	320	127
Maputaland-Pondoland-Albany	274	25	8,100	631	202
Coastal Forests of Eastern Africa	291	10	4,050	633	198
Eastern Afromontane	1,018	11	7,600	1,300	490
Indian Ocean Islandsa	601	10	13,500	503	211
Horn of Africa	1,659	5	5,000	697	220

Source: Mittermeyer et al., 2004; https://www.cepf.net/our-work/biodiversity-hotspots.

a Includes Madagascar and Mascarene islands

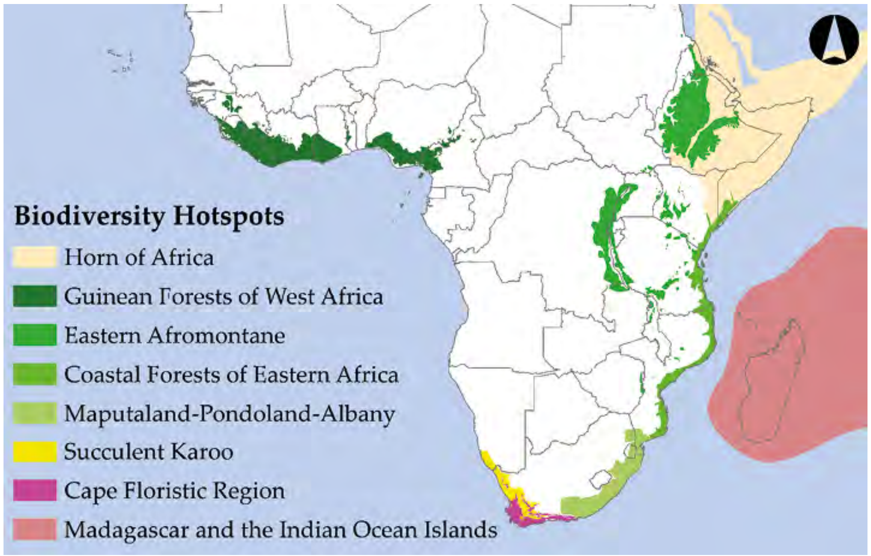

Figure 13.3 Sub-Saharan Africa’s eight Global Biodiversity Hotspots. These areas are targets for protection because of their high biodiversity, endemism, and significant threat of imminent extinctions. After Mittermeier et al., 2005. Map by Johnny Wilson, CC BY 4.0.

While the Global Biodiversity Hotspots highlight some of the most important global conservation priorities, none of these Hotspots are small enough to be contained in a single protected area—in fact, most of these Hotspots identify whole regions, not projects, requiring conservationists to still make decisions for prioritising protection within them. To create actionable priorities from within regional hotspots, several initiatives aim to identify local hotspots of species richness that can be conserved as one protected area of a manageable size. One such approach is the Key Biodiversity Areas (KBA) program (Eken et al., 2004), which identifies conservation priorities using standardised criteria and thresholds that account for concentrations of threatened species and/or globally significant population aggregations. The KBA program functions as an umbrella designation for several taxon-specific approaches, most prominently BirdLife International’s Important Bird and Biodiversity Areas (IBA) program (Fishpool and Evans, 2011). Other KBA programs include PlantLife International’s Important Plant Areas program (e.g. Smith and Smith 2004), as well as the Important Sites for Freshwater Biodiversity program (Darwall et al., 2005). One example from Guinea used KBA criteria and thresholds regarding threatened mammals to provide suggestions for expanding the country’s protected areas network (Brugiere and Kormos, 2009).

Gap analysis approach

Assessing the performance of existing protected areas can be done by spatially comparing their footprint to prioritised conservation areas (as above). Such an assessment offers not only an assessment of existing protected areas performance, but also offers a means to identify conservation gaps—important areas that still need to be protected to meet broader conservation goals. Such assessments, which systematically evaluate whether different aspects of biodiversity are adequately protected, are collectively known as systematic conservation planning assessments (McIntosh et al., 2017). Perhaps the most popular systematic conservation planning method is gap analysis, during which scientists overlay maps of species (or ecosystem) distributions with maps of protected areas to identify species (called gap species, see also Figure 10.3) or ecosystems that are not adequately protected in existing protected areas networks (Box 13.2).

> Gap analysis enables conservation planners to identify species or ecosystems that are not adequately protected in existing protected areas networks.

Box 13.2 Identifying Key Sites for Conservation in the Albertine Rift

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The Albertine Rift is one of the richest regions on Earth for vertebrate diversity (Figure 13.B). Spanning about 100 km either side of the international border of the eastern DRC, it includes forests, wetlands and savannahs from eastern DRC and western Uganda, Rwanda, Burundi, and Tanzania, and runs from the northern end of Lake Albert to the southern end of Lake Tanganyika. It contains more than 40% of Africa's mammals, 52% of Africa's birds, as well as 19% of its amphibians and plants, in only 1% of the continent's surface area. It also contains more endemic and globally threatened species than any other ecoregion in Africa (Plumptre et al., 2007). Endemic large charismatic species include the eastern gorilla (Gorilla beringei, CR), golden monkey (Cercopithecus kandti, EN), Congo bay owl (Phodilus prigoginei, EN), and Ruwenzori turaco (Ruwenzorornis johnstoni, LC). The lakes in the Albertine Rift each also contain several hundred unique fish species. Unfortunately, this rich biodiversity also occurs in one of the most densely populated parts of Africa, and the threats to existing protected areas are high.

Figure 13.B (Top) Mubwindi Swamp, in Bwindi Impenetrable National Park, an important site for mountain gorillas and the Albertine Rift endemic Grauer's Rush Warbler (Bradypterus grayeri, EN). (Bottom) A Grauer's gorilla, the largest of the four gorilla subspecies and a flagship for conservation efforts in the Albertine Rift. Photographs by A.J. Plumptre/WCS, CC BY 4.0.

The Wildlife Conservation Society (WCS) has been working to support the conservation of six key landscapes in the Albertine Rift (ARCOS, 2004), particularly focusing on (a) identifying critical areas for conservation of threatened and endemic species; (b) undertaking research and monitoring of species and key landscapes; and (c) supporting the conservation of critical sites and the creation of new protected areas to conserve large and small mammals, birds, reptiles, amphibians and plants in all protected areas, as well as sites where new protected areas might be established. These surveys have identified critically important areas in eastern DRC, such as the Itombwe and Kabobo Massifs where new species have been identified and some species were rediscovered, having been last seen more than 50 years ago. Working with local communities, the surveys have been used to design the boundaries of newly established protected areas to ensure that they capture as much of the biodiversity as feasible. Once the local people in the area are presented with survey results and options for protection discussed, they often realise the importance of their site and propose more stringent protection measures than conservationists initially thought possible.

Using species distribution models (SDM) of the region's endemic and globally threatened species, WCS gained an understanding of where these species should occur both now and under future climate change scenarios (Ayebare et al., 2018). Using Marxan software (Possingham et al., 2000), WCS then identified those areas that would conserve all the species of conservation interest at minimum cost (Plumptre et al., 2019). This procedure identified the Itombwe and Kabobo Massifs together with the Sitebi Hills east of Mahale Mountains National Park in western Tanzania as being critical for conservation of species that are currently not adequately protected (Figure 13.C).

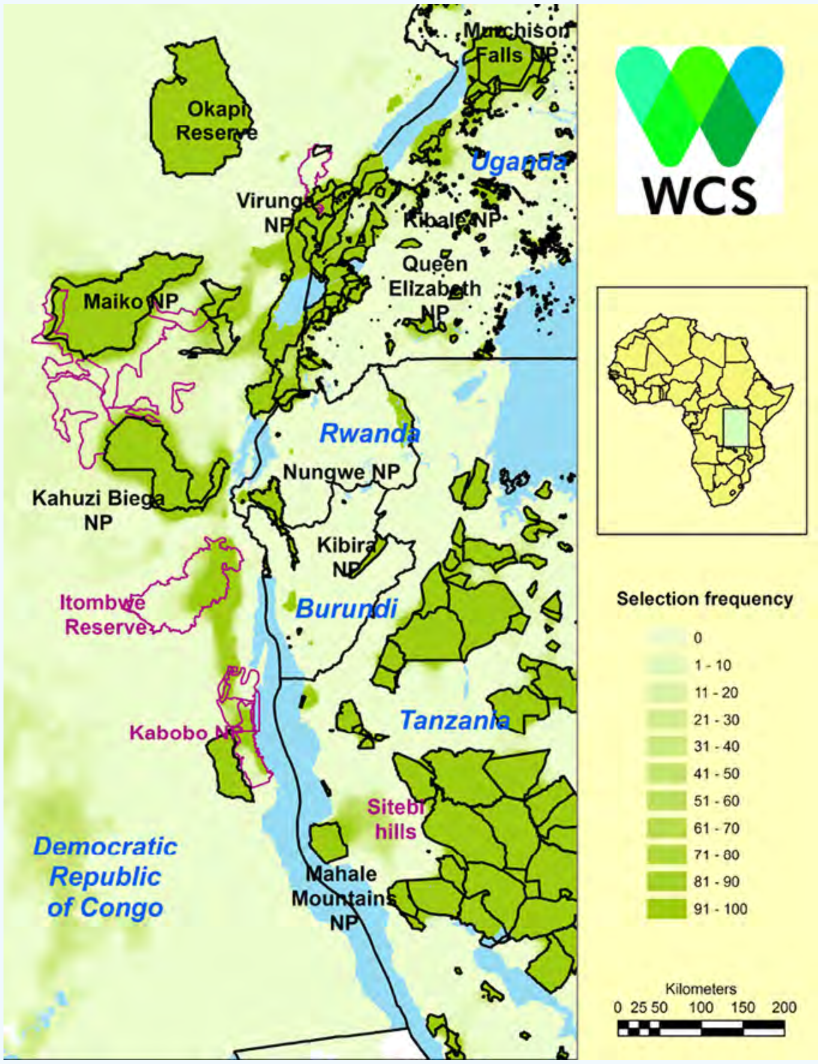

Figure 13.C Selection frequency of 5 km2 cells in the Albertine Rift from Marxan analysis, indicating priority areas for the conservation of endemic and threatened mammals, birds, reptiles, amphibians, and plants. Existing protected areas (all highlighted) were locked in but proposed protected areas such as Itombwe and Kabobo and community reserves (purple boundary) were not. Darker green areas indicate priority conservation sites. Image courtesy of WCS Albertine Rift Program, CC BY 4.0.

These results were used to develop an Albertine Rift Action Plan (Plumptre et al., 2016), together with detailed conservation action plans for the preservation of the six core landscapes and their unique and threatened species, both inside and outside of protected areas, now and into future.

When identifying conservation gaps, it is important to think carefully about the taxa or ecosystem used to make the assessment. Many conservation assessments assume that one well-known species group can act as a biodiversity indicator (also known as a biodiversity surrogate or surrogate species) for lesser-known taxa, so establishing a protected area to protect one gap species will also afford protection to other under-protected taxa. While this is true to some level, several studies have shown that this may not always be the case (Rodrigues and Brooks, 2007; Carwardine et al., 2008; Jones et al., 2016).

Optimization Approach

Prioritisation efforts typically need to consider multiple factors in addition to biodiversity, such as cost-effectiveness, socio-economics, site condition, and potential threats that may impact a proposed protected area. Technical computer software known as “decision support tools” are providing a new way to identify conservation priorities that meet a suite of conservation objectives. One of the most popular packages is Marxan (http://marxan.org), a freely available program that identifies the optimal location for protected areas based on flexible user-defined criteria (Watts et al., 2009). The user-defined criteria can be complex; for example, one can set the model parameters to choose the areas that best protect certain aspects of biodiversity (e.g. protect at least 25% of each vegetation type) while reducing costs and minimising impact on other stakeholders; model input can include measured data, as well as expert input. In one such example, conservation biologists from South Africa, Eswatini, and Mozambique used Marxan to identify potential locations for new protected areas in the Maputaland Centre of Endemism which the three countries share. They found that adding 4,291 km2 to the existing protected areas network could generate US $18.8 million in revenues while fulfilling their conservation objectives: protecting 44 landcover types, 53 species, and 14 ecological processes (Smith et al., 2008).

> Decision support tools help identify conservation priorities that meet a suite of objectives, including cost-effectiveness, socio-economics, and site condition.

Regardless which prioritization approach one follows, it is important to remember that prioritising species and ecosystems in need of protection does not amount to “doing conservation”. Real conservation only happens when a conservation plan that will implement those suggestions is drawn up and put in place. A review of eight different systematic conservation assessments in South Africa provides a good foundation to guide conservation biologists in the process from prioritization to implementation (Knight et al., 2006).

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Review

Slow Lives in the Fast Landscape: Conservation and Management of Plethodontid Salamanders in Production Forests of the United States

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Abstract: Intensively-managed forest (IMF) ecosystems support environmental processes, retain biodiversity and reduce pressure to extract wood products from other forests, but may affect species, such as plethodontid salamanders, that are associated with closed canopies and possess limited vagility. We describe: (1) critical aspects of IMF ecosystems; (2) effectiveness of plethodontid salamanders as barometers of forest change; (3) two case studies of relationships between salamanders and coarse woody debris (CWD); and (4) research needs for effective management of salamanders in IMF ecosystems. Although plethodontid salamanders are sensitive to microclimate changes, their role as ecological indicators rarely have been evaluated quantitatively. Our case studies of CWD and salamanders in western and eastern forests demonstrated effects of species, region and spatial scale on the existence and strength of relationships between plethodontid species and a "critical" microhabitat variable. Oregon slender salamanders (*Batrachoseps wrighti*) were more strongly associated with abundance of CWD in managed second growth forests than ensatina salamanders (*Ensatina eschscholtzii*). Similarly, CWD was not an important predictor of abundance of Appalachian salamanders in managed hardwood forest.

Gaining knowledge of salamanders in IMF ecosystems is critical to reconciling ecological and economic objectives of intensive forest management, but faces challenges in design and implementation.

Keywords: intensive forest management; plantation; plethodontid; production forestry; salamander; woody debris

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1. Introduction

Forest ecosystems are critical for global primary productivity, carbon storage and sustaining human populations [1]. Increased anthropogenic use, however, threatens forest ecosystems worldwide. In many areas, traditional forms of exploitation, such as over-harvesting of trees, remain primary threats [2]. However, additional forms of utilization (e.g., biomass harvesting, bushmeat hunting, firewood and food collection), conversion to alternative production uses (food crops for humans or livestock), loss (to expanding residential/commercial infrastructure) and inability to match regional supply and demand represent contemporary, acute stresses on forest ecosystems [2–7]. These changes have profound implications, as forests provide habitat for an estimated 80% of the world's terrestrial biodiversity [8] and contribute ecosystem services (e.g., clean air and water, recreation sites) valued at more than 16 trillion dollars USD [9].

Intensively-managed forest ecosystems (IMF) represent an opportunity to alleviate pressure on natural forests from existing or future stresses, especially relative to supplying wood products [10–12]. To realize the potential benefits of IMF ecosystems, an increased understanding is required of how specific practices affect habitat structures and cover types that support biological diversity [13]. For example, to produce commodities under contemporary restrictions on the expansion of area under management, IMF ecosystems in the "wood baskets" of the Pacific Northwest (PNW) and Southeastern (SE) United States depend on rapid regeneration of harvested stands and frequent stand turnover. Site preparation, planting of specific genotypes or clones and control of competing vegetation are critical tools for meeting production targets [14,15]. Implementing these tools at the harvest unit (*i.e.*, stand) scale, combined with a reduction or absence of natural disturbances and application of forest practice regulations, creates novel landscape patterns that differ substantially from those created by historic disturbance regimes alone [16–18]. Increasing intensity of forest management, which compresses successional development, particularly the duration in which stands reside in mature structural stages, has unique implications for organisms with limited dispersal capabilities and slow rates of population growth, such as salamanders.

Salamanders in the family Plethodontidae occur primarily in forests of the New World and reach their greatest diversity in the United States in southeastern and northwestern forests [19,20]. Plethodontid salamanders are often abundant, constitute biomass in amounts comparable to mammals or birds, have broad functional roles as predator and prey and are morphologically and physiologically linked to cool and moist microhabitat types [20–23]. Collectively, these characteristics have garnered the attention of biologists interested in understanding broader ecological effects of forest disturbances, including timber harvesting and management. For example, given their ecology and life history characteristics, salamanders are often proposed as indicator species of environmental health, although a limited amount of empirical information is available to evaluate this claim [24–26]. Numerous experimental and observational studies have reported declines in abundance following a range of forest regeneration methods across North America (reviewed in [27,28]). Presumably, this response derives from a warmer and dryer microclimate after canopy removal, indicating that salamanders can be sensitive to a range of anthropogenic habitat alterations [29]. However, unified understanding of causes of variation in inter-specific and inter-eco-regional responses of salamanders to silvicultural practices or to specific attributes of managed forests is lacking [28,30].

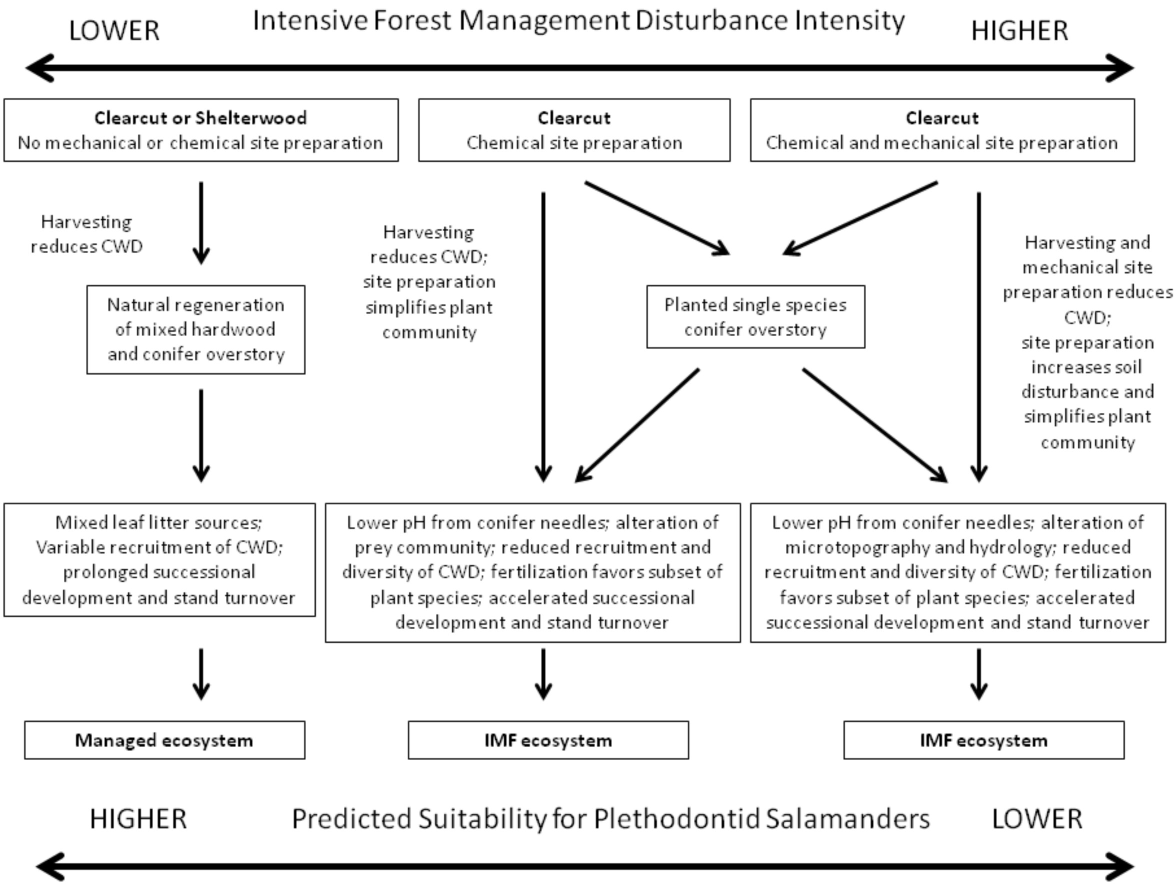
Here, we describe contemporary patterns in IMF ecosystems and evaluate the use of plethodontid salamanders as effective barometers of how forest management affects conservation of biological resources [31,32]. To address this issue, we focus our discussion on temperate forests of the PNW and SE United States, given their global significance for wood production and as hotspots for terrestrial salamander diversity [20]. We explore use of plethodontid salamanders as indicators of forest ecosystem condition, including response to anthropogenic disturbances. Next, we present two case studies to explore responses of salamander populations to coarse woody debris (CWD), a structural component that is important for maintenance of many salamander populations and that is often altered by forest management. Given our findings, we make specific recommendations about research questions and appropriate study designs to inform policy development for best management practices and sustainability targets.

2. IMF Ecosystems in Time and Space

Historically, production of wood commodities relied on unregulated harvesting and passive regeneration practices distributed broadly across the forested land base [33,34]. In addition, given the volume of supply relative to demand, the harvesting pressure and levels of utilization were extremely variable spatially and temporally [35]. In contrast, IMF ecosystems produce wood from relatively uniform operational practices deployed across a restricted portion of the available forested land base [15]. Intensively-managed forest ecosystems often occur in landscapes with multiple ownerships that pursue different management objectives and, consequently, differing land use practices. Finally, regulations governing forestry operations and requirements of third-party sustainability certification programs interact with management objectives to shape the distribution of habitat structures and cover types across time and space in IMF landscapes [36]. To be successful, contemporary conservation and management initiatives must address these factors and the dynamic economic environment in which IMF ecosystems develop [10,37].

Regeneration practices in both the PNW and SE are orchestrated to produce high volume stands at harvest. Southern pine (loblolly pine, *Pinus taeda*; slash pine, *P. elliottii*; shortleaf pine; *P. echinata*; and longleaf pine, *P. palustris*) are common plantation species in the SE, whereas Douglas-fir (*Pseudotsuga menziesii*) is preferred in the PNW, with these species grown primarily for structural lumber (southern pine is used also for pulp). Loblolly pine is the most common SE planted species, with ≈14 million ha in plantations, due to their wide tolerance and high growth rates [15,38]. Stand turnover in pine plantations is more frequent in the SE due to the rapid maturity of southern pine stands (~25–35 years *vs.* ~50 years for Douglas-fir). In both regions, selected nursery stock is planted at relatively uniform densities to achieve desired stocking rates and fertilized to accelerate growth. Although herbicides are used for site preparation and vegetation control in Douglas-fir plantations, combinations of both herbicide and mechanical site preparation frequently occur in southern pine plantations [15]. Hardwood management is common in production forests of the Appalachian Mountains and portions of the Piedmont region in the SE, and these forests are managed with a wide range of silvicultural options, including selection harvests or clearcutting to promote the natural regeneration of high quality species, including oaks (*Quercus* spp.) [37,39,40]. Shearing of stumps, bedding or other mechanical site preparation techniques remain a critical regional difference (Figure 1) in regeneration practices that could affect organisms, such as salamanders, that use CWD, leaf litter and upper portions of the soil profile, but more investigation is needed [25,28,41].

Figure 1. Habitat conditions for plethodontid salamanders is influenced by silvicultural regimes and relative management intensity, which influences proximate habitat characteristics, leading to a range of predicted suitability. Adapted from Ramovs and Roberts (2003) [42]. CWD, coarse woody debris.



Across both regions, the application of forestry and water quality best management practices (BMPs; [43]), state forest practice rules (e.g., stipulating riparian/upland set-asides and green-tree retention) and requirements for sustainable forestry certification (spatial and temporal distributions of coarse woody debris, a broad distribution of age classes) create heterogeneity at the stand scale [44–46]. However, uniform application of BMPs, voluntary certification guidelines, regulations, including limits to harvest unit size and adjacency requirements (time or structural characteristics required before neighboring stands can be harvested), and uniform operational prescriptions across IMF ownerships likely contribute to homogeneity at the landscape scale, regardless of region [47]. Available information is insufficient to determine how organisms respond to novel stand and landscape conditions created by the interactions of operational practices and forest practice regulations. Further, whether and/or how the presence of unmanaged forest within the same landscapes as IMF mediates organismal responses, including those of plethodontid salamanders, merits research attention [48].

Ownership patterns also create substantial variation, both across and within the PNW and SE. In the PNW, private, state and federal ownerships are distributed as both large, consolidated holdings and dispersed parcels. In addition, historic allocations from public to private ownership resulted in “checker-boarding” patterns (where every other square mile (259 ha) parcel is owned by a private or public entity) [48]. In the SE, nearly 86% of forest is privately owned and distributed broadly across parcel sizes from a few-ha family woodlot to millions of ha managed as a timberland real estate investment trust [37]. In either case, multiple ownerships, characterized by different economic and environmental objectives, can result in highly variable stand conditions and fragmented landscapes [49]. Across ownerships, IMF ecosystems are less likely to occur along a gradient of management intensity, but are in opposition to forests that no longer are subject to any form of management (except the suppression of natural disturbance agents, such as fire and insect outbreaks) [50]. Finally, the potential for rapid conversion of large private parcels to other land uses represents an underappreciated, growing challenge to both conservation of biological diversity and sustainable provisioning of commodities [6,51].

The ability of IMF ecosystems to sustain levels of biological diversity compared to unmanaged forests, whereas providing goods for human consumption, is questioned regularly [11,14,52]. Two emerging concepts, emulation of natural disturbance (END; [53]) prescriptions and land sharing *vs.* land sparing allocations [54,55], are potentially profitable means to advance current debates about IMF ecosystems. Prescriptions based on END conserve ecosystem structure and function by encompassing historical ranges of variation in ecosystem conditions across multiple spatial and temporal scales [53]. Emulation of natural disturbance is designed to increase resiliency of ecosystems against future natural and anthropogenic disturbances [56]. To do so, prescriptions aim to capture similar structural and functional diversity compared to what inevitably occurs after natural disturbances [16,17,57]. Land sharing *vs.* land sparing allocations emerged from debates about sustainable production of food commodities [54]. Land sparing separates conservation and production land (although both types can occur in the same landscape, [58]), as intensive management precludes the need to produce commodities on all of the land base. Under a land-sharing scenario, low-intensity production occurs across a larger percentage of the landscape (e.g., [59]). We note that IMF ecosystems in the PNW, as shaped by current operational practices and regulatory requirements, and by sharing regional landscapes with publicly owned, unmanaged forest ecosystems, resemble a land-sparing allocation [49]. Finally, although embedding END prescriptions within a land-sharing framework is appealing from a conservation perspective, we are unaware of any studies that examined economic consequences of this strategy for IMF ecosystems. That is, how much does net area under management increase to meet current and future demand for commodities, and is a significant increase in biological diversity achieved [60]? This broad variation in design and composition of IMF landscapes presents a challenge to understanding the effects of habitat alteration on biological diversity. An alternative and/or complementary approach to guide sustainable forest management is to select indicator species for a greater range of organisms or structural conditions [61,62]. However, as we note in the next section, this approach has both positive and negative aspects that require careful consideration.

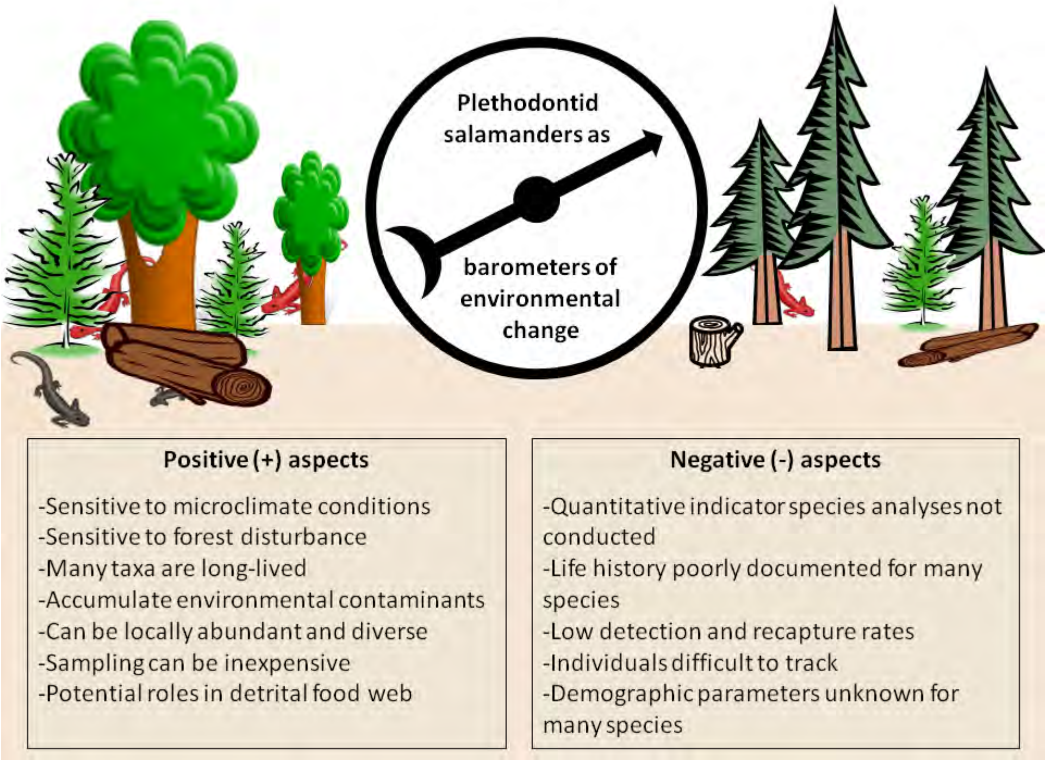
3. Salamanders as Barometers of Forest Condition

Plethodontid salamanders have numerous ecological, physiological and morphological characteristics linking them to a “slow life” in forest ecosystems. Many long-lived species do not reach reproductive maturity for several years and have low reproductive rates that limit population growth. For example, eastern red-backed salamanders (*Plethodon cinereus*), a common and well-studied species in eastern North America, may not reach reproductive maturity until 3–4 years old, and females only produce about seven eggs biennially [29,63,64]. All plethodontids lack lungs and, as adults, respire cutaneously [65]. Cool, moist microclimates facilitate gas exchange across the skin of salamanders and, because they are ectotherms, reduce metabolic costs and affect physiological processes from digestion to growth [66–68]. Increased metabolic costs resulting from warmer and potentially dryer forests following forest harvesting may be a key contributor to observed declines in salamander abundances and changes to demography [69]. However, quantification of actual energetic costs of salamanders in harvested and unharvested units has not been conducted [25]. Approximately 70% of North American salamanders lay terrestrial eggs that develop directly into adults without an aquatic larval stage, a reproductive strategy that also requires humid and cool conditions for egg survival. Additionally, stream-associated salamanders with aquatic larvae need suitable microclimatic conditions to support dispersal of juveniles post-metamorphosis and of adults into upland habitats [70,71]. Consequently, the slow lives of salamanders likely necessitate decades-long recovery periods for population declines caused by anthropogenic habitat alterations, such as forest harvesting [29,72,73].

In addition to life history requirements tied to narrow microclimate conditions in forest ecosystems, plethodontid salamanders exemplify other characteristics of potential indicator species [24]. Plethodontid salamanders are sensitive to environmental changes, including acidification [74] and heavy metals contamination [75,76], and can transfer contaminants to their young [77]. They are often locally abundant and relatively simple and inexpensive to sample, conferring statistical advantages for subsequent analyses [78], but see [79]. Further, energetic efficiency is a paramount attribute of salamanders, as plethodontids convert a greater proportion of assimilated energy into biomass than other taxa, including other herpetofauna [21]. This biomass, or stored energy, is available to numerous predators of these small, euryphagic consumers of invertebrates and contributes to energy flow within an ecosystem. As such, salamanders have been termed keystone species of the forest floor and lauded for their ability to regulate invertebrate prey, litter decomposition rates and even contribute to carbon sequestration [80,81]. However, examination of this body of work suggests substantial variability exists in response metrics among salamander, invertebrate and overstory tree (litter source) species [24,82–84]. Whereas researchers have documented top-down regulation of some invertebrate taxa and/or leaf litter decomposition, results are inconsistent across invertebrate guilds and, in some cases, experimental designs (*i.e.*, field plots, semi-natural mesocosms, laboratory microcosm) [80–82,85]. In other experiments, salamanders did not influence ecosystem functions or potential prey, further complicating any broader understanding of ecological roles of salamanders in the complex detrital food web [83].

Due to these attributes and observed declines in abundance associated with many silvicultural systems, researchers and managers have cited salamanders as a barometer of forest condition (Figure 2), with an abundant and diverse salamander community taken as evidence of a resistant, intact and healthy ecosystem [19]. However, prior to selection, indicator species should be tested rigorously to ensure consistent relationships with other taxa, disturbances or responses of interest at the appropriate spatial and temporal scales [61,86,87]. Little research exists that has compared characteristics of plethodontid salamanders to other forest-dependent species or evaluated empirically whether population status of these taxa provides an acceptable surrogate for the complex ecological processes that they are presumed to represent [61,88,89]. For example, in an experiment in Appalachian hardwood forest, the herbaceous plant community responded to a gradient of overstory removal harvests as expected, with greater compositional change with greater treatment intensity [90]. On the same experimental research sites, compositional stability of plethodontid salamanders varied through time post-harvest and across treatments, suggesting a lack of cross-taxa congruence between plants and salamanders (Homyack, J.A. [91]). Thus, without close examination of relationships between these groups, one could have interpreted erroneously that a pattern existed in salamander responses to a broader context, including herbaceous plants. Finally, despite their abundance, terrestrial salamanders often have low or variable detection and recapture rates, as well as small body sizes, factors that complicate tracking individuals and quantifying population responses accurately [79,92–95].

Figure 2. Characteristics of plethodontid salamanders make them both desirable and problematic for serving as barometers of forest change and ecosystem health in intensively-managed forest (IMF) ecosystems.



Positive (+) aspects

- Sensitive to microclimate conditions
- Sensitive to forest disturbance
- Many taxa are long-lived
- Accumulate environmental contaminants
- Can be locally abundant and diverse
- Sampling can be inexpensive
- Potential roles in detrital food web

Negative (-) aspects

- Quantitative indicator species analyses not conducted
- Life history poorly documented for many species
- Low detection and recapture rates
- Individuals difficult to track
- Demographic parameters unknown for many species

4. Case Studies of Salamanders in Managed Forests

Often, observed relationships between salamanders and forest structure from a specific research study have been expanded across species, ecosystems and forest management regimes without detailed examinations of mechanisms or meta-analyses to detect broader patterns (but see [27,28]). For example, coarse woody debris is undoubtedly a critical element of forests for numerous terrestrial salamanders [96]. Logs, stumps and root masses on the forest floor provide a means of subterranean entry, refugia from inhospitable microclimates, habitat and foraging surfaces for invertebrate prey, access to mates and habitat conditions for brooding eggs, and many species establish territories under woody debris [20]. Intensively-managed forest ecosystems may provide lower volumes, piece sizes and decay classes of CWD, as large, well-decayed debris is often redistributed or rarer after forest harvest and tends to decline after multiple rotations [28,97–99]. Also, many management guidelines recommend adequate recruitment and retention of woody debris in managed forests to maintain or enhance salamander communities [28,100]. Emerging biofuel markets that rely on woody debris or formerly non-merchantable material gleaned following harvests may reduce woody debris below thresholds required by salamanders or other wildlife [96,101]. Thus, documenting variability in existence and strength of relationships between salamanders and CWD is necessary for developing an understanding of increased intensity of forest management.

To describe potential variation in the strength and direction of the relationships between plethodontid salamanders and CWD in IMF ecosystems, we developed two case studies. First, we illustrate relationships between plethodontid salamanders and CWD in intensively-managed Douglas-fir forests of the Pacific Northwest. Second, we modeled microhabitat relationships and examined patterns of substrate use by plethodontid salamanders in mixed-hardwood forests in Virginia and West Virginia. Although the Appalachian hardwood forest is extraneous to our prior discussion of southern pine plantations, much of our current knowledge regarding plethodontid salamanders and CWD is from this region, and the example is appropriate for illustrating variability.

4.1. Douglas-Fir Regeneration and Plethodontid Salamanders in Cascades Forest

We studied Oregon slender (*Batrachoseps wrighti*) and ensatina (*Ensatina eschscholtzii*) salamanders at 66 forested harvest units (*i.e.*, stands) in the Cascade Range, Oregon, USA. The harvest unit age ranged from 35 to 90 (average = 60; SD = 8) years and from 20 to 183 ha (average = 79; SD = 33). Harvest units occurred in one of two study blocks: Clackamas (Clackamas County, OR, USA) or Snow Peak (Linn County, OR, USA). All units were regenerated from previous clearcut harvests. We selected harvest units randomly for inclusion within a long-term experimental study of salamander responses to contemporary management prescriptions.

To understand how Oregon slender and ensatina salamanders responded to variation in the amount of CWD, we estimated both occupancy and abundance with hierarchical models [102,103]. Within each harvest unit, we sampled seven 81-m^2 (9 × 9 m) plots in 2013–2014. Each 81-m^2 plot was selected randomly and sampled over three consecutive 10-min intervals in a single day (sampling occurred between 08:00 and 16:00 and from April to June). Spatial and temporal replication was necessary to estimate and incorporate detection into estimates of occupancy and abundance [104]. Observers employed a □light touch□ methodology in which all surface objects, including leaf litter and moss blankets on logs, were turned over to observe salamanders. We followed a □removal□sampling protocol in which sampling stopped once both species were observed in a plot [105]. During sampling, observers quantified the amount of CWD (all logs >25 cm DBH (small end) and >1 m in length). To estimate occupancy and abundance, we fit hierarchical models within a Bayesian framework [106]. We used a multi-scale model that allowed us to estimate occupancy at both the harvest unit and plot levels [102,107]. For abundance, we fit a model presented by Royle and Nichols [103]. We allowed occupancy and abundance to vary with block and amount of CWD in an 81-m^2 plot and detection probability to vary with sampling date.

We detected 149 and 133 Oregon slender and 64 and 83 ensatina salamanders in 2013 and 2014, respectively (60 harvest units in 2013 and 55 harvest units in 2014). Oregon slender salamanders were detected in 53/420 (34%) plots and 101/378 (27%) plots in 2013 and 2014, respectively; ensatina salamanders were detected in 144/420 (13%) plots and 73/378 (19%) plots in 2013 and 2014, respectively. Abundance of Oregon slender salamanders was strongly associated with amount of CWD (Figure 3). For Oregon slender salamanders, average occupancy was greater at Clackamas than Snow Peak in both 2013 and 2014 (Table 1). For ensatina salamanders, we did not find evidence of an association between abundance and CWD (Figure 3). Further, average occupancy was greater at Snow Peak than Clackamas in both 2013 and 2014. Harvest unit occupancy was □1 for both species.

Figure 3. Association between abundance of Oregon slender (*Batrachoseps wrighti*) (OSS) and ensatina (*Ensatina eschscholtzii*) (ENES) salamanders and downed wood, Cascade Range, OR, USA, 2013□2014. All logs >25 cm DBH (small end) and >1 m in length were included in coarse woody debris (CWD) counts. CRI = credibility interval.

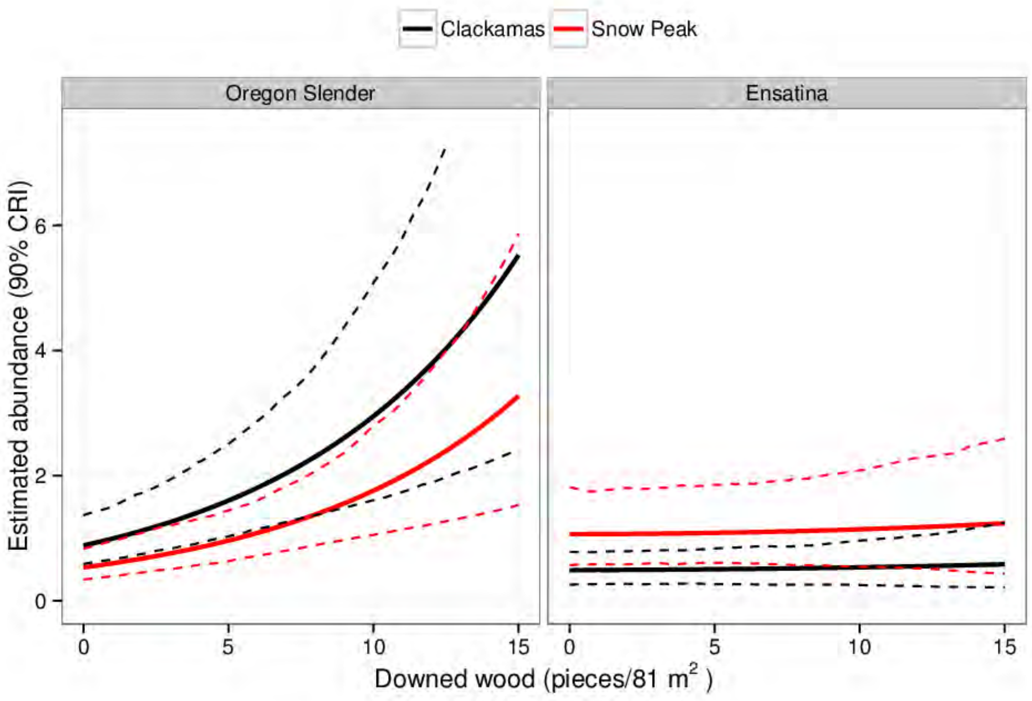


Table 1. Occupancy and abundance of Oregon slender (*Batrachoseps wrighti*) and ensatina (*Ensatina eschscholtzii*) salamanders by study block and year at average amounts of CWD (Clackamas: 2.4, SD = 2.2, range = 0□10; Snow Peak: 3.6, SD = 2.7, range = 0□15), Cascade Range, OR, USA, 2013□2014. All logs >25 cm DBH (small end) and >1 m in length were included in CWD counts.

Species	Block	Year	Occupancy Estimate	95% CI	Abundance Estimate	95% CI
Oregon slender	Snow Peak	2013	0.59	0.45, 0.75	0.91	0.60, 1.38
	Clackamas	2013	0.76	0.63, 0.90	1.5	0.99, 2.32
	Snow Peak	2014	0.45	0.33, 0.63	0.62	0.41, 0.99
	Clackamas	2014	0.63	0.48, 0.80	1.03	0.66, 1.62
Ensatina	Snow Peak	2013	0.61	0.41, 0.82	1.0	0.52, 1.7
	Clackamas	2013	0.36	0.21, 0.53	0.46	0.24, 0.76
	Snow Peak	2014	0.66	0.47, 0.86	1.15	0.64, 1.95
	Clackamas	2014	0.25	0.34, 0.56	0.53	0.28, 0.83

We found that Oregon slender and ensatina salamanders were common in the second growth forest stands we sampled. These results suggest that both species either persisted in units following harvest and/or were able to recolonize units as they regenerated over time. However, we did not find an association between CWD and ensatina responses. This result was not surprising, given the species has a broad geographic distribution and occurs in a wide range of habitat types [108]. In contrast, Oregon slender salamander responses were associated strongly with the amount of CWD. General observations suggest this species uses CWD for both foraging and rearing young [108]. Additionally, most of its geographic distribution occurs in forests of the western Cascade Range that contain substantial amounts of CWD and standing biomass [109]. However, our results suggest that within these forests, population size is positively associated with the amount of CWD. Therefore, providing adequate amounts of CWD may be required to support persistence of Oregon slender salamanders in IMF ecosystems, but we have not identified a minimum threshold. Finally, the inter-specific variation in this case study exemplifies concerns about the dogmatic application of similar prescriptions to provide suitable habitat for all plethodontid species.

4.2. Relationships of Plethodontid Salamanders with Woody Debris in Appalachian Forest

We studied plethodontid salamanders and associations with CWD in managed forests across six study sites located in mixed hardwood forest in the Jefferson National Forest, Virginia ($n = 4$), and private, industrial forest, West Virginia, USA ($n = 2$) [110]. Silvicultural prescriptions were applied randomly to seven, 2-ha treatment plots at each site during 1994□1998. In order of increasing overstory removal, prescriptions included a control, a mid-story removal with herbicide, group selection, high-leave shelterwood, leave-tree harvest, commercial clearcut and silvicultural clearcut [29]. Treatments represented a gradient of typical silvicultural options for oak regeneration.

To understand whether CWD or other micro- or macro-scale characteristics predicted relative abundance of plethodontid salamanders across spatial scales, we quantified the relative abundance of plethodontid salamanders, measured a suite of habitat characteristics, developed *a priori* candidate models representing biological or environmental hypotheses relating salamanders to forest structure and evaluated relative evidence for each model with an information-theoretic approach [110]. First, we quantified habitat characteristics and relative abundances of plethodontid salamanders on sampling grids of 2 × 15-m transects established within each treatment plot at each study site. We quantified the relative abundances of surface-active salamanders with searches of one randomly-selected transect per treatment plot/site during warm (>4 °C), rainy (leaf litter remained moist) nights, April–August, 2007. Observers crawled along transects, hand-captured salamanders and marked the point of capture with an individually-numbered pin-flag. We also recorded the substrate (e.g., log, stump, leaf litter) where salamanders were observed and identified captures to species.

After salamander sampling, we quantified habitat characteristics at the: (1) treatment plot scale; (2) scale of the entire 30-m^2 transect; or (3) a smaller, sub-transect (2 m × 5 m) scale of 10 m^2 for all sampled transects. We quantified correlates or predictors of salamander abundance, including basal area, leaf litter depth, number, volume and decomposition class of CWD [111], densities of trees and shrubs and the percent cover of leaf litter, vegetation, rocks, moss, bare soil and CWD, all of which are altered by forest harvesting. Our candidate model set contained habitat covariates that described forest structure related to large-scale habitat conditions (basal area), foraging, refugia and nest sites, retention of ground-level moisture and combinations of the three factors, in addition to a global model. We used simple linear regression and applied the candidate set of models to data collected at the 30-m^2 transect and a randomly selected 10-m^2 nested sub-transect and evaluated the balance between goodness-of-fit and statistical parsimony using AIC, model weights (w_i) and R^2 values. During 2006–2008, we qualitatively evaluated the proportion of salamander captures on CWD and compared them to other substrate types and across species and evaluated whether the plots were harvested (control and herbicide *vs.* other treatments).

In 2007, we sampled 155 transects and captured 653 salamanders of 10 species (*P. glutinosus* and *P. cylindraceus* were combined into slimy salamanders), with 0–22 salamanders/transect and 0–10 salamanders/sub-transect. Salamander counts were dominated by eastern red-backed salamanders, Allegheny mountain dusky salamanders (*Desmognathus ochrophaeus*), slimy salamanders and ravine salamanders (*P. richmondi*). Abundances of plethodontid salamanders at the scale of the 30-m^2 transect were best (ΔAIC = 0, w_i = 0.58; Table 2) described by a positive association with basal area and herbaceous cover and a negative association with litter depth. However, some evidence existed for a positive relationship between salamanders and basal area at the plot scale (ΔAIC = 1.40, w_i = 0.29). At the sub-transect scale, the model describing basal area at the plot scale had the greatest model weight (ΔAIC = 0, w_i = 0.59), but models describing overhead canopy conditions (ΔAIC = 1.98, w_i = 0.22) and overhead canopy conditions and foraging habitat (ΔAIC = 2.38, w_i = 0.18) received support. Secondly, across 2480 salamanders captured across three years, most (75%) surface-active salamanders were captured on leaf litter. The proportion of salamanders captured on leaf litter varied from 51% to 86% across species (Table 3). Salamanders found on coarse or fine woody debris (FWD) accounted for 10% of total captures and 8%–22% by species. Within a species or across all species, harvest history appeared to have relatively small effects on use of CWD or FWD, except for the small sample of Wehrle’s salamanders (*P. wehrlei*). Wehrle’s salamanders in unharvested plots primarily were captured on leaf litter (59%) or trees and saplings (26%), but those captured on harvested plots were mostly on CWD/FWD (44%) or leaf litter (39%).

Table 2. Parameter estimates and strength of evidence (AIC) from linear regression relating relative abundances of plethodontid salamanders from surface counts to forest structural characteristics across spatial scales. We present data from models with ΔAIC < 2.0 at both the 30- and 10-m^2 scale. Abundances of salamanders and habitat characteristics were quantified in mixed-oak forest on six study sites in southwestern VA and north-central WV, USA, April–August, 2007.

Model	ΔAIC	Model Weight (w_i)	R^2	β	SE	Direction
Overhead Canopy Conditions and Foraging (30-m^2 scale)	0	0.58	0.29			
Intercept				2.724	1.039	
Basal Area				0.146	0.030	+
Understory Density				0.003	0.014	+
Leaf Litter Depth				−0.410	0.147	−
Percent Herbaceous Cover				0.061	0.029	+
Large-Scale Habitat Conditions (30-m^2 scale)	1.40	0.29	0.20			
Intercept				1.435	0.536	
Basal Area				0.158	0.025	+
Large-Scale Habitat Conditions (10-m^2 scale)	0.00	0.59	0.15			
Intercept				0.421	0.246	
Basal Area				0.059	0.011	+
Overhead Canopy Conditions (10-m^2 scale)	1.98	0.22	0.15			
Intercept				0.371	0.359	
Basal Area				0.060	0.013	+
Understory Density				0.003	0.015	+

For direction, + refers to a positive association and − refers to a negative association with the variable.

Similar to recent work in Missouri hardwood forest [112], our results suggest that foraging or active salamanders likely did not solely rely on CWD during wet conditions in our study area at night. Our modeling of salamander abundance indicated that broad-scale metrics describing overstory conditions had greater predictive power than local-scale metrics describing CWD or other ground-level characteristics. Further, across □2500 captures of salamanders (dominated by eastern red-backed salamanders) during night sampling, most were observed on leaf litter rather than available CWD. Observed relationships generally were consistent across species regardless of recent (<14 years) harvesting activity, but Wehrle's salamanders may have been more reliant on CWD in harvested plots. Low sample sizes preclude strong inferences, but suggest avenues for future empirical research to understand inter-specific variability in associations with CWD or other microhabitat characteristics of suspected biological importance.

Table 3. Percentage of salamander captures by substrate type when captured during night-time area constrained searches of plots in experimentally treated Appalachian hardwood forest, VA and WV, USA, 2006–2008. No harvest treatments included untreated control plots and a mid-story herbicide release plot. Harvest treatments included a group selection, shelterwood harvest, leave tree, commercial clearcut and a silvicultural clearcut.

Category	Species	CWD/FWD [a]	Leaf Litter	Tree/Sapling	Herbaceous	Rock	Bare Ground	Other	Number of Captures
All Treatments	Total Salamanders	10.0	75.0	6.1	1.3	2.3	0.8	4.5	2480
No Harvest Treatments		7.5	77.0	6.3	1.7	2.3	0.9	4.4	1311
Harvest Treatments		12.8	72.7	5.9	0.9	2.4	0.8	4.5	1169
All Treatments	*Plethodon cinereus*	7.7	75.9	6.7	1.5	3.2	1.5	3.5	1107
No Harvest Treatments		6.3	76.9	6.3	1.9	3.7	1.8	3.0	567
Harvest Treatments		9.1	74.8	7.0	1.1	2.6	1.3	4.1	540
All Treatments	*Desmognathus* spp.	9.4	75.5	7.0	1.7	0.2	0.2	6.1	641
No Harvest Treatments		7.3	77.3	7.6	1.7	0.0	0.0	6.1	423
Harvest Treatments		13.3	72.0	6.0	1.8	0.5	0.5	6.0	218
All Treatments	*P. glutinosus complex*	16.8	69.3	3.2	0.4	4.1	0.6	5.4	463
No Harvest Treatments		13.2	69.5	3.6	0.6	4.2	1.2	7.8	167
Harvest Treatments		13.3	72.0	6.0	1.8	0.5	0.5	6.0	218
All Treatments	*P. richmondi*	5.2	86.0	4.1	1.6	0.0	0.0	3.1	193
No Harvest Treatments		5.4	89.3	0.9	2.7	0.0	0.0	1.8	112
Harvest Treatments		4.9	81.5	8.6	0.0	0.0	0.0	4.9	81
All Treatments	*P. wehrlei*	22.2	51.1	17.8	0.0	6.7	0.0	2.2	45
No Harvest Treatments		7.4	59.3	25.9	0.0	7.4	0.0	0.0	27
Harvest Treatments		44.4	38.9	5.6	0.0	5.6	0.0	5.6	18

[a] CWD/FWD refers to coarse and fine woody debris, including logs, stumps, root masses and sticks.

5. Conclusions

5.1. Beyond CWD

The direct linkages among physiology and reproduction of plethodontid salamanders and CWD and research linking the presence of salamanders to logs, stumps and root masses has led to a simplistic view of habitat requirements for this family. Efforts to retain persistent populations of plethodontid salamanders on IMF ecosystems should move beyond a default focus on CWD to integrate documented variability among salamander species, regions, topographical features and ecosystems to maximize conservation benefits for the greatest number of species. DeMaynadier and Hunter [28] summarized published relationships between plethodontid salamanders and microhabitat characteristics, and associations with rocks, litter depth, understory vegetation, canopy closure and moisture often were more frequently positively associated with salamanders than CWD. As the previous case studies and the work of DeMaynadier and Hunter [28] illustrate, a universal relationship between downed woody material and salamanders does not exist, but rather, a myriad of factors impart significant variation to relationships between plethodontids and their habitat. Similarly, other herpetofaunal species have exhibited variable relationships with experimental manipulations of woody debris [25,113–115], but manipulative research from the PNW is notably lacking.

Further, many North American plethodontid salamanders are habitat specialists that rely on features other than CWD to meet life history requirements. For example, green salamanders (*Aneides aneides*) are associated with boulders, rock crevices or tall deciduous trees; Red Hills salamanders (*Phaeognathus hubrichti*) occur only on steep hardwood ravines of a few select geologic formations; and talus slopes formed from lava flows are primary habitat for Larch Mountain salamanders (*Plethodon larselli*). As we have described, IMF ecosystems present different conservation challenges than other landscapes, and developing management recommendations without recognizing needs of habitat specialists may not minimize effects of harvesting and regeneration activities for many species, including those of immediate conservation concern. Thus, integrating a combination of filter grains, from coarse- to fine-scale approaches to conservation, will be necessary for conservation of salamanders in IMF ecosystems [116,117].

5.2. Opportunities for Future Research

Given the need for IMF ecosystems to provide ecosystem services in conjunction with sustainable production of commodities, an increase in understanding of mechanisms behind population trends of salamanders in production forests is integral for mitigating negative effects where and when they occur. We encourage additional research in IMF ecosystems to document mechanisms of population change, such as whether reduced reproduction and abundance occur from proximate (e.g., prey abundance) or ultimate (e.g., pH, moisture) factors after harvest and regeneration [28]. When developing future monitoring and research programs, practitioners and scientists could consider experimental designs and best practices that meet clearly-defined research objectives. For terrestrial salamanders in IMF ecosystems, these practices could include incorporating methods to estimate detection probabilities to reduce bias, pre- and post-treatment estimates of abundance or occupancy, long-term data collection to capture population and demographic trends over a complete turnover of these relatively long-lived species to capture variability across anticipated stand rotation lengths and designing studies to examine potential impacts of silvicultural activities at micro- to landscape scales [96,118,119]. Meta-analyses (e.g., [27]) that examine inter-specific, regional and spatio-temporal differences in population responses to manipulation of habitat characteristics in IMF ecosystems are critical to drawing strong inferences about the broader impacts of forest management on plethodontid salamanders.

Likewise, describing individual-, population- and community-level responses of salamanders to experimentally manipulated habitat features in IMF ecosystems will develop a mechanistic understanding of how forest management influences habitat quality [120]. Researchers could integrate salamander responses with quantifying changes to environmental metrics, such as soil moisture, micro- and macro-topography or CWD, as much remains to be learned. For example, altered abundance, piece sizes and decay classes of CWD occur after most types of forest harvesting or other silvicultural activities [96,121–123], but little is known about CWD and FWD dynamics across IMF ecosystems or silvicultural regimes [124,125]. Finally, many plethodontid salamanders lack basic population, demographic or natural history information, limiting the use of models to estimate population growth rates, to characterize resistance and resiliency to environmental perturbations and to evaluate rigorously their use as indicator species.

As with agriculture [126], appropriately managed intensification of forest management can meet human demands and conserve biological diversity. Creative, collaborative relationships among forest industry, state and federal agencies, universities and non-profit organizations are necessary to navigate the difficulties of securing long-term funds and access for monitoring and research of plethodontid salamanders [118]. Successful research programs will identify the types and amounts of forest structural attributes that promote persistence and re-colonization of unoccupied sites through rotations. However, these labors will only be realized when knowledge gains are translated and implemented as management activities that promote population persistence of target species while maintaining intensive forestry as a profitable and necessary enterprise in multi-ownership geographies.

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Author Contributions

Both authors contributed to the development and writing of this paper.

Conflicts of Interest

The authors declare no conflict of interest.

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The Southern Appalachian Assessment was accomplished through the cooperation of federal and state natural resource agency specialists. This page displays the logos of the agencies involved. The strong emphasis placed on working together toward a common goal is increasingly recognized as essential to effective government operation. Teamwork has strengthened our interagency understanding and communication. With the assessment as a framework for future action, government policy and management can become more consistent and better coordinated.

The assessment employs the latest technology in geographic information systems and computer communication. These tools make the information more useful to analysts and decision-makers. They should also facilitate future networking and information sharing among government agencies, educators, and the public.

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THE SOUTHERN APPALACHIAN ASSESSMENT

TERRESTRIAL

TECHNICAL REPORT

Prepared by Federal and State Agencies

Coordinated through
Southern Appalachian Man and
the Biosphere Cooperative

July 1996

REPORT
5 OF 5

Abstract

This report examines the condition of two important Southern Appalachian ecosystem elements: terrestrial plant and animal resources and forest health. Topics include broad landscape habitat and land cover patterns, federally listed threatened and endangered species, rare species and communities, popular game species, possible national forest old-growth forest, oak decline, exotic pests and diseases, disturbance, biological diversity, fragmentation, black bear habitat, genetic conservation programs, and neotropical migratory birds. The goal was to build an information base for defining resource management objectives, desired future conditions, standards, guidelines, and management directions. Results will be used in national forest plan revisions and other planning efforts.

Cover photos are by Bill Lea©, Asheville, North Carolina; report designs and layout are by Project Center, Atlanta, Georgia; maps are by agency Geographic Information Systems; other graphics and tables are by Blue Line, Inc., Roanoke, Virginia.

Preface

Our vision for the Southern Appalachian region is an environment for natural resources management that applies the best available knowledge about the land, air, water, and people of the region. Applied on public lands, this knowledge would provide a sustainable balance among biological diversity, economic uses, and cultural values. All would be achieved through information gathering and sharing, integrated assessments, and demonstration projects.

The Southern Appalachian Assessment takes a major step toward fulfillment of that vision. It is an ecological assessment – a description of conditions that goes beyond state, federal, or private boundaries. In using Southern Appalachian Assessment data, land managers can base their decisions on the natural boundaries of ecosystems rather than on the artificial boundaries of counties, states, or national forests and parks.

The assessment was accomplished through the cooperation of federal and state natural resource agencies within the Southern Appalachian region. It was coordinated through the auspices of the Southern Appalachian Man and Biosphere (SAMAB) cooperative. Members of the cooperative are: U.S. Department of Agriculture, Forest Service; Tennessee Valley Authority; U.S. Environmental Protection Agency; U.S. Department of the Interior, Geological Survey, National Park Service, National Biological Service, Fish and Wildlife Service; Appalachian Regional Commission; U.S. Army Corps of Engineers; Georgia Department of Natural Resources; North Carolina Department of Environment, Health, and Natural Resources; Tennessee Department of Environment and Conservation; U.S. Department of Commerce, Economic Development Administration; and the U.S. Department of Energy, Oak Ridge National Laboratory. This cooperation significantly expanded the scope and depth of analysis that might have been achieved by separate initiatives. It also avoided duplicating work that might have been necessary if each agency had acted independently. The findings in this assessment do not reflect unanimous (unqualified) views of all agencies involved on all points.

Although the Southern Appalachian Assessment is broad and comprehensive in subject matter and geographic scope, there are many opportunities to further expand the analyses based on this data. Urgent demands for the assessment data restricted our timeframe. So identifying data gaps became as important a task as identifying and gathering existing data. The Southern Appalachian Assessment serves as both a useful reference and as a benchmark for future analyses.

There was no specific statutory requirement for the assessment. However, national forest land and resource management plans authorized under the 1976 National Forest Management Act have been in place for almost 10 years and are therefore subject to revision. Due to the relationship of the national *forests and other federal lands to the biological, social, and economic conditions in the assessment area, more comprehensive and more scientifically credible data are needed to facilitate land management planning. This assessment supports individual forest plans by determining how the lands, resources, people, and management of the national forests interrelate within the larger context of the surrounding lands. The broadly identified pollutants and impacts of concern are not intended as a source of information upon which to base future regulatory or permitting action.*

This report is one of five that document the results of the Southern Appalachian Assessment. The reports include a summary report, atmospheric, social/ cultural/economic, terrestrial, and aquatic reports.

The five reports are available in printed form and via the Internet. By providing direct access to assessment materials via Internet, we hope that users can obtain information more quickly and at a lower cost than would have been possible otherwise. As with most reference documents, users will need only a small portion of the assessment for their specific projects at any given time. Moreover, an Internet document can be revised or updated when the occasion arises.

In-depth versions of data are available on the SAMAB, Forest Service, and Info South Home Pages on the World-Wide Web (WWW). These versions can be accessed at http://www.lib.utk.edu/samab for SAMAB's Southern Appalachian Home Page, at http://www.fs.fed.us/ for the Forest Service Home Page and at http://wwwfs.libs.uga.edu for the Info South Home Page. Additional materials such as maps and data that support the assessment are described and referenced in each report.

The Southern Appalachian Assessment is presented in five separate reports. They can be cited as follows:

Southern Appalachian Man and the Biosphere (SAMAB). 1996. The Southern Appalachian Assessment Summary Report. Report 1 of 5. Atlanta: U.S. Department of Agriculture, Forest Service, Southern Region.

Southern Appalachian Man and the Biosphere (SAMAB). 1996. The Southern Appalachian Assessment Aquatic Technical Report. Report 2 of 5. Atlanta: U.S. Department of Agriculture, Forest Service, Southern Region.

Southern Appalachian Man and the Biosphere (SAMAB). 1996. The Southern Appalachian Assessment Atmospheric Technical Report. Report 3 of 5. Atlanta: U.S. Department of Agriculture, Forest Service, Southern Region.

Southern Appalachian Man and the Biosphere (SAMAB). 1996. The Southern Appalachian Assessment Social/Cultural/Economic Technical Report. Report 4 of 5. Atlanta: U.S. Department of Agriculture, Forest Service, Southern Region.

Southern Appalachian Man and the Biosphere (SAMAB). 1996. The Southern Appalachian Assessment Terrestrial Technical Report. Report 5 of 5. Atlanta: U.S. Department of Agriculture, Forest Service, Southern Region.

Executive Summary

The Southern Appalachian ecosystem is widely recognized as one of the most diverse in a temperate region. The headwaters of nine major rivers lie within the boundaries of the Southern Appalachians, making it a source of drinking water for much of the Southeast.

The assessment area (fig. 1) includes parts of the Appalachian Mountains and Shenandoah Valley extending southward from the Potomac River to northern Georgia and the northeastern corner of Alabama. It includes seven states, 135 counties, and covers approximately 37 million acres. The Southern Appalachians are one of the world's finest remaining ecological regions. Early in the 20th century, the Appalachian landscape and natural resources were being exploited; croplands, pastures, and hillsides were eroding; and timberlands were being cut with little thought for sustaining the resources. National forests and national parks were created to preserve and restore the natural resources in the region. The seven national forests in conjunction with three national parks, the Blue Ridge Parkway, and the Appalachian Trail form the largest contiguous block of public lands east of the Mississippi River.

This comprehensive, interagency assessment began in the summer of 1994 and was completed in March 1996. It was designed to collect and analyze ecological, social, and economic data. The information provided will facilitate an ecosystem-based approach to management of the natural resources on public lands within the assessment area.

Public participation has been, and will continue to be, an important part of the assessment. One of the first actions of the assessment was to conduct a series of town hall meetings at which the public gave suggestions on the major themes and questions to be addressed. These questions, embellished by additional concerns expressed by land managers and policy makers, form the structure for the assessment.

The Terrestrial Team for the SAA examined the status and trends in forest health and in terrestrial plant and animal resources on 37.4 million acres in seven southeastern states. The information was gathered to help land managers and landowners make planning decisions.

The assessment was designed to answer eight questions, four concerned wildlife and botanical resources and four concerned forest health. Findings are summarized as responses to those eight questions.

Question 1:

Based on available information and reference material, what plant and animal species occur in the SAA area, and what are their habitat associations?

More than 20,000 species of plants and animals may occur in the Southern Appalachians. No effort was made to list all of them. Instead, a list of species was compiled that are of particular interest for various reasons. The total includes 51 federally listed, threatened and endangered (T&E) species, 366 species whose viability is of concern (VC), 38 species of high interest to natural resource managers and the public, 10 game species, and 7 other species with demanding habitat requirements. The short list includes 225 plants, 155 invertebrates, 47 birds, 23 amphibians and reptiles, and 22 mammals.

Sixteen land cover classes were defined for analysis of SAA ecosystem status and trends. These included nine forest cover types, agricultural pasture, agricultural cropland, grass/forb early successional, developed, barren, wetland, and water. There were four forest successional classes defined for each forest cover type.

Thirty-one rare community types were identified as occurring in the SAA area.

Habitat associations were determined for 442 of the 472 species on the short list. Based on habitat associations and habitat suitability models, the special list of species was placed in 19 groups. The assessment focused on these groups.

Question 2:

What are the status, trends, and spatial distributions of terrestrial habitats and wildlife and plant populations for:

Federal T&E species?

Viability Concern (VC) species?

Rare communities?

Wildlife species that are hunted, viewed, or photographed?

Species for which there is high management and public interest?

Species with special or demanding habitat needs?

Species considered to be true ecological indicators?

Of the 26 million acres of forest in the Southern Appalachians, 67 percent is in deciduous forests, 17 percent is in evergreen forests, and about 16 percent is in mixed deciduous and evergreen forests. The acreages occupied by the major forest type groups in the region are:

Forest Type Group	Million Acres
Oak	17.6
Southern yellow pine	3.8
Mixed pine-hardwood	3.2
Mixed mesophytic hardwood	3.1
White pine-hemlock-hardwood	0.8
Northern hardwoods	0.6
White pine-hemlock	0.7
Bottomland hardwood	0.4
Montane spruce-fir	0.09

The percentage distribution of forest acres among types of owners is:

Type of Owner	Percent
Private	77
National forest	17
National park	3
State	2
Other federal	1

Forest acreage has decreased by 2 percent since the mid-1970s. The loss is occurring primarily on private land, and is expected to continue at the same pace though the year 2010. Since 1980, large urban areas have grown by 35 percent and small urban areas by 53 percent. Acreage of cultivated cropland has diminished by 25 percent.

The percentage distribution of forestland by forest succession class is: early successional, 8 percent; sapling/pole, 22 percent; middle successional, 52 percent; and late successional, 18 percent.

In an initial inventory, approximately 1.1 million acres of possible old growth have been identified on national forest lands.

Rare Communities

Thirty-one rare communities were identified in the study area. Each of the communities occupies less than 1 percent of the land in the Southern Appalachians. Almost three-fourths of the rare terrestrial plant and animal species are found in at least 1 of the 31 rare communities. Some rare communities are concentrated on federal land where T&E and VC species can be nurtured under existing programs. Many are on private land where special cooperative efforts may be needed to conserve the species.

T&E and VC Species

A list of 51 federal T&E species and 366 VC species was compiled from information provided by the U.S. Fish and Wildlife Service, state natural heritage programs, and peer review of the original species list.

The highest number of occurrences for federally listed species (300) and VC species (1,929) is in the Blue Ridge Mountain section with most of these occurring in the Southern Blue Ridge Mountain subsection.

Private lands had 493 of 788 (63 percent) occurrences of federal T&E species, followed by NFS lands with 154 of the 788 (20 percent), national parks with 90 occurrences (11 percent), and other federal lands with four occurrences.

Private lands contain 1,802 out of 3,243 (56 percent) occurrences of species with viability concern, followed by NFS lands with 952 (29 percent) occurrences, national park lands with 315 (10 percent), state lands with 113 (3 percent), and other federal lands with 53 occurrences.

Game Species

Populations of white-tailed deer and wild turkeys have increased greatly in the Southern Appalachians since 1970. Populations of black bears have also increased, but the species is absent from many areas. Ruffed grouse densities are generally low to medium. National forests and national parks contain the highest densities. Populations have declined since 1970, possibly due to a decrease in acreage in the sapling/pole successional class. Bobwhite quail populations also have decreased since 1970. This species depends heavily on agricultural, grass and shrub habitats. A continued decline in the species is expected as the acreage of suitable habitat continues to decrease.

Landscape Habitat Suitability Analysis

Spruce-Fir/Northern Hardwood Habitats

Potential habitat for 23 spruce-fir/northern hardwood associated species (of which 4 species are federally listed, 18 species are VC, and 1 species is high-interest) is estimated at 184,000 acres. Forty-seven percent of these acres is located in national parks and 32 percent is located in national forests.

The outlook for this community and the 23 species associated with these high-elevation habitats is uncertain due to the negative effects caused by air pollution and exotic pests. A downward trend for these habitats is expected over the next 15 years.

High-Elevation Balds

There are an estimated 27,000 acres of high-elevation grassy balds and grass/shrub early successional habitat in the SAA area. Eighteen species were identified as being associated with these habitats. Approximately 86 percent of this habitat is located in the Blue Ridge Mountains section, 73 percent on private lands, and 25 percent on NFS lands. Approximately one-half of these early successional habitats is greater than 20 acres in size.

The outlook overall is for these habitats to remain near, or slightly above, the current levels over the next 15 years. However, the effects from air pollution on these communities could adversely affect quality of the remaining habitat. Populations of the rare species associated with this habitat will continue at low levels.

General High-Elevation Forest Habitats

Of the 350,000 acres of high-elevation, mid- and late-successional forest, 150,000 acres (42 percent) are in tracts larger than 5,000 acres and have the potential to support all seven general high-elevation forest species. Approximately 90 percent of total acres is interior forest habitat. The national parks contain 74 percent of the total habitats in 5,000 acre and larger tracts, followed by NFS lands with 17 percent.

The outlook for these forest communities and the seven species associated with these general high-elevation habitats is uncertain due to the negative effects caused by air pollution and exotic pests. A downward trend for these habitats is probable over the next 15 years.

Early Successional Habitats

There are an estimated 1.5 million acres of early successional habitat in the SAA area. Ten species were identified as associated with this habitat. Much of this habitat is located in the Southern Cumberland Plateau, Southern Ridge and Valley, and Southern Appalachian Piedmont sections. Approximately 97 percent of this habitat is located on private lands, with 2 percent located on NFS lands. About half of these early successional habitats are greater than 20 acres in size.

Riparian Habitats

A total of 2.3 million acres of riparian habitat was identified, with 1.5 million of these acres in forest riparian habitat. Forty-nine plant and animal species are associated with these seeps, springs, and streamside habitats. Approximately 80 percent of the forested riparian habitat is located on private lands.

Mid- to Late-Successional Deciduous Forest Habitats

There are an estimated 17 million acres of mid- and late-successional closed-canopy deciduous forest habitats in the SAA area. There are 66 species associated with these habitats (does not include species identified in other species groups). Over 50 percent of these habitats occurs in the Blue Ridge Mountain section. Around 71 percent of this habitat is on private lands. National forest lands contain 23 percent of these habitats

Habitats for Area Sensitive Species Associated with Mid- to Late-Successional Deciduous Forests

A total of 15.8 million acres of suitable habitat was identified for mid- to late-deciduous forest species with some area sensitivity requirement. About half of this area is in tracts greater than 5,000 acres in size. It is thought that these larger tracts have the potential to support all 16 area sensitive bird species associated with this species group. The majority of these habitats is located in the Blue Ridge Mountains, the Northern Ridge and Valley, Allegheny Mountains, and the Northern Cumberland Mountains. Approximately 51 percent of the larger tracts occurs on private lands, followed by national forest lands with 39 percent.

About 66 percent of the total acres is suitable forest interior habitat for the 10 interior bird species included in this group.

Based on past trends in land use, overall habitat acres in larger tract sizes and associated forest interior habitats will continue to decrease over the next 15 years due to loss of forestland to other land uses such as agricultural pasture and development. These decreases may continue to be most evident in the section/section groups currently with less than 70 percent of the area forested. These decreases should be seen primarily on private lands.

Black Bear Habitat

Approximately 21 million acres of potential bear habitat were identified in the SAA. Of these acres, 51 percent had total road density less than 1.6 miles per square mile. Approximately 75 percent of the total potential acres is located on private lands, followed by 19 percent of the acres on NFS lands. Around 91 percent of national forest land, 84 percent of state lands, 78 percent of national park land, and 51 percent of private lands contain suitable bear habitat. Approximately 70 percent of the Northern Cumberland Plateau, Southern Cumberland Plateau, and the Blue Ridge Mountains contains potential habitat. The forecast is for potential habitat to remain stable on public ownership, with expected decreases in the amount of potential habitat on private lands due to continued loss of forested habitats and increased development.

Question 3:

What habitat types, habitat parameters, and management activities are important for maintaining viable populations of the species on the "short list" of plants and animals?

And

Question 4:

Based on our current knowledge of ecological land unit capabilities in the Southern Appalachians, what are the conditions needed to:

Recover T&E species?

Conserve populations of VC species?

Maintain existing species and community diversity?

Provide suitable populations on national forests?

The rare communities that were identified are keys to conserving many of the region's plant and animal species. The report provides management considerations for:

- Cave communities
- Mountain bog communities
- Fen or pond communities
- High-elevation balds
- High pH or mafic balds
- Rock outcrop and cliff habitats
- Montane spruce-fir forests
- Seeps, springs, and streamside habitats
- Mountain longleaf forests

The report also provides management considerations for mid- to late-successional forest habitats, early successional habitats, and black bear habitat.

Question 5:

What changes or trends in forest vegetation are occurring in response to human-caused disturbances or natural processes?

Disturbances can be broadly grouped into those resulting from human influence and those not caused by humans. Human-caused changes, such as introduction of exotic plants and diseases, extirpation/extinction of species, or utilization of natural resources, raise particular concern because their long-term consequences often are unknown. Natural processes of disturbance that currently affect ecosystems may be similar to past processes, whereas human processes of disturbance are much different and much greater in magnitude than at any previous time.

Logging and other land uses of the past have particularly affected age-class distribution on national forests. Currently, this distribution of age consists of a large percentage of stands aged in the 60- to 90-year-old age classes. This condition may exacerbate the severity of insect and disease outbreaks in some forest types. Current rates of disturbance from timber harvesting and other forest management activities may be low when compared to estimates of pre-European settlement early successional vegetation trends and the descriptions of historic land use patterns of the late 19th and early 20th centuries.

Future vegetation is likely to be greatly affected by the direct and indirect impacts of exotic pest organisms. Some factors affecting vegetation are: (1) the amount of distribution of older-age forest stands, (2) fire suppression, (3) air pollutants, and (4) the introduction of new pests or other unforeseen causes. A principal source of human-caused disturbances in forests are silvicultural activities that are designed to manage vegetation and regenerate commercially valuable tree species.

Question 6:

What are the potential effects of the presence or absence of fire on forest health?

Fire is perhaps the most common form of major natural disturbance in most of the ecosystems of the Southern Appalachians. Fire is particularly important in systems dominated by southern yellow pines, and its ecological effects in those systems are well understood. Effects on xeric deciduous forests are also important but are less understood. Fire certainly appears to be a major factor in the development of upland oak forests. Light burning appears to increase the amount of oak regeneration beneath maturing stands of mixed hardwoods. Periodic fire probably also checks plant succession in oak forests, because later successional species, such as red maple, have low resistance to fire damage.

In the absence of fire, two rare forest communities in the Southern Appalachians–mountain longleaf pine woodlands and table mountain pine–pitch pine woodlands–are being replaced by hardwoods and loblolly pine. Judicious use of fire is needed to halt the decline of these communities.

Fire also is important for regeneration and maintenance in many other forest types and plant communities in the Southern Appalachians. Additional information is needed on the precise effects of prescribed burning in the mountains and on the risks associated with its use.

Question 7:

How is the health of the forest ecosystems being affected by native and exotic pests?

Many tree species in the Southern Appalachians are being severely affected by native and exotic pests.

Flowering dogwood is affected by dogwood anthracnose. The hemlock woolly adelgid will impact the future of Carolina and eastern hemlocks. The balsam woolly adelgid has damaged Fraser fir. Butternut canker could eliminate butternut from the area’s forests.

American chestnut and Allegheny chinquapin have almost been eliminated as tree species by chestnut blight. American elms in the area's forests are killed by Dutch elm disease, but losses in the forest are noticed less than loss of shadetrees in cities. Table mountain pine is disappearing from the Southern Appalachians primarily because fire exclusion is preventing reproduction.

Gypsy moth is one of a combination of factors contributing to oak decline. The effects of the decline complex appear to be accelerating in North Carolina and Virginia. Oak leaves are a favored food of the European gypsy moth, which is steadily advancing southward through the Appalachians. The Asiatic gypsy moth could be an even greater threat because females of that species can fly and because it has a much wider host range than the European gypsy moth. In 1995, an infestation of Asiatic gypsy moths in North Carolina was aggressively treated at great cost. Whether they were eradicated remains to be seen.

Question 8:

How are current and past management practices affecting the health and integrity of forest vegetation in the Southern Appalachians?

Reforestation, watershed improvement, erosion control, and fire protection were the primary management activities on the area's national forests in the first half of the century. Selective logging occurred until the 1960s. In efforts to reproduce desired tree species, the USDA Forest Service (FS) began to clearcut in the 1960s. The policy of extinguishing wildfires continues.

In response to public objections, the FS has severely curtailed clearcutting, and it is doing some prescribed burning. The agency's focus is now on management of ecosystems. Timber harvests on national forests peaked in 1985 and have declined rapidly since then. Current harvesting levels are comparable to those in the 1970s.

On average, national forest land in the region is at higher elevations and is less productive than private land. National forest stands are logged less frequently, so they have higher average timber inventory per acre, less removals, less growth, and slightly higher mortality rates than private land. While they encompass only 17 percent of the timberland in the Southern Appalachians, the national forests there have much larger proportions of the highest quality sawtimber. (SAMAB 1996C)

Oak decline appears to be a major threat. Its effects might be reduced with treatments to improve the vigor of individual trees. Evolving markets for low-quality trees and rising prices for high-quality oak sawtimber could provide profitable opportunities to improve the health of oak stands.

Integrated pest management is a program that could be used to reduce the impacts of pests such as the gypsy moth and the southern pine beetle.

Genetic conservation is one option for preserving species threatened by exotic pests. Species at risk include American chestnut, Allegheny chinquapin, butternut, Fraser fir, flowering dogwood, and eastern and Carolina hemlock. Hybridizing with closely related species and backcrossing could preserve genetic resources.

Introduction and Overview

Background

Early in the 20th century, the Southern Appalachian landscape reflected short sighted management practices in use at that time. Logging was done with little thought for sustaining resources for future generations. Cropland and pastures were eroding, threatening the productivity of the land. The Southern Appalachians were in big trouble, but help was on the way.

Through the leadership of such people as Theodore Roosevelt, Gifford Pinchot, and Aldo Leopold, the country began moving toward wiser use of its natural resources. National forests were established in the area to protect the headwaters of major rivers. National parks were created to preserve special places. The national forests and parks in the Southern Appalachians now make up the largest concentration of federal land in the eastern United States. During the Great Depression, the Tennessee Valley Authority was created to assist in the development and protection of the region. State officials worked closely with private landowners to restore their depleted forests and wildlife populations.

Today, the Southern Appalachians testify to the great conservation efforts of the past century. The land is once again predominately forested. There are many economic opportunities to use natural resources. Once again, the ecosystems are among the most biologically diverse in the world. Populations of deer and turkey are large and growing. People are moving to the region in greater numbers to enjoy the surroundings and to take advantage of economic opportunities. The restoration of the Southern Appalachians is a great story, but a new generation of conservationists is concerned about new threats to the region’s natural resources.

In early 1994, member agencies in the Southern Appalachian Man and the Biosphere (SAMAB) program began discussions about conducting a broad-scale assessment of the Southern Appalachian region. One factor that motivated these discussions was the recognition that ecosystem management as a principle for the planning and management of natural resources. This region is rich in natural resources, and there are many ideas about how to best manage these resources. Most national forests are approaching the time for revision of their forest plans, which guide most activities on the forests. The National Forest Management Act of 1976 requires such revisions every 10 to 15 years. SAMAB conducted a series of open meetings in the Southern Appalachians to identify major public concerns about public land management. Eight general questions related to forest health and terrestrial plant and animal resources were developed. A team of scientists and land managers was formed to address these questions.

Scope and Purpose

The Terrestrial Team of the Southern Appalachian Assessment (SAA) examined the condition of two important ecosystem elements: forest health and terrestrial plant and animal resources. The study area includes about 37.4 million acres in 7 states, 135 counties, 7 national forests, and 2 national parks (fig. 1). Assessment topics included broad landscape habitat and land cover patterns, federally listed threatened and endangered (T&E) species, rare species and communities, popular game species, possible national forest old-growth forest, oak decline, exotic pests and diseases, disturbance, biological diversity, fragmentation, black bear, genetic conservation programs, and neotropical migratory birds.

The Terrestrial Team consisted of a Plant and Animal Resources Subteam and a Forest Health Subteam. Team members included wildlife biologists, foresters, ecologists, botanists, research scientists, plant pathologists, entomologists, economists, silviculturists, public information specialists, Geographic Information System (GIS) analysts, and editor/writers. Team members represented the USDA Forest Service, U.S. Fish and Wildlife Service, National Park

Service, Oak Ridge National Laboratory, and the Tennessee Valley Authority. The team members served in a part-time capacity while performing their regular duties.

The Terrestrial Plant and Animal Resources Subteam used existing spatial and quantitative information to ascertain the current status and trends of terrestrial indigenous plant and animal resources in the Southern Appalachians. The assessment of aquatic species and habitats is included in the Aquatics Technical Report (SAMAB 1996a). Included were federally listed T&E species; other rare species; underrepresented plant and biological communities, including old-growth forests; hunted, viewed, or photographed wildlife; species with high public or management interest; and species with demanding habitat needs. These categories culminated in a "short list" that includes individual species, groups of species, and plant communities. Habitat conditions meaningful to species on the short list were described. The information provides a basis for consistent planning for terrestrial wildlife resources.

The objectives of the Forest Health Subteam were to describe present conditions and to identify changes and trends in the health of the region's forests. In its simplified approach to the assessment of forest health, the subteam addressed such elements as growth and mortality, reproductive success, and distribution of trees.

With both subteams, the assessment process was open and accessible to all governmental agencies, organizations, partners, and individuals. The long-term goal was to build an information base for defining resource management objectives, desired future conditions, standards, guidelines and management directions. Results will be used in national forest plan revisions and other planning efforts.

Terrestrial Plant and Animal Resource Questions

Four questions were assigned to the Terrestrial Plant and Animal Resources Subteam:

1. Based on available information and referenced material, what plant and animal species occur within the range of the SAA area and what are their habitat associations? (Chapter 2)
2. What are the status, trends, and spatial distributions of populations and habitats in the Southern Appalachian Assessment area for:
   - Federally listed threatened and endangered species?
   - Species with viability concern?
   - Unique or underrepresented communities?
   - Wildlife species that are hunted, viewed, or photographed?
   - Species for which there is high management or public interest?
   - Species having special or demanding habitat needs?
   - Species considered true ecological indicators?

   The answer to this question is provided in Chapter 3.
3. What habitat types, habitat parameters, and management activities are important in providing the distribution and types of habitats to sustain viable populations and/or desired habitat capability for the "short list" of wildlife and plants?

   and
4. Based on current knowledge of ecological unit land capabilities for the Southern Appalachians, what are the general habitat mixes and conditions needed to:
   - Recover federally listed threatened and endangered species?
   - Conserve populations of species with viability concern?
   - Maintain the existing species and community diversity that will not result in the loss of viability of any plant or animal species (in the context of the entire Southern Appalachian region)?
   - Provide sustainable levels of species populations at desired levels on national forests?

The answers to these questions are provided in Chapter 4.

Terrestrial Forest Health Questions

Four questions were assigned to the Terrestrial Forest Health Subteam:

5. What changes and/or trends in forest vegetation are occurring in the Southern Appalachians in response to human-caused disturbances or natural processes? (Chapter 5)
6. What are the potential effects of the presence and absence of fire on forest health? (Chapter 5)
7. How is the health of the forest ecosystem being affected by native and exotic pests? (Chapter 6)
8. How are current and past management practices affecting the health and integrity of forest vegetation in the Southern Appalachians? (Chapter 7)

Data Sources

Sources of data on the current status and past trends for broad land cover/vegetation types, communities, habitats, populations, and components of forest health were:

- LANDSAT Thematic Mapper Spectral Data
- Forest Inventory and Analysis (FIA) – Southern Reseach Station, USDA Forest Service
- Southern Forest Health Atlas – Southern Region, USDA Forest Service
- Continuous Inventory of Stand Conditions (CISC) – Southern Region, USDA Forest Service
- Species Element Occurrence (EOR) data – state natural heritage programs
- Forest Health Monitoring Program – Southern Research Station, USDA Forest Service
- County density estimates for game species – state fish and wildlife agencies
- 1:250,000 Digital Line Graph (DLG) ownership coverage – U.S. Geological Survey
- 1:100,000 DLG water/stream reaches – Environmental Protection Agency
- 1:100,000 DLG road coverage – U.S. Geological Survey
- 1:100,000 Digital Elevation Model (DEM) – Department of Defense Mapping Agency
- Natural Resources Inventory (NRI) – Natural Resources Conservation Service
- Ecological Mapping Units – Southern Region, USDA Forest Service
- National Interagency Fire Management Integrated Data Base (NIFMED) – participating state and federal agencies

These data sources are described briefly in appendix A. Data analysis and interpretation relied heavily upon a (GIS) for data storage, retrieval, analysis, and display. Scientists and experts reviewed selected analyses and narratives throughout the assessment.

Wildlife and Plant Species and Important Habitats

Question 1:

Based on available information and referenced material, what plant and animal species occur within the range of the Southern Appalachian Assessment (SAA) area, and what are their habitat associations?

The Southern Appalachian area contains an estimated 80 species of amphibians and reptiles, 175 species of land birds, 65 species of mammals, 2,250 species of vascular plants, and possibly as many as 25,000 species of invertebrates (Boone and Aplet 1994, USDA FS 1993b, Hamel 1992). It was not possible to identify and develop habitat relationships for all of these individual species. As an alternative, the team used an approach that has been likened to coarse and fine filtration (The Nature Conservancy 1982, Noss 1987). Hunter (1990) describes this approach.

The coarse filter component looks at broad habitat types in various stages of succession, rare communities, and ecological units. The purpose is to identify the full range of habitat types across the region. The underlying theory is that most plant and animal species in a region can be maintained by providing an appropriate mixture of habitats. Coarse filtration avoids the need to fully examine every species–a nearly impossible assignment. Two problems with the coarse filter component are that some species requirements may not be adequately addressed, and species of particular interest to the public may be omitted. A fine filter component was used to identify individual species and special habitat parameters. The coarse-fine filter approach resulted in a list of special individual plant and animal species; a list of broad vegetation classes; a list of ecological section and subsection units; and a list of rare communities. A detailed species/habitat matrix was developed to relate the special individual species to various habitat elements. With this information, the individual species were then organized into groups based on habitat associations to simplify the assessment.

Broad Vegetation Classes

To help describe the structure of SAA ecosystems, 16 broad land cover classes were identified to characterize “macro” habitats across the SAA area (table 2.1, column 2).These included nine forest classes and seven non-forest classes. Brief descriptions of these 16 classes are included in appendix C. Each forest class was subdivided into four successional stages because individual species often require a particular successional stage of a habitat. The four successional stages recognized in the analysis were: (1) grass, shrubs, and seedlings; (2) saplings and poletimber; (3) mid-succession; and (4) late-succession, including old forests. Criteria for placement into successional stages are shown in table 2.2. An analysis for possible old growth on National Forest System (NFS) lands was performed using classes of old-growth forests based on classes developed by Nowacki (1993). Table 2.3 shows the relationship of these old-growth classes (column 3) to broad vegetation classes (column 1) and USDA Forest Service (FS), Region 8, forest cover types (column 2).

Rare Community Types

In cooperation with The Nature Conservancy, 31 rare ecological groups (rare communities) were identified in the SAA area. Abbreviated descriptions are included in appendix C. A detailed description of each is included in the SAA process file. These 31 communities are broad “umbrella” descriptions of groups of communities and do not replace finer scale community units described by state natural heritage programs or those developed as part of The Nature Conservancy national classification. They do not detail all the

able 2.1 The relationship of the Southern Appalachian Assessment (SAA) area remote sensing imagery vegetation classes with the habitat groups in the SAA plant nd animal species/habitat matrix and Southern National Forest forest–type codes.

errestrial Habitat Groups Used in ant and Animal Species/Habitat Matrix	Classes Mapped in SAA Remote Sensing Imagery	Classes[1]	Code	Southern Region NFS Forest Types Name
orthern Hardwood Forests	Northern Hardwood Forests	(1) (6)	81	Sugar Maple–Beech–Yellow Birch
ixed Mesophytic Hardwood Forests	Mixed Mesophytic Hardwood Forests	(1) (6)	50	Yellow Poplar
			56	Yellow Poplar–White Oak–Red Oak
eric Oak Forests	Oak Forests	(1) (6)	51	Post Oak–Black Oak
			52	Chestnut Oak
			57	Scrub Oaks
			59	Scarlet Oak
			60	Chestnut Oak–Scarlet Oak
esic Oak Forest	Oak Forests	(1) (6)	53	White Oak-Red Oak-Hickory
			54	White Oak
			55	Northern Red Oak–Hickory
ottomland Hardwood Forests	Bottomland Hardwood Forests	(1) (6)	58	Sweetgum–Yellow Poplar
			71	Black Ash–American Elm-Red Maple
			72	River Birch-Sycamore
			73	Cottonwood
			76	Silver Maple-American Elm
ontane Spruce–Fir Forests	Montane Spruce–Fir Forests	(1) (7)	6	Fraser Fir
			7	Red Spruce-Fraser Fir
	White Pine–Hemlock–Hardwood Forest	(1) (8)	17	Red Spruce-Northern Hardwoods
uthern Yellow Pine Forests	Southern Yellow Pine Forests	(1) (7)	31	Loblolly Pine
			32	Shortleaf Pine
			33	Virginia Pine
			38	Pitch Pine
			39	Table Mountain Pine
ountain Longleaf Pine Forests	Southern Yellow Pine Forest	(1) (7)	21	Longleaf Pine
	Mixed Pine–Hardwood Forests	(1) (8	26	Longleaf Pine
hite Pine–Hemlock–Hardwood Forests	White Pine–Hemlock Forests	(1) (7)	3	White Pine
			4	White Pine–Hemlock
			5	Hemlock
	White Pine–Hemlock-Hardwood Forests	(1) (8)	8	Hemlock–Hardwood
			9	White Pine–Cove Hardwoods
			10	White Pine–Upland Hardwoods
			41	Cove Hardwoods–White Pine-Hemlock
			42	Upland Hardwoods–White Pine

able 2.1 (cont.) The relationship of the Southern Appalachian Assessment (SAA) area remote sensing imagery vegetation classes with the habitat groups in the SAA lant and animal species/habitat matrix and Southern National Forest forest–type codes.

errestrial Habitat Groups Used in ant and Animal Species/Habitat Matrix	Classes Mapped in SAA Remote Sensing Imagery	Classes	Southern Region NFS Forest Types Code	Name
eric Mixed Pine–Hardwood Forests	Mixed Pine-Hardwood Forests	(1) (8)	12	Shortleaf Pine–Oaks
			15	Pitch Pine–Oaks
			16	Virginia Pine–Oaks
			20	Table Mountain Pine–Hardwoods
			44	Southern Red Oak–Yellow Pine
			45	Chestnut Oak–Scarlet Oak–Yellow Pine
			49	Bear Oak–Southern Scrub Oaks–Yellow Pine
lesic Mixed Pine–Hardwood Forests	Mixed Pine-Hardwood Forests	(1) (8)	13	Loblolly Pine–Hardwoods
			46	Bottomland Hardwoods–Yellow Pine
			47	White Oak–Black Oak–Yellow Pine
			48	Northern Red Oak–Hickory–Yellow Pine
edar Woodlands	Mixed Pine-Hardwood Forests	(1) (8)	11	Eastern Redcedar–Hardwoods
	Southern Yellow Pine Forest	(1) (7)	35	Eastern Redcedar
	Mixed Pine–Hardwood Forests	(1) (8)	43	Oaks-Eastern Redcedar
ld Fields	Grass, Forbs, Cedar Woodlands, Early Successional			
igh Elevation Grasslands	with less than 25% woody vegetation.	(2)		
igh Elevation Shrubs				
arly Successional with <25% woody vegetation				
ock Outcrops	Rock Outcrops, Bare Soil	(3)		
allus Slopes				
aves				
oray Cliffs				
are Soil, Sand, Gravel				
rchards, Croplands	Croplands			
astures	Pastures			
nergent Wetlands	Wetland	(4)		
lountain Bogs				
oring Heads and Springs				
0% or more Synthetic Surface	Developed	(5)		
akes, Rivers, Ponds	Water			

emote Sensing Imagery Classes:
= Greater than 25% is woody vegetation.
= Consists of grasses, forbs, woody vegetation with less than 25% woody vegetation
= Greater than 25% is non-vegetation (bare soil, rock, sand).
= Land where the vegetation is less than 60% AND water is present.
5 = Where vegetation is less than 25%, and more than 50% is synthetic surface.
6 = Of the vegetation present, 70% or more is deciduous.
7 = Of the vegetation present, 70% or more is evergreen
8 = Of the vegetation present, 31 to 69% or is a mix of deciduous and evergreen.

Table 2.2 The ages for successional classes of the 9 forest classes in the Southern Appalachian Assessment area.

Forest Classes Mapped in Remote Sensing Imagery	Grass/ Shrub/ Seedling	Sapling /Pole	Mid Successional	Late Successional including Old Forests	Old Forest are Believed to be About
Northern Hardwood Forests	0-10 years	11-40 years	41-90 years	91+ years	180 years
Mixed Mesophytic Hardwood Forests	0-10 years	11-40 years	41-80 years	81+ years	130 years
Oak Forests	0-10 years	11-40 years	41-80 years	81+ years	130 years
Bottomland Hardwood Forests	0-10 years	11-20 years	21-60 years	61+ years	100 years
White Pine–Hemlock Forests	0-10 years	11-40 years	41-90 years	91+ years	180 years
Montane Spruce–Fir Forests	0-10 years	11-40 years	41-80 years	81+ years	130 years
Southern Yellow Pine forests	0-10 years	11-20 years	21-60 years	61+ years	100 years
White Pine–Hemlock–Hardwood Forests	0-10 years	11-30 years	31-90 years	91+ years	180 years
Mixed Pine–Hardwood Forests	0-10 years	11-40 years	41-80 years	81+ years	130 years

(Source: Developed by SAA TPAR team, in coordination with Southern Station Forest Inventory and Analysis (FIA) Unit)

variation in the relatively broad groups; but, where possible, this variation is addressed in the description. The 31 communities are: beaver pond and wetland complexes; beech gap forests; boulder fields (forested); calcareous cliffs; calcareous woodlands and glades; Carolina hemlock forest; caves; granitic domes; granitic flatrocks; grassy balds; heath balds; high-elevation rocky summits; mafic and calcareous fens; mafic cliffs; mafic woodlands and glades; mountain lakes; mountain longleaf pine woodlands; mountain ponds; river gravel cobble bars; sandstone cliffs; seasonally dry sinkhole ponds; serpentine woodlands and glades; shale barrens; sinkholes and karstlands; sphagnum and shrub bogs; spray cliffs; spruce-fir forests; swamp forest-bog complexes; Table Mountain pine-pitch pine woodlands; talus slopes (nonforested); and wet prairies.

The Selection of Wildlife and Plant Species for Emphasis in the SAA

List of Special Species

The fine filtration resulted in a list of special species. To be included, a species had to meet one of six criteria:

1. Federally proposed threatened and endangered species (T&E)
2. Other species with viability concerns (VC), including federal candidate species in

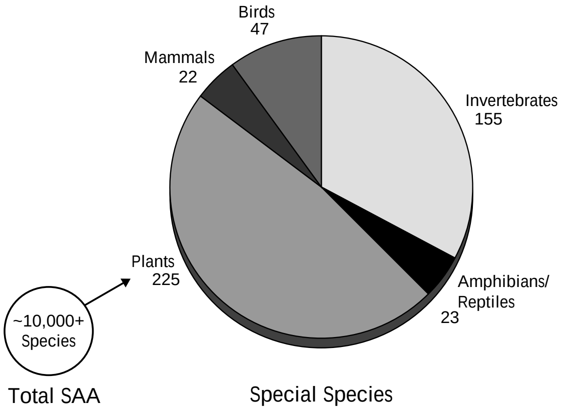

Figure 2.1 A taxonomic summary of the terrestrial plant and animal short list for the Southern Appalachian Assessment (SAA)

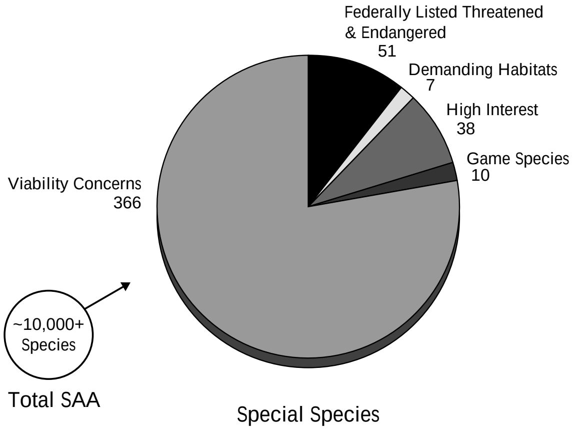

Figure 2.2 The number of terrestrial plant and animal species from the short list for the Southern Appalachian Assessment (SAA)

able 2.3 The relationship of the Southern Appalachian Assessment area remote sensing imagery vegetation classes with the old growth forest type groups and the outhern National Forest forest–type codes.

rest Classes Mapped in emote Sensing Imagery	NFS Southern Region's Forest Types		Southern Region's Old Growth Forest Type Groups [1]	
	Code	Name	Code	Name
orthern Hardwood Forests	81	Sugar Maple-Beech-Yellow Birch	1	Northern Hardwood Forests
ixed Mesophytic Hardwood	50	Yellow Poplar	5a	Mixed Mesophytic forests
	56	Yellow Poplar-White Oak-Red Oak	5a	Mixed Mesophytic forests
ak Forests	Xeric:			
	51	Post Oak–Black Oak	22b	Dry and Xeric Oak Forests
	52	Chestnut Oak	21, 22b	Dry-Mesic/Dry & Xeric Oak Forests
	57	Scrub Oaks	22b, 22c	Dry and Xeric Oak Forest
	59	Scarlet Oak	21, 22b	Dry-Mesic Oak/ Dry & Xeric Oak Forests
	60	Chestnut Oak-Scarlet Oak	21,22b	Dry-Mesic Oak/ Dry & Xeric Oak Forests
	Mesic:			
	53	White Oak-Red Oak-Hickory	21	Dry-Mesic Oak Forests
	54	White Oak	21	Dry-Mesic Oak Forests
	55	Northern Red Oak-Hickory	21	Dry-Mesic Oak Forests
	N/A	Mixed Oaks that we should code as 53	21	Dry-Mesic Oak Forests
ottomland Hardwood Forests	58	Sweetgum-Yellow Poplar	13	River Floodplain Hardwood Forests
	71	Black Ash-American Elm-Red Maple	10	Hardwood (Elm-Ash-Maple) Wetland Forests
	72	River Birch-Sycamore	28	Eastern Riverfront Forests
	73	Cottonwood	28	Eastern Riverfront Forests
	76	Silver Maple-American Elm	28	Eastern Riverfront Forests
hite Pine-Hemlock Forests	3	Eastern White Pine	2b	Conifer-Northern Hardwoods
	4	Eastern White Pine-Hemlock	2a, 2b	Conifer-Northern Hardwoods
	5	Eastern Hemlock	2a, 5a	Conifer-Northern Hardwoods/Mixed Mesophytic Forests
ontane Spruce-Fir Forests	6	Fraser Fir	31	Montane and Allied Spruce-Fir Forests
	7	Red Spruce-Fraser Fir	31	Montane and Allied Spruce-Fir Forests
uthern Yellow Pine Forests	21	Longleaf Pine	26	Upland Longleaf Pine Forests
	31	Loblolly Pine	25	Dry and Dry-Mesic Oak-Pine Forests
	32	Shortleaf Pine	24, 25	Xeric Pine & Pine-Oak/Dry & Dry-Mesic Oak-Pine Forests
	33	Virginia Pine	24, 25	Xeric Pine & Pine-Oak/Dry & Dry-Mesic Oak-Pine Forests
	38	Pitch Pine	24	Xeric Pine and Pine-Oak Forests
	39	Table Mountain Pine	24	Xeric Pine and Pine-Oak Forests
hite Pine-Hemlock-Hardwood forests	8	Hemlock-Hardwood	2a	Conifer-Northern Hardwoods
	9	White Pine-Cove Hardwoods	5a	Mixed Mesophytic Forests
	10	White Pine-Upland Hardwoods	24, 25	Xeric Pine & Pine-Oak/Dry & Dry-Mesic Oak-Pine Forests
	17	Red Spruce-Northern Hardwoods	2c	Conifer-Northern Hardwoods
	41	Cove Hardwoods-White Pine-Hemlock	5a	Mixed Mesophytic Forests
ixed Pine-hardwood Forests	Xeric:			
	11	Eastern Redcedar-Hardwoods	37	Rocky, thin-soiled, excessively drained cedar woodlands
	12	Shortleaf Pine-Oaks	24	Xeric Pine and Pine-Oak Forests
	15	Pitch Pine-Oak	24, 25	Xeric Pine & Pine-Oak/Dry & Dry-Mesic Oak-Pine Forests
	16	Virginia Pine-Oak	24, 25	Xeric Pine & Pine-Oak/Dry & Dry-Mesic Oak-Pine Forests
	20	Table Mountain Pine-Hardwoods	24, 25	Xeric Pine & Pine-Oak/Dry & Dry-Mesic Oak-Pine Forests
	26	Longleaf Pine-Hardwoods	24, 25	Xeric Pine & Pine-Oak/Dry & Dry-Mesic Oak-Pine Forests
	43	Oaks-Eastern Redcedar	37	Rocky, thin-soiled, excessively drained cedar woodlands
	44	Southern Red Oak-Yellow Pine	24, 25	Xeric Pine & Pine-Oak/Dry & Dry-Mesic Oak-Pine Forests
	45	Chestnut Oak-Scarlet Oak-Yellow Pine	24, 25	Xeric Pine & Pine-Oak/Dry & Dry-Mesic Oak-Pine Forests
	49	Bear Oak-Southern Scrub Oaks-Yellow Pine	22b, 22c, 24	Dry and Xeric Oak/Xeric Pine & Pine-Oak Forests
	Mesic:			
	13	Loblolly Pine-Hardwoods	25	Dry and Dry-Mesic Oak-Pine Forests
	42	Upland Hardwoods-White Pine	24, 25	Xeric Pine & Pine-Oak/Dry & Dry-Mesic Oak-Pine Forests
	46	Bottomland Hardwoods-Yellow Pine	13	River Floodplain Hardwood Forests
	47	White Oak-Black Oak-Yellow Pine	25	Dry and Dry-Mesic Oak-Pine Forests
	48	Northern Red Oak-Hickory-Yellow Pine	24, 25	Xeric Pine & Pine-Oak/Dry & Dry-Mesic Oak-Pine Forests

Group codes and names are the same as used by the Nature Conservancy categories 1 or 2 and species with a Nature Conservancy global rank of 1, 2, or 3

3. Game species
4. Species with high management or public interest
5. Species with demanding habitat parameters
6. Keystone species

The list of special species includes species of terrestrial plants and animals (table B-1). Among these are 225 plants, 47 birds, 22 mammals, 21 amphibians, 2 reptiles, and 155 invertebrates (fig. 2.1). Federal T&E species and species with viability concerns account for 88 percent of the species identified (fig. 2.2).

A matrix was created to develop habitat associations and relationships for the special species. The SAA species/habitat matrix was developed on spreadsheet software and is available in the SAA CD-Rom.

Species/Habitat Matrix

The matrix used 12 forest habitats and 31 rare communities. Table 2.1 lists some of them in the left column. The center column indicates how each habitat group is identified in remote sensing imagery. The right column lists the FS forest types in each habitat group. The forest habitats were subdivided into six successional classes for the matrix.

To define conditions adequately for certain species, 15 special habitat characteristics were recognized:

- Remoteness (for species sensitive to human disturbance)
- Tract size (for species needing large, contiguous tracts)
- Open canopy
- Closed canopy
- Forest interior
- Riparian
- Springs, heads, and seeps (small wet habitats)
- Water (flowing, standing, or both)
- Large trees (18+ inches in diameter at breast height)
- Trees and snags for cavity nesters
- Large snags
- Downed trees
- Leaf litter
- Elevation class:
  - Greater than 4,500 feet
  - Greater than 3,500 feet
  - Greater than 2,500 feet
  - Less than 4,500 feet
  - Less than 3,500 feet
  - Less than 2,500 feet
- Aspect (north or south)

Species Groups Based on Habitat Association

To simplify the assessment process the 472 selected species were assigned to groups based on habitat associations using information in the species/habitat matrix. Thirty species could not be associated with some type of habitat parameter due to lack of information on habitat relationships. Nineteen species groups were defined:

- Cave Species: 122 species associated with cave habitats.
- Mountain Bog Species: Eighteen species associated with swamp forest-bog complexes and/or sphagnum and shrub bog rare communities.
- Spray Cliff species: Nineteen species associated with spray cliffs.
- Fen or Pond Wetland Species: Six species associated with nonforested habitat and primarily with mafic and calcareous fens, wet prairies, seasonally dry sinkhole ponds, mountain ponds, mountain lakes, and beaver pond and wetland complexes.
- High-Elevation Bald and Rocky Summit Species: Twenty species associated primarily with grassy balds, heath balds, and high-elevation rocky summits. This species group is associated with high-elevation, early successional habitats.
- High pH or Mafic Species: Thirty-six species associated with the cedar woodlands calcareous woodlands and glades, calcareous cliffs, sinkholes and karstlands, and mafic woodlands and glades.
- Rock Outcrop and Cliff Species: Thirty-six species associated with shale barrens, granitic domes, mafic cliffs, boulder fields, talus slopes, and granitic flatrocks.
- Early Successional Grass/Shrub Species: Ten

species associated with the grass/shrub/seedling successional class. Other associated habitat groups include old fields, improved pastures, and agricultural crops.

- Wide-Ranging Area-Sensitive Species: The red wolf (*Canis rufus*), eastern cougar (*Felis concolor cougar*) and black bear (*Ursus americanus*). The red wolf occurs in the Southern Appalachians only in re-introduced populations. The eastern cougar is probably extirpated since there have been no confirmed sightings for several years. These species are associated with most forest types and successional stages.
- Mid- to Late-Successional Deciduous Forest Species: Seven species associated with deciduous forest habitats in the following successional classes: Sapling pole, mid-successional, and late-successional. The group includes two salamanders, one plant, two squirrels, and two birds.
- Seep, Spring, and Streamside Species: Fifty-one species associated with the same five rare communities found in the fen or pond wetland species group but generally associated with forested habitat and/or spring heads, seeps, some type of flowing water, or other riparian habitat. This group also includes species associated with the river gravel cobble bars.
- Habitat Generalist Species: Seven species associated with a variety of forest habitat groups and successional stages and not closely associated with a particular rare community. Three game species (ruffed grouse, turkey, and deer), three birds, and a plant comprise this group.
- Area-Sensitive Mid- to Late-Successional Deciduous Forest Species: Sixteen bird species comprise this group. All are area-sensitive, and many are forest interior species. They are associated with sapling pole, mid-successional, and late-successional deciduous forest habitats.
- General High-Elevation Forest Species: Seven species at elevations greater than 3,500 feet and not associated with a particular forest type or rare community.
- High-Elevation Spruce-Fir Forest Species: Twenty-three species associated with montane spruce-fir forests and the spruce-fir forests rare community. Two species, a fern and a moth, are associated with northern hardwood forests and with the spruce-fir forests rare community. All species are at elevations greater than 3,500 feet, and many require elevations greater than 4,500 feet.
- Bottomland Forest Species: Two species found primarily in bottomland hardwood forests: the Virginia cup-plant (*Silphium connatum*) and the prothonotory warbler (*Protonotaria citrea*).
- Southern Yellow Pine Forest Species: Two bird species, the red cockaded woodpecker (*Picoides borealis*) and the brown-headed nuthatch (*Sitta pusilla*), dependent on southern yellow pine forests, especially longleaf pine (*Pinus palustris*).
- Mixed Mesic Species: Forty-six plant and invertebrate species primarily associated with mesic mixed pine/hardwood forests, mesic oak forests, northern hardwood forests, or mixed mesophytic hardwood forests.
- Mixed Xeric Species: Twelve species associated with xeric oak, xeric mixed pine-hardwood, and southern yellow pine forest habitat groups. Ten are plants, one is an invertebrate, and one is a reptile, the northern pine snake (*Pituophis m. melanoleveus*).

Ecological Mapping Units of the SAA

The National Hierarchical Framework of Ecological Units is a classification and mapping system developed to provide a scientific basis for ecosystem management at multiple geographic scales (USDA FS 1993a). The framework was designed to assist scientists and managers in addressing scale-related resource planning and management questions and to evaluate potential uses for land and water resources. Lands within the SAA area have been classified to five levels from domain to subsection of the National Hierarchical Framework of Ecological Units. The ecological units are representations of an association of biological and environmental factors that directly affect or indirectly express energy, moisture and nutrient gradients which regulate the structure and function of ecosystems. Ecological units at all levels are defined by a combination of physical and biological components including climate, geology, soils, geomorphology, hydrology and vegetation. One domain, 2 divisions, 3 provinces, 10 sections, and 29 subsections are in the SAA area (fig. 2.3). A brief description of these ecological units is given in appendix D. A more detailed description of each unit is available in the SAA process file. The ecological units for the SAA are:

- Humid Temperate Domain (200)
  - Hot Continental Division (220)
    - Eastern Broadleaf Forest Province (221)
      - Northern Cumberland Plateau Section (221H)
        - Southwestern Escarpment Subsection (221Hc)
        - Sesquatchie Valley Northern Subsection (221Hd)
      - Southern Cumberland Mountains Section (221I)
        - Pine Mountain Thrust Block Subsection (221Ia)
        - Cleveland Subsection (221Ib)
      - Central Ridge and Valley Section (221J)
        - Rolling Limestone Hills Subsection (221Ja)
        - Sandstone Hills Subsection (221Jb)
        - Holston Valley Subsection (221Jc)
    - Central Appalachian Broadleaf Forest–Coniferous Forest–Meadows Province (M221)
      - Northern Ridge and Valley Section (M221A)
        - Appalachian Ridges Subsection (M221Aa)
        - Great Valley of Virginia Subsection (M221Ab)
      - Allegheny Mountains Section (M221B)
      - Northern Cumberland Mountains Section (M221C)
        - Central Coalfields Subsection (M221Ca)
      - Blue Ridge Mountains Section (M221D)
        - Northern Blue Ridge Mountains Subsection (M221Da)
        - Central Blue Ridge Mountains Subsection (M221Db)
        - Southern Blue Ridge Mountains Subsection (M221Dc)
        - Metasedimentary Mountains Subsection (M221Dd)
  - Subtropical Division (230)
    - Southeastern Mixed Forest Province (231)
      - Southern Appalachian Piedmont Section (231A)
        - Midland Plateau Central Uplands Subsection (231Aa)
        - Piedmont Ridge Subsection (231Ab)
        - Schist Plains Subsection (231Ac)
        - Lower Foothills Subsection (231Ad)
        - Schist Hills Subsection (231 Ag)
        - Lynchburg Belt Subsection (231Ak)

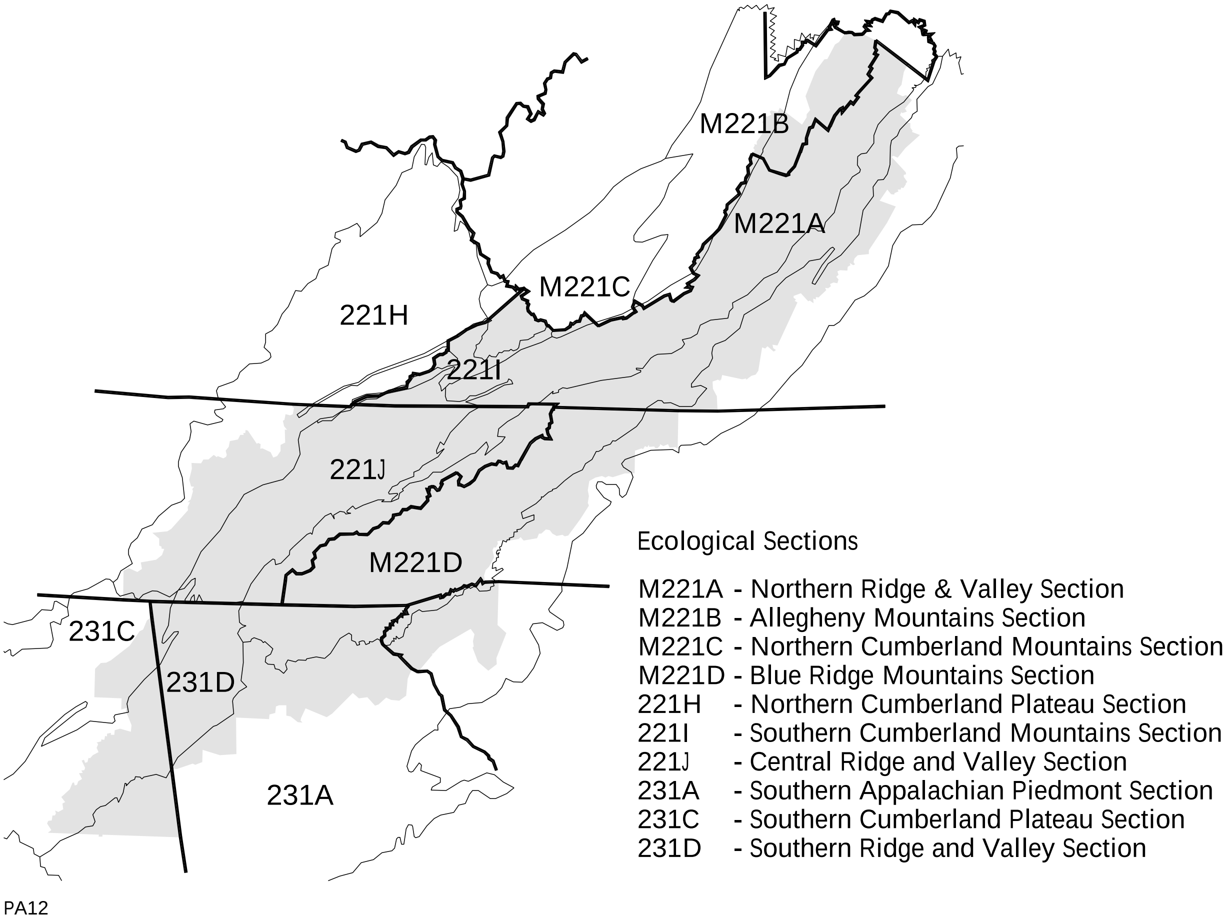

Figure 2.3 National hierarchical framework of ecological units from domain to subsection for the Southern Appalachian Assessment Area.

- Northern Piedmont Subsection (231Al)
- Triassic Basins Subsection (231Ap)

Southern Cumberland Plateau Section (231C)

- Table Plateau Subsection (231Cc)
- Southern Cumberland Valleys Subsection (231Cf)

Southern Ridge and Valley Section (231D)

- Chert Valley Subsection (231Da)
- Sandstone, Shale and Chert Ridge Subsection (231Db)
- Sandstone Ridge Subsection (231Dc)
- Quartzite and Talledega Slate Ridge Subsection (231Dd)
- Shaley Limestone Valley Subsection (231De)

Status, Trends, and Spatial Distribution of Terrestrial Habitats and Wildlife and Plant Populations

Question 2:

What are the status, trends, and spatial distribution of populations and habitats in the Southern Appalachians for federal threatened and endangered (T&E) species; species with viability concerns; unique under represented communities (including areas that have potential for old-growth); wildlife species that are hunted, viewed, or photographed; species for which there is high management or public interest; species having special or demanding habitat needs; and species considered to be true ecological indicators?

This chapter attempts to quantify current conditions, past trends, and possible future trends for the resource elements identified using the course-fine filter approach discussed in Chapter 2.

The first two sections (Forest and Nonforest Ecosystems, and Rare Communities) describe the broad forest and nonforest ecosystems identified during the course filter approach. Detailed geographical information system (GIS) analyses reveal the current status and past trends for the identified habitats for the total Southern Appalachian Assessment (SAA) area, by ownerships, and ecological units. Also provided are the locations of 31 rare communities and the initial inventory for possible old-growth forests on national forests. While results of this analysis were important on their own, they also served as intermediate data for landscape habitat suitability analysis. (This landscape analysis is described in the sixth section in this chapter, Landscape-level Habitat Suitability Analysis for Selected Species Groupings.)

The remaining sections provide results of analysis for the special list of wildlife and plant species identified during the fine filter screening approach. These sections include analysis of populations and habitats for the 19 species groupings. Again, the current status, past trends, and possible future trends are discussed where information was available.

The third section (Federally Listed Threatened and Endangered Terrestrial Species) shows the results of the analysis conducted for 51 federally listed terrestrial species. The fourth section (Terrestrial Species with Viability Concern) provides the findings for the viability concern (VC) species. The fifth section (Major Game Species) reports the results for the current status and 25-year trends for 10 of the major game species identified on the special species list. The sixth section (Landscape-level Habitat Suitability Analysis for Selected Species Groupings) provides an analysis of landscape-level habitat suitability for 10 groups of species associated with broad habitat types.

Forest and Nonforest Ecosystems

The status and trends for forest and nonforest ecosystems are described. Forest habitat types were first grouped into three broad tree categories: (1) deciduous, (2) evergreen, and (3) mixed evergreen-deciduous. The deciduous group was further stratified into: (1) northern hardwood, (2) mixed mesophytic hardwood, (3) oak, and (4) bottomland hardwood. The evergreen category was stratified into: (5) white pine-hemlock, (6) montane spruce-fir, and (7) southern yellow pine. The mixed evergreen-deciduous was stratified into: (8) white pine-hemlock-hardwood and (9) mixed (yellow) pine-hardwood. In addition, forest habitats were described using the successional stages: (1) grass-seedling-forbs, (2) sapling-pole timber, (3) mid-successional seral stage, and

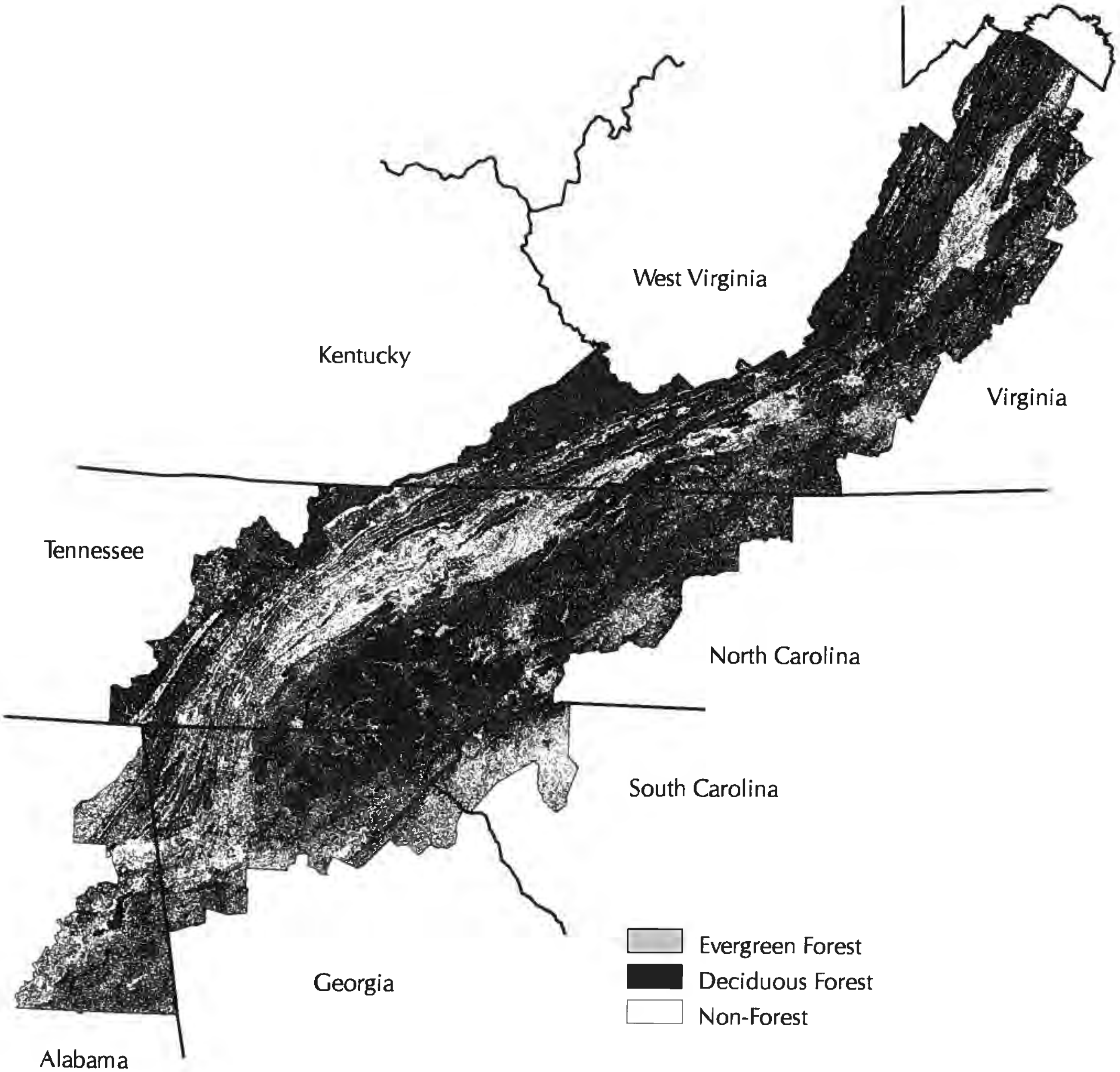

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Figure 3.1 The spatial distribution of forest and nonforest land cover in the SAA area as determined by LANDSAT remote sensing imagery.

(4) late-successional seral stage. These stages are shown in table 2.2 as the range of ages for the overstory.

Nonforest ecosystems for terrestrial plant and animal species are combined into seven categories: (1) grass, forbs, cedar woodland, early successional areas with less than 25 percent woody vegetation; (2) cropland; (3) pastures; (4) wetland; (5) human developed areas; (6) water; and (7) rock outcrops and bare soil. These are used with the nine forest habitat types in an attempt to classify all land and water areas in the SAA area.

Because of the different processes used by various sources, it was not always possible to reconcile all sources data to the same total acres. This is the principal reason for focusing on relationships with much of the data. The primary sources of data were LANDSAT remote sensing, Forest Inventory and Analysis (FIA), Continuous Inventory of Stand Conditions (CISC), and Natural Resources Inventory (NRI). The application of the LANDSAT, FIA, and NRI data at finer scales then used in the SAA is not appropriate in most cases.

Current Status

The current status of forest and nonforest ecosystems is described below for the following: total SAA, ecological units, and ownerships.

Current Status – Total SAA

Within the SAA area's 37,419,400 acres, about 70 percent is forested, as determined by LANDSAT remotely sensed data (fig. 3.1). Based on FIA data, most of the forest ecosystems are deciduous, with evergreen forests and mixed evergreen-deciduous forests occupying smaller proportions. Within the deciduous portion, oak forests are the dominant type. Southern yellow pine forests constitute the large majority of the evergreen type, followed by white pine-hemlock and montane spruce-fir forests. The largest portion of mixed evergreen and deciduous forests consists of mixed yellow pine-hardwood (table 3.1). When considering all forests combined, 70 percent are mid- or late-successional habitats. Currently, about 8 percent of the forested land is in grass-seedling

Table 3.1 The acreage summary of the current Southern Appalachian Assessment (SAA) area vegetation and landcover types as determined by FIA and LANDSAT remote sensing imagery.

Landcover Classes[1]	Total Acres	% of Total SAA
Forest Cover Types	26,172,425	70
Deciduous Types	17,621,894	47.1
Northern Hardwood	615,004	1.6
Mixed Mesophytic Hardwood	3,126,124	8.4
Oak Forests	13,427,883	35.9
Bottomland Hardwood	452,883	1.2
Evergreen Types	4,514,743	12.1
White Pine-Hemlock	665,925	1.8
Montane Spruce-Fir	90,101	0.2
Southern Yellow Pine	3,758,717	10.1
Mixed Types	4,035,743	10.8
White Pine-Hemock-Hardwood	830,565	2.2
Mixed Pine-Hardwood	3,205,223	8.6
Nonforest Cover Types	11,233,231	30
Grass/Shrub, Old Fields	1,528,350	4.1
Agricultural Cropland	1,271,222	3.4
Agricultural Pasture	6,522,433	17.4
Developed	1,169,798	3.1
Barren	112,529	0.3
Water	556,237	1.5
Wetlands	72,662	0.2
Totals	37,419,400	100

[1] Forest acreage is estimated using FIA data in combination with LANDSAT data. Nonforest acreage is estimated using LANDSAT data.

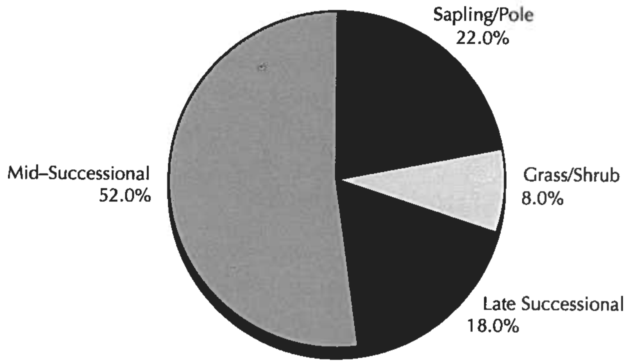


Figure 3.2 The current distribution of successional stages in the Southern Appalachian Assessment area.

successional stage (fig. 3.2). The distribution of successional habitats by forest cover is shown in table C-1.

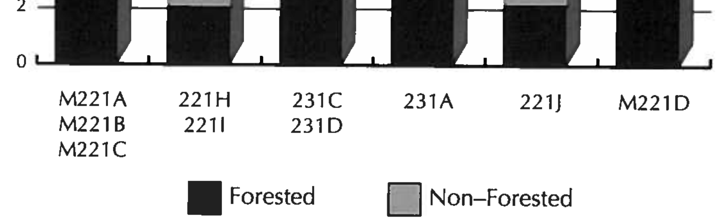
LANDSAT remotely sensed data shows about 30 percent of the SAA area is nonforested (fig. 3.1). The pastureland category accounts for the largest proportion, followed by grass-forbs-cedar woodland-early successional category, cropland, developed land, water, barren, and wetland (table C-2)

Current Status – Ecological Units

The Blue Ridge Mountains section accounts for 28 percent of the land area in the SAA area. The largest portion of the SAA area's forested ecosystems, almost nine million acres, is in the Blue Ridge Mountains section. This section has a forest-dominated landscape, with 84 percent in forestland and 16 percent in nonforest land (fig. 3.3, table C-3). Oak forest is the dominant forest type, with pastureland as the major nonforest type. This section contains most of the montane spruce-fir forest occurring in the SAA area.

The other section group considered as a forest-dominated landscape is the combined Northern Cumberland Plateau-Southern Cumberland Mountains sections. This section group is 79 percent forested (fig. 3.3, table C-3). Oak forest is the dominant forest type, and pastureland is the dominant nonforest type.

The combined Northern Ridge and Valley-Allegheny Mountains-Northern Cumberland Mountains sections account for 22 percent of the land in the SAA area. Approximately a third of this section group is in nonforest types (fig. 3.3, table C-3). Most of the nonforest type is pasture. A majority of the forestland is deciduous, with oak forest being the major type.



Section Groups

M221A – Northern Ridge & Valley
M221B – Allegheny Mountains
M221C – Northern Cumberland Mountains
M221D – Blue Ridge Mountains
231C – Southern Cumberland Plateau

221H – Northern Cumberland Plateau
221I – Southern Cumberland Mountains
221J – Central Ridge & Valley
231A – Southern Appalachian Piedmont
231D – Southern Ridge & Valley

Figure 3.3 A summary of forest and non–forest land by ecological sections in the Southern Appalachian Assessment area.

The Southern Appalachian Piedmont section is similarly distributed in forest and nonforest proportions. A difference in the Piedmont is that southern pine forests and mixed pine-hardwood forests are more common in forested lands.

The Central Ridge and Valley section has the largest proportion in nonforest land, about 51 percent, with the Southern Ridge and Valley-Southern Cumberland Plateau next at about 42 percent. A large proportion of these nonforest habitats is pastures, old fields, farms, and urban areas. Mixed pine-hardwood and southern yellow pine forests are more common in forested lands when compared to the SAA as a whole (fig. 3.3, table C-3).

Current Status by Ownerships

The majority of land within the SAA area, about 84 percent, is privately held. Public holdings account for the remaining 16 percent. Private ownership was not further stratified because additional GIS ownership data was not available (i.e., Tennessee Valley Authority (TVA), local county ownerships). About three-fourths of the publicly owned land in the SAA area is National Forest System (NFS) land. National parks occupy about 14 percent of public land; state-owned land occupies about 9 percent; and other federal land occupies about 2 percent (table 3.2).

Public ownerships contain most of the high-elevation montane spruce-fir forests and northern hardwood forests, while private ownerships contain the majority of the remaining forest and nonforest types. Of the 23 percent of forested land in public ownership, national forests have the largest proportion, followed by national parks, and state lands (table 3.3). Tables C-4, C-5, C-6, and C-7 provide detailed information regarding the distribution of forest types by ownership using FIA data.

Private ownership currently contains the majority of the forest grass-shrub, sapling-pole, and mid-successional habitats (figs. 3.4, 3.5, 3.6). National forests contain the majority of the late successional forest habitats in the SAA area (fig. 3.7). The current forest successional stage distribution within ownerships deviates from

Table 3.2 The distribution of current Southern Appalachian Assessment (SAA) area acres by ownerships.

Ownership	Acres	Percent by Owner Group	Percent within Owner Group	Percent of Total SAA
Private	31,343,760	83.8		
Cherokee Reservation	45,437		1	1
Other Private	31,298,323		99	83.7
Public	6,075,640	16.2		
National Forests	4,553,637		74.9	12.2
National Parks	840,687		13.8	2.2
State Owned	574,622		9.5	1.5
Other Federal	106,694		1.8	0.3

(Source: Derived from USGS and national forest stand cover layers for the Southern Appalachian Assessment)

Table 3.3 The distribution of current Southern Appalachian Assessment (SAA) area acres by forest and non–forest according to ownerships based on LANDSAT data.

	Forest		Non–Forest	
Ownership	Acres	Percent	Acres	Percent
Private	20,268,893	77.4	11,063,968	98.5
Cherokee Reservation	42,033	0.2	3,404	0.0
Other Private	20,226,860	77.3	11,060,564	98.5
Public	5,903,532	22.6	169,268	1.5
National Forests	4,468,835	17.1	82,896	0.7
National Parks	820,127	3.1	20,560	0.2
State Owned	531,144	2.0	43,050	0.4
Other Federal	83,426	0.3	22,762	0.2

(Source: Derived from remotely sensed data for the Southern Appalachian Assessment)

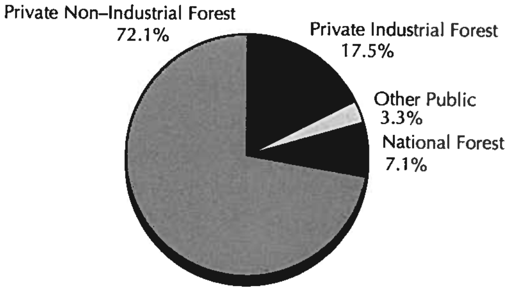


Figure 3.4 The current distribution of grass–shrub early successional forest habitats by ownership in the Southern Appalachian Assessment area.

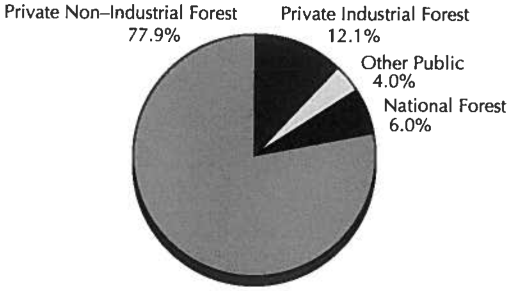


Figure 3.5 The current distribution of sapling–pole successional forest habitats by ownership in the Southern Appalachian Assessment area.

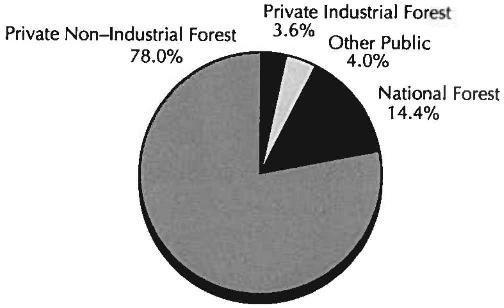


Figure 3.6 The current distribution of mid–successional forest habitats by ownership in the Southern Appalachian Assessment area.

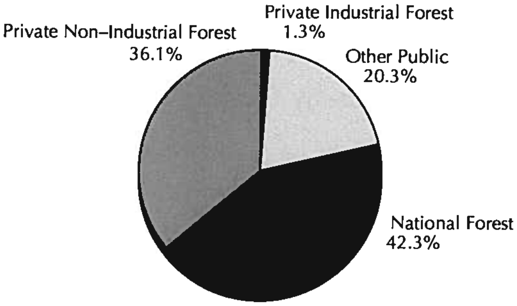


Figure 3.7 The current distribution of late successional forest habitats by ownership in the Southern Appalachian Assessment area.

the SAA area as a whole. For example, smaller percentages of NFS land are in grass-seedling habitats, sapling-pole, and mid-successional stages than compared to the total SAA area. However, the 45 percent of the NFS forest in late-successional habitats is more than twice that of the total SAA area.

Tables C-4, C-5, C-6, C-7, and C-8 contain additional detailed information regarding forest and nonforest habitats according to ownership.

TRENDS

Changes in acreages were estimated for eight forest habitat types over the past 20 to 25 years. The FIA data permitted analysis of forest habitats that included: (1) maple-beech-birch forests, (2) oak-mixed mesophytic hardwood forests, (3) elm-ash-cottonwood forests, (4) white pine-hemlock forests, (5) spruce-fir forests, (6) southern yellow pine forests, (7) longleaf pine forests, and (8) oak-pine forests. In addition to habitat types, FIA data permitted assessment of trends for four ownership classes: (1) NFS land, (2) other public land, (3) private industry and leased land, and (4) nonindustrial private land. The NRI database permitted some assessment of trends of several nonforest ecosystems.

Trends – Total SAA

The total acreage in the forest ecosystems represented by the FIA's timberland within the SAA area has decreased about 2 percent since the mid-1970s, however this decrease has not been uniform throughout the eight forest groups. Decreases have occurred in oak-mixed mesophytic hardwoods, elm-ash-cottonwood, and southern yellow pine; but, increases registered in acreage of maple-beech-birch, white pine-hemlock, spruce-fir, longleaf pine, and oak-pine forests. The sharpest decreases occurred in southern yellow pine, down 16 percent, and in elm-ash-cottonwood, down 9 percent. The largest gains were made by white pine-hemlock, up 39 percent, and in longleaf pine, up 24 percent (table 3.4). This loss in forest acres is occurring on private lands from development and conversion to agricultural land uses.

For the SAA area, data shows that between 1982 and 1992 large urban areas increased by 35 percent, and small urban areas by 53 percent. Cultivated cropland fell 25 percent. Non-cultivated cropland (orchards, etc.), however, increased 9 percent. Grass pasture decreased 3 percent, but legume pasture increased 38 percent. When these trends are examined by state, rather than for the SAA area as a whole, the same pattern seems to hold. Cultivated cropland decreased, noncultivated cropland increased, and both large and small urban developed areas grew (table C-9).

Trends By Ownerships

Forest Habitat Types. During the past two decades, timberland acreage appears to have increased on public land while decreasing on private holdings. NFS timberland increased by 5 percent and other types of public timberland increased 8 percent. On private industry and leased land and nonindustrial private timberland, the decreases in timbered acreage were 5 percent and 2 percent, respectively. On NFS timberland, the largest increase occurred in longleaf pine forests and in maple-beech-birch. Both these increases probably came at the expense of oak forest and mixed mesophytic hardwood forests, which decreased about a percentage point. On other public timberland, white pine-hemlock and maple-beech-birch, more than doubled. Acreage decreases for this ownership class were in elm-ash-cottonwood and oak-pine forest (table C-10).

For private industry and leased timberland, increases came in white pine-hemlock forest, elm-ash-cottonwood forest, and oak-pine forest. Decreases occurred in maple-beech-birch and oak-mixed mesophytic hardwoods. On nonindustrial private timberland, increases occurred in white pine-hemlock forests, with the largest acreage decrease coming in spruce-fir and southern yellow pine. Interpretations of the data for these forest groups, however, should include consideration that the largest percentage changes are due primarily to large changes in small acreages and, therefore, should not be considered as showing significant changes in the proportions of forest ecosystems (table C-10).

Successional Stages. In 20 years, the acreage in grass-seedling-shrubs stage increased about 26 percent in the SAA area as a whole. However, there were important differences among ownerships. Increases were 185 percent for other public, 11 percent for land owned or leased by forest industry, and 33 percent for nonindustrial private land, while national forest acreage decreased by 4 percent (table 3.5).

Acreage in sapling-pole decreased 24 percent on the SAA area as a whole, but again there was considerable variance in this stage by ownerships. NFS acreage increased about 12 percent, but other public land decreased approximately 17 percent. Industry-leased private acreage increased about 10 percent, but nonindustry private acreage decreased some 30 percent (table 3.5).

Acreage in mid-successional stage increased about 3 percent overall. NFS acreage in this stage decreased about 6 percent, other public land increased about 12 percent, industry-leased private acreage decreased about 24 percent, and nonindustry private

Table 3.4 The 20 year trends (mid 1970s to 1995) for forest habitats for the total Southern Appalachian Assessment area based on FIA data.

	Acres of Timberland		
FIA Forest Type Group	Mid '70s Acres	1995 Acres	% Change
Maple–Beech–Birch Forests	508,861	552,152	9
Oak–Hickory Forests	15,283,985	15,100,804	–1
Elm–Ash–Cottonwood Forests	205,462	185,999	–9
White Pine–Hemlock Forests	432,193	598,929	39
Spruce–Fir Forests	12,714	13,130	3
Southern Yellow Pine Forests	4,077,348	3,412,977	–16
Longleaf Pine Forests	33,121	40,916	24
Oak–Pine Forests	3,426,563	3,626,484	6
Totals	23,980,247	23,531,391	–2

(Source: USDA Forest Service, Southern Research Station, Forest Inventory and Analysis Unit)

Table 3.5 The 20 year trends (mid 1970s to 1995) for forest successional classes according to ownerships in the Southern Appalachian Assessment area based on FIA data.

	Acres of Timberland					
	Grass/Seedling/Shrub Stage			Sapling/Pole Stage		
Ownership	Mid '70s Acres	1995 Acres	% Change	Mid '70s Acres	1995 Acres	% Change
National Forest	237,299	227,744	–4	479,013	534,486	12
Other Public	22,024	62,802	185	266,283	221,744	–17
Private Industry & Leased	297,252	330,219	11	550,323	605,171	10
Non–Industrial Private	1,022,383	1,363,230	33	6,429,017	4,502,565	–30
Totals	1,578,958	1,983,995	26	7,724,636	5,863,966	–24

	Acres of Timberland					
	Mid Successional Stage			Late Successional Stage		
Ownership	Mid '70s Acres	1995 Acres	% Change	Mid '70s Acres	1995 Acres	% Change
National Forest	2,272,935	2,127,135	–6	847,079	1,133,935	34
Other Public	449,664	502,433	12	77,829	94,559	21
Private Industry & Leased	612,553	462,524	–24	59,604	56,148	–6
Non–Industrial Private	9,301,433	9,917,351	7	1,055,556	1,615,980	53
Totals	12,636,585	13,009,443	3	2,040,068	2,900,622	42

(Source: USDA, Forest Service, Southern Research Station, Forest Inventory and Analysis Unit)

acreage increased about 7 percent (table 3.5).

Acreage in late-successional stage increased about 42 percent on all ownerships. Forest industry acreage dropped 6 percent. NFS acreage increased by about 34 percent, other public increased by about 21 percent, and non-industrial private increased approximately 53 percent (table 3.5).

Tables C-11, C-12, C-13, C-14, and C-15 contain additional detailed information regarding trends in forest successional classes by forest types for ownerships in the SAA.

National Forest Land Inventoried for Possible Old-Growth Forest Goals

Part of the assessment involved inventorying national forest land for areas having some of the physical characteristics of an old-growth forest. This inventory is a starting point. The areas identified in this inventory may or may not be managed to meet a standard for old-growth forests. The upcoming forest plan revisions for national forests in the SAA will determine areas to be managed as old-growth.

Criteria were developed for identifying the initial inventory of areas (1994). These criteria included:

- forest stands that are greater than 100 years old
- forest stands included within designated wilderness
- supplemental inventory based on local knowledge that might include undisturbed riparian forest, undisturbed stands for a number of decades, stands of low site productivity with little disturbance, and past inventories.

Efforts were made to ensure that proxy attributes were used consistently among the forests as much as possible. Some variances were expected because of the need to include on-the-ground judgments about the areas and because of limits in the existing data sources. For example, some areas of forest vegetation could fit into several old-growth forest type groups, but an on-the-ground examination permits the area to be assigned to the most precise group. Another example is where some forest communities mature before age 100. The proxy attributes permitted their inclusion. Another case is younger areas that are partially or fully surrounded by old forest communities.

Some 13,050 areas comprising almost 1.1 million acres were identified in this initial inventory of national forests (fig. 3.8, figs. C-1 to C-37, table C-16). The average size of these areas was about 84 acres, but size varied from a single acre to 13,000 acres. This inventory found sites in 11 different old-growth forest

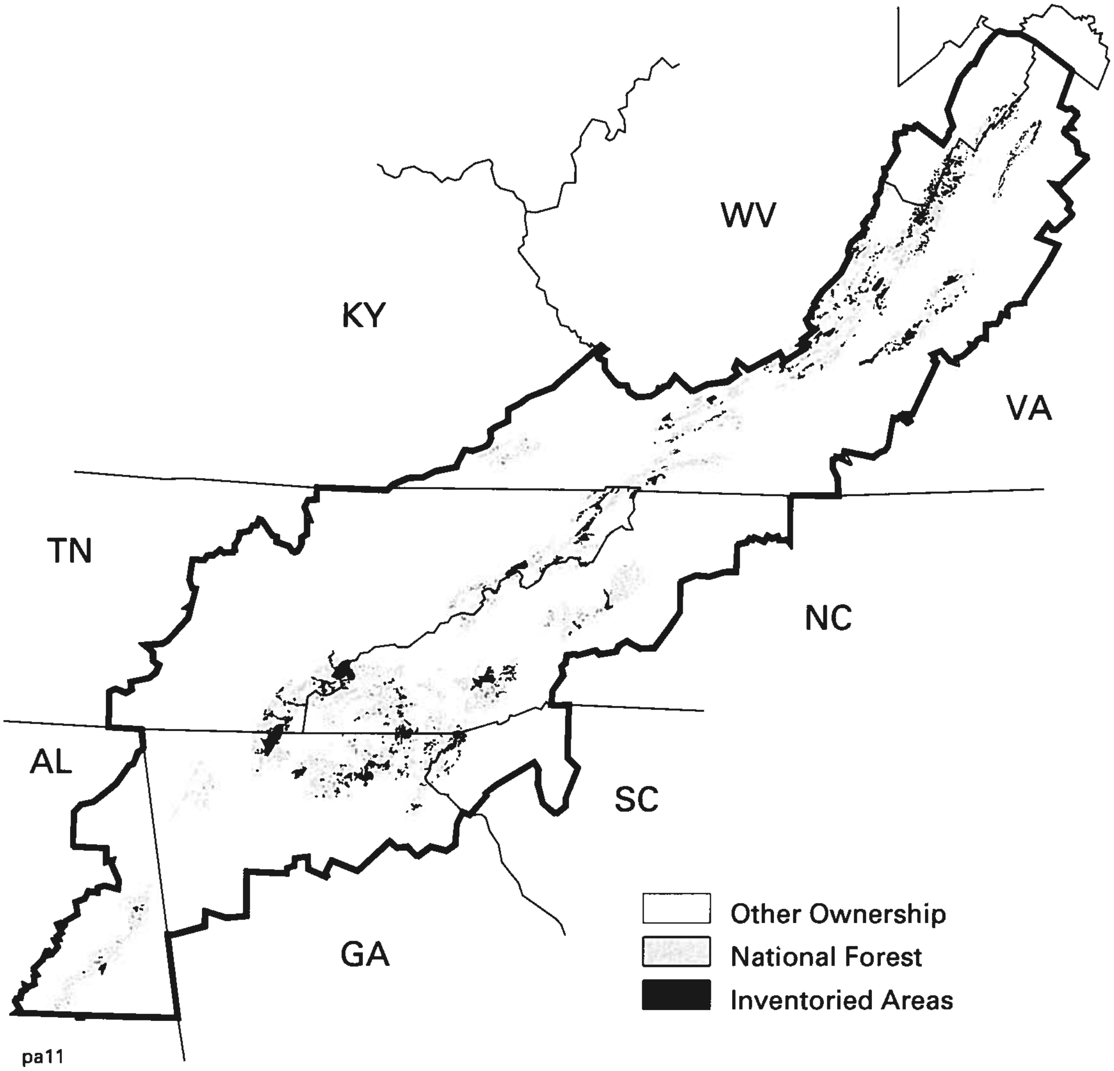


Figure 3.8 The distribution of stands identified in an initial inventory of possible old-growth forest on National Forest System lands in the SAA.

type groups represented in the SAA area (table 2.3). The two not represented are hardwood wetland forests and the rocky, thin-soil, excessively drained, cedar woodlands. Of the 11 groups, 4 old-growth forest type groups dominate the representation. The dry-mesic oak forest has nearly half the acreage. The next most represented communities are the mixed mesophytic forests; the dry and dry-mesic, oak-pine forests; the dry/xeric oak and xeric oak; and the xeric pine and pine-oak forests. In about 20 percent of the inventoried areas, there were insufficient data to assign an old-growth forest type code. A large part of this percentage is in wildernesses (table C-16).

Almost 428,000 acres, about 39 percent, identified in the initial inventory are currently being managed for timber production (table C-17). The largest part of this is dry-mesic oak forests (about 63 percent). The average size of the stands currently managed for timber production is 58 acres, varying in size from 1 acre to about 2,100 acres. The distribution differs on the 671,000 acres of NFS land not managed for timber production that were included in the inventory. Most of the high-elevation forests (montane spruce-fir and northern hardwood) and the mixed mesophytic hardwood forests are in areas not managed for timber production. The average size of the stands not managed for timber production is about 118 acres. These stands varied in size from about an acre to 6,800 acres.

Rare Communities

Occurrence information was compiled for 31 identified rare communities. The information came from state natural heritage programs and agencies including TVA, the USDA Forest Service (FS), the U.S. Fish and Wildlife Service (FWS), and others. Because definitions of communities vary among sources, the process for assigning occurrence to communities was not precise. Rare communities and indicator species are listed in table C-18.

In some cases, an occurrence point for a rare community is the same as an indicator species. So, what may show as two occurrences in the tables, may, in reality, be only one. Thus, numbers of occurrences are inflated for some communities. Acreage data for rare communities are available only for spruce-fir forests. Other rare community occurrences are known only as points. What this means is that an enormous amount of work is needed to determine how much area of rare communities we have in the Southern Appalachians.

CURRENT STATUS

Two thousand and eighty-seven rare community and/or indicator species occurrences were assembled for this analysis. Based on the known size of some occurrences and the approximate size of others, the best estimate is that the total aggregate acreage of these rare communities across the SAA area is less than 3 percent of total acreage. Only about a third of this area – about 1 percent of total SAA area acreage – represents high-quality examples of the communities.

Table C-19 provides a breakdown of rare community occurrences by state. Within the SAA area, Virginia and North Carolina contain the highest numbers of occurrences with more than 750 for each state. A distant third in number of occurrences is Tennessee with slightly more than 200. Alabama, Georgia, South Carolina, and West Virginia show less than 100 each. These figures do not necessarily reflect the states as a whole, only that portion of the state that lies within the SAA area. For example, only a few counties of West Virginia fall inside the SAA area boundary. In addition, inventory efforts in some states have been more intense than in others. However, some of the underlying reasons for greater occurrence numbers in Virginia, North Carolina, and Tennessee are due to the mountains of the Blue Ridge and Ridge and Valley sections (fig. 3.9).

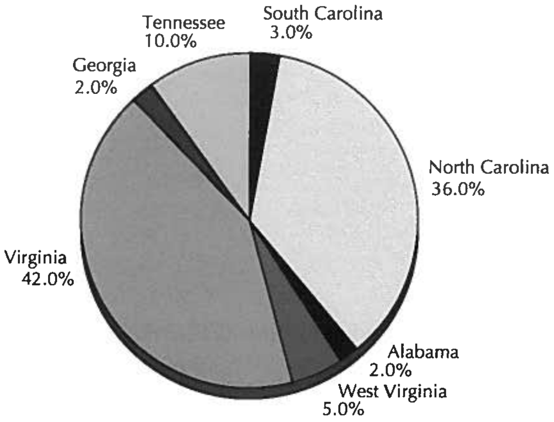
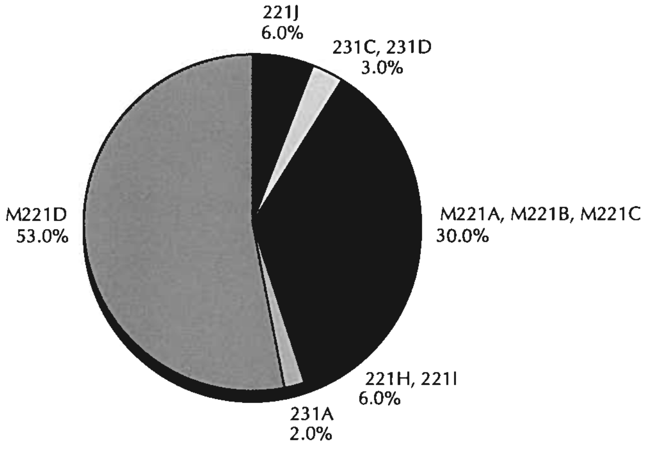


Figure 3.9 The distribution of the 31 rare communities by states in the Southern Appalachian Assessment area.



Section Groups

M221A –	Northern Ridge & Valley	221H –	Northern Cumberland Plateau
M221B –	Allegheny Mountains	221I –	Southern Cumberland Plateau
M221C –	Northern Cumberland Mountains	221J –	Central Ridge & Valley
M221D –	Blue Ridge Mountains	231A –	Southern Appalachian Piedmont
231C –	Southern Cumberland Plateau	231D –	Southern Ridge & Valley

Figure 3.10 The distribution of the 31 rare communities by ecological units in the Southern Appalachian Assessment area.

Table C-20 shows the occurrence of rare community by ecological unit. The Blue Ridge Mountains have the greatest number of occurrences with slightly more than 1,100. The Northern Ridge and Valley section has almost 600. Taken together, these two sections have more than 80 percent of the rare community occurrences within the SAA area, yet they occupy less than half the total acreage. This pattern is probably due to the wide range of conditions that favor development of specialized habitats (fig. 3.10).

Public land contains about 38 percent of the occurrences of rare communities (fig. 3.11). Except for a few high-elevation and mountain-associated rare communities, all occurrences are more numerous on private land. In the case of caves and sphagnum-shrub bog rare communities, virtually all occurrences are on private land. Most rare communities are found in valley settings. Table C-21 provides a breakdown of rare community occurrences by ownership.

TRENDS

Specific trend information is lacking for most rare communities in the SAA area. In general, however, rare communities seem to be declining in acreage and quality. For instance, sphagnum-shrub bogs, along with most other wetland types, have declined by more than 90 percent (Noss and others 1995). Others have suffered slight to moderate declines in acreage, but their integrity and quality are being severely impacted by naturalized exotic plants and animals. Communities with these impacts include beech gap forests, calcareous woodlands and glades, Carolina hemlock forests and spruce-fir forests. Still others, such as caves and granitic domes, are being impacted by increased recreational activities. Of the 31 rare communities, only the beaver ponds and wetland complex may be showing an upward trend in acreage.

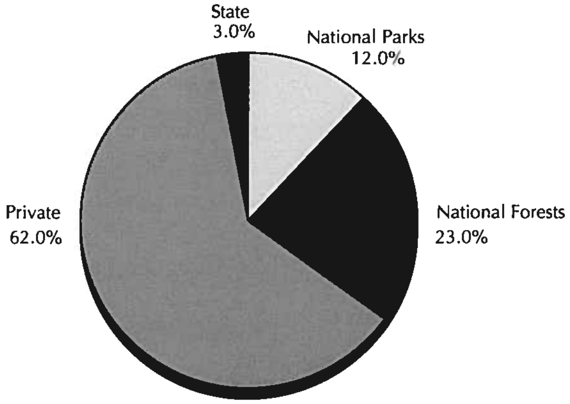


Figure 3.11 The distribution of the 31 rare communities by ownership in the Southern Appalachian Assessment area.

Table 3.6 The federally listed threatened and endangered terrestrial plant and animal species of the Southern Appalachian Assessment area.

Scientific Name	Common Name	Taxa	Federal Status	Global Rank	Species Group[1]
Plethodon nettingi	Cheat Mountain Salamander	Amphibian	T	3	15
Plethodon shenandoah	Shenandoah Salamander	Amphibian	E	1	7
Falco peregrinus anatum	American Peregrine Falcon	Bird	E		7
Haliaeetus leucocephalus	Bald Eagle	Bird	T		11
Picoides borealis	Red Cockaded Woodpecker	Bird	E		17
Antrolana lira	Madison Cave isopod	Invertebrate	T	1	1
Lirceus usdagalun	Lee County Cave Isopod	Invertebrate	E		1
Microhexura montivaga	Spruce-Fir Moss Spider	Invertebrate	E		15
Patera clarki nantahala	Noonday globe snail	Invertebrate	T		16
Polygyriscus virginicus	Virginia Fringed Mountain Snail	Invertebrate	E		16
Canis rufus	Red Wolf	Mammal	E		9
Corynorhinus townsendii virginianus	Virginia Big-eared Bat	Mammal	E		1
Felis concolor cougar	Eastern Cougar	Mammal	E		9
Glaucomys sabrinus coloratus	Carolina Northern Flying Squirrel	Mammal	E		15
Glaucomys sabrunus fuscus	Virginia Northern Flying Squirrel	Mammal	E		15
Myotis grisescens	Gray Bat	Mammal	E		1
Myotis sodalis	Indiana Bat	Mammal	E		1
Amphianthus pusillus	Pool Sprite	Plant	T		7
Apios priceana	Price's potato-bean	Plant	T		7
Arabis serotina	Shale barren rock cress	Plant	E	2	7
Arenaria cumberlandensis	Cumberland sandwort	Plant	E		7
Asplenium scolopendrium var american	Hart's tongue fern	Plant	T	1	6
Betula uber	Virginia round-leaf birch	Plant	T	1	11
Cardamine micranthera	Small anthered bittercress	Plant	E	1	11
Clematis socialis	Alabama leather-flower	Plant	E	1	11
Conradina verticillata	Cumberland rosemary	Plant	T		11
Echinacea laevigata	Smooth Coneflower	Plant	E	3	6
Geum radiatum	Spreading avens	Plant	E	1	5
Gymnoderma lineare	Rock gnome lichen	Plant	E	2	7
Hedyotis purpurea var. montana	Roan mountain bluet	Plant	E	2	5
Helonias bullata	Swamp pink	Plant	T	3	2
Hexastylis naniflora	Dwarf-flowered heartleaf	Plant	T	2	18
Hudsonia montana	Mountain gloden heather	Plant	T	1	7
Iliamna corei	Peter's mountain mallow	Plant	E	1	7
Isotria medeoloides	Small whorled pogonia	Plant	E	3	18
Liatris helleri	Heller's blazing star	Plant	T	1	5
Marshallia morhii	Morh's Barbara's buttons	Plant	T		11
Pityopsis ruthii	Ruth's golden aster	Plant	E	1	11
Platanthera leucophaea	Eastern prarie fringed orchid	Plant	T	2	4
Ptilimnium nodosum	Harperella	Plant	E	2	11
Sagittaria fasciculata	Bunched arrowhead	Plant	E	1	11
Sagittaria secundifolia	Kral's water-plaantain	Plant	T		2
Sarracenia jonesii	Mountain sweet pitcherplant	Plant	E	1	2
Sarracenia oreophila	Green pitcher plant	Plant	E	2	2
Scirpus ancistrochaetus	Northeastern bullrush=Barbed bullrush	Plant	E	2	4
Scutellaria montana	Large-flowered skullcap	Plant	E	2	18
Sisyrinchium dichotomum	White irisette	Plant	E	1	6
Solidago spithamaea	Blue Ridge goldenrod	Plant	T	1	5
Spiraea virginiana	Virginia spiraea	Plant	T	1	11
Trillium persistens	Persistent trillium	Plant	E		18
Xyris tennesseensis	Tennessee yellow-eyed grass	Plant	E		6

[1] Species Group Codes

1 = Cave Habitats
2 = Mountain Bogs
3 = Spray Cliffs
4 = Fen or Pond Wetlands
S = High Elevation Balds
6 = High pH or Mafic Habitats
7 = Rock Outcrop and Cliffs
8 = Early Successional Habitats
9 = Wide Ranging Area Sensitive Species
10 = Mid– to Late–Successional Forest Species
11 = Seep, Spring, and Streamside Habitat
12 = Habitat Generalist
13 = Area Sensitive Deciduous Forest
14 - General High Elevation Habitats
15 = High Elevation Spruce–Fir Forest
16 = Bottomland Forests
17 = Southern Yellow Pine Habitats
18 = Mixed Mesic Habitats
19 = Mixed Xeric Habitats

Federally Listed Threatened and Endangered Terrestrial Species

An important component of the assessment was the determination of the status of the federally listed T&E species. The list of 51 federally listed species was based on information from the U.S. Fish and Wildlife Service, the state natural heritatge programs, and peer review of a draft species list. Habitat relationships were determined for all T&E species. These species/habitat associations also received peer review. It should be noted, however, that much of the information on species/habitat relationships is still subjective.

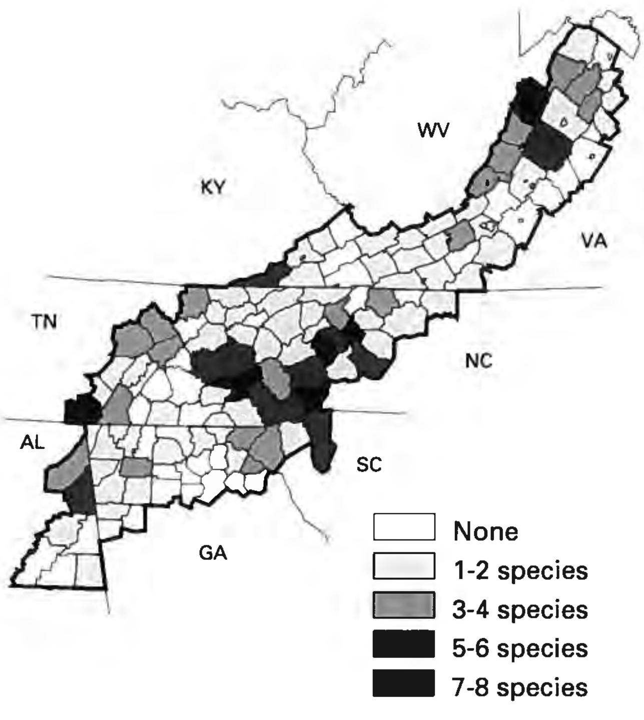
Based on the analysis of species/habitat relationships, around 65 percent of these species is associated with rare communities. The proportion rises to 84 percent when riparian communities are included. These species, for the most part, were not suited for broad-scale analysis of habitat suitability. This section provides the analysis of current status, expressed as spatial occurrences. These occurrences were taken from Element Occurrence Records (EOR) obtained from the seven state natural heritage programs in the SAA. Both analysis of occurrence data and habitat suitability (Chapter 3, Habitat Suitability section) were provided for some species.

Current Status – Total SAA

The distribution maps of T&E terrestrial species occurrence records described below are based on data provided by state heritage programs. Some of these data are quite old, and, in some cases, the species may no longer be present at the sites indicated. For the majority of these species, occurrence records were not derived from systematic surveys and, therefore, probably do not provide a complete picture of their ranges. Still, these are the best data available for many of these species.

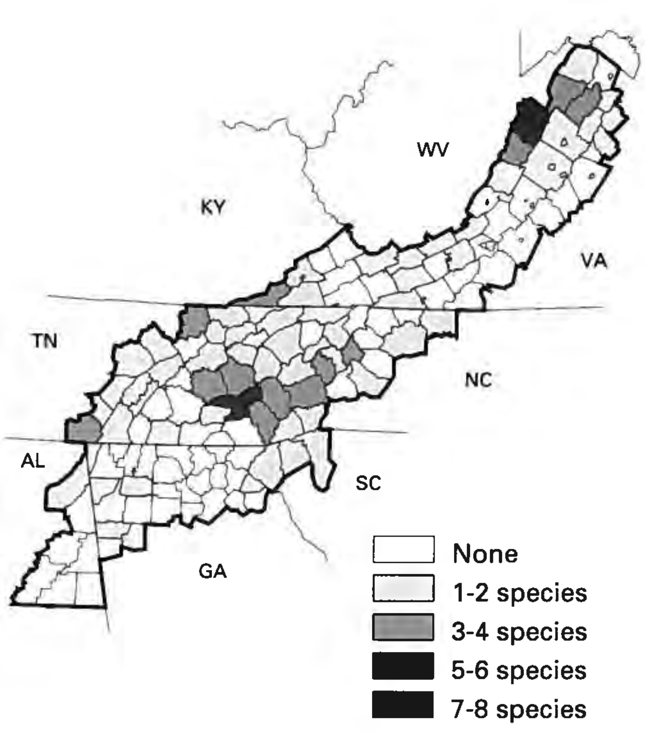
Of 51 federal T&E species in the SAA area, 17 are animals and 34 are plants (table 3.6). Of the animal species, seven are mammals, three are birds, two are amphibians, and five are invertebrates. No species are proposed for addition to the federal T&E species list as of late 1995.

Fifty-three counties within the SAA area contain T&E terrestrial animals and 55 contain T&E plants (figs. 3.12, 3.13, 3.14). Two counties



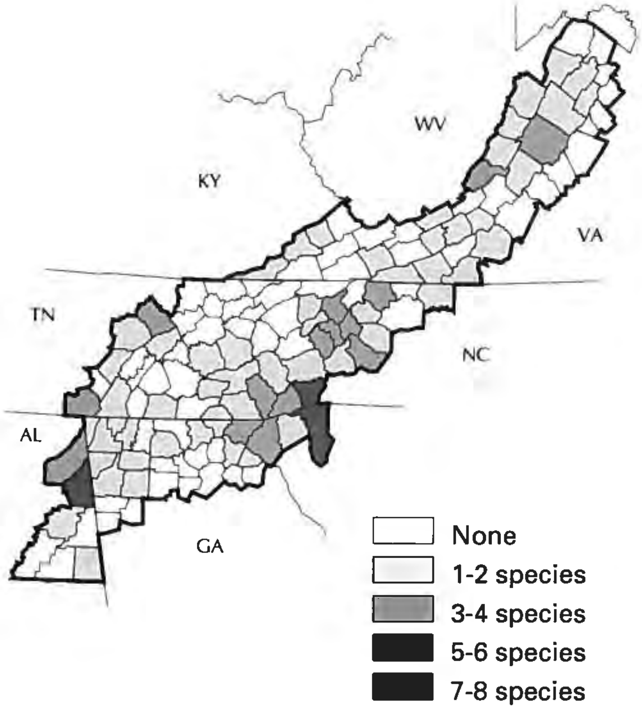
pa2b

Figure 3.12 The spatial distribution for the number of federally listed threatened and endangered terrestrial species by county in the SAA area.



pa2a

Figure 3.13 The spatial distribution for the number of federally listed threatened and endangered terrestrial animal species by county in the SAA area.



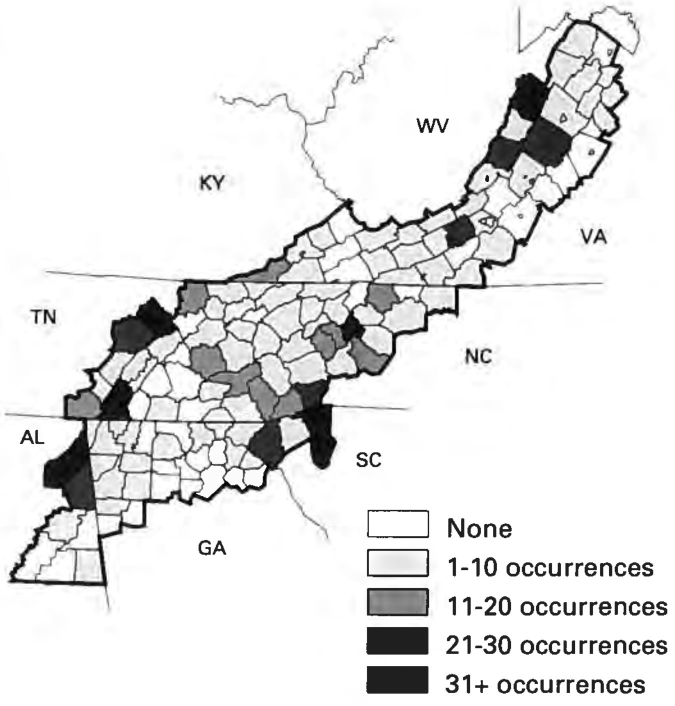
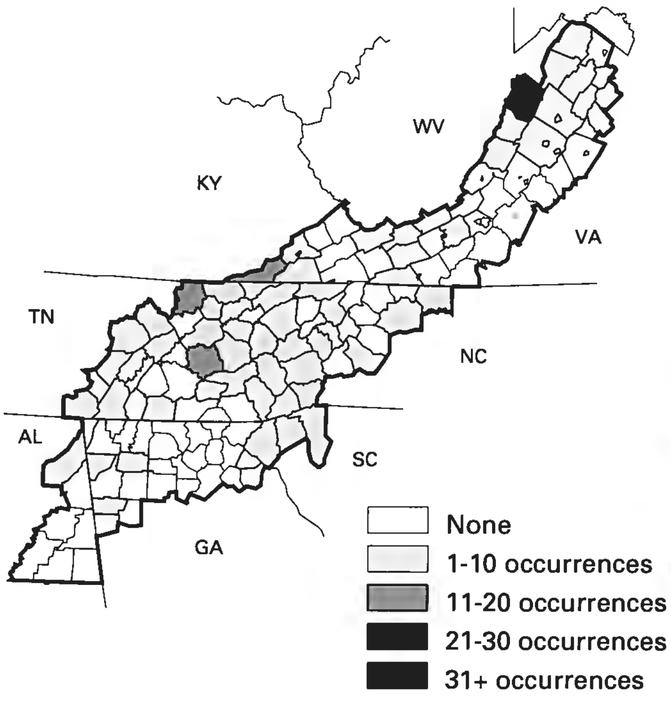


Figure 3.14 The spatial distribution for the number of federally listed threatened and endangered terrestrial plant species by county in the SAA area.

Figure 3.15 The spatial distribution for the number of occurrences of federally listed threatened and endangered terrestrial species by county in the SAA area.



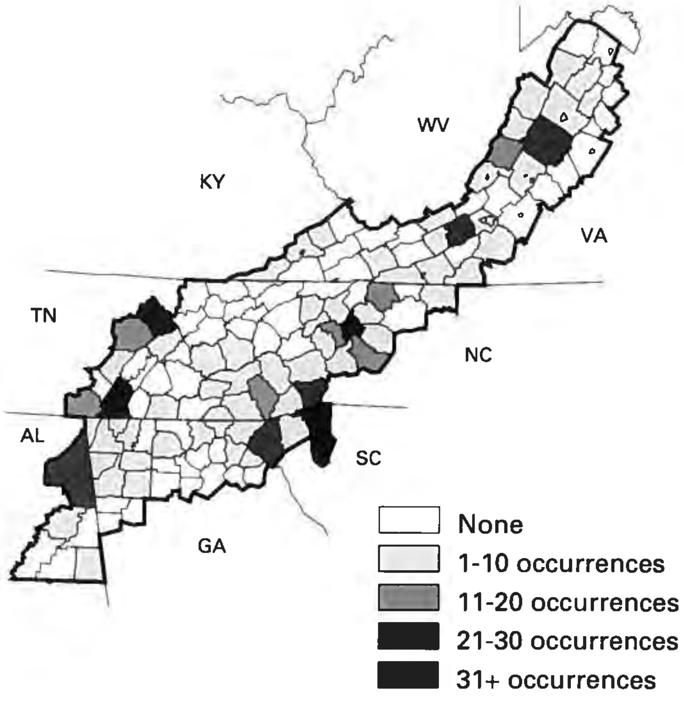


Figure 3.16 The spatial distribution for the number of occurrences of federally listed threatened and endangered terrestrial animal species by county in the SAA area.

Figure 3.17 The spatial distribution for the number of occurrences of federally listed threatened and endangered terrestrial plant species by county in the SAA area.

Table 3.7 The number of federally threatened and endangered terrestrial species and number of occurrences by species group in the Southern Appalachian Assessment area.

	Number of Species			Number of Occurrences		
Species Group	Plant	Animal	Total	Plant	Animal	Total
Cave	0	5	5	0	129	129
Mountain Bog	4	0	4	88	0	88
Spray Cliff	0	0	0	0	0	0
Fen or Pond Wetland	2	0	2	8	0	8
High Elevation Bald	4	0	4	58	0	58
High pH or Mafic	4	2	6	76	3	79
Rock Outcrop and Cliff	6	2	8	87	29	116
Early Successional Grass/Shrub	0	0	0	0	0	0
Wide Ranging Area Sensitive	0	2	2	0	10	10
Mid to Late Successional Deciduous Forest	0	0	0	0	0	0
Seeps, Springs, and Streamside	9	2	11	124	21	145
Habitat Generalist	0	0	0	0	0	0
Area Sensitive, Mid to Late Successional Deciduous	0	0	0	0	0	0
General High Elevation Forest	0	0	0	0	0	0
High Elevation Spruce–Fir	0	4	4	0	41	41
Bottomland Forest	0	0	0	0	0	0
Southern Yellow Pine Forest	0	1	1	0	24	24
Mixed Mesic	5	0	5	90	0	90
Mixed Xeric	0	0	0	0	0	0

have five to six T&E plants and four have five to six T&E animals. No county has more than eight animal or plant T&E species.

Thirty-seven counties have no occurrence records for T&E species (fig. 3.15). This does not necessarily mean there are no such species in these counties. It means that there are no records in the state heritage databases for them. Additional surveys may reveal T&E species in counties not now counted as having them. Most counties (64 for animals, 42 for plants) have 10 or fewer occurrence records for T&E species (figs. 3.15, 3.16, 3.17).

The T&E species were organized into 11 groups based on habitat associations. As stated previously, most of the T&E terrestrial species are associated with rare communities. Eight of the 19 species groups contain no T&E species (table 3.7). Of the remaining, 5 have 5 to 10 species (caves, 5; high pH or mafic species, 6; rock outcrop and cliff species, 8; seeps, springs, and streamside species, 11; and mixed mesic species, 5). For those groups with T&E species, the number of occurrences ranges from a low of 8 (fen or pond wetland species) to more than 100 (cave species, 129; rock outcrop and cliff species, 116; seeps, springs, and streamside species, 145). Individual T&E terrestrial species are listed by species group in Table E-1.

Current Status by Ecological Units

The distribution of T&E terrestrial species by physiographic section varies from four to 31 (table 3.8). The Blue Ridge Mountain section (M221D) contains the highest number with 13 animal and 18 plant T&E terrestrial species. Physiographic subsection totals vary, with most having fewer than 12 species (table 3.8). M221Dc (Southern Blue Ridge Mountains subsection) has the highest number with 25 (8 animal, 17 plant).

Current Status by Ownership

National park land contains 17 T&E terrestrial species with 90 occurrence records (table 3.9). National forest lands hold 26 species (10 animal, 16 plant) and 154 records (38 animals and 116 plant occurrences). Other federal ownerships have three T&E species and four occurrences. Thirteen T&E species are found on state lands and contain 47 records. Other ownerships have the highest number of T&E species (45, of which 13 are animal and 32 are plant species). The majority of occurrence records (493 records or 62.5 percent) are noted from other ownerships (table 3.9).

The number of T&E terrestrial species on

Table 3.8 The number of federally threatened and endangered terrestrial species and number of occurrences by ecological section and subsection in the Southern Appalachian Assessment area.

	Number of Species			Number of Occurrences			
Section/Subsection	Animal	Plant	Total	Animal	Plant	Total	Acres
221H	5	7	12	27	69	96	2,089,915
221Hc	4	7	11	15	65	80	1,295,014
221Hd	2	1	3	4	3	7	236,597
221He	3	1	4	8	1	9	558,304
221I	2	2	4	5	4	9	533,365
221Ib	2	2	4	5	4	9	533,365
221J	6	2	8	43	9	52	4,519,923
221Ja	6	2	8	42	8	50	2,822,951
221Jb	0	1	1	0	1	1	1,162,501
221Jc	1	0	1	1	0	1	474,471
231A	1	8	9	2	60	62	6,943,194
231Aa	1	4	5	1	21	22	2,059,243
231Ab	0	1	1	0	4	4	197,264
231Ac	0	1	1	0	3	3	1,670,486
231Ad	1	3	4	1	28	29	965,570
231Ag	0	1	1	0	1	1	549,331
231Ai	0	1	1	0	1	1	48,588
231Ak	0	1	1	0	2	2	1,023,886
231C	2	4	6	6	44	50	791,057
231Cc	1	4	5	3	43	50	791,057
231Cf	2	1	3	3	1	4	181,869
231D	2	8	10	4	51	55	3,504,025
231Da	0	5	5	0	28	28	925,691
231Db	0	3	3	0	7	7	591,862
231Dc	1	0	1	1	0	1	462,060
231Dd	1	0	1	1	0	1	482,093
231De	2	5	7	2	16	18	1,042,319
M221A	7	6	13	65	85	150	7,711,967
M221Aa	5	4	11	54	46	100	4,643,426
M221Ab	6	5	11	11	39	50	3,068,542
M221B	3	1	4	12	1	1	217,690
M221Ba	3	0	3	10	0	10	65,172
M221Bb	1	1	2	2	1	3	152,518
M221C	0	1	1	0	1	1	481,891
M221D	13	18	31	87	213	300	10,626,358
M221Da	2	1	3	6	12	18	1,258,648
M221Db	0	1	1	0	8	8	1,305,965
M221Dc	8	17	25	42	177	219	5,226,770
M221Dd	10	5	15	39	16	55	2,834,975

(Source: Table based on 1995 EOR data furnished by the state heritage programs.)

Table 3.9 The distribution of federally listed threatened and endangered terrestrial species by ownership in the Southern Appalachian Assessment area.

	Number of Species			Number of Occurrences		
Ownership	Animal	Plant	Total	Animal	Plant	Total
National Parks	8	9	17	42	48	90
National Forest	10	16	26	38	116	154
Other Federal	1	2	3	1	3	4
State	7	6	13	25	22	47
Private	13	32	45	145	348	493
Total				251	537	788

Table 3.10 The distribution of federally listed threatened and endangered terrestrial species by National Forest in the Southern Appalachian Assessment area.

	Number of Species			Number of Occurrences		
National Forest	Animal	Plant	Total	Animal	Plant	Total
George Washington	3	3	6	4	37	41
Jefferson	2	1	3	2	2	4
Monongahela	3	0	3	10	0	10
Nantahala/Pisgah	5	8	13	16	38	54
Sumter	1	2	3	1	17	18
Cherokee	4	6	10	5	11	16
Chattahoochee	0	5	5	0	11	11
Talladega	1	0	1			

Table 3.11 The number of federal threatened and endangered terrestrial species and occurrences by species groups and land ownership in the Southern Appalachian Assessment area.

	National Parks		National Forests		Other Federal		State Lands		Private	
Species Group	#Sp.	#Oc.	#Sp.	#Oc.	#Sp.	#Oc.	#Sp.	#Oc.	#Sp.	#Oc.
Cave Habitats	1	6	3	15	0	0	2	7	5	101
Mountain Bogs	2	18	2	15	0	0	1	1	4	54
Fen/Pond Habitat	0	0	1	2	0	0	0	0	2	6
High Elevation Balds	2	6	4	14	0	0	1	1	3	37
Mafic Habitats	0	0	2	17	1	2	0	0	5	60
Rock Outcrop/ Cliff Habitats	3	16	4	50	0	0	2	3	6	47
Wide Ranging Species	2	5	1	1	1	1	1	3	0	0
Seeps, Springs, Streamside Habitat	2	3	4	15	1	1	2	11	10	115
Spruce-Fir Forests	2	15	2	11	0	0	2	2	4	13
Southern Yellow Pine Forest	1	7	1	1	0	0	1	1	1	15
Mixed Mesic Forests	2	14	2	13	0	0	1	8	5	55
Total	17	90	26	154	3	4	13	37	45	503

#Sp. = number of species
#Oc. = number of occurrences

national forests within the SAA area varies from a low of one species for the Talladega to a high of 13 species for the Pisgah-Nantahala (table 3.10). Occurrence records are highest for the Pisgah-Nantahala (54 records), followed by the George Washington with 41 records (table 3.10).

In comparing distribution of species by land ownership and species group, the number of T&E terrestrial species is highest within the private ownership category for species groups 1, 2, 4, 6, 7, 11, 15, and 18 (table 3.11). These groups correspond to cave; mountain bog; fen or pond wetland; high pH or mafic; rock outcrop and cliff; seeps, springs, and streamside; high-elevation spruce-fir; and mixed mesic species groups. National forest lands have more species (four) in species group 5 (high-elevation bald) than any other landowner. For the most part, ownerships with the highest number of species for a particular species group also have the most occurrences (table 3.11).

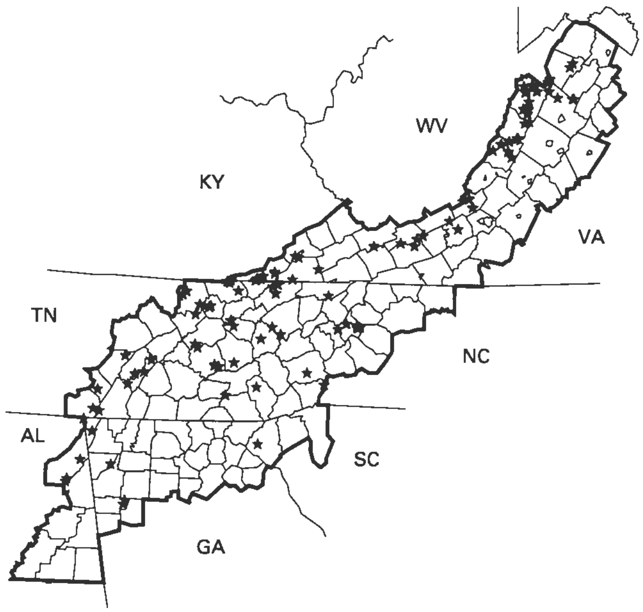


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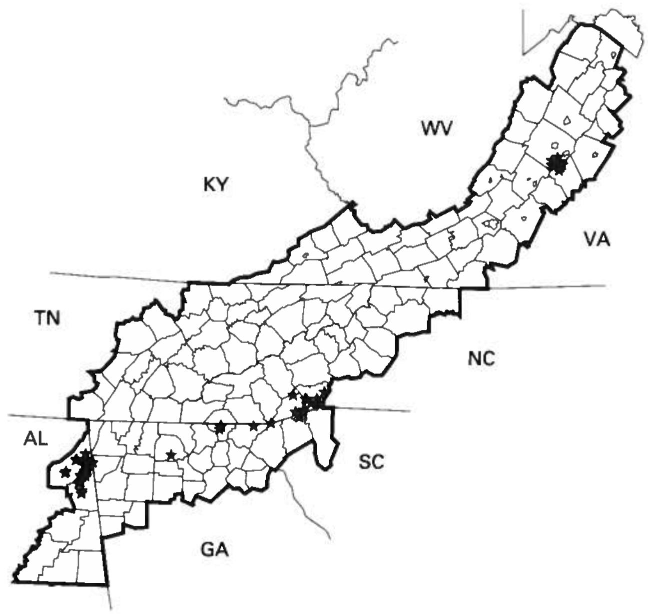
Figure 3.18 The spatial distribution for the number of occurrences for federally listed threatened and endangered terrestrial species associated with cave habitats.



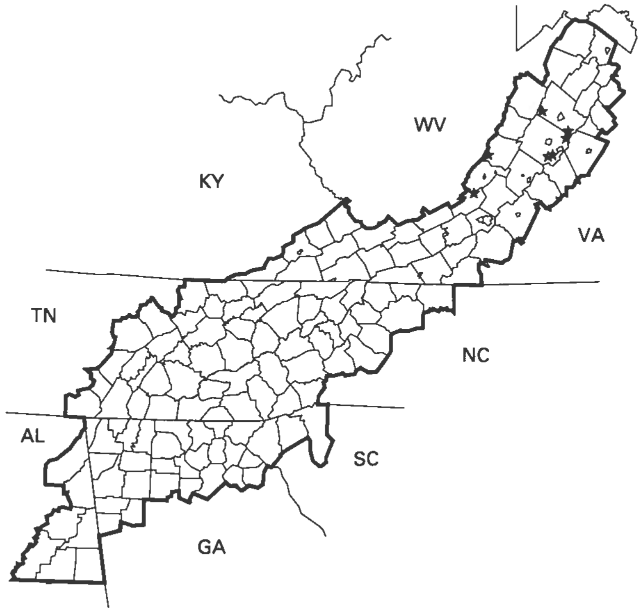
pa302

Figure 3.19 The spatial distribution for the number of occurrences for federally listed threatened and endangered terrestrial species associated with mountain bog habitats.

Current Status by Species Group

Occurrence records provided by the state heritage programs in 1995 for T&E terrestrial species were used to generate figures 3.18 through 3.27. Each figure represents the location records for a particular species group. Eight of the 19 species groups were not regarded as the primary group of any T&E species. Therefore, there are no species in these eight groups, and the number of occurrences from the state heritage databases are reported as zero. These species groups are: early successional grass-shrub, mid- to late-successional deciduous forest, habitat generalist, area-sensitive mid- to late-successional, general high-elevation, bottomland hardwood, and mixed xeric. The seeps, springs, and streamside species group has the highest number of species (nine plant, two animal). Of the remaining species groups, eight groups have from one to five species, and two groups (high pH or mafic species, figure 3.22; and rock outcrop and cliff species, figure 3.23) have six species. For those species groups that contain T&E species, the number of occurrences ranges from 8 (group 4, fen or pond wetland species) to 145 (group 11, seeps, springs, and streamside species). In general, as the number of species in a group increases, so does the number of occurrence records.

In distribution of occurrence records within the SAA area, locations for two species groups (cave; and seeps, springs, and streamsides; figures 3.18 and 3.25, respectively) are more or less evenly distributed throughout the assessment area. High-elevation bald species (fig. 3.21) and high-elevation spruce-fir species (fig. 3.26) are concentrated in North Carolina and eastern Tennessee. Most observations of high pH or mafic species (fig. 3.22) and rock outcrop species (fig. 3.23) were in Virginia and either South or North Carolina. The limited number of observations of wide-ranging area-sensitive species (fig. 3.24) was primarily in North and South Carolina. Fen or pond wetland species (fig. 3.20) were noted in the northern part of the assessment area in Virginia. All observations in the southern yellow pine species group were in Tennessee. However, there were observations of red-cockaded woodpeckers in Alabama (Talladega and Shoal Creek Ranger Districts) that did not appear in the database. Mountain bog species



pa304

Figure 3.20 The spatial distribution for the number of occurrences for federally listed threatened and endangered terrestrial species associated with fen or pond wetland habitats.

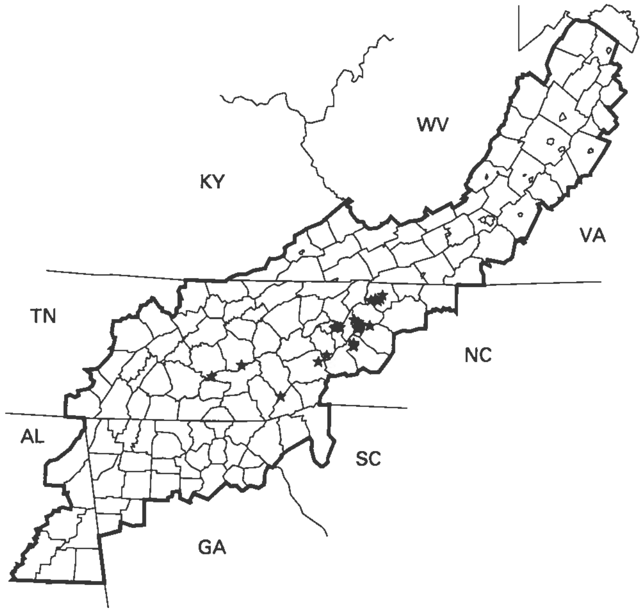


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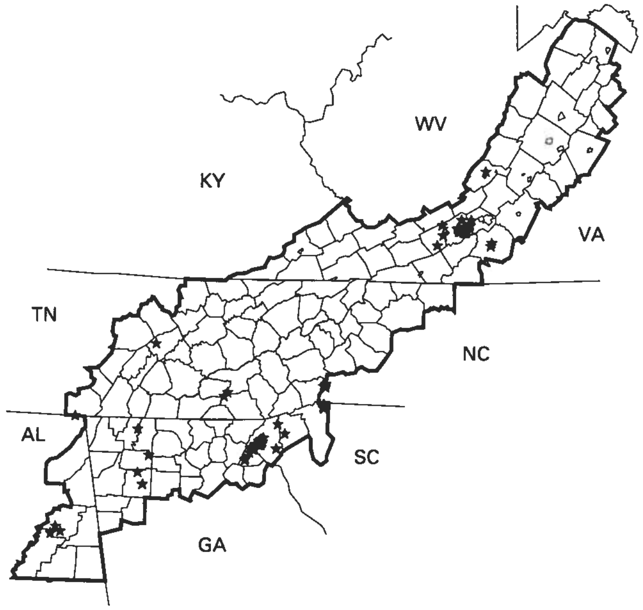
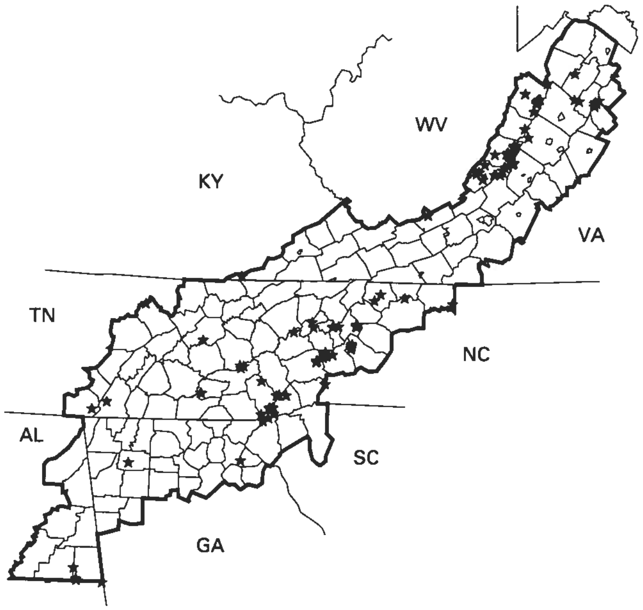
Figure 3.21 The spatial distribution for the number of occurrences for federally listed threatened and endangered terrestrial species associated with high-elevation bald/early successional habitats.



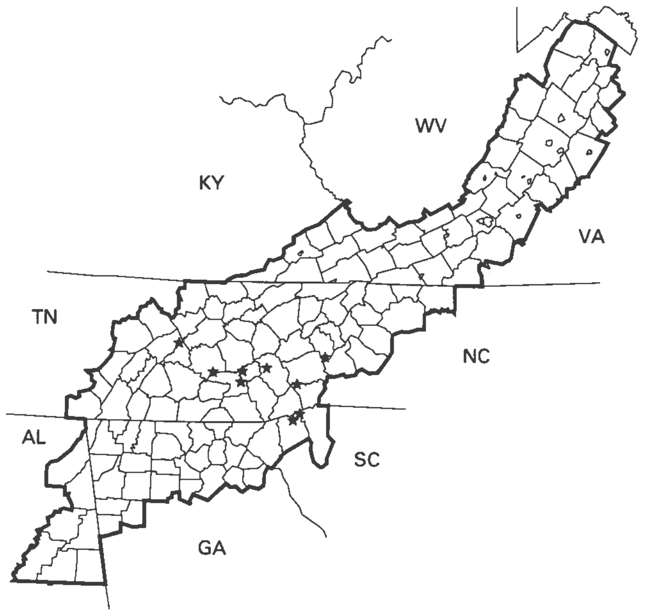
pa306

Figure 3.22 The spatial distribution for the number of occurrences for federally listed threatened and endangered terrestrial species associated with high pH or mafic habitats.



pa307

Figure 3.23 The spatial distribution for the number of occurrences for federally listed threatened and endangered terrestrial species associated with rock outcrop and cliff habitats.



pa309

Figure 3.24 The spatial distribution for the number of occurrences for federally listed threatened and endangered terrestrial wide ranging area sensitive species.

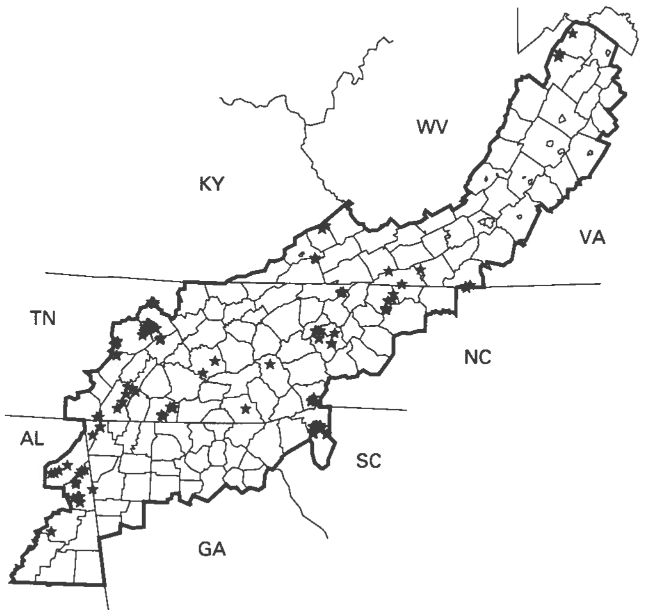


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Figure 3.25 The spatial distribution for the number of occurrences for federally listed threatened and endangered terrestrial species associated with seeps, springs, and streamside habitats.

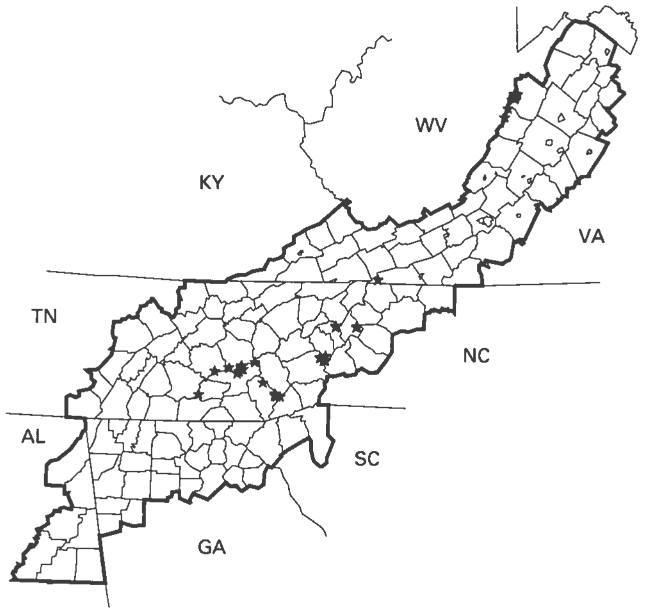

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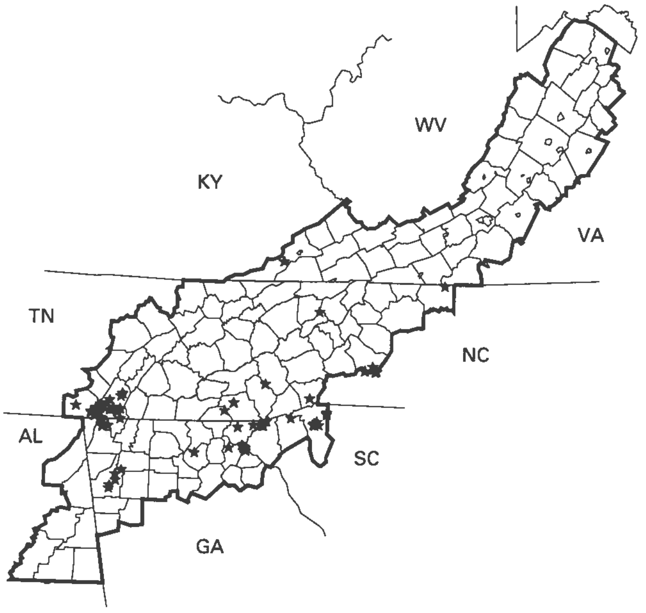
Figure 3.26 The spatial distribution for the number of occurrences for federally listed threatened and endangered terrestrial species associated with high-elevation spruce-fir/northern hardwood forest habitats.



pa318

Figure 3.27 The spatial distribution for the number of occurrences for federally listed threatened and endangered terrestrial species associated with mixed mesic forest habitats.

(fig. 3.19) are heavily concentrated in small areas within Virginia, North Carolina, and Alabama. Members of the mesic species group (fig. 3.27) are somewhat concentrated along the Tennessee/Georgia and South Carolina/Georgia borders.

Terrestrial Species with Viability Concern

Species whose viability is of concern (VC) were defined as globally ranked 1, 2, or 3 and federal category 1 or 2 terrestrial plant and animal species. Spatial and quantitative information for these species were obtained from state heritage biological and conservation data, state heritage sitebasic records, and FS occurrence records. The list of 366 viability concern species occurring in the SAA area was compiled from information supplied by the FWS, the state natural heritage programs, and peer review of the initial species lists. Habitat relationships were determined for all but 30 plant species. The species/habitat associations for all species received peer review. Much of the information on species/habitat relationships is based on expert opinion.

Based on the analysis of species/habitat relationships, about two-thirds of these species are associated with rare communities. This proportion rises to 74 percent when riparian habitats are included. These species, for the most part, are not amenable for broad-scale analysis of habitat suitability. The analysis of current status focused primarily on the spatial occurrences. These occurrences were based on EOR obtained from the seven state natural heritage programs in the SAA. Analysis of both occurrence data and habitat suitability (Chapter 3, Habitat Suitability section) was provided for the remaining species.

Current Status – Total SAA

Currently, there are occurrence records for 318 plant and animal VC species in the SAA area. Of these species, 156 are animals and 162 are plants. Twelve counties in the SAA area had no records of VC species (neither plant nor animal); 79 counties had 1 to 10 species; 29 counties had 11 to 20 species; 12 counties had 21 to 30 species; and 3 counties had more than 31 species (figs. 3.28, 3.29, 3.30). It should be noted that of the 156 terrestrial animal species,

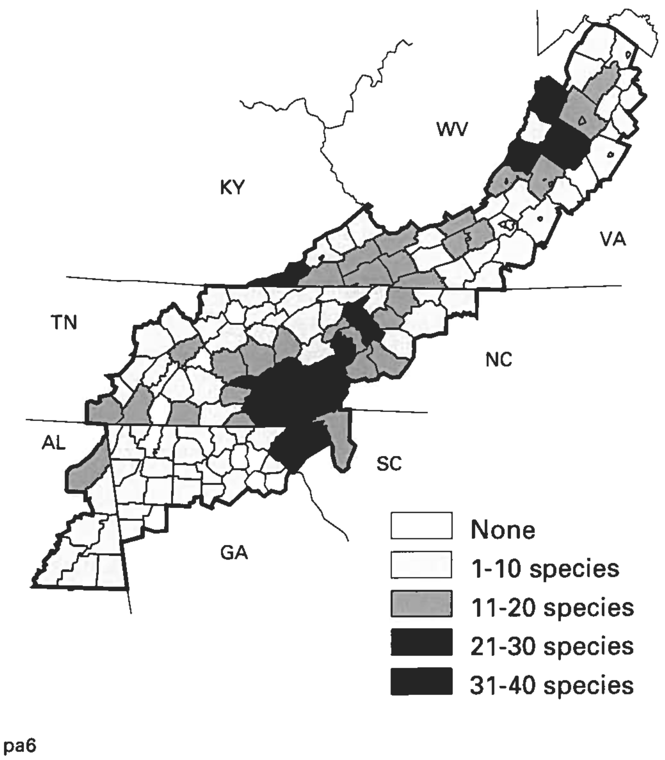


Figure 3.28 The spatial distribution for the number of terrestrial species with viability concern by county in the SAA area.

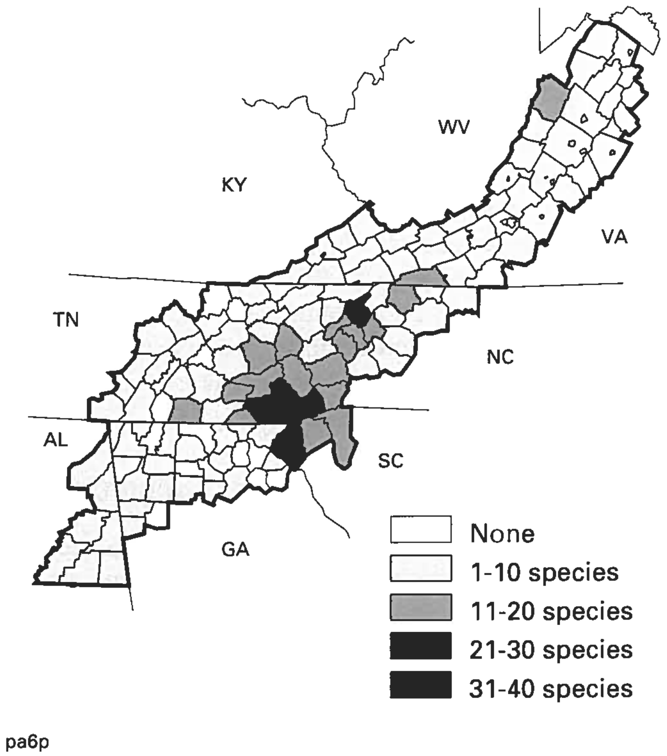


Figure 3.29 The spatial distribution for the number of terrestrial plant species with viability concern by county in the SAA area.

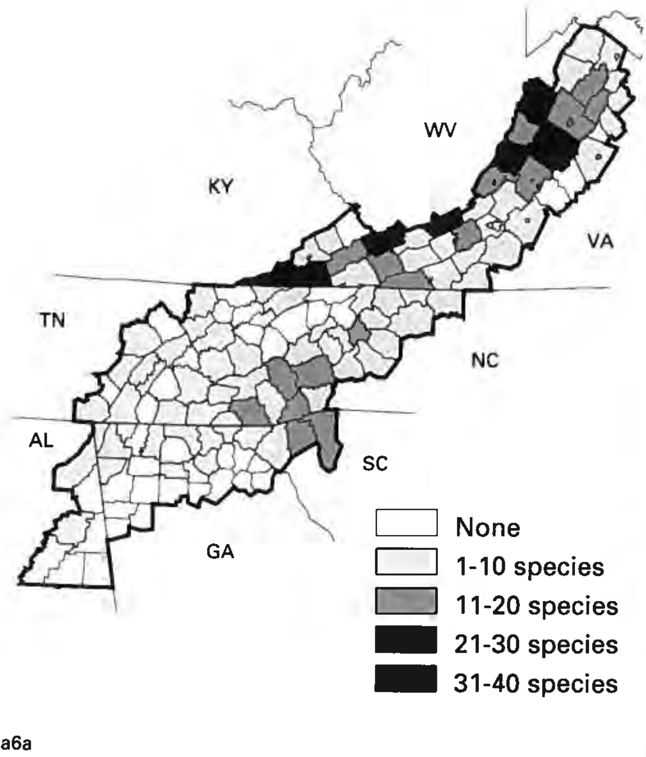


Figure 3.30 The spatial distribution for the number of terrestrial animal species with viability concern by county in the SAA area.

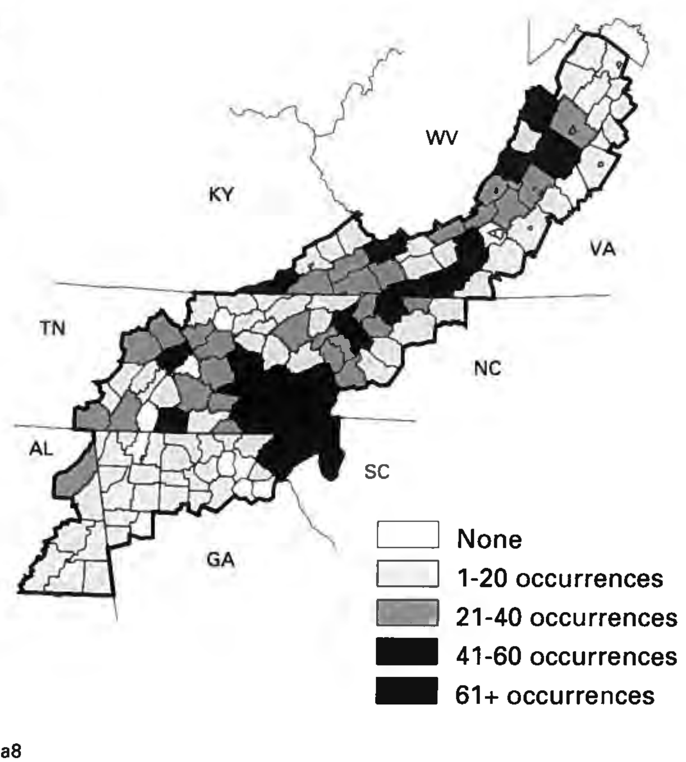


Figure 3.31 The spatial distribution for the number of occurrences of terrestrial species with viability concern by county in the SAA area.

110 are invertebrates. Although some counties show no species, this could change as databases are updated, records are verified, and surveys are implemented.

Of the 318 total plant and animal species, there are 3,243 occurrences of terrestrial plant and animal VC species in the SAA area. Of these occurrences, 2,335 are plants and 908 are animals. Seventy-one counties had 1 to 20 occurrences; 29 counties had 21 to 40 occurrences; 10 counties had 41 to 60 occurrences; and 13 counties had more than 60 occurrences (figs. 3.31, 3.32, 3.33).

Current Status by Species Group (SG)

Species with viability concern as they occur by groups based on habitat association, are summarized in table 3.12 and shown spatially on figures 3.34 through 3.49. There were no wide-ranging sensitive species (SG 9) or southern yellow pine forest species (SG 17). The highest number of species and occurrences, both plant and animal, is in the cave and mixed mesophytic groups. The lowest number of species and occurrences, both plant and animal, is in the habitat generalist and bottomland forest groups (table F-1).

Current Status by Ecological Unit

The number of terrestrial species with viability concern by ecological section and subsection is shown in table 3.13. The highest number of both plant and animal species is in the Blue Ridge Mountains and Northern Ridge and Valley, respectively. The lowest number of species is in the Allegheny Mountains. The number of occurrences ranges from a high of 1,929 (447 animal, 1,482 plant) in Blue Ridge Mountains to a low of 16 occurrences (11 animal, 5 plant) in the Allegheny Mountains (table 3.13).

Current Status by Ownership

Private lands contain around 56 percent of the occurrences for VC species, followed by national forest lands with 29 percent, national park lands with 10 percent, state lands with 3 percent, and other federal lands with 2 percent (table 3.14). The same general patterns follow for viability concern plants and animals. The

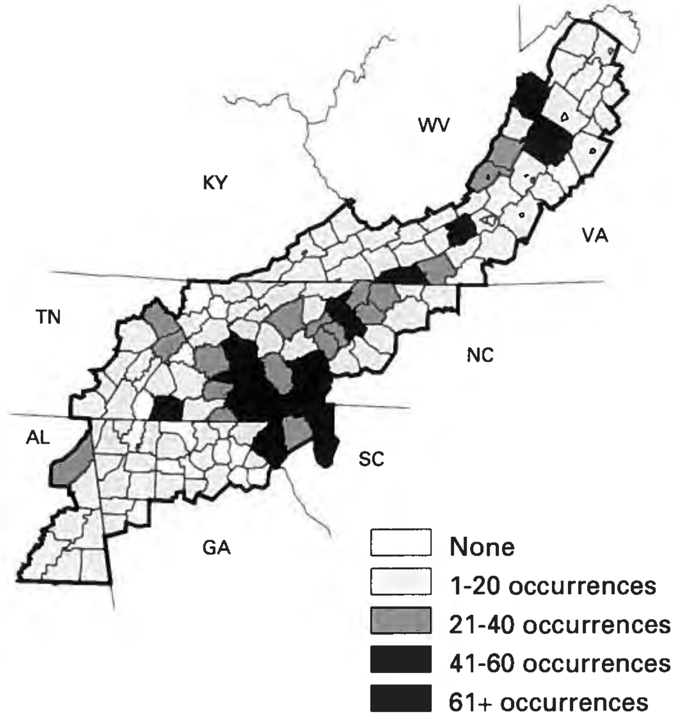


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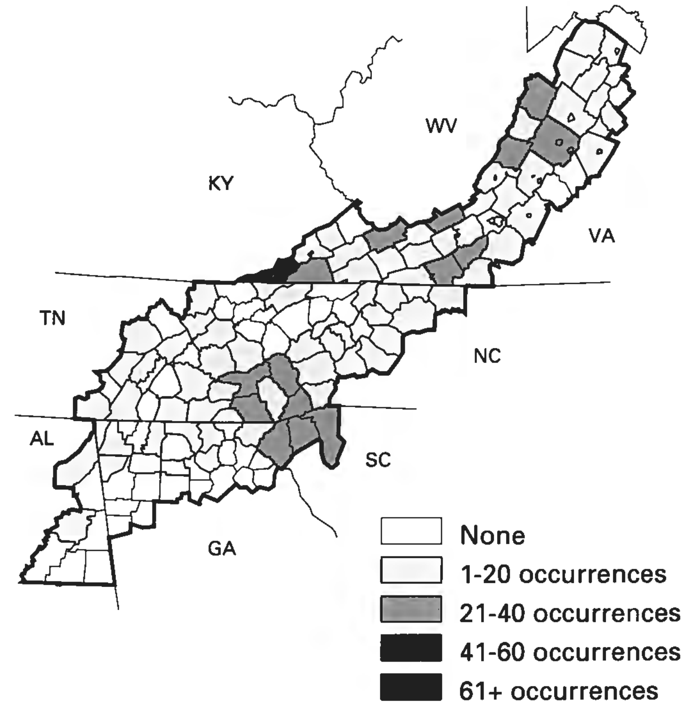
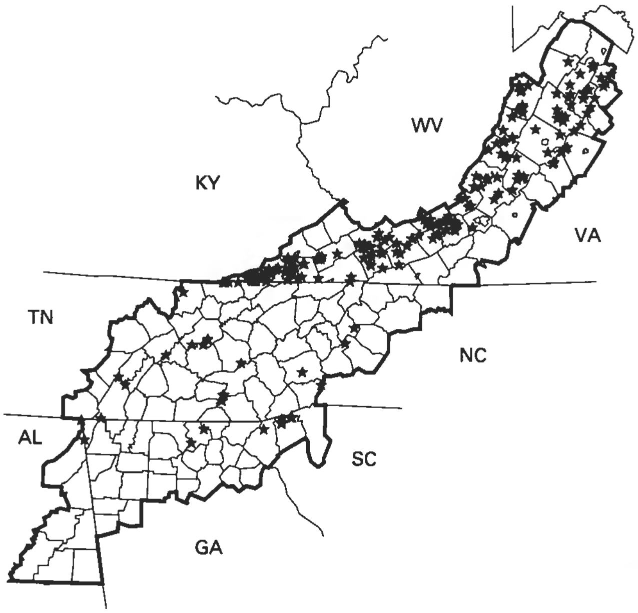
Figure 3.32 The spatial distribution for the number of occurrences of terrestrial plant species with viability concern by county in the SAA area.



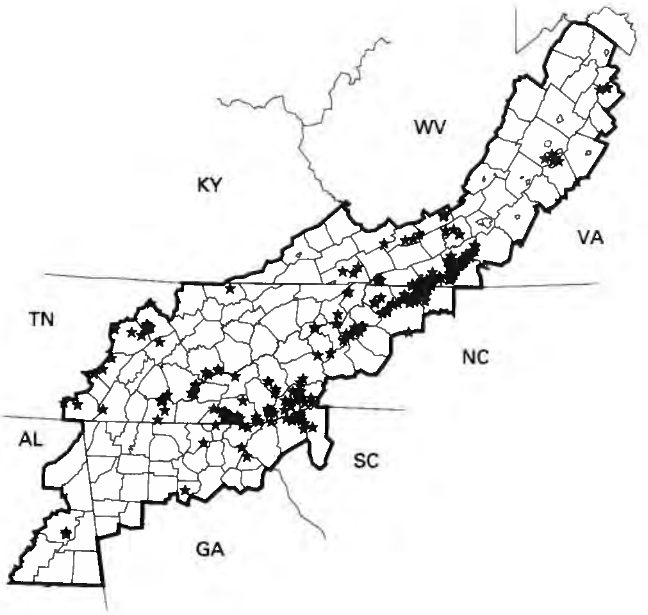
pa8a

Figure 3.33 The spatial distribution for the number of occurrences of terrestrial animal species with viability concern by county in the SAA area.



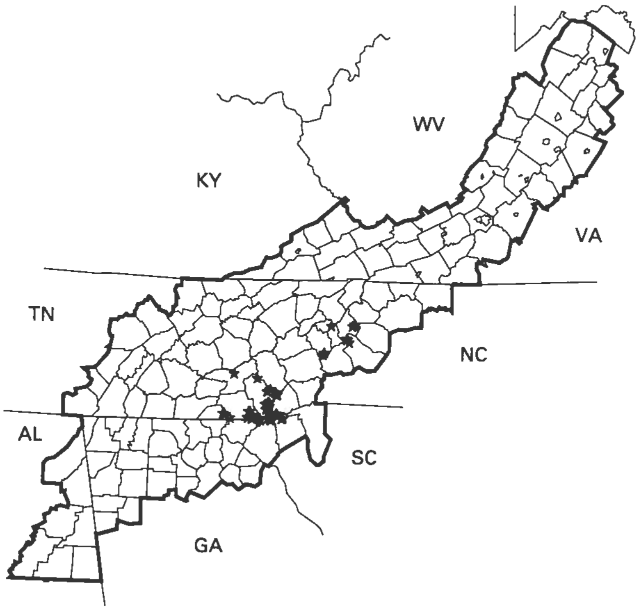
pa401

Figure 3.34 The spatial distribution for the number of occurrences for terrestrial species with viability concern associated with cave habitats in the SAA.



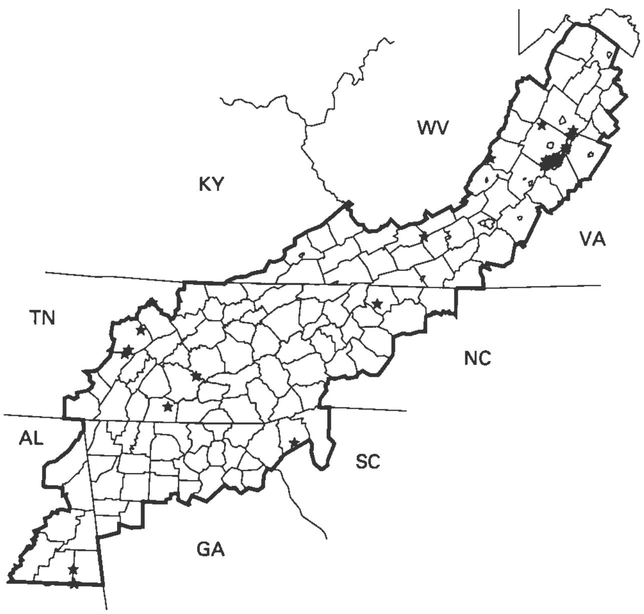
pa402

Figure 3.35 The spatial distribution for the number of occurrences for terrestrial species with viability concern associated with mountain bog habitats in the SAA.



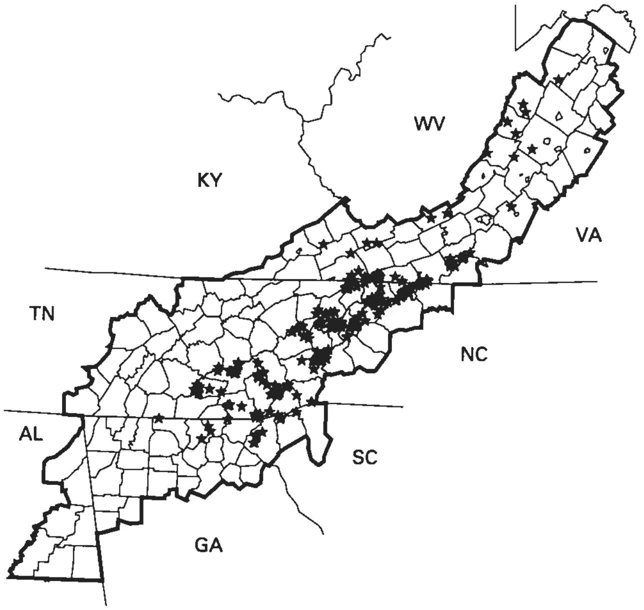
pa403

Figure 3.36 The spatial distribution for the number of occurrences for terrestrial species with viability concern associated with spray cliff habitats in the SAA.



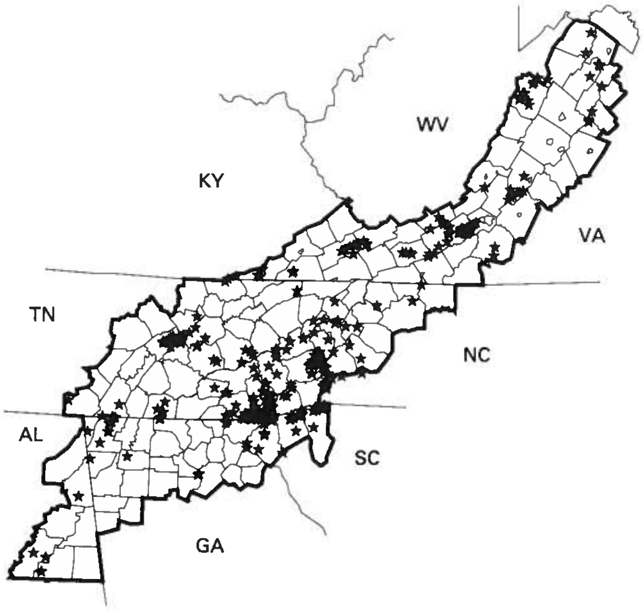
pa404

Figure 3.37 The spatial distribution for the number of occurrences for terrestrial species with viability concern associated with fen or pond wetland habitats in the SAA.



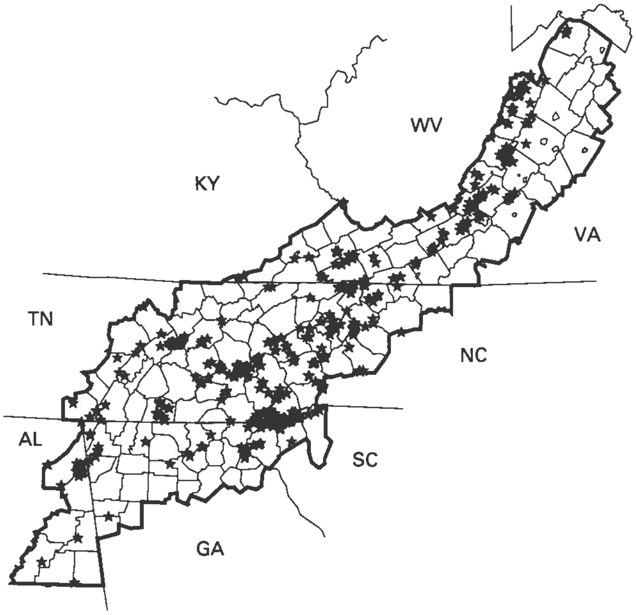
pa405

Figure 3.38 The spatial distribution for the number of occurrences for terrestrial species with viability concern associated with high-elevation bald/early successional habitats in the SAA.



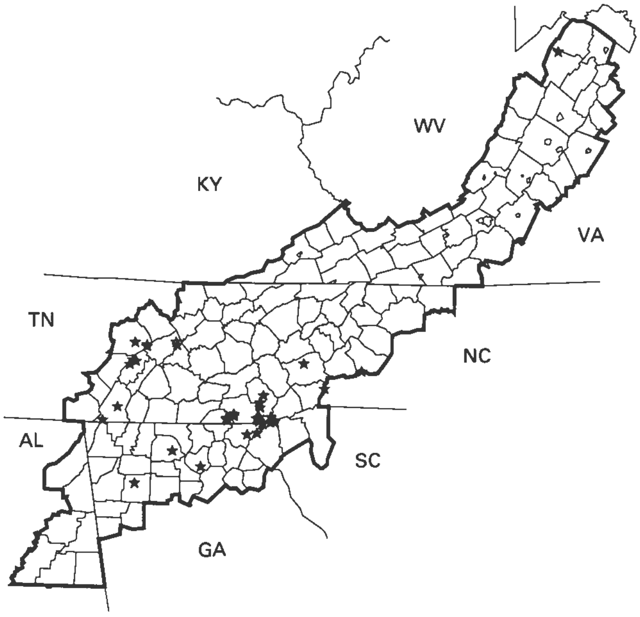
pa406

Figure 3.39 The spatial distribution for the number of occurrences for terrestrial species with viability concern associated with high pH or mafic habitats in the SAA.



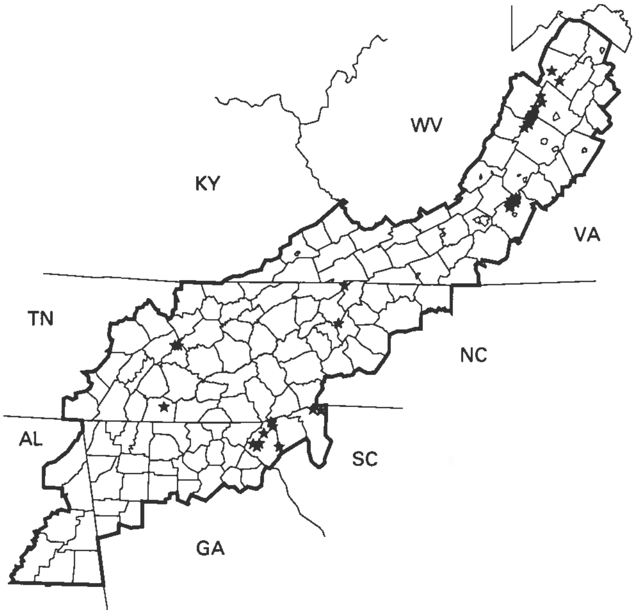
pa407

Figure 3.40 The spatial distribution for the number of occurrences for terrestrial species with viability concern associated with rock outcrop and cliff habitats in the SAA.



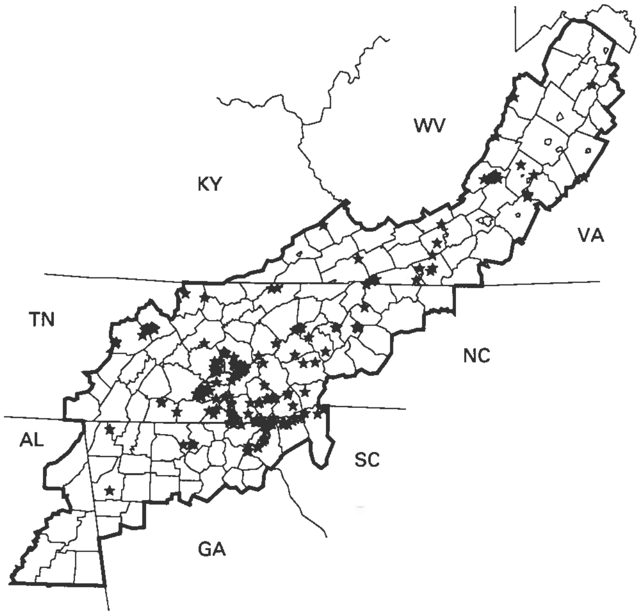
pa408

Figure 3.41 The spatial distribution for the number of occurrences for terrestrial species with viability concern associated with early successional/grass-shrub habitats in the SAA.



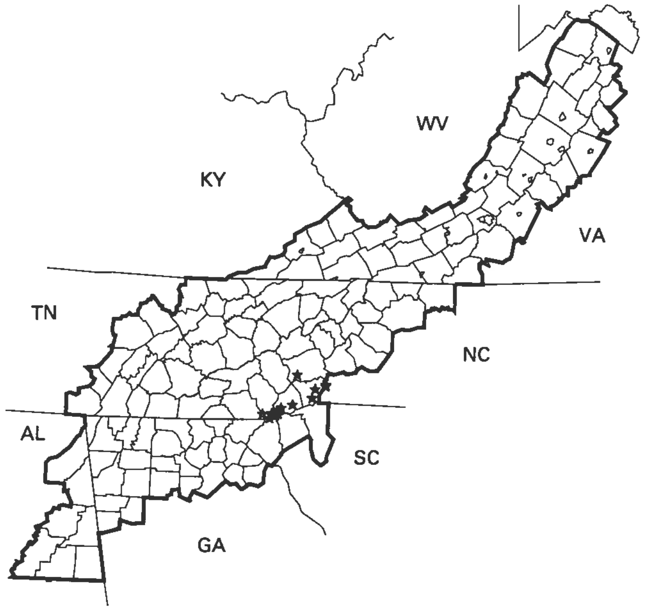
pa410

Figure 3.42 The spatial distribution for the number of occurrences for terrestrial species with viability concern associated with mid- to late-successional forest habitats in the SAA.



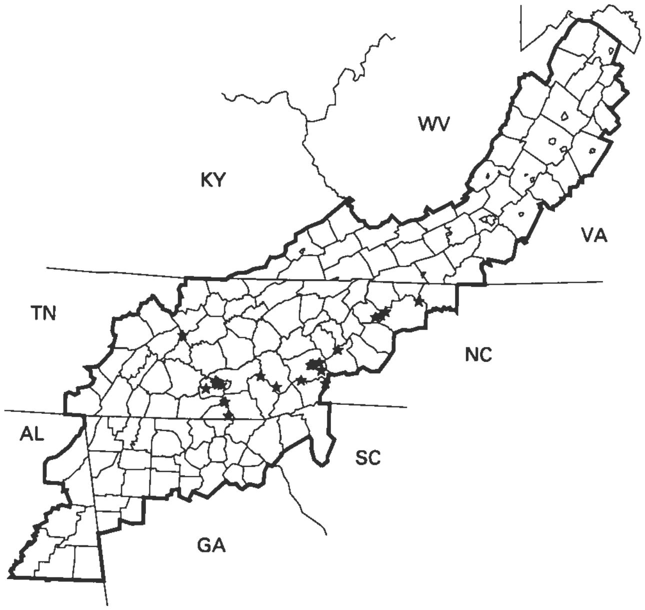
pa411

Figure 3.43 The spatial distribution for the number of occurrences for terrestrial species with viability concern associated with seeps, springs, and streamside habitats in the SAA.



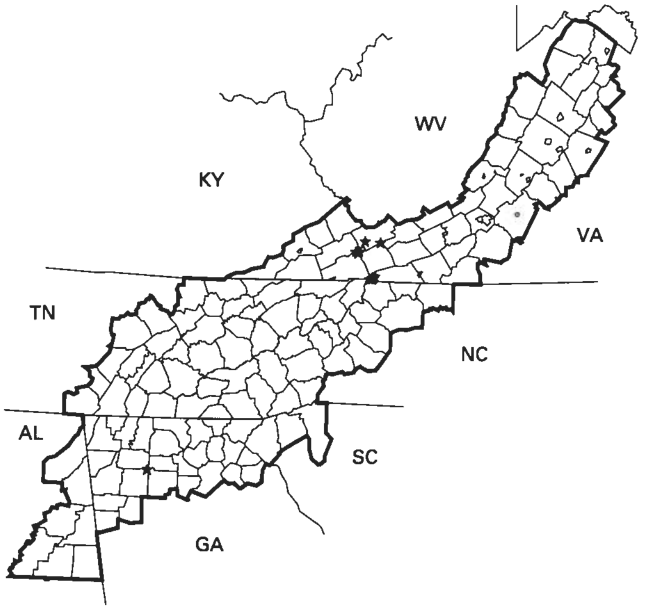
pa412

Figure 3.44 The spatial distribution for the number of occurrences for terrestrial species with viability concern considered habitat generalist in the SAA.



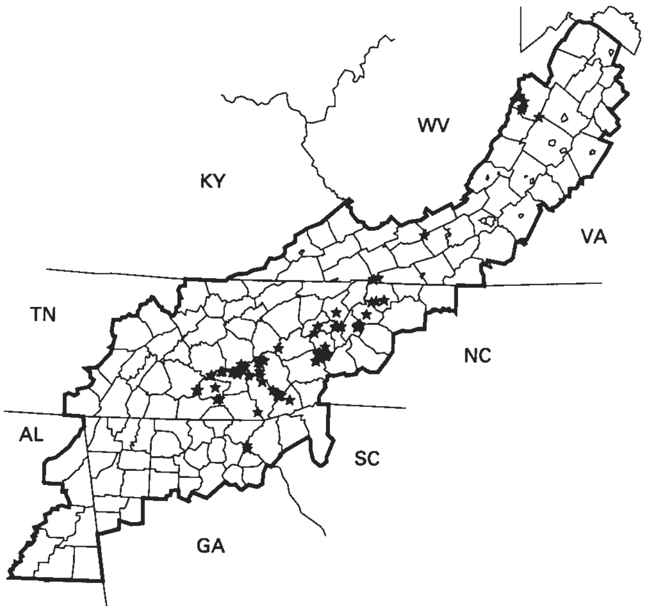
pa413

Figure 3.45 The spatial distribution for the number of occurrences for terrestrial species with viability concern associated with mid- to late-deciduous forest and considered to have area size requirements in the SAA.



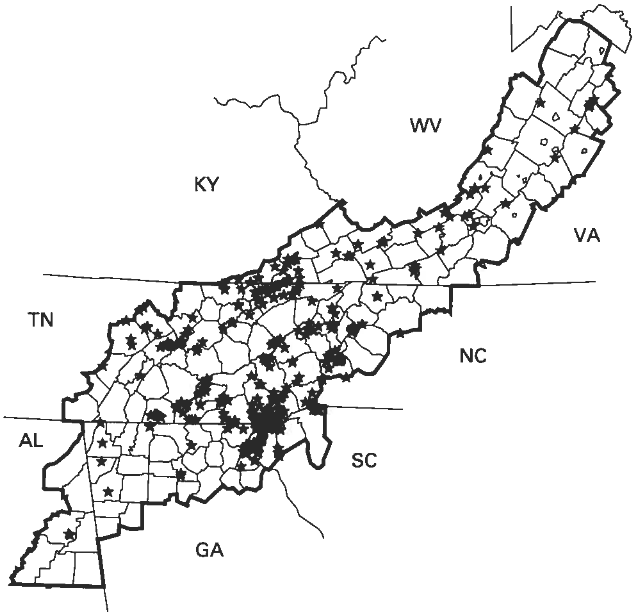
pa414

Figure 3.46 The spatial distribution for the number of occurrences for terrestrial species with viability concern associated with general high-elevation forest habitats in the SAA.



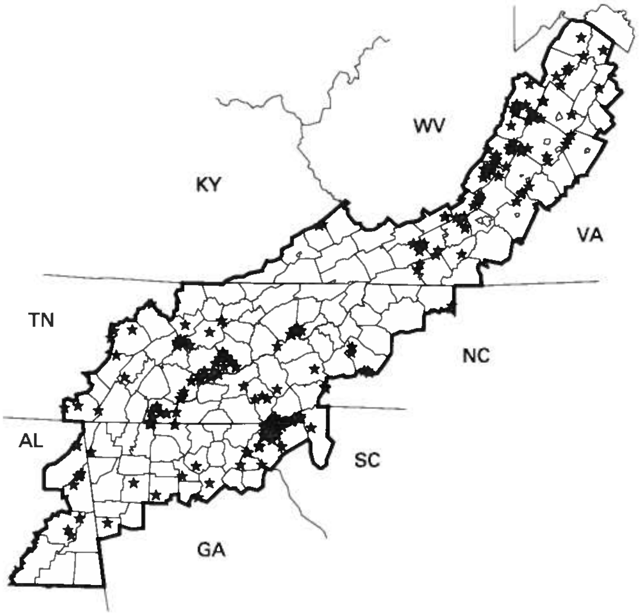
pa415

Figure 3.47 The spatial distribution for the number of occurrences for terrestrial species with viability concern associated with high-elevation spruce-fir/northern hardwood habitats in the SAA.



pa418

Figure 3.48 The spatial distribution for the number of occurrences for terrestrial species with viability concern associated with mixed mesic forest habitats in the SAA.



pa419

Figure 3.49 The spatial distribution for the number of occurrences for terrestrial species with viability concern associated with mixed xeric forest habitats in the SAA.

Table 3.12 The number of species and occurrences by species group for terrestrial species with viability concern in the Southern Appalachian Assessment area.

	Number of Species			Number of Occurences		
Species Group	Animal	Plant	Total	Animal	Plant	Total
Cave	110	0	110	360	0	360
Mountain Bog	1	12	13	143	167	310
Spray Cliffs	0	16	16	0	88	88
Fen or Pond Wetland	0	5	5	0	46	46
High Elevation Balds and Rock Summits	3	7	10	64	233	297
High ph or Mafic	0	25	25	0	371	371
Rock Outcrop and Cliffs	5	24	29	137	376	513
Early Successional Grass/Shrub	2	1	3	13	28	41
Mid to Late Successional Deciduous Forests	2	1	3	29	15	44
Seeps, Springs, and Streamside	3	15	18	50	163	213
Habitat Generalists	0	1	1	0	14	14
Area Sensitive Mid to Late Deciduous Forest	1	0	1	23	0	23
General High Elevation Forest	0	1	1	0	15	15
High Elevation Spruce-Fir Forests	2	9	11	3	99	102
Bottomland Hardwood	0	1	1	0	1	1
Mixed Mesic Forest	16	21	37	32	420	452
Mixed Xeric Forest	2	10	12	31	204	235

Table 3.13 The number of species with viability concern and the number of occurrences by ecological section and subsection in the Southern Appalachian Assessment area.

		Number of Species			Number of Occurrences		
Sections	Subsection	Animals	Plants	Total	Animals	Plants	Total
221H		6	19	25	19	85	104
	221Hc	3	19	22	14	82	96
	221Hd	2	0	2	3	0	3
	221He	2	3	5	2	3	5
221I		13	6	19	21	16	37
	221Ib	13	6	19	21	16	37
221J		24	19	43	80	132	212
	221Ja	24	17	41	76	112	188
	221Jb	2	6	8	4	15	19
	221Jc	0	2	2	0	5	5
231A		4	36	40	19	88	107
	231Aa	2	13	15	4	18	22
	231Ab	0	3	3	0	3	3
	231Ac	0	7	7	0	10	10
	231Ad	3	18	21	15	47	62
	231Ag	0	5	5	0	7	7
	231Ai	0	0	0	0	0	0
	231Ak	0	3	3	0	3	3
231C		5	17	22	12	53	65
	231Cc	2	15	17	4	50	54
	231Cf	4	3	7	8	3	11
231D		6	27	33	13	59	72
	231Da	4	16	20	9	28	37
	231Db	2	8	10	4	9	13
	231Dc	0	3	3	0	3	3
	231Dd	0	5	5	0	8	8
	231De	0	7	7	0	11	11
M221A		97	42	139	271	314	585
	M221Aa	70	34	104	192	229	421
	M221Ab	40	19	59	77	83	160
	M221Ac	2	2	4	2	2	4
M221B		7	3	10	11	5	16
	M221Ba	6	2	8	9	4	13
	M221Bd	2	1	3	2	1	3
M221D		36	122	158	447	1,482	1,929
	M221Da	9	14	23	36	30	66
	M221Db	6	15	21	94	96	190
	M221Dc	22	106	128	248	1,002	1,250
	M221Dd	12	52	64	69	354	423

Table 3.14 The number of terrestrial species with viability and occurrences by land ownership in the Southern Appalachian Assessment area.

	Species with Viability Concerns	
Ownership Category	# Species	# Occurrences
National Forests		
Chattahoochee	15	28
Cherokee	34	200
George Washington	32	78
Jefferson	33	80
Monongahela	10	16
Nantahala/Pisgah	78	417
Sumter	20	128
Talladega	3	5
National Parks	74	315
Private	278	1,802
Other Federal	16	53
Cherokee Indian Reservation	8	8
State	54	113

Table 3.15 The number of species and occurrences for species with viability concern by species group and land ownership in the Southern Appalachian Assessment area.

	Land Ownership									
	National Parks		National Forests		Other Federal		State Lands		Private	
Species Group	#Sp.	#Oc.	#Sp.	#Oc.	#Sp.	#Oc.	#Sp.	#Oc.	#Sp.	#Oc.
Cave Habitats	6	14	15	26	0	0	2	2	105	318
Mountain Bogs	6	35	8	49	1	2	6	11	13	213
Spray Cliffs	3	6	14	45	2	2	1	1	11	34
Fen/Pond Habitat	0	0	2	5	0	0	1	1	5	40
High Elevation Balds	7	37	19	119	2	2	4	10	9	129
Mafic Habitats	7	24	12	94	3	19	8	12	24	222
Rock Outcrop/Cliff Habitats	9	65	19	128	2	14	10	31	27	275
Early Succession.Habitat	0	0	1	16	1	2	0	0	3	23
Mid- to Late-Successional Deciduous Forest	1	3	3	29	1	1	0	0	2	11
Seeps, Springs, Streamside Habitat	8	36	12	82	1	1	4	8	16	92
Habitat Generalist	0	0	1	6	0	0	0	0	1	8
Area Sensitive Mid- to Late Deciduous Forest	1	3	1	9	1	2	0	0	1	9
General High Elevation	0	0	1	7	0	0	1	3	1	5
Spruce-Fir Forests	5	40	11	33	0	0	3	4	7	25
Mixed Mesic Forests	7	21	29	209	3	6	6	9	19	206
Mixed Xeric Forests	6	17	12	75	2	6	5	13	9	122

#Sp. = number of species
#Oc. = number of occurrences

Table 3.16 The density class definitions for 10 of the major game species in the Southern Appalachian Assessment area.

	Density Class		
Species	Low	Medium	High
White–tailed Deer	<15/square mile	15–30/square mile	>30/square mile
Eastern Wild Turkey	< 6/square mile	6–15/square mile	>15/square mile
Black Bear	<1/1,500 acres	1/1,500–1/1,000 acres	>1/1,000 acres
Gray Squirrel	<1/10 acres	1/10–1/3 acres	>1/3 acres
Fox Squirrel	<1/10 acres	1/10–1/3 acres	>1/3 acres
Eastern Cottontail	<1/20 acres	1/20–1/10 acres	>1/10 acres
Raccoon	<5/square mile	5–10/square mile	>10/square mile
Ruffed Grouse	<5/square mile	5–10/square mile	>10/square mile
Bobwhite Quail	<1/100 acres	1/100–1/10 acres	>1/10 acres
American Woodcock	<1/500 acres	1/500–1/100 acres	>1/100 acres

largest number of viability concern species associated with high-elevation habitats such as balds, montane spruce-fir, and general forest occur on national parks and national forests. Also relative to land area, public lands contain a high portion of occurrences for species associated with mid- to late-deciduous forests (including those needing large forest tracts), mesic forests, xeric forests, seeps and streamside habitats, early successional habitats, and spray cliff habitats (table 3.15).

Major Game Species

Estimates of current and historical (1970) population densities for 10 major game species were provided by state wildlife agencies and included in the assessment area. Because of the importance of acorns to numerous species of wildlife in the Southern Appalachians, oak mast capability was also estimated.

For each of the 10 species, state agency biologists were asked to classify each county by one of four density classes: absent, low, medium, or high. Specific population densities corresponding to each density class were provided to the state agency biologists and are shown in table 3.16. Although population densities often vary within a county, for purposes of this broadscale analysis, counties were classified by average density for the county as a whole. Density estimates were derived from harvest and survey data where available, as well as from professional judgment by the appropriate state agency biologists.

County estimates were stratified by ecological section group (table 3.17), state (table 3.18), and ownership (table 3.19). Where a boundary between two section groups fell within a county, the county was assigned to the section group comprising the largest proportion of the county. The ownership stratification was accomplished by overlaying county density maps with ownership coverage and determining the proportion of each ownership category in each of the four density classes.

To examine the relationship between current population density and land use for each species, land cover data from satellite imagery were stratified by county density class (table 3.20). In addition, satellite imagery and FIA data were stratified by section group, state, and ownership and compared to current density estimates. Trends in game population densities were compared with trends in land use derived from FIA and NRI land-use data.

Oak Mast Capability Estimates

Because of the importance of acorns to numerous species of wildlife in the Southern Appalachians, oak mast capability was estimated for each forest type by successional stage (mid-successional, late-successional) and section group. Data used in these calculations included: (1) tree counts by species and diameter class for each forest type and successional class and successional stage proportions for each forest type by section group derived from FIA statistics; (2) total acres for each forest type, total forest acres and total land area by section group from satellite imagery; and (3) acorn yield coefficients by oak species and diameter class found in the FS, Southern Region Wildlife Habitat Management Handbook (USDA FS 1980).

Oak mast capability for each forest type and successional stage was estimated by multiplying the number of trees of each species and diameter class by the appropriate acorn yield coefficient. To determine acorn yield for each section group, acres of each forest type by successional class were first calculated by multiplying total acres of each forest type from satellite imagery by the appropriate successional stage proportion from FIA data. These values were then multiplied by the appropriate oak mast capability coefficient to determine the total acorn yield for each forest type-successional stage combination. These values were summed to determine the total oak mast capability for each section group. Then, these values were divided by the acres of forestland and total land area for each section group to determine acorn capability in pounds per acre of forestland area and pounds per acre of total section area, respectively.

Estimated oak mast capability was highest in the Blue Ridge Mountains (section group 2) and in the Northern Ridge and Valley, Allegheny Mountains and Northern Cumberland Plateau (section group 1) (table 3.21). Because of the low proportion of the region in acorn-bearing forest types and the low proportion in mid- to late-successional stages, estimated acorn capability was lowest in the Southern Cumberland Plateau (section group 5).

White-tailed Deer

White-tailed deer are present throughout the assessment area (fig. 3.50). Population densities generally are medium to high in the Northern Ridge and Valley, Allegheny Mountains and Northern Cumberland Mountains (section group 1) and the Southern Appalachian Piedmont (section group 6). Densities generally are low to medium in the remainder of the assessment area. High deer densities are associated with greater amounts of cropland and lesser amounts of developed and coniferous forestland (table 3.20). Current densities generally are higher on private land, national forest, and state lands than on the remaining ownerships (table 3.19).

Although deer were present in essentially all portions of the assessment area in 1970, densities have greatly increased in the last 25 years. In 1970, approximately 70 percent of counties had a low deer density and none had a high deer density. Today nearly 70 percent of counties has a medium to high density of deer. This pattern of increase generally is consistent throughout the assessment area and within ownerships. This increase probably is related to both nonhabitat factors such as extensive restoration efforts, protection, and conservative

Table 3.17 The trends in the proportion of counties in each density class by section group for the 10 major game species in the Southern Appalachian Assessment area.

	Section Group 1[1] Density Classes[2]				Section Group 2[1] Density Classes[2]				Section Group 3[1] Density Classes[2]				Section Group 4[1] Density Classes[2]			
	A	L	M	H	A	L	M	H	A	L	M	H	A	L	M	H
White-tailed Deer																
1970	0	34	65	0	0	89	11	0	0	100	0	0	0	100	0	0
1995	0	10	69	21	0	59	38	3	0	63	38	0	0	44	50	6
Wild Turkey																
1970	3	48	45	3	46	51	3	0	63	25	0	12	83	17	0	0
1995	0	28	55	17	0	46	49	5	0	38	63	0	0	56	39	6
Black Bear																
1970	17	72	10	0	19	78	3	0	100	0	0	0	89	11	0	0
1995	0	79	21	0	0	46	46	8	88	13	0	0	83	6	6	6
Gray Squirrel																
1970	0	0	100	0	0	0	95	5	0	0	100	0	0	0	100	0
1995	0	0	100	0	0	0	92	8	0	0	100	0	0	11	89	0
Fox Squirrel																
1970	7	90	3	0	62	38	0	0	13	50	38	0	0	67	11	22
1995	7	83	10	0	62	38	0	0	13	50	38	0	0	67	11	22
Cottontail																
1970	0	34	24	41	0	32	54	14	0	38	63	0	0	28	44	28
1995	0	52	7	41	0	35	51	14	0	63	38	0	0	44	50	6
Raccoon																
1970	0	55	45	0	0	84	16	0	0	100	0	0	0	67	33	0
1995	0	10	83	7	0	0	100	0	0	13	88	0	0	6	67	28
Ruffed Grouse																
1970	0	24	21	55	0	11	67	22	0	25	50	25	0	39	39	22
1995	0	34	58	7	0	8	83	8	0	25	63	13	0	39	44	17
Bobwhite Quail																
1970	0	52	48	0	0	83	8	8	0	75	25	0	0	33	67	0
1995	0	100	0	0	0	95	5	0	0	75	25	0	0	50	50	0
Woodcock																
1970	0	100	0	0	3	89	8	0	0	88	13	0	0	83	16	0
1995	0	100	0	0	3	97	0	0	0	88	13	0	0	83	16	0

[1] Section Groups:
1 = Northern Ridge and Valley, Allegheny Mountains, Northern Cumberland Mountains (n = 29)
2 = Blue Ridge Mountains (n = 37)
3 = Northern Cumberland Plateau, Southern Cumberland Mountains (n = 8)
4 = Central Ridge and Valley (n = 18)

Table 3.17 (cont.) The trends in the proportion of counties in each density class by section group for the 10 major game species in the Southern Appalachian Assessment area.

	Section Group 5[1]				Section Group 6[1]				SAA Area			
	Density Classes[2]				Density Classes[2]				Density Classes[2]			
	A	L	M	H	A	L	M	H	A	L	M	H
White-tailed Deer												
1970	0	53	47	0	4	57	39	0	1	69	30	0
1995	0	20	80	0	0	11	68	21	0	33	57	10
Wild Turkey												
1970	47	53	0	0	43	50	7	0	42	44	12	1
1995	0	20	80	0	0	7	82	11	0	32	60	8
Black Bear												
1970	80	20	0	0	46	50	4	0	45	51	4	0
1995	73	27	0	0	32	60	7	0	31	47	19	3
Gray Squirrel												
1970	0	20	73	7	0	14	82	4	0	5	92	3
1995	0	13	80	7	0	18	71	11	0	7	88	5
Fox Squirrel												
1970	0	80	20	0	64	36	0	0	33	58	7	3
1995	0	100	0	0	61	39	0	0	32	59	6	3
Cottontail												
1970	0	13	67	20	0	21	43	36	0	28	46	26
1995	0	7	93	0	0	43	43	14	0	40	44	16
Raccoon												
1970	0	0	100	0	0	36	64	0	0	57	43	0
1995	0	0	100	0	0	0	100	0	0	4	91	5
Ruffed Grouse												
1970	67	33	0	0	25	64	11	0	13	32	33	22
1995	60	40	0	0	25	61	14	0	12	33	48	7
Bobwhite Quail												
1970	0	7	87	7	0	21	68	11	0	48	47	5
1995	0	27	73	0	0	54	46	0	0	72	28	0
Woodcock												
1970	7	33	60	0	0	57	43	0	1	78	21	0
1995	20	30	60	0	0	79	21	0	3	83	14	0

[1] Section Groups:
5 = Southern Cumberland Plateau, Southern Ridge and Valley
6 = Southern Appalachian Piedmont

[2] Density Classes:
A = Absent
L = Low
M = Medium
H = High

Table 3.18 The trends in the proportion of counties in each density class by state for the 10 major game species in the Southern Appalachian Assessment area.

	Alabama[1]				Georgia[1]				North Carolina[1]				South Carolina[1]			
	Density Classes[2]				Density Classes[2]				Density Classes[2]				Density Classes[2]			
	A	L	M	H	A	L	M	H	A	L	M	H	A	L	M	H
White–tailed Deer																
1970	0	86	14	0	0	54	46	0	0	95	5	0	0	100	0	0
1995	0	0	100	0	0	25	54	21	0	82	14	5	0	0	100	0
Wild Turkey																
1970	0	100	0	0	57	43	0	0	74	27	0	0	0	100	0	0
1995	0	0	100	0	0	11	79	11	0	77	14	9	0	0	100	0
Black Bear																
1970	100	0	0	0	54	46	0	0	14	86	0	0	0	100	0	0
1995	100	0	0	0	46	25	29	0	0	64	36	0	0	100	0	0
Gray Squirrel																
1970	0	0	100	0	0	25	61	14	0	0	100	0	0	0	100	0
1995	0	14	86	0	0	21	64	14	0	0	100	0	0	0	100	0
Fox Squirrel																
1970	0	57	43	0	43	57	0	0	100	0	0	0	100	0	0	0
1995	0	100	0	0	39	61	0	0	100	0	0	0	100	0	0	0
Cottontail																
1970	0	57	0	43	0	21	57	21	0	0	100	0	0	0	100	0
1995	0	43	57	0	0	36	64	0	0	0	100	0	0	100	0	0
Raccoon																
1970	0	0	100	0	0	21	79	0	0	100	0	0	0	0	100	0
1995	0	0	100	0	0	0	100	0	0	0	100	0	0	0	100	0
Ruffed Grouse																
1970	100	0	0	0	36	32	21	11	0	23	77	0	0	100	0	0
1995	100	0	0	0	32	36	32	0	0	23	77	0	0	100	0	0
Bobwhite Quail																
1970	0	0	86	14	0	36	64	0	0	77	23	0	0	0	100	0
1995	0	29	71	0	0	57	43	0	0	77	23	0	0	100	0	0
Woodcock																
1970	14	86	0	0	0	39	61	0	0	100	0	0	0	100	0	0
1995	43	57	0	0	0	46	54	0	0	100	0	0	0	100	0	0

Table 3.18 (cont.) The trends in the proportion of counties in each density class by state for the 10 major game species in the Southern Appalachian Assessment area.

	Tennessee[1]				Virginia[1]				West Virginia[1]			
	Density Classes[2]				Density Classes[2]				Density Classes[2]			
	A	L	M	H	A	L	M	H	A	L	M	H
Whitetailed Deer												
1970	0	100	0	0	3	36	62	0	0	33	67	0
1995	0	48	48	3	0	8	85	8	0	0	0	100
Wild Turkey												
1970	76	24	0	0	0	54	41	5	0	100	0	0
1995	0	42	55	3	0	21	66	13	0	33	67	0
Black Bear												
1970	79	21	0	0	26	62	13	0	0	100	0	0
1995	67	9	12	12	0	85	15	0	0	100	0	0
Gray Squirrel												
1970	0	0	100	0	0	0	100	0	0	0	100	0
1995	0	6	94	0	0	0	92	8	0	0	100	0
Fox Squirrel												
1970	0	73	15	12	18	82	0	0	0	67	33	0
1995	0	73	15	12	18	82	0	0	0	0	100	0
Cottontail												
1970	0	42	45	12	0	36	8	56	0	0	100	0
1995	0	64	36	0	0	36	8	56	0	100	0	0
Raccoon												
1970	0	73	27	0	0	64	36	0	0	0	100	0
1995	0	0	79	21	0	13	87	0	0	0	100	0
Ruffed Grouse												
1970	0	30	42	27	0	41	13	46	0	0	100	0
1995	0	30	42	27	0	36	64	0	0	100	0	0
Bobwhite Quail												
1970	0	55	45	0	0	44	41	15	0	100	0	0
1995	0	70	30	0	0	85	15	0	0	100	0	0
Woodcock												
1970	0	88	12	0	3	79	18	0	0	100	0	0
1995	0	88	12	0	3	97	0	0	0	100	0	0

[1] Sample Sizes:
Alabama = 7
Georgia = 28
North Carolina = 22
South Carolina = 3
Tennessee = 33
Virginia = 39
West Virginia = 3

[2] Density Classes:
A = Absent
L = Low
M = Medium
H = High

Table 3.19 The trends in the proportion of each ownership in each density class for 10 of the major game species in the Southern Appalachian Assessment area.

	Deer				Turkey				Bear				Gray Squirrel				Fox Squirrel			
	Density Classes[1]				Density Classes[1]				Density Classes[1]				Density Classes[1]				Density Classes[1]			
Ownership	A	L	M	H	A	L	M	H	A	L	M	H	A	L	M	H	A	L	M	H
National Forests																				
In Alabama																				
1970	0	58	42	0	0	100	0	0	100	0	0	0	0	0	100	0	0	37	63	0
1995	0	0	100	0	0	0	100	0	100	0	0	0	0	0	100	0	0	100	0	0
Chattahoochee																				
1970	0	71	29	0	8	92	0	0	8	92	0	0	0	9	70	21	63	37	0	0
1995	0	38	58	4	0	8	88	3	4	15	80	0	0	2	77	21	62	38	0	0
In North Carolina																				
1970	0	100	0	0	55	45	0	0	<1	100	0	0	0	0	100	0	100	0	0	0
1995	<1	100	0	0	0	100	<1	0	0	38	62	<1	0	0	100	0	100	0	0	0
Sumter																				
1970	0	100	0	0	0	100	0	0	0	100	0	0	0	0	100	0	100	0	0	0
1995	0	0	100	0	0	0	100	0	0	100	0	0	0	0	100	0	100	0	0	0
Cherokee																				
1970	0	100	0	0	47	53	0	0	31	69	0	0	0	0	100	0	<1	100	0	0
1995	0	50	50	0	0	16	83	<1	<1	17	51	32	0	0	100	0	<1	100	0	0
George Washington																				
1970	0	5	95	0	0	13	87	0	0	92	8	0	0	0	100	0	7	83	10	0
1995	0	0	68	32	0	21	75	5	0	99	1	0	0	0	93	7	7	88	5	0
Jefferson																				
1970	0	34	66	0	<1	30	60	10	18	46	36	0	0	0	100	0	6	94	0	0
1995	0	6	94	0	0	22	69	8	0	43	57	0	0	0	100	0	6	94	0	0
Monongahela																				
1970	0	100	0	0	0	100	0	0	0	100	0	0	0	0	100	0	0	0	100	0
1995	0	0	0	100	0	100	0	0	0	100	0	0	0	0	100	0	0	0	100	0
Total National Forests																				
1970	0	61	39	0	20	49	29	2	13	79	7	0	0	1	95	1	37	56	6	0
1995	0	36	53	10	0	36	61	3	6	47	43	4	0	<1	95	5	37	59	4	0
National Parks																				
1970	4	79	18	0	42	50	8	0	8	91	1	0	0	<1	99	<1	40	58	2	<1
1995	0	64	35	1	0	73	26	1	4	33	36	26	0	1	98	1	40	59	<1	<1
Cherokee Indian Reservation																				
1970	0	100	0	0	100	0	0	0	0	100	0	0	0	0	100	0	0	100	0	0
1995	0	100	0	0	0	100	0	0	0	43	57	0	0	0	100	0	0	100	0	0
Other Federal																				
1970	0	94	6	0	35	65	0	0	93	7	0	0	0	0	100	0	0	77	23	0
1995	0	35	64	1	0	36	64	0	93	3	4	0	0	0	100	0	0	77	23	0
State																				
1970	0	73	27	0	36	54	10	<1	51	49	<1	0	0	2	84	13	13	62	21	4
1995	0	47	43	11	0	15	80	5	46	42	11	<1	0	2	84	13	10	61	24	4
Private																				
1970	0	68	31	0	38	47	13	2	48	49	3	0	0	4	94	2	30	58	9	3
1975	<1	29	61	10	0	30	62	8	33	51	13	3	0	7	89	4	30	60	8	3

Table 3.19 (cont.) The trends in the proportion of each ownership in each density class for 10 of the major game species in the Southern Appalachian Assessment area.

	Cottontail Density Classes[1]				Raccoon Density Classes[1]				Grouse Density Classes[1]				Quail Density Classes[1]				Woodcock Density Classes[1]			
Ownership	A	L	M	H	A	L	M	H	A	L	M	H	A	L	M	H	A	L	M	H
National Forests																				
In Alabama																				
1970	0	37	0	63	0	0	100	0	100	0	0	0	0	0	99	1	9	91	0	0
1995	0	28	72	0	0	0	100	0	100	0	0	0	0	9	91	0	51	49	0	0
Chattahoochee																				
1970	0	62	34	4	0	59	41	0	2	17	40	40	0	80	20	0	0	81	19	0
1995	0	84	16	0	0	0	100	0	<1	20	80	0	0	85	15	0	0	84	16	0
In North Carolina																				
1970	0	<1	100	0	0	100	0	0	0	16	84	<1	0	84	16	0	0	100	0	0
1995	0	<1	100	0	0	0	100	0	0	16	84	<1	0	84	16	0	0	100	0	0
Sumter																				
1970	0	0	100	0	0	0	100	0	0	100	0	0	0	0	100	0	0	100	0	0
1995	0	100	0	0	0	0	100	0	0	100	0	0	0	100	0	0	0	100	0	0
Cherokee																				
1970	0	77	23	<1	0	70	30	0	0	24	37	39	0	85	15	0	0	100	0	0
1995	0	86	14	<1	0	0	91	9	0	24	37	39	0	94	6	0	0	100	0	0
George Washington																				
1970	0	82	10	8	0	8	92	0	0	54	45	1	0	92	8	0	2	93	5	0
1995	0	92	0	8	0	0	100	0	0	56	44	0	0	93	7	0	2	98	0	0
Jefferson																				
1970	0	3	6	91	0	97	3	0	0	6	1	93	0	9	82	9	0	91	9	0
1995	0	3	6	91	0	8	92	<1	0	6	94	<1	0	95	5	0	0	100	0	0
Monongahela																				
1970	0	0	100	0	0	0	100	0	0	0	100	0	0	100	0	0	0	100	0	0
1995	0	100	0	0	0	0	100	0	0	100	0	0	0	100	0	0	0	100	0	0
Total																				
1970	0	42	38	20	0	59	41	0	5	25	43	26	0	69	30	1	1	93	6	0
1995	0	53	32	16	0	1	98	1	5	28	62	5	0	86	14	0	3	94	3	0
National Parks																				
1970	0	52	42	6	0	72	28	0	3	28	67	2	0	91	6	4	<1	96	4	0
1995	0	52	43	4	0	1	87	12	3	28	69	<1	0	94	6	0	1	98	1	0
Cherokee Indian Reservation																				
1970	0	0	100	0	0	100	0	0	0	0	100	0	0	100	0	0	0	100	0	0
1995	0	0	100	0	0	0	100	0	0	0	100	0	0	100	0	0	0	100	0	0
Other Federal																				
1970	0	70	23	6	0	29	71	0	58	23	13	6	0	13	87	0	58	20	23	0
1995	0	13	80	6	0	0	87	13	58	23	19	0	0	77	23	0	58	20	23	0
State																				
1970	0	40	48	11	0	44	56	0	11	31	47	11	0	51	47	2	<1	80	20	0
1995	0	56	36	8	0	2	96	2	8	38	51	4	0	68	32	0	<1	82	17	0
Private																				
1970	0	27	46	27	0	56	44	0	14	36	30	20	0	43	50	8	2	78	20	
1995	0	40	41	19	0	4	90	6	14	38	42	6	0	68	32	0	4	84	12	0

[1] Density Classes: A = Absent, L = Low, M = Medium, H = High

Table 3.20 The current land cover by population density class for 10 of the major game species in the Southern Appalachian Assessment area.

	Deer				Turkey				Bear				Gray Squirrel				Fox Squirrel			
	Density Classes[1]				Density Classes[1]				Density Classes[1]				Density Classes[1]				Density Classes[1]			
	A	L	M	H	A	L	M	H	A	L	M	H	A	L	M	H	A	L	M	H
	Number of Counties				Number of Counties				Number of Counties				Number of Counties				Number of Counties			
Land Use Category	0	4	77	14	0	43	81	11	42	63	26	4	0	9	119	7	3	80	8	4
Northern Hardwoods		1	<1	2		1	<1	<1	0	1	1	1		0	<1	<1	<1	<1	3	0
Mixed Mesophytic Hardwoods		4	2	1		4	1	3	<1	2	4	4		1	2	<1	4	1	<1	4
Oak		51	47	52		48	48	55	39	53	56	38		30	49	66	53	46	57	37
Bottomland Hardwoods		<1	<1	<1		<1	<1	<1	<1	<1	<1	<1		<1	<1	<1	<1	<1	<1	<1
Total Deciduous		55	49	56		53	50	58	39	56	60	42		31	52	66	57	48	61	41
White Pine/Hemlock		<1	<1	<1		<1	<1	<1	<1	<1	1	<1		<1	<1	1	<1	<1	<1	<1
Montane Spruce/Fir		1	<1	<1		1	<1	<1	0	<1	1	2		0	<1	0	1	<1	<1	0
Southern Yellow Pine		5	4	2		4	5	1	8	2	2	8		8	4	2	2	5	3	6
Total Coniferous		6	4	3		5	5	2	8	3	3	10		8	5	3	3	5	3	6
White Pine/Hemlock/Hardwoods		3	1	2		3	1	2	<1	3	3	2		<1	2	1	4	1	1	1
Mixed Pine/Hardwoods		11	12	13		9	13	11	14	10	12	16		18	11	14	14	11	9	10
Total Mixed		14	13	15		12	15	13	14	13	16	18		18	13	14	18	12	10	11
Total Forest Land		75	67	73		71	69	72	62	72	79	71		57	70	83	78	65	74	58
Herbaceous		3	5	4		3	5	3	7	3	1	1		9	4	3	3	5	3	1
Barren		<1	<1	<1		<1	<1	<1	<1	<1	<1	<1		1	<1	<1	<1	<1	<1	<1
Pasture		14	20	15		17	18	17	20	17	15	19		16	18	9	11	21	17	31
Cropland		2	4	5		3	3	5	3	4	1	3		4	3	2	3	4	4	3
Wetland		<1	<1	<1		<1	<1	<1	<1	<1	<1	<1		<1	<1	<1	<1	<1	<1	<1
Developed		4	3	2		4	3	2	4	3	2	4		8	3	1	3	4	1	2
Water		2	1	2		1	2	1	3	1	1	1		4	1	1	1	1	1	4
Total Nonforest		25	33	27		29	31	28	38	28	21	29		43	30	17	22	35	26	42

Table 3.20 (cont.) The current land cover by population density class for 10 of the major game species in the Southern Appalachian Assessment area.

	Cottontail				Raccoon				Grouse				Quail				Woodcock			
	Density Classes[1]				Density Classes[1]				Density Classes[1]				Density Classes[1]				Density Classes[1]			
	A	L	M	H	A	L	M	H	A	L	M	H	A	L	M	H	A	L	M	H
	Number of Counties				Number of Counties				Number of Counties				Number of Counties				Number of Counties			
Land Use Category	0	4	77	14	0	43	81	11	42	63	26	4	0	9	119	7	3	80	8	4
Northern Hardwoods		1	<1	<1		0	<1	<1	0	1	<1	<1		1	<1		0	<1	0	
Mixed Mesophytic Hardwoods		2	3	1		2	2	3	<1	1	3	8		3	1		<1	3	<1	
Oak		48	45	58		74	49	30	35	45	57	41		52	41		47	51	32	
BottomlandHardwoods		<1	<1	<1		<1	<1	<1	<1	<1	<1	<1		<1	<1		<1	<1	<1	
Total Deciduous		51	49	60		76	52	30	35	46	61	49		55	43		47	54	32	
White Pine/Hemlock		<1	<1	<1		<1	<1	<1	0	<1	<1	<1		<1	<1		<1	<1	<1	
Montane Spruce/Fir		<1	<1	<1		0	<1	0	0	<1	1	<1		<1	<1		0	<1	0	
Southern Yellow Pine		4	6	1		1	4	7	10	5	2	3		3	7		7	3	11	
Total Coniferous		5	6	1		1	5	8	10	5	3	3		4	7		7	4	11	
White Pine/Hemlock/Hardwoods		2	2	1		1	2	1	<1	2	2	3		2	1		<1	2	<1	
Mixed Pine/Harddwoods		13	12	7		2	12	11	18	14	9	9		11	14		11	11	17	
Total Mixed		15	15	8		3	14	12	18	16	11	12		13	15		11	13	18	
Total Forest Land		70	70	69		80	71	52	64	67	75	64		72	65		64	71	61	
Herbaceous		4	6	1		3	4	3	11	5	2	1		3	7		8	3	11	
Barren		<1	<1	<1		1	<1	<1	1	<1	<1	<1		<1	1		<1	<1	<1	
Pasture		15	17	25		14	17	27	16	16	18	25		16	20		21	17	19	
Cropland		5	3	1		1	3	4	3	6	2	3		4	3		3	3	3	
Wetland		<1	<1	<1		<1	<1	<1	1	<1	<1	<1		<1	<1		<1	<1	<1	
Developed		4	3	3		1	3	11	2	4	2	5		3	2		2	3	3	
Water		1	2	1		<1	1	3	2	2	1	2		1	2		1	1	3	
Total Nonforest		30	30	31		20	29	48	36	22	25	36		28	35		36	29	39	

[1] Density Classes:
A = Absent
L = Low
M = Medium
H = High

Table 3.21 The estimated oak mast capability based on related land cover variables by section groups for the Southern Appalachian Assessment area.

	Section Group[1]					
Variable	1	2	3	4	5	6
Percent Deciduous Forest [2]	58	67	69	29	34	39
Percent Mixed Pine/Hardwood Forest [2]	8	15	7	12	13	24
Percent Coniferous Forest [2]	1	3	3	9	11	6
Percent Mid to Late Successional Forest [3]	81	77	56	65	46	59
Percent Nonforest [2]	33	16	21	51	42	32
Oak Mast Capability (lb/ac)						
Forest Land Area	131	139	81	72	46	64
Total Land Area	88	117	64	36	27	44

[1] Section Groups:
1 = Northern Ridge and Valley, Allegheny Mountains, Northern Cumberland Mountains
2 = Blue Ridge Mountains
3 = Northern Cumberland Plateau, Southern Cumberland Mountains
4 = Central Ridge and Valley
5 = Southern Cumberland Plateau, Southern Ridge and Valley
6 = Southern Appalachian Piedmont

[2] Based on satellite imagery

[3] Based on FIA statistics

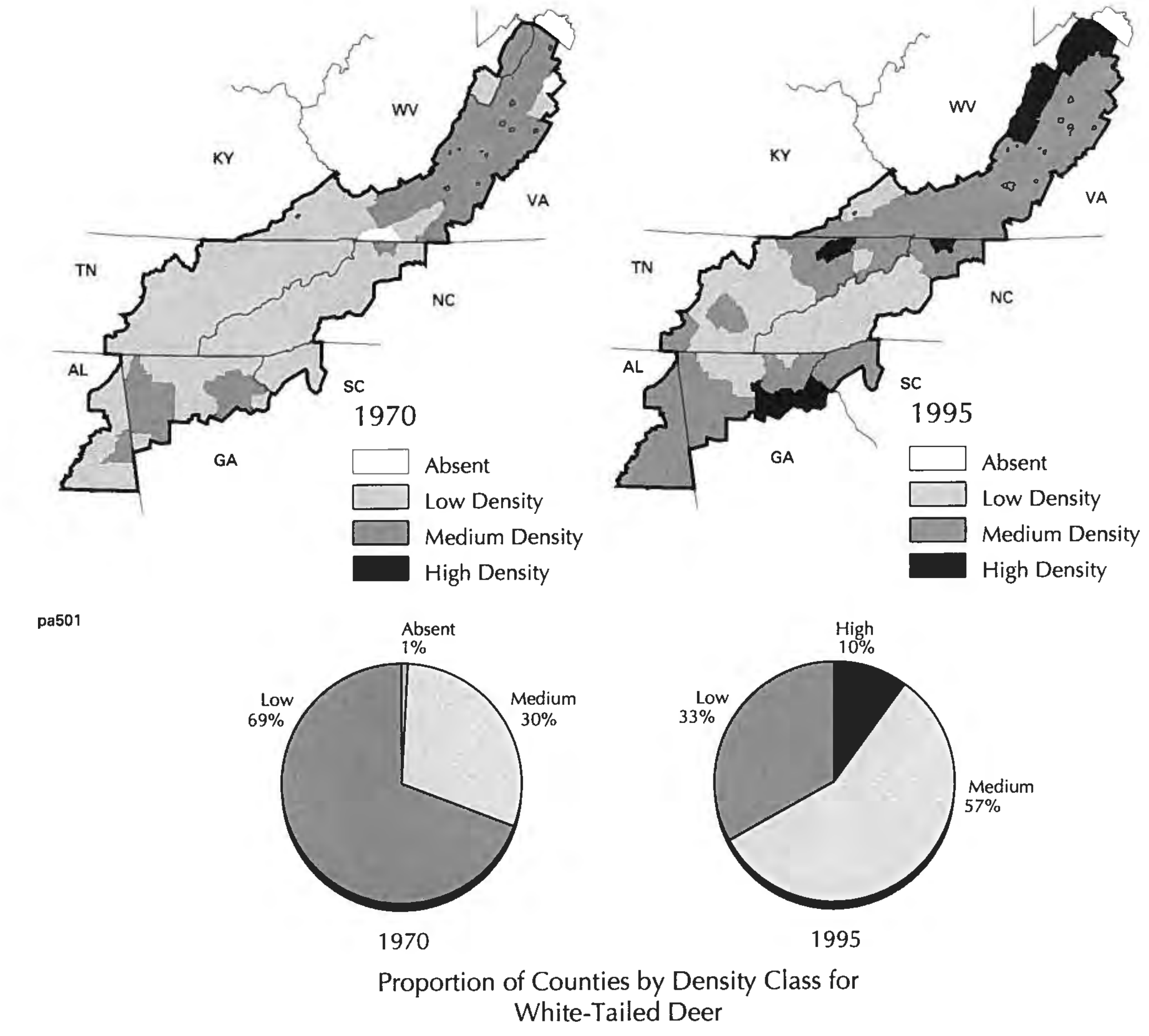


Figure 3.50 The spatial distribution of white-tailed deer county population density estimates for 1970 and 1995 for the Southern Appalachian Assessment area.

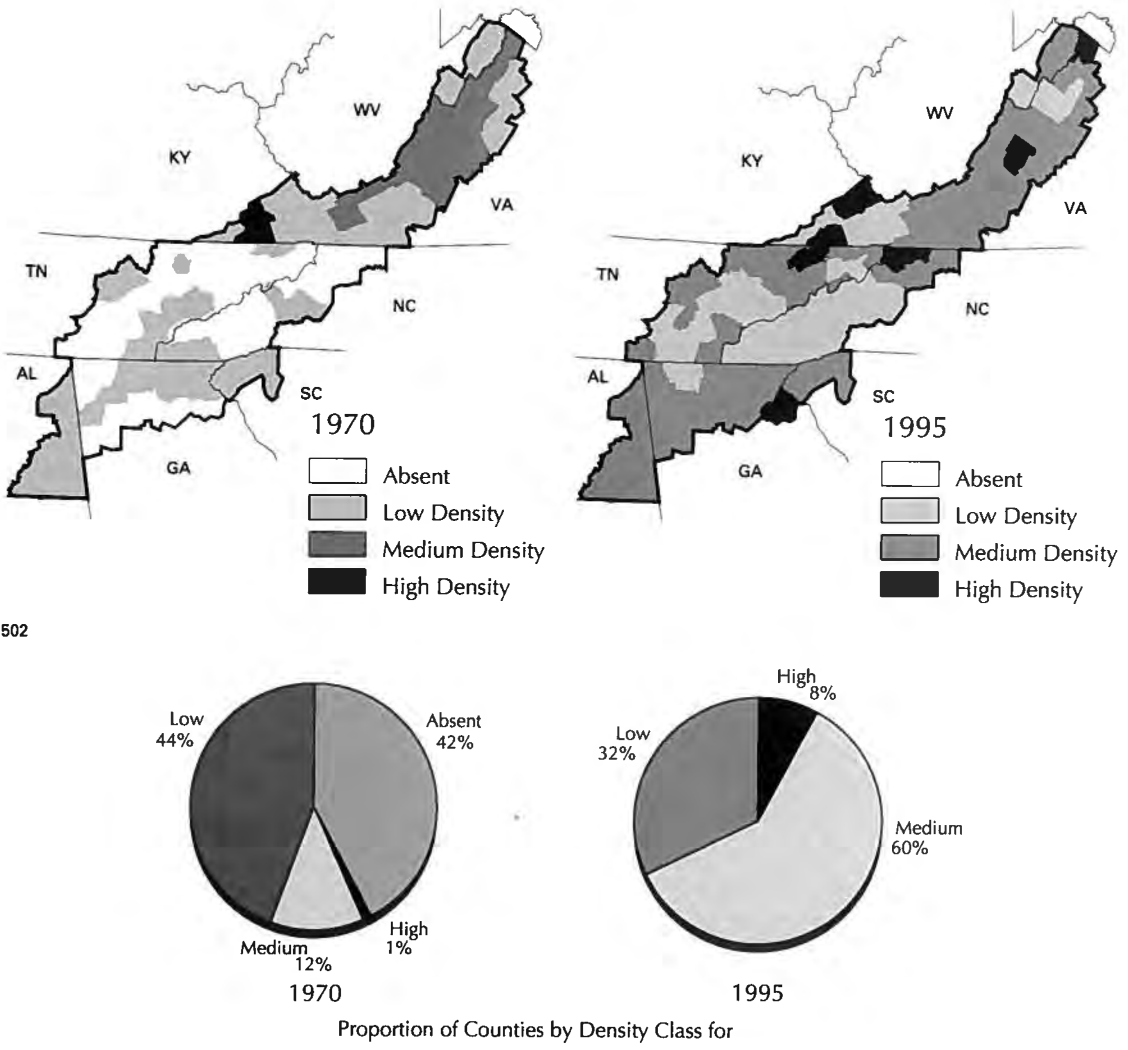
harvest strategies as well as increased acorn capability resulting from the increase in mid- to late-successional oak forests.

The outlook for this species is for current population trends to level off. Opportunities for hunting on private lands will be increasingly based on a willingness or ability to pay for leases to hunt big game species. Demand for big game hunting/viewing opportunities on national forests and state lands should continue to increase slightly over the next 15 years due to decreasing hunting access to private lands for the general public.

Eastern Wild Turkey

Wild turkeys are present throughout the assessment area (fig. 3.51). Population densities generally are medium to high in the Northern Ridge and Valley, Allegheny Mountains, and Northern Cumberland Mountains (section group 1) and the Southern Appalachian Piedmont (section group 6). Densities generally are low to medium in the remainder of the assessment area. Counties with high turkey densities generally contain greater amounts of oak forest and cropland and lesser amounts of developed and coniferous forestland than do counties with a low or medium density (table 3.20). Current densities generally are higher on private land, state, and national forest land than on the remaining ownerships (table 3.19).

Wild turkey populations have greatly expanded in range and density in the last 25 years. In 1970, turkeys were absent from approximately 42 percent of the counties in the



Proportion of Counties by Density Class for Wild Turkey

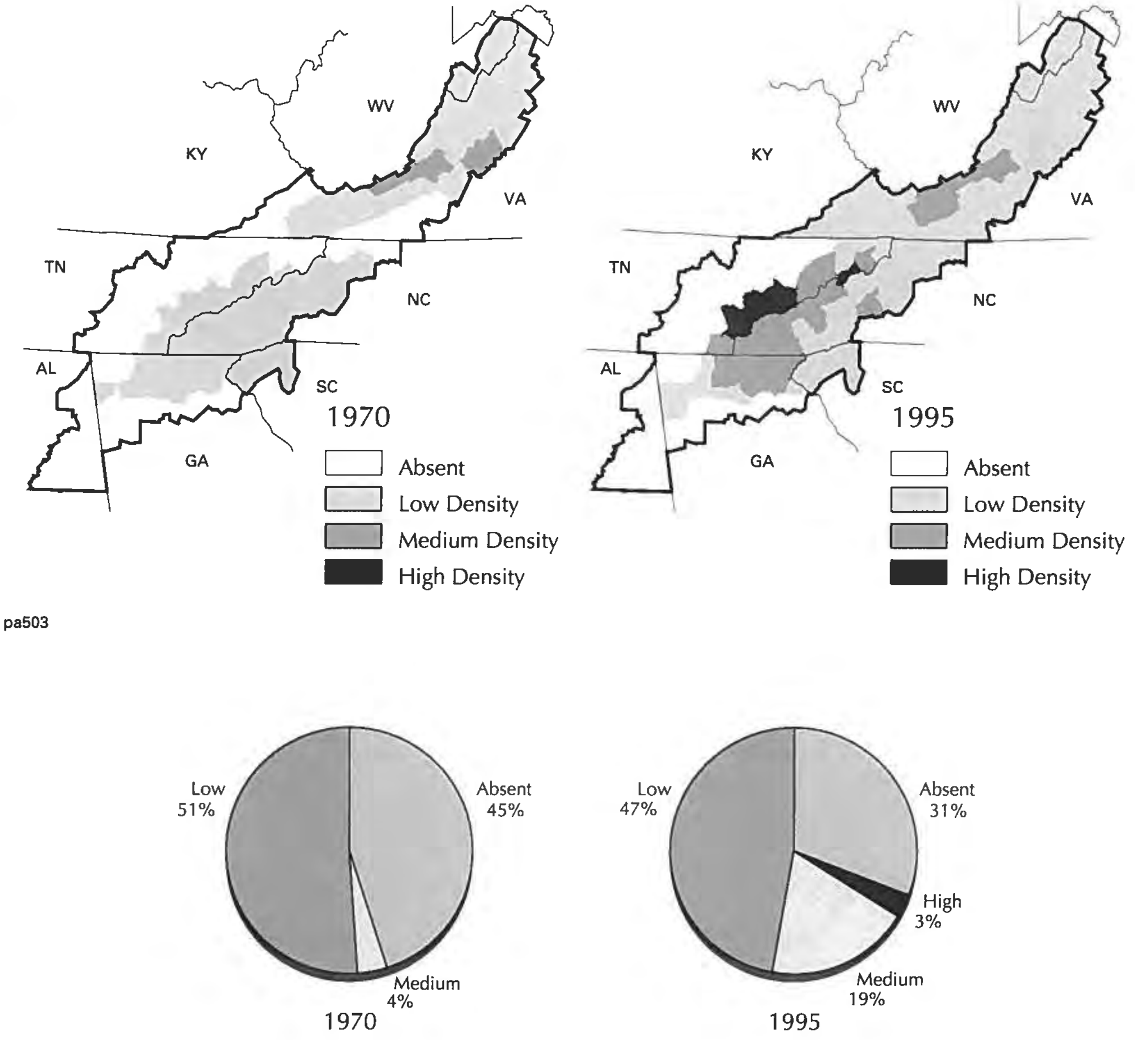
Figure 3.51 The spatial distribution of eastern wild turkey county population density estimates for 1970 and 1995 for the Southern Appalachian Assessment area.

assessment area and only 13 percent of the counties had a medium or high density of turkeys. Today turkeys are present in all counties of the assessment area and nearly 70 percent of the counties have a medium to high density of turkeys. This pattern of increase generally is consistent throughout the assessment area and within ownerships. As with deer, this increase probably is related to both nonhabitat factors such as extensive restoration efforts, protection and conservative harvest as well as increased acorn capability resulting from the increase in mid- to late-successional oak forests.

The outlook for this species is for current population trends to level off. Hunting on private land will be increasingly based on a willingness or ability to pay for leases to hunt big game species. Demand for big game hunting and viewing opportunities on national forests and state lands should continue to increase slightly over the next 15 years due to decreased public access to private land for hunting.

Black Bear

Black bears are generally present in low to medium densities in the Northern Ridge and Valley, Allegheny Mountains, and Northern Cumberland mountains (section group 1) and Blue Ridge Mountains (section group 2) (fig. 3.52). These are regions with highest acorn capability (table 3.21). Bears are absent from much of the Northern Cumberland Plateau and Southern Cumberland Mountains (section group 3); the Central Ridge and Valley (section



Proportion of Counties by Density Class for Black Bear

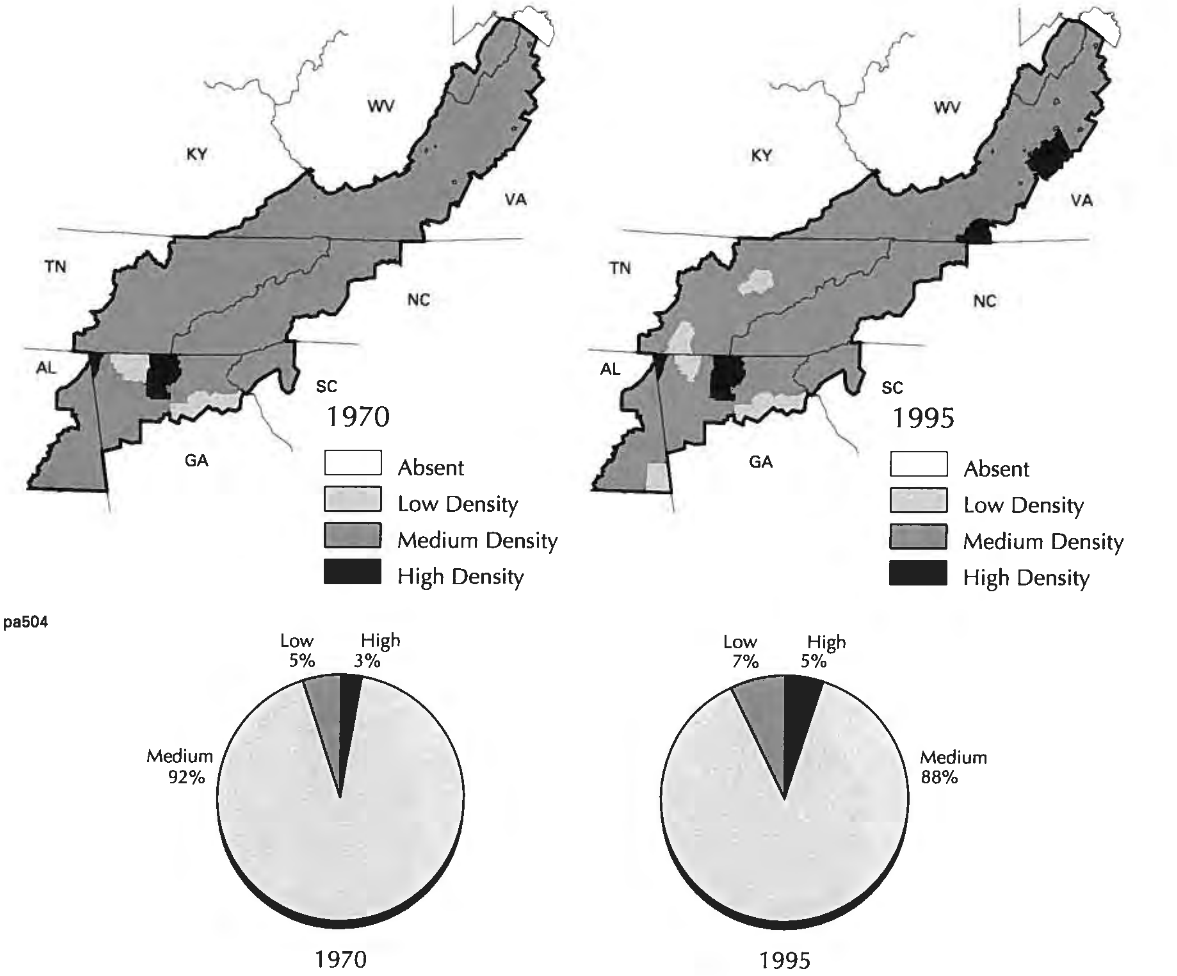
Figure 3.52 The spatial distribution of black bear county population density estimates for 1970 and 1995 for the Southern Appalachian Assessment area.

group 4); the Southern Cumberland Plateau and Southern Ridge and Valley (section group 5); and the Southern Appalachian Piedmont (section group 6), particularly in the extreme southern and western portions of the assessment area. The absence of bears from some counties appears to be related largely to the presence of large amounts of nonforest habitats and limited forested habitat (table 3.20). Bear population densities generally are higher on national park land and, to a lesser extent on national forest land and the Cherokee Indian Reservation, than on the remaining ownerships (table 3.19).

Black bears have made moderate range expansions since 1970, particularly in southern Virginia and northern Tennessee and North Carolina. As a result, the northern and southern population centers are now linked. There also has been a moderate increase in population densities. In 1970, 4 percent of counties had a medium bear density and none had high bear densities. Today, approximately 19 percent have medium densities and 3 percent have high densities. This pattern of increase generally is consistent throughout the assessment area and within ownerships. This increase likely is related to both nonhabitat factors such as protection and conservative harvest and to the increased acorn capability resulting from the increase in mid- to late-successional oak forests.

Gray Squirrel

Gray squirrels are found throughout the assessment area generally at medium population densities (fig. 3.53). High squirrel population densities are associated with a high



Proportion of Counties by Density Class for Gray Squirrel

Figure 3.53 The spatial distribution of gray squirrel county population density estimates for 1970 and 1995 for the Southern Appalachian Assessment area.

proportion of oak forest and lower proportions of coniferous forest, pasture, cropland, and developed land cover (table 3.20). Gray squirrel densities are similar among the various ownerships in the assessment area (table 3.19).

Gray squirrel population densities have remained very stable during the last 25 years. Although not reflected in the density estimates, gray squirrel populations likely have benefited from increased acorn capability resulting from maturation of oak forests.

Fox Squirrel

Fox squirrels are absent from approximately one third of the counties in the assessment area (fig. 3.54). They are absent from much of the Blue Ridge Mountains (section group 2) and the Southern Appalachian Piedmont (section group 6). Where they are present at all, their densities are often low. Medium to high densities are present in a few counties in the Northern Cumberland Plateau and Southern Cumberland Mountains (section group 3), the Central Ridge and Valley (section group 4), and to a lesser extent the Northern Ridge and Valley, Allegheny Mountains, and Northern Cumberland Plateau (section group 1). Fox squirrels appear to be less strongly tied to deciduous forest than are gray squirrels. High fox squirrel densities are associated with greater amounts of nonforest habitat, particularly agricultural land (table 3.20). Population densities generally are lower on national forests and parks than on the remaining ownerships (table 3.19). Fox squirrel population densities have remained very stable over the last 25 years.

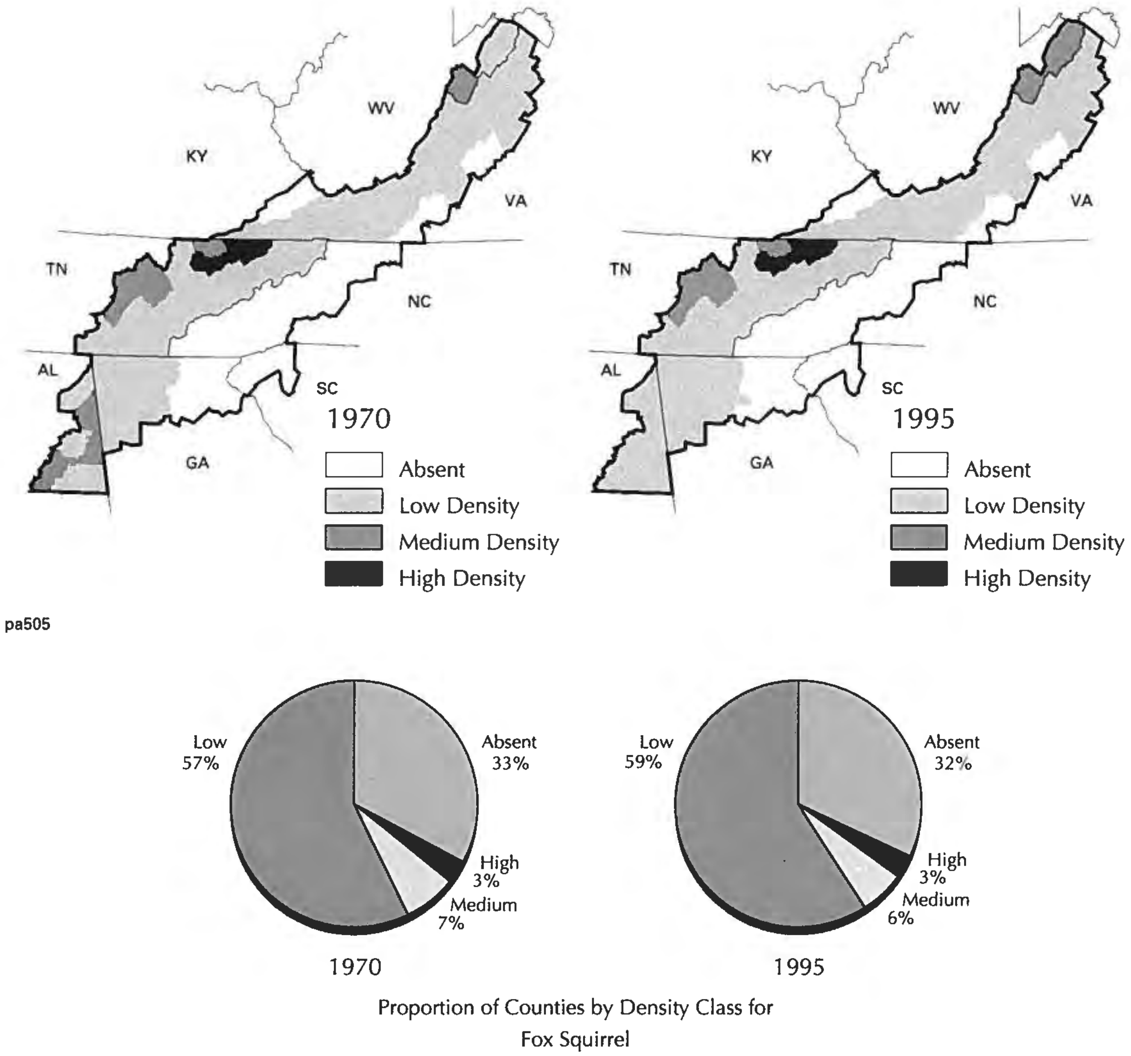


Figure 3.54 The spatial distribution of fox squirrel county population density estimates for 1970 and 1995 for the Southern Appalachian Assessment area.

Eastern Cottontail

Cottontails are present throughout the assessment area (fig. 3.55). Population densities generally are low to medium, but high densities are reported for southwestern Virginia, primarily in the Northern Ridge and Valley, Allegheny Mountains, and Northern Cumberland Mountains (section group 1). High population densities are associated with high proportions of agricultural land, especially pastures (table 3.20). Rabbit population densities generally are lower on national parks and state land than on the remaining ownerships (table 3.19).

With the exception of Virginia, where populations have remained stable, cottontail densities have declined during the last 25 years. The proportion of counties with a high density declined from 26 percent to 16 percent from 1970 to the present. Over the same period, counties with a low density increased from 28 percent to 40 percent. This decline likely is attributable to the reduction in agricultural land in the assessment area. All ownerships experienced a decline over this period.

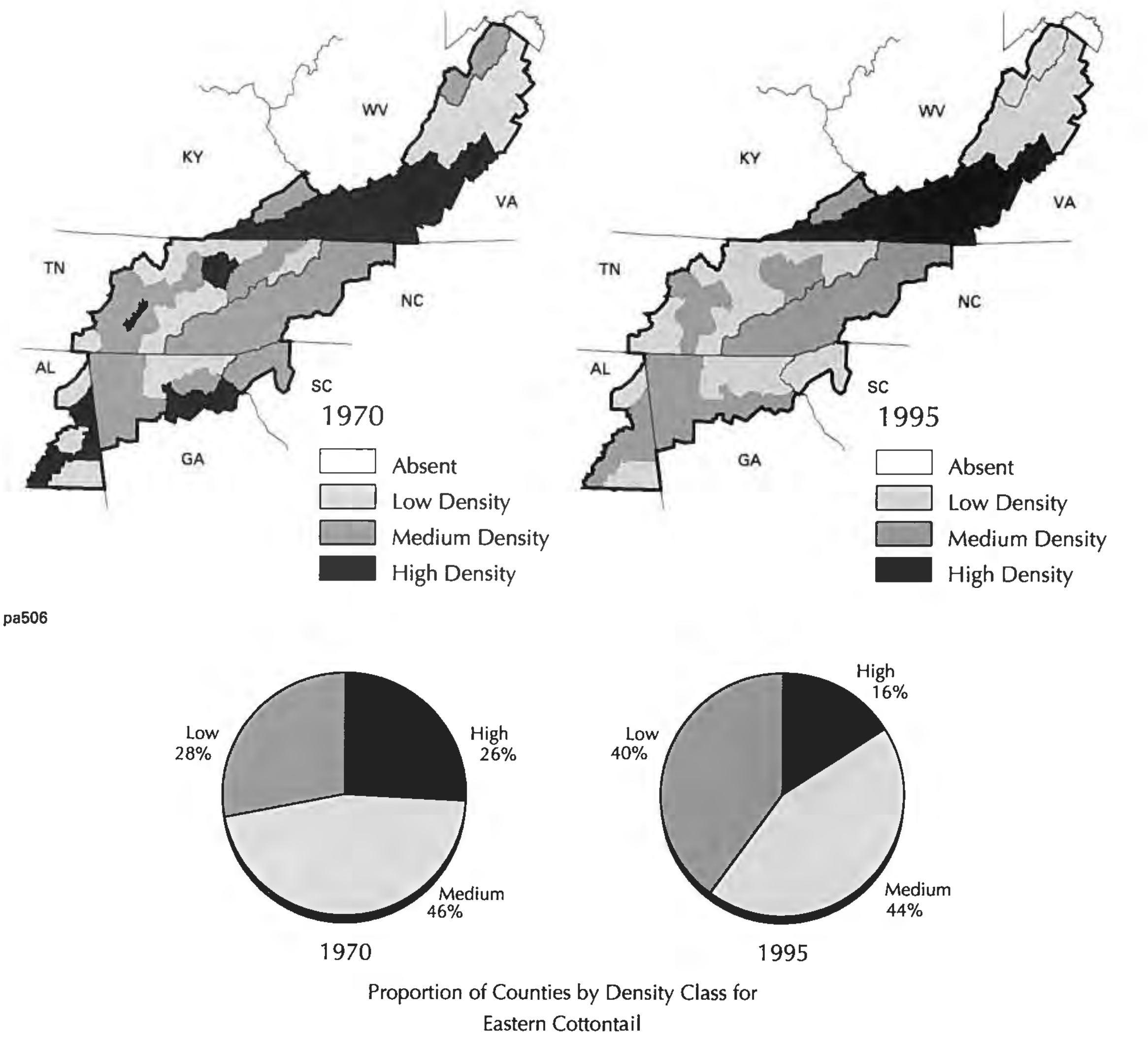
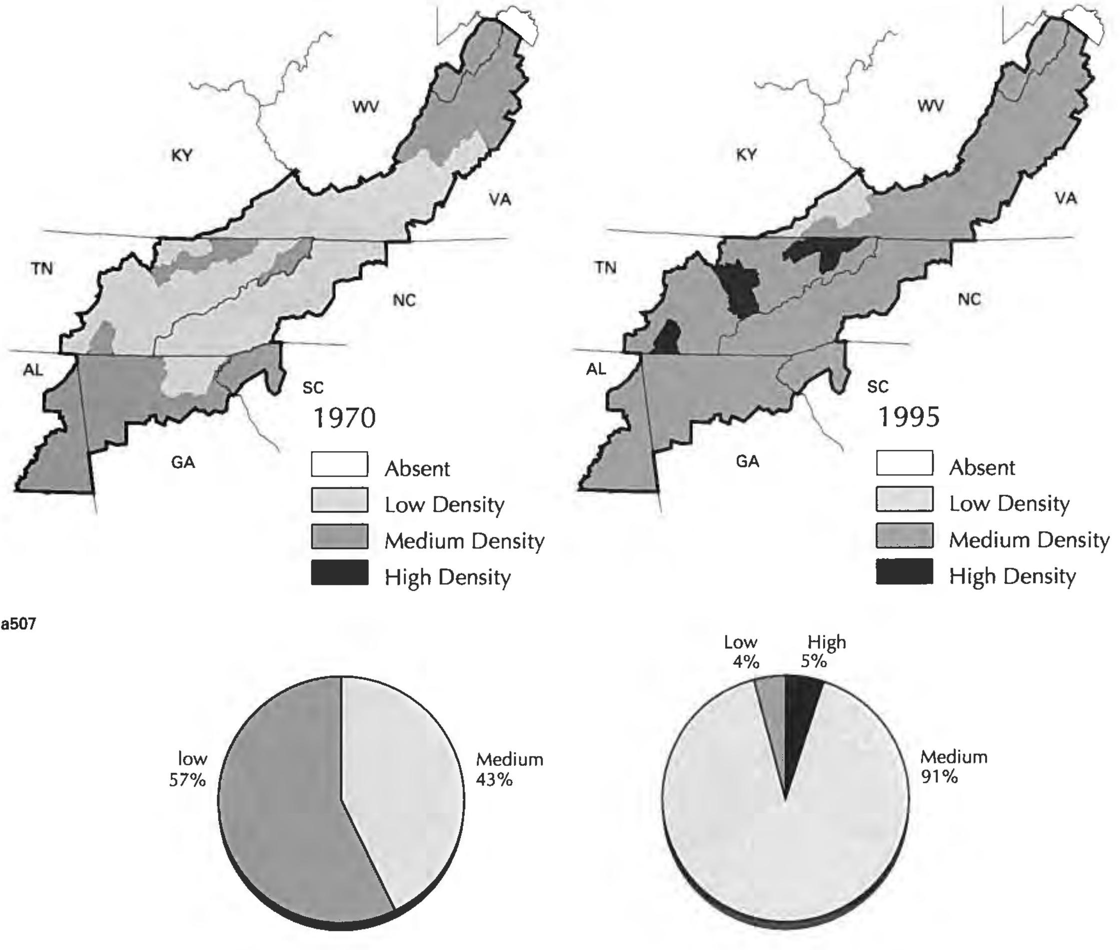


Figure 3.55 The spatial distribution of eastern cottontail county population density estimates for 1970 and 1995 for the Southern Appalachian Assessment area.

Raccoon

Raccoons are present in medium densities of the assessment area (fig. 3.56). High population densities are reported for portions of the Central Ridge and Valley (section group 4) in eastern Tennessee. Water comprises a greater proportion of this section group than the remainder of the assessment area. High raccoon populations are also associated with greater proportions of nonforest land, including pasture, cropland, developed land, and water (table 3.20). Population densities are similar among all ownership categories (table 3.19).

Raccoon population densities have increased throughout the assessment area in the last 25 years. In 1970, approximately 57 percent of counties had a low raccoon density and none had a high raccoon density. Today approximately 96 percent of counties have a medium to high density of raccoons. This pattern of increase generally is consistent throughout the assessment area and within ownerships. This trend probably is a result of nonhabitat factors, including protection and conservative harvest. Because of their adaptability, raccoons may have benefited from the increased food supply associated with human development.



Proportion of Counties by Density Class for Raccoon

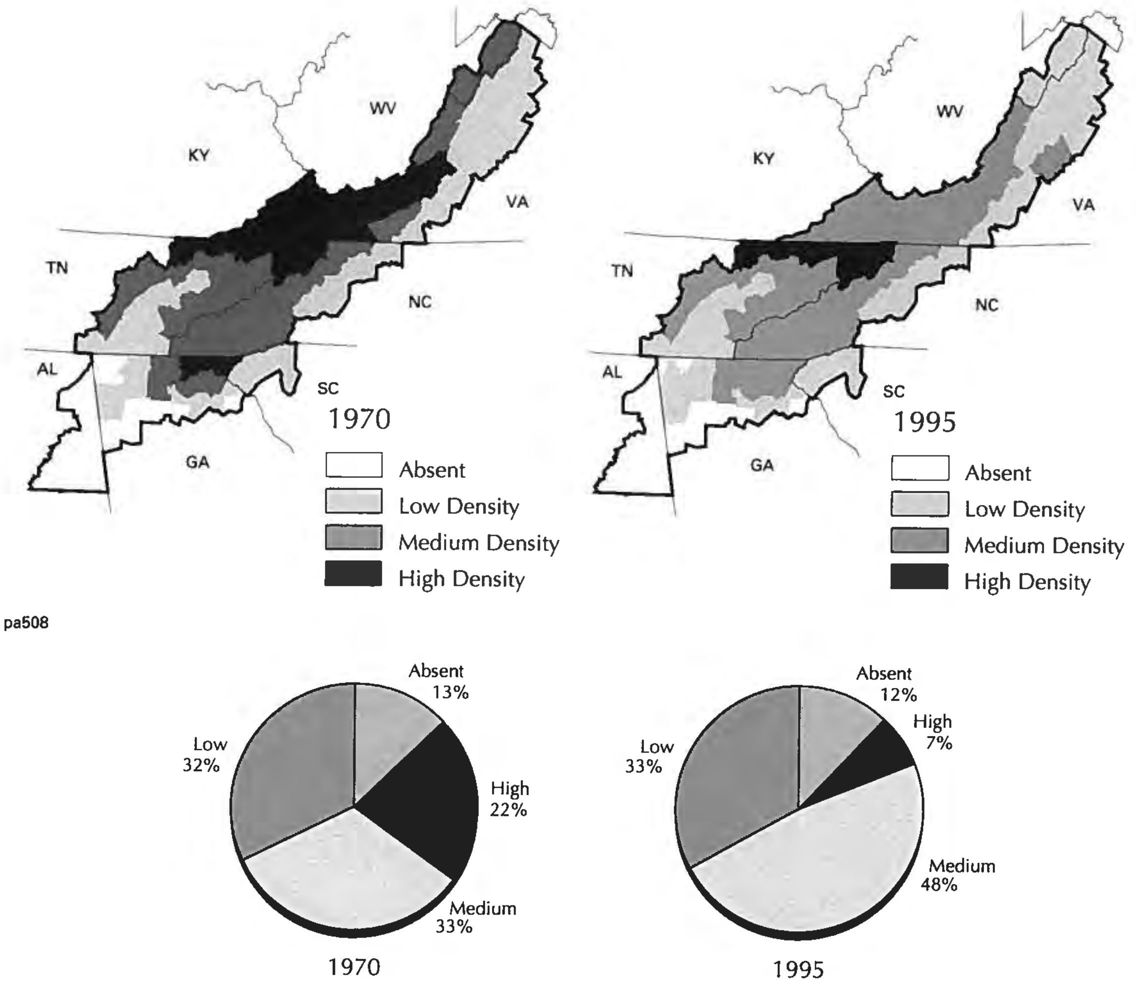
Figure 3.56 The spatial distribution of raccoon county population density estimates for 1970 and 1995 for the Southern Appalachian Assessment area.

Ruffed Grouse

Grouse are present throughout the assessment area except for portions of the Southern Cumberland Plateau and Southern Ridge and Valley (section group 5) and the Southern Appalachian Piedmont (section group 6) in Georgia and Alabama (fig. 3.57). Population densities generally are medium in the Blue Ridge Mountains (section group 2) and medium to low in the remaining portions of the assessment area. High population densities are reported for nine counties in northern Tennessee. Counties in which grouse are absent are characterized by low proportions of deciduous forest cover and high proportions of coniferous and mixed forest cover and herbaceous cover (table 3.20). Current grouse populations generally are higher on national forest lands, national parks, and the Cherokee Indian Reservation than on remaining ownerships (table 3.19).

Grouse population densities have declined in the assessment area since 1970. In 1970, approximately 22 percent of counties had a high grouse density, while only 7 percent have a high grouse density today. This pattern of decrease generally is consistent throughout the assessment area, but slight increases are reported in northwestern Georgia and the Virginia Piedmont. The declining trend probably is largely a result of the reduction of forest cover in the sapling-pole successional class which is important to this species. The declines have occurred on all ownerships.

National forests will continue to provide the major source of grouse habitat and opportunities to hunt this species. While demand



Proportion of Counties by Density Class for Ruffed Grouse

Figure 3.57 The spatial distribution of ruffed grouse county population density estimates for 1970 and 1995 for the Southern Appalachian Assessment area.

is expected to remain near current levels on national forests, populations and habitat quality are expected to decrease through the year 2010.

Bobwhite Quail

Bobwhite populations are low throughout the Northern Ridge and Valley, Allegheny Mountains, and Northern Cumberland Mountains (section group 1); the majority of the Blue Ridge Mountains (section group 2); and the Northern Cumberland Plateau and Southern Cumberland Mountains (section group 3) (fig. 3.58). Medium population densities are found in much of the remaining portions of the assessment area. Medium densities are associated with a greater proportion of nonforest cover, especially herbaceous cover and pastureland (table 3.20). Quail populations are slightly lower on national forest lands, national parks, and the Cherokee Indian Reservation than on remaining ownerships (table 3.19).

Bobwhite population densities have declined during the last 25 years. In 1970, less than 50 percent of counties had a low bobwhite density. Five percent had a high density. Today over 70 percent of counties has a low density and none has a high density. This pattern of decrease generally is consistent throughout the assessment area and within ownerships. The decline in quail populations probably is largely a result of the loss of agricultural land in the region as well as changes in agricultural practices.

It is expected that habitat for quail will continue to decrease due to shifts of agricultural

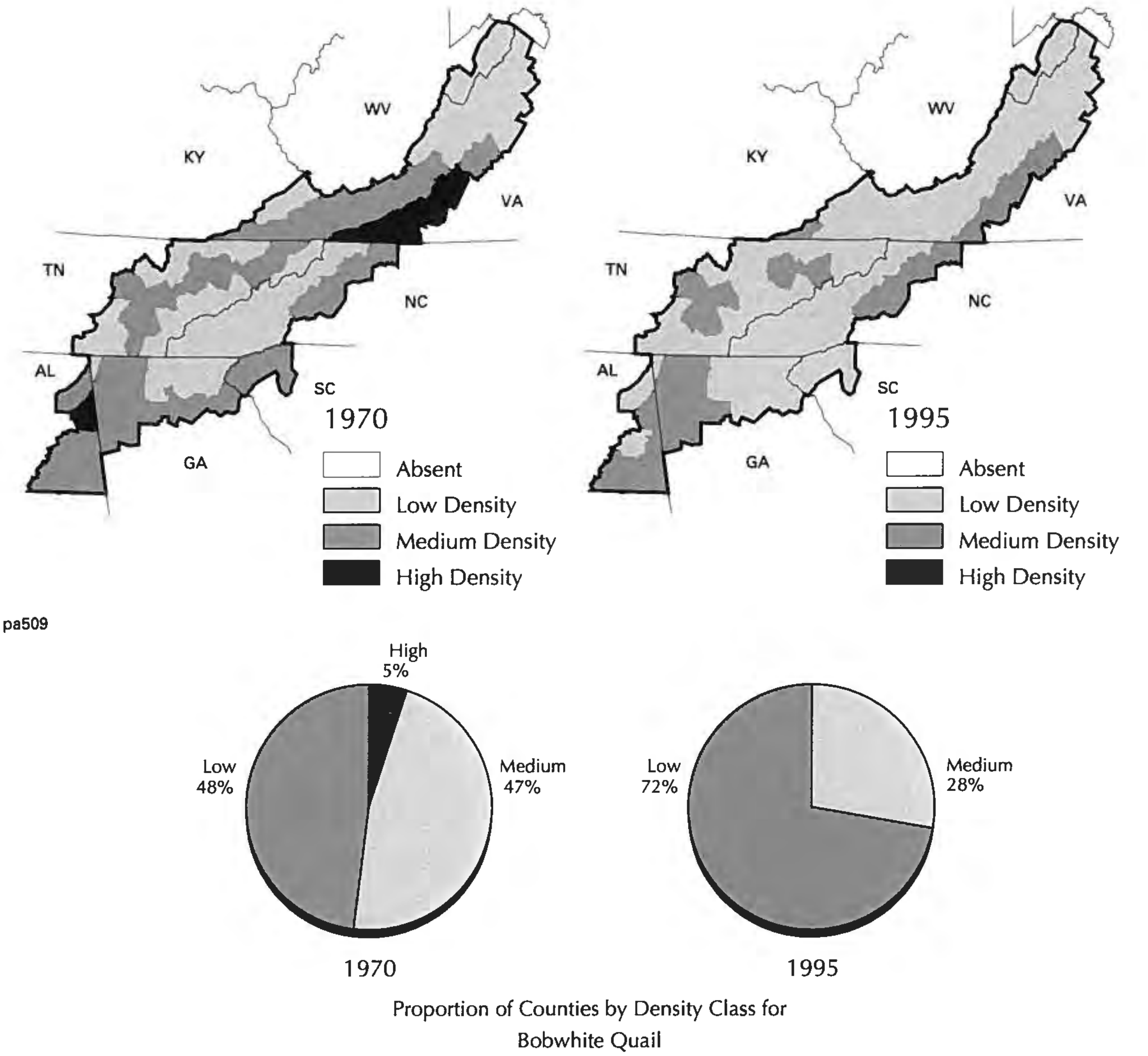


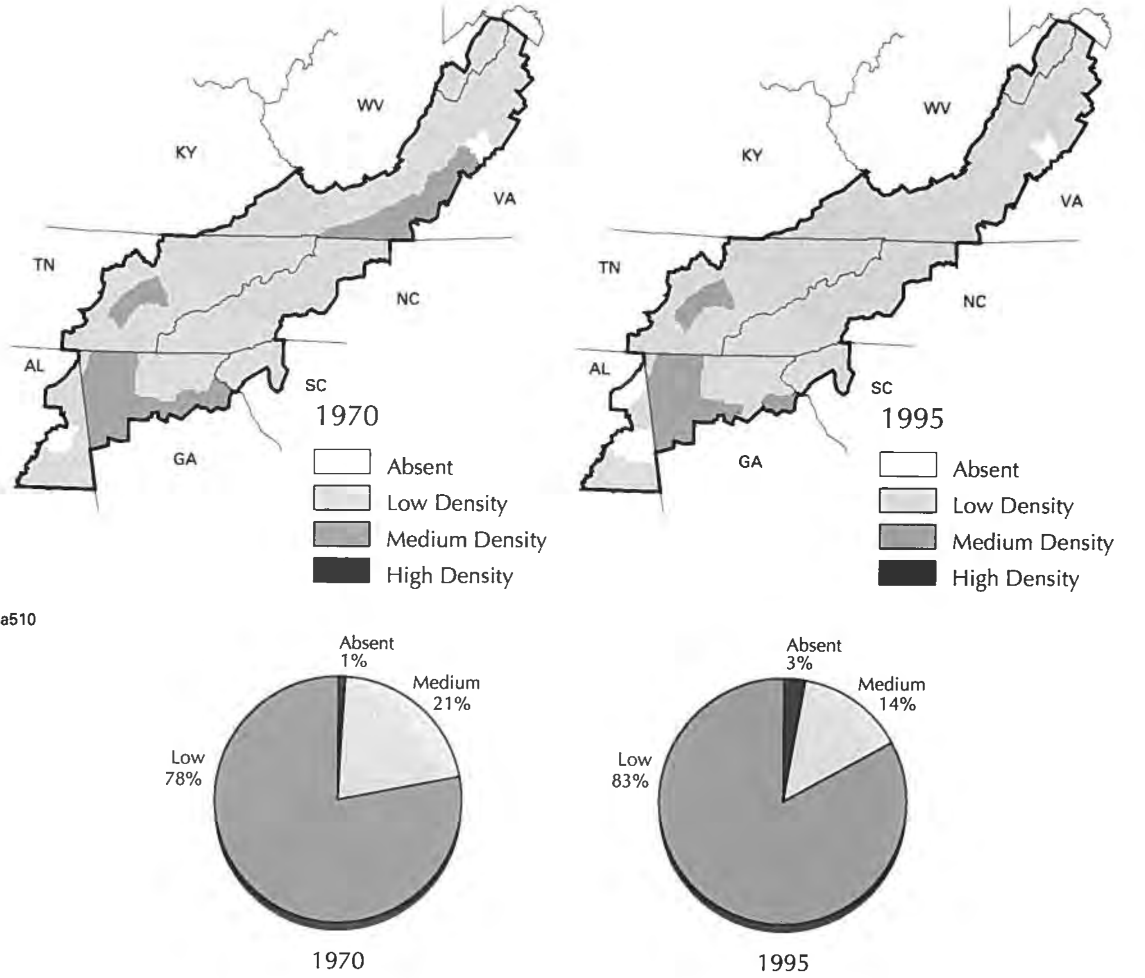
Figure 3.58 The spatial distribution of bobwhite quail county population density estimates for 1970 and 1995 for the Southern Appalachian Assessment area.

lands to improved pasture and a continuing isolation of suitable early successional grass/shrub and cropland habitats.

American Woodcock

Woodcock populations generally are low in most of the assessment area (fig. 3.59). Only the Southern Cumberland Plateau and Southern Ridge and Valley (section group 5) and the Southern Appalachian Piedmont (section group 6) contain a substantial number of counties with medium population densities. Woodcock densities do not appear to be strongly related to any particular land use pattern, but medium density populations are associated with a greater proportion in water (table 3.20). Woodcock densities generally are similar among ownerships with the exception of other federal lands which contain limited woodcock populations (table 3.19).

Woodcock population densities have declined slightly since 1970. The proportion of counties with medium density decreased from 21 percent to 14 percent during the last 25 years while the number of counties where woodcock are absent increased slightly. This pattern generally is consistent throughout the assessment area. Loss of agricultural land may have contributed to this decline, but the effects appear to be much less than with other small game such as cottontails and quail. Slight declines have occurred in most of the ownership categories.



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Proportion of Counties by Density Class for American Woodcock

Figure 3.59 The spatial distribution of American woodcock county population density estimates for 1970 and 1995 for the Southern Appalachian Assessment area.

Landscape-Level Habitat Suitability Analysis for Selected Species Groupings

To identify broadscale habitat patterns within the assessment area, spatial analysis of habitat suitability was conducted for 10 of the 19 species groups. These species groups were selected because their habitat associations lend themselves to broad, landscape-level analysis using remote sensing data. Given the scale of analysis and available data, suitability analysis was not attempted for species groups with either highly specific habitat requirements (e.g., spray cliff species, high pH, or mafic species) or very general requirements (e.g., habitat generalist species). Seven habitat suitability models were developed: (1) Area-sensitive mid to late-successional deciduous forest species (SG 13); (2) General high-elevation forest species (SG 14); (3) Seeps, springs, and streamside species (SG 11); (4) High-elevation bald–early successional species–early successional grass-shrub species (SG 5 & 8); (5) Closed canopy deciduous forest species (SG 10, 16, and 18); (6) High-elevation spruce-fir/northern hardwood forest species (SG 15); and (7) black bear (SG 9).

It should be noted that these landscape-level models represent only gross habitat suitability based on general habitat requirements. Many species included have very specific, micro-habitat requirements not discernible in a broadscale analysis. Therefore, results of the suitability models should be viewed as providing a regional-scale picture of habitat potential among ownerships and ecological units rather than an indication of site-specific presence or absence of a particular species or group.

The assumptions and decision rules developed for each habitat suitability model were based on information contained in the species habitat matrix and pertinent literature sources. The primary data source for modeling was the 17-class, land-use data derived from LANDSAT Thematic Mapper™ imagery. Supplementary data included 1:100,000 scale ownership, road, and elevation coverages and water-stream reach coverage. The remote sensing and supplementary data provided a spatial analysis of habitat suitability at the landscape scale. However, as forest successional stages could not be determined with the imagery data, the acres of suitable habitat derived from the models were often overestimated. This was especially true for species groups utilizing mid- to late-successional forest habitat since the suitable acres derived from imagery data included all successional stages from late-successional forest down to older seedling-shrub stands. To compensate, all tabular data was adjusted for successional stage distribution using CISC data for national forest lands and FIA data for other ownerships. Because of the length of time since establishment, all forestland on national parks property was assumed to be in mid- to late-successional stages, and the forest acres derived from imagery data were not adjusted on these lands. Similar successional stage adjustments were made for the tabular summaries of suitable acres by section group using FIA successional stage information for each section group.

Area-sensitive Mid- to Late-Successional Deciduous Forest Species (SG 13)

This species group includes 16 birds associated with mid- to late-successional deciduous forests, including many neo-tropical migrant species (table 3.22). All the species included in this group are considered to be area-sensitive, requiring continuous forested tracts ranging in size from 2 to 4,325 acres. Many also avoid forest edges during nesting and, therefore, are considered forest interior species.

Model Development

Based on habitat associations presented in the habitat matrix, this species group is primarily associated with northern hardwood, mixed mesophytic hardwood, oak, mixed pine-hardwood, and bottomland hardwood forests. Forest stands of these types were selected from the remote sensing data and classified as suitable habitat. To represent significant canopy breaks undetected in the imagery data, these suitable forest areas were then overlain with images of major roads (Class 1, 2, and 3) and railroad and power line rights-of-way to define suitable tracts. Suitable habitat tracts were stratified by tract size class (<50, 50 to 99, 100 to 999, 1,000 to 4,999, 5,000+ acres) for spatial display. The data, adjusted for successional stage distribution as discussed above, were further stratified by ownership and section group to produce the tabular summaries.

Table 3.22 The area sensitive, mid- to late-successional deciduous forest species (species group 13) for the Southern Appalachian Assessment area.

Common Name	Scientific Name
Cerulean warbler	*Dendroica cerulea*
Black–billed cuckoo	*Coccyzus erythropthalmus*
Black–throated green warbler	*Dendroica virens*
Worm–eating warbler	*Helmitheros vermivorus*
Wood thrush	*Hylocichla mustelina*
Swainson's warbler	*Limnothlypis swainsonii*
Kentucky warbler	*Oporornis formosus*
Northern parula	*Parula americana*
Scarlet tanager	*Piranga olivacea*
Summer tanager	*Piranga rubra*
Ovenbird	*Seiurus aurocapillus*
Yellow–throated vireo	*Vireo flavifrons*
Hooded warbler	*Wilsonia citrina*
Pileated woodpecker	*Dryocopus pileatus*
Red–bellied woodpecker	*Melanerpes carolinus*
Hairy woodpecker	*Picoides villosus*

Two approaches were used to examine the effects of edge on habitat suitability. In the first approach, the degree of edge effects was varied between section groups based on the dominant landscape (forest vs. agricultural) in a section group. Levels of nest parasitism and predation have been shown to be negatively related to the amount of forest cover in the landscape (Robinson and others 1995). More significant edge effects can be expected in highly fragmented landscapes. For this analysis, each section group was classified as either forest dominated (>75 percent forest, <15 percent agriculture) or agricultural dominated (>15 percent agriculture, <75 percent forest) based on remote sensing data. The Blue Ridge Mountains (section group 2) and the Northern Cumberland Plateau and Northern Cumberland Mountains (section group 3) met the forest dominated criteria (84 percent and 79 percent forested, 11 percent and 14 percent agricultural, respectively). All other section groups were classified as agricultural dominated (18 to 35 percent agriculture, 49 to 68 percent forested).

In the agriculture dominated landscapes, edge habitat was defined as a buffer of approximately 300 feet (one 90 m cell) on each side of all roads (Classes 1 to 4), railroad and power line rights-of-way and all lands classified as herbaceous, cropland, pasture, developed, or barren from the imagery data. In the forest-dominated landscapes, edge habitat was defined as a similar buffer of land classified as cropland, pasture, developed, or barren only. Edge habitats were subtracted from suitable habitat tracts to define interior habitat.

In the second approach, edge effects were held constant across the assessment area, irrespective of the dominant landscape. Edge and interior habitats for the entire area were defined using the criteria utilized for agricultural dominated landscapes above.

Analysis and Results

There are approximately 15.8 million acres of suitable habitat for mid- to late-successional deciduous forest species in the SAA area (fig. 3.60, table 3.23). About 8.2 million acres (52 percent) are in tracts greater than 5,000 acres. Approximately 70 percent of suitable habitat and 51 percent of the largest tracts are on private land, while 23 percent of suitable habitat and 39 percent of the habitat in tracts greater than 5,000 acres are on national forest land. The majority of suitable habitat for this species group is in the Blue Ridge Mountains (section group 2) and the Northern Ridge and Valley, Allegheny Mountains, and Northern Cumberland Mountains (section group 2) (table 3.24).

The proportion of suitable habitat in forest edge habitat is highest on private land (other) and DOE/military lands (other federal) and lowest on national park and national forest land

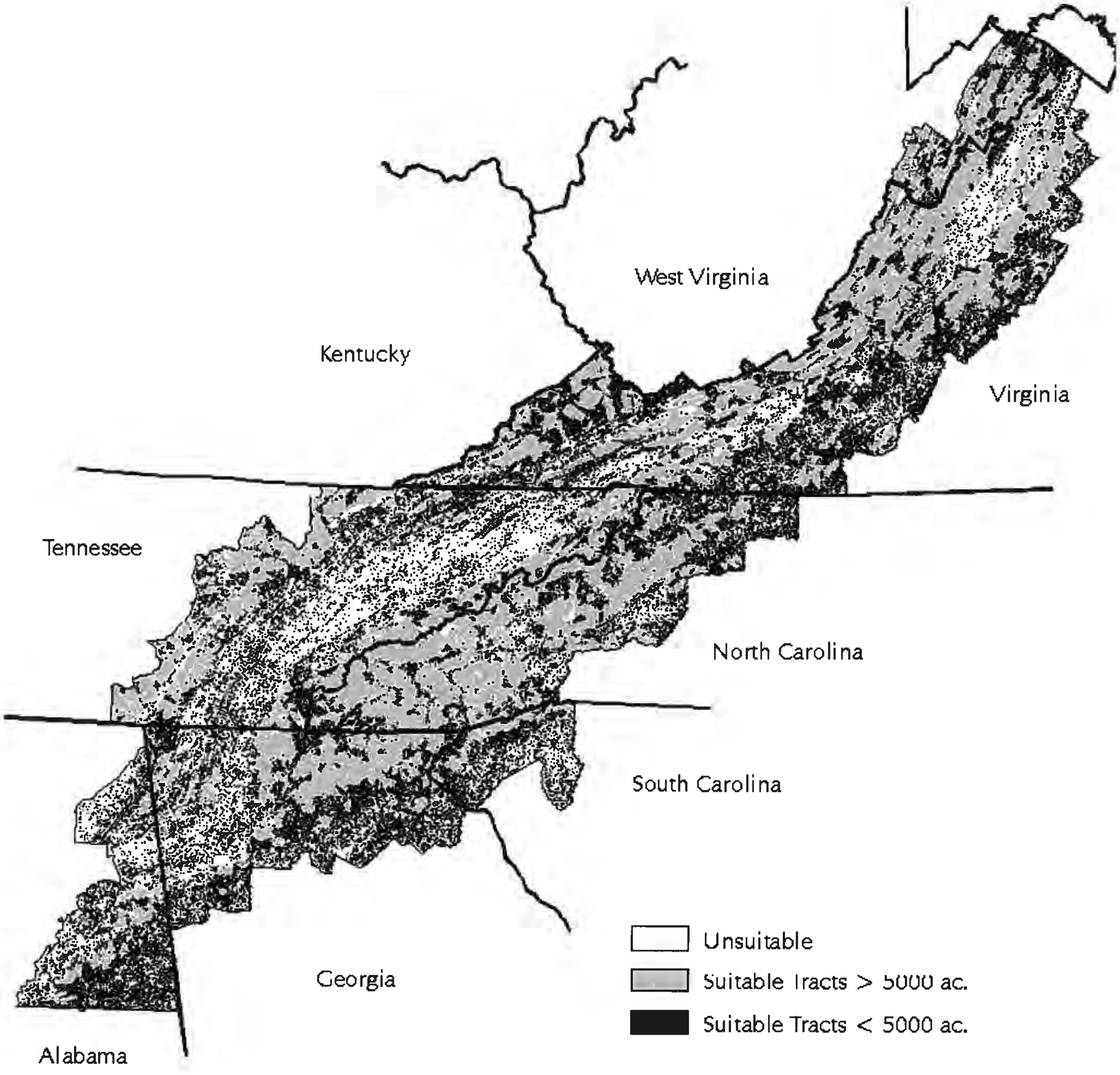

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Figure 3.60 Spatial distribution of suitable habitat for area-sensitive, mid- to late-successional deciduous forest species (species group 13) by tract size classes for the SAA area.

Table 3.23 The acres of suitable habitat for area sensitive, mid–late successional deciduous forest species (species group 13) and proportion in edge habitat by ownership for the Southern Appalachian Assessment Area.

Ownership	Tract Size Class <50 Acres	50–100 Acres	100–1000 Acres	1000–5000 Acres	>5000 Acres	Total
National Forest						
Talladega	2,894	538	6,583	47,650	105,308	162,973
Chattahoochee	3,727	544	10,280	42,311	532,227	589,089
Pisgah/Nantahala	3,444	1,098	12,751	66,592	762,880	846,765
Sumter	1,006	217	2,266	17,689	37,034	58,211
Cherokee	4,638	1,294	17,310	72,743	396,373	492,358
George Washington	1,976	224	9,611	94,588	801,084	907,483
Jefferson	917	448	5,915	76,659	510,043	593,983
Monongahela	2,087	382	4,873	8,352	46,808	62,502
Total National Forest	20,689	4,745	69,589	426,584	3,191,757	3,713,364
National Parks	7,890	3,359	16,339	22,752	611,176	661,516
Cherokee Indian Reservation	259	109	657	6,872	15,424	23,321
Other Federal	3,286	1,527	10,806	10,412	9,204	35,235
State	4,338	1,959	19,584	51,408	212,406	289,695
Private	884,407	359,478	2,729,867	2,903,839	4,208,506	11,086,097
Total	920,869	371,177	2,846,842	3,421,867	8,248,473	15,809,228

Ownership	% Edge Variable[1]	Constant[2]
National Forest		
Talladega	21	21
Chattahoochee	6	25
Pisgah/Nantahala	3	22
Sumter	5	32
Cherokee	4	32
George Washington	19	21
Jefferson	12	18
Monongahela	41	41
Total National Forest	10	23
National Parks	3	24
Cherokee Indian Reservation	13	29
Other Federal	51	51
State	20	28
Private	42	49
Total	34	43

[1] Variable edge effects among section groups (see text for further explanation).

[2] Constant edge effects among section groups.

(table 3.23). When edge effects were varied based on the dominant landscape, the proportion of suitable habitat in forest interior habitat ranged from 97 percent (three percent edge habitat) on national parks to 49 percent (51 percent edge habitat) on other federal lands. When edge effects were held constant across the assessment area, the proportion of suitable habitat in forest interior habitat ranged from 77 percent (23 percent edge habitat) on national forest lands to 49 percent (51 percent edge habitat) on other federal lands. The proportion of suitable habitat in forest edge was highest in the Central Ridge and Valley (section group 4) and lowest in the Blue Ridge Mountains (section group 2) and Northern Cumberland Plateau and Southern Cumberland Mountains (section group 3) for both approaches (table 3.24).

Based on past trends in land use, it is expected that, over the next 15 years, suitable acreage in large tract sizes and associated forest interior habitats will continue to decrease due to loss of forestland to other land uses such as agricultural pasture and development. These decreases may continue to be most evident in the section/section groups currently with less

Table 3.24 The acres of suitable habitat for area sensitive, mid–late successional deciduous forest species (species group 13) and proportion in edge habitat by section group for the Southern Appalachian Assessment area.

Section Group[1]	Tract Size Class <50 Acres	50–100 Acres	100–1000 Acres	1000–5000 Acres	>5000 Acres	Total
Section Group 1	195,167	80,579	573,139	922,917	2,831,189	4,602,991
Section Group 2	191,252	66,246	647,306	1,107,890	4,041,319	6,054,013
Section Group 3	27,626	10,208	94,951	197,670	726,900	1,057,355
Section Group 4	192,049	67,037	387,884	245,288	168,302	1,060,560
Section Group 5	110,901	47,360	248,687	199,966	179,191	786,105
Section Group 6	203,874	99,747	894,875	748,137	301,571	2,248,204

Section Group[1]	% Edge Variable[2]	Constant[3]
Section Group 1	37	37
Section Group 2	17	36
Section Group 3	18	35
Section Group 4	63	63
Section Group 5	52	52
Section Group 6	56	56

[1] Section Groups:
1 = Northern Ridge and Valley, Allegheny Mountains, Northern Cumberland Mountains
2 = Blue Ridge Mountains
3 = Northern Cumberland Plateau, Southern Cumberland Mountains
4 = Central Ridge and Valley
5 = Southern Cumberland Plateau, Southern Ridge and Valley
6 = Southern Appalachian Piedmont

[2] Variable edge effects among section groups (see text for further explanation).

[3] Constant edge effects among section groups.

than 70 percent of the area forested. These decreases should be seen primarily on other private lands.

General High-Elevation Forest Species (SG 14)

This group includes seven species associated with high-elevation forests (table 3.25). Included are three area-sensitive birds. This species group is primarily associated with mid- to late-successional montane spruce-fir, northern hardwood, white pine-hemlock-hardwood, and mixed mesophytic hardwood forests.

Model Development

Because of the confounding influences of latitude and elevation on distribution of plant species and wildlife habitat, a latitudinally adjusted elevation was used to define the elevation breakpoint for the high-elevation class. Based on field knowledge, it was defined as 3,500 feet at the northern end of Great Smoky Mountains National Park and 2,800 feet at the northern end of Shenandoah National Park. These data points were used to develop a linear equation defining high-elevation habitat at any latitude in the assessment area. Values of the

Table 3.25 The general high elevation forest species (species group 14) for the Southern Appalachian Assessment area.

Common Name	Scientific Name
Fragile supercoil	*Glyphyalina clingmani*
Roan supercoil	*Paravitrea varidens*
Fringed scorpion–weed	*Phacelia fimbriata*
Black–throated blue warbler	*Dendroica caerulescens*
Blackburnian warbler	*Dendroica fusca*
Red crossbill	*Loxia curvirostra*
Canada warbler	*Wilsonia canadensis*

Table 3.26 The acres of suitable habitat for general high elevation forest species (species group 14) and proportion in edge habitat by ownership for the Southern Appalachian Assessment Area.

	Tract Size Class					
Ownership	<50 Acres	50–100 Acres	100–1000 Acres	1000–5000 Acres	>5000 Acres	Total
National Forest						
Talladega	0	0	0	0	0	0
Chattahoochee	498	0	0	0	0	498
Pisgah/Nantahala	22,452	2,500	4,586	9,549	6,883	45,970
Sumter	0	0	0	0	0	0
Cherokee	12,543	1,913	1,590	451	0	16,497
George Washington	3,377	49	0	0	0	3,426
Jefferson	3,198	147	944	2,126	0	6,415
Monongahela	1,340	109	1,332	3,697	18,534	25,012
Total National Forest	43,408	4,718	8,452	15,823	25,417	97,818
National Parks	12,946	1,933	8,760	3,965	110,020	137,624
Cherokee Indian Reservation	1,348	525	1,427	203	10	3,512
Other Federal	0	0	0	0	0	0
State	1,192	21	0	5	858	2,076
Private	75,302	7,072	9,348	9,803	12,917	114,444
Total	134,196	14,269	27,987	29,799	149,222	355,474

	% Edge	
Ownership	Variable[1]	Constant[2]
National Forest		
Talladega		
Chattahoochee	2	22
Pisgah/Nantahala	2	16
Sumter		
Cherokee	3	22
George Washington	7	10
Jefferson	3	21
Monongahela	27	27
Total National Forest	9	20
National Parks	<1	7
Cherokee Indian Reservation	2	28
Other Federal	–	–
State	3	20
Private	20	37
Total	10	22

[1] Variable edge effects among section groups (see text for further explanation).

[2] Constant edge effects among section groups.

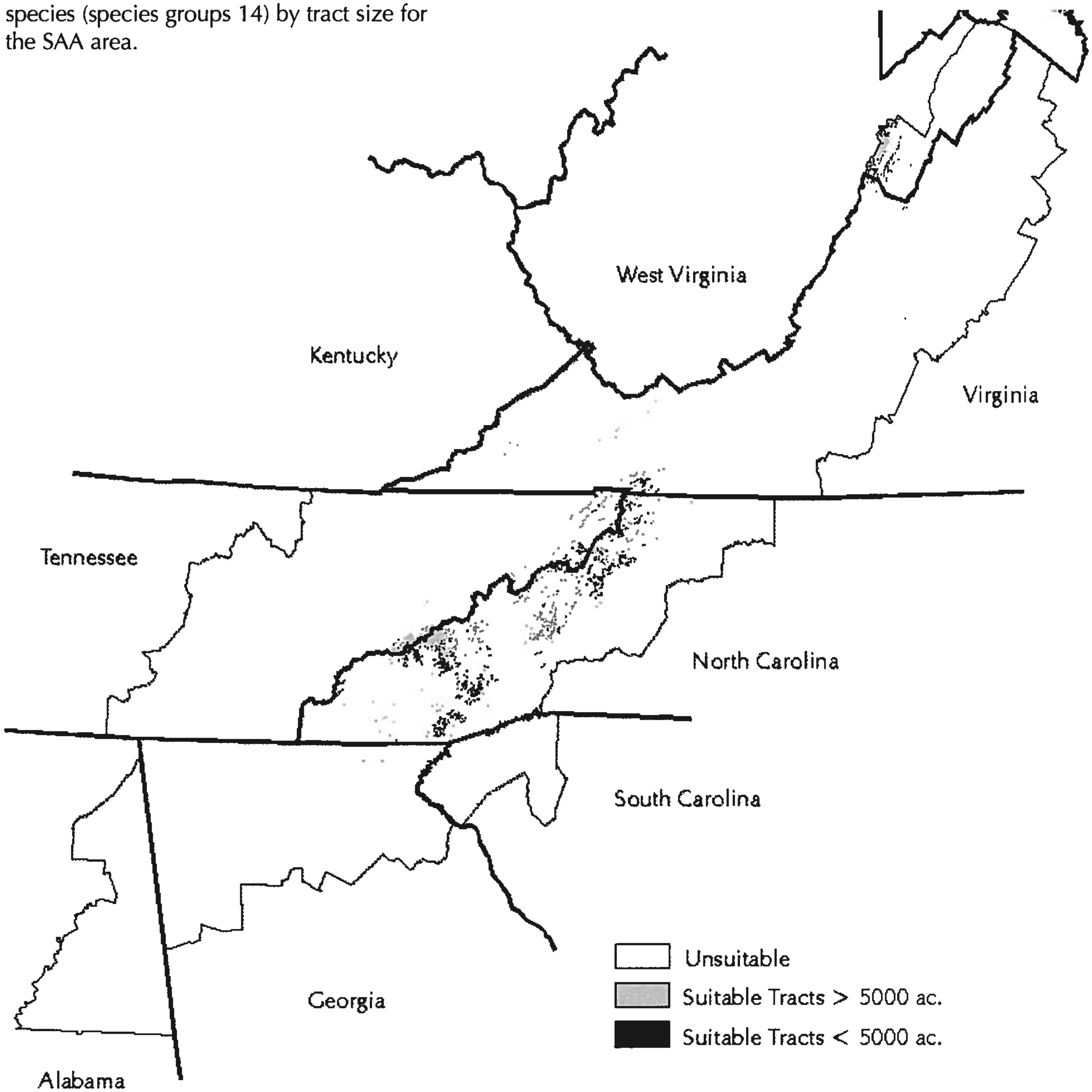
elevation breakpoints ranged from 3,970 feet at the extreme southern end of the assessment area to 2,660 at the extreme northern end.

Suitable habitat was defined as forest stands of the appropriate types occurring at higher elevations as defined above. To represent significant canopy breaks undetected by the imagery data, these suitable forest areas were then overlain with major roads (Class 1, 2, and 3) and railroad and power line rights-of-way to define suitable tracts. These were stratified by tract size-class (<50, 50 to 99, 100 to 999, 1,000 to 4,999, 5,000+ acres) for spatial display. The data, adjusted for successional stage distribution, were further stratified by ownership and section group to produce tabular summaries. Edge effects were examined using the two approaches outlined above.

Analysis and Results

Approximately 355,000 acres of high-elevation forest are in the assessment area, of which 149,000 acres (42 percent) are in tracts larger than 5,000 acres (table 3.26, fig. 3.61). These large tracts have potential to support all seven general high-elevation forest species.

Figure 3.61 Spatial distribution of suitable habitat for general high-elevation forest species (species groups 14) by tract size for the SAA area.



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Approximately 74 percent of the area in these large tracts is in national parks. National forests contain approximately 28 percent of this habitat type and 17 percent of the area in tracts greater than 5000 acres. The majority (83 percent) of high-elevation forest is in the Blue Ridge Mountains (section group 2) (table 3.27).

The proportion of suitable habitat in edge was highest on private land and lowest on national parks (table 3.26). When edge effects were varied based on the dominant landscape, the proportion of suitable habitat in edge ranged from <1 percent on national park land to 20 percent on private land. When edge effects were held constant across the assessment area, the proportion of suitable habitat in edge ranged from 7 percent on national forest land to 37 percent on private.

The outlook for these forest communities and the seven species associated with these general high-elevation habitats is uncertain due to the negative effects caused by air pollution and exotic pests. A downward trend for these habitats is probable over the next 15 years.

Table 3.27 The acres of suitable habitat for general high elevation forest species (species group 14) and proportion in edge habitat by section group for the Southern Appalachian Assessment area.

	Tract Size Class					
Section Group[1]	<50 Acres	50–100 Acres	100–1000 Acres	1000–5000 Acres	>5000 Acres	Total
Section Group 1	15,293	1,189	3,254	8,455	30,735	58,92
Section Group 2	118,156	13,080	24,733	21,344	118,487	295,800
Section Group 3	734	0	0	0	0	734
Section Group 4	12	0	0	0	0	12

	% Edge	
Section Group[1]	Variable[2]	Constant[3]
Section Group 1	25	25
Section Group 2	7	22
Section Group 3	3	32
Section Group 4	66	66

[1] Section Groups:
1 = Northern Ridge and Valley, Allegheny Mountains, Northern Cumberland Mountains
2 = Blue Ridge Mountains
3 = Northern Cumberland Plateau, Southern Cumberland Mountains
4 = Central Ridge and Valley

[2] Variable edge effects among section groups (see text for further explanation).

[3] Constant edge effects among section groups.

Seeps, Springs, and Streamside Species (SG 11)

This group includes species associated with forested riparian areas as well as those found in spring-heads, seeps, and river gravel bars (table 3.28). Due to limitations of the remote sensing data, habitat suitability modeling was attempted only for forested riparian habitat. Species associated with forested riparian habitat included salamanders and fewer numbers of plants, birds, and mammals.

Model Development

Using the water-stream reach coverage, riparian habitat was defined as the area approximately 100 feet (one 30 m pixel) on each side of stream segments and 100 feet along the shoreline of water bodies. This riparian buffer was overlain with imagery data to determine acres of riparian habitat by land-use class. Acres of forested riparian habitat and proportion of total riparian habitat in forest cover were stratified by ownership and section group to produce the tabular summaries.

Analysis and Results

There are approximately 2.3 million acres of riparian habitat in the assessment area, 1.5 million acres (65 percent) of which is in forest cover (table 3.29). Approximately 80 percent of the forested riparian habitat is on private (other) land. The proportion of riparian habitat in forest cover is highest on national forest and park land (97 percent and 94 percent) and lowest on private and other federal land (60 percent and 65 percent). By section group, the proportion of riparian habitat in forest cover ranged from 79 percent in the Northern Cumberland Plateau and Southern Cumberland Mountains (section group 3) to 43 percent in the Central Ridge and Valley (section group 4) (table 3.30).

High-Elevation Bald/Early Successional Grass-Shrub Species (SG 5 & 8)

These two groups include species associated with open conditions, including early successional forests, grassy and heath balds, and old fields (tables 3.31 and 3.32).

Model Development

Since the grass-forb, early successional class from the remote sensing imagery best represents these habitat types, the two species groups were combined for analysis. The results are, at best, a conservative estimate of actual

Table 3.28 The riparian species from the seeps, springs, and streamside species group (species group 11) for the Southern Appalachian Assessment area.

Common Name	Scientific Name
A hornwort	*Aspiromitus appalachianus*
Virginia round–leaf birch	*Betula uber*
Seepage salamander	*Desmognathus aeneus*
Dark–sided (Brownback) salamander	*Eurycea aquatica*
Junaluska salamander	*Eurycea junaluska*
Southern water shrew	*Sorex palustris punctulatus*
Bittercress	*Cardamine flagellifera*
Sweet indian plantain	*Hasteola suaveolens*
Broadleaf Barbara's buttons	*Marshallia trinervia*
Ruth's golden aster	*Pityopsis ruthii*
Heart–leaf plantain	*Plantago cordata*
Harperella	*Ptilimnium nodosum*
Rock goldenrod	*Solidago rupestris*
Virginia spiraea	*Spiraea virginiana*
Arrowwood	*Viburnum bracteatum*
Sand grape	*Vitus rupestris*
American woodcock	*Scolopax minor*
Raccoon	*Procyon lotor*
Imitator salamander	*Desmognathus imitator*
Blackbelly salamander	*Desmognathus quadramaculatus*
Santeetlah dusky salamander	*Desmognathus santeetlah*
Black Mountain salamander	*Desmognathus welteri*
Pigmy salamander	*Desmognathus wrighti*
Blue Ridge two–lined salamander	*Eurycea wilderae*
Shovelnose salamander	*Leurognathus marmoratus*
Jordan's salamander	*Plethodon jordani*
Cumberland Plateau salamander	*Plethodon kentucki*
Yonahlossee salamander	*Plethodon yonahlossee*
Acadian flycatcher	*Empidonax virescens*
Louisiana waterthrush	*Seiurus motacilla*
Beaver	*Castor canadensis*

Table 3.29 The acres of suitable habitat for forest riparian species (species group 11) and proportion of total riparian habitat in forest cover by ownership for the Southern Appalachian Assessment Area.

Ownership	Acres	% of Riparian Habitat in Forest Cover
National Forest		
Talladega	9,802	95
Chattahoochee	37,595	98
Pisgah/Nantahala	50,620	95
Sumter	4,851	97
Cherokee	37,621	96
George Washington	50,353	98
Jefferson	32,131	98
Monongahela	3,094	82
Total National Forest	226,005	97
National Parks	41,935	94
Cherokee Indian Reservation.	1,960	81
Other Federal	4,320	65
State	30,312	84
Private	1,186,090	60
Total	1,490,622	65

Table 3.30 The acres of suitable habitat for forest riparian species (species group 11) and proportion of total riparian habitat in forest cover by section group for the Southern Appalachian Assessment Area.

Section Group[1]	Acres	% of Riparian Habitat in Forest Cover
Section Group 1	309,666	64
Section Group 2	478,914	76
Section Group 3	117,350	79
Section Group 4	135,488	43
Section Group 5	148,948	56
Section Group 6	300,256	69

[1] Section Groups:
1 = Northern Ridge and Valley, Allegheny Mountains, Northern Cumberland Mountains
2 = Blue Ridge Mountains
3 = Northern Cumberland Plateau, Southern Cumberland Mountains
4 = Central Ridge and Valley
5 = Southern Cumberland Plateau, Southern Ridge and Valley
6 = Southern Appalachian Piedmont

Table 3.31 The high elevation bald species (species group 5) for the Southern Appalachian Assessment area.

Common Name	Scientific Name
Allegheny onion	*Allium allegheniense*s
Appalachian Bewick's wren	*Thryomanes bewickii altus*
Appalachian cottontail	*Sylvilagus obscurus*
Appalachian gentian	*Gentiana austromontana*
Bent avens	*Geum geniculatum*
Spreading avens	*Geum radiatum*
Roan Mountain bluet	*Hedyotis purpurea var. montana*
Blue Ridge St. John's–wort	*Hypericum buckleyi*
Mountain St. John's–wort	*Hypericum graveolens*
Mitchell's St. John's–wort	*Hypericum mitchellianum*
Heller's blazing star	*Liatris helleri*
Gray's lily	*Lilium grayi*
Roan rattlesnakeroot	*Prenanthes roanensis*
Carolina rhododendron	*Rhododendron carolinianum*
Cumberland azalea	*Rhododendron cumberlandense*
Clammy locust	*Robinia viscosa var. viscosa*
Blue Ridge goldenrod	*Solidago spithamaea*
Chestnut–sided warbler	*Dendroica pensylvavica*

Table 3.32 The early successional grass/shrub species (species group 8) for the Southern Appalachian Assessment area.

Common Name	Scientific Name
Blue Ridge bindweed	*Calystegia catesbiana spp. sericata*
Bachman's sparrow	*Aimophila aestivalis*
Henslow's sparrow	*Ammodramus henslowii*
Loggerhead shrike	*Lanius ludovicianus*
Northern bobwhite	*Colinus virginianus*
Eastern cottontail	*Sylvilagus floridanus*
Prairie warbler	*Dendroica discolor*
Field sparrow	*Spizella pusilla*
Golden–winged warbler	*Vermivora chrysoptera*
Blue–winged warbler	*Vermivora pinus*

Table 3.33 The acres of suitable habitat for high elevation bald (species group 5) and early successional grass/shrub species (species group 8) by ownership for the Southern Appalachian Assessment area.

Ownership	Elevation Class[1]	Tract Size Class					
		<5ac	5–10ac	10–20ac	20–100ac	>100ac	Total
National Forest							
Talladega	<3500'	712	652	887	1,584	144	3,979
	>3500'	0	0	0	0	0	0
Chattahoochee	<3500'	1,479	1,286	1,968	3,700	83	8,516
	>3500'	19	5	0	24	0	48
Pisgah/Nantahala	<3500'	992	576	667	709	0	2,944
	>3500'	516	359	372	513	118	1,878
Sumter	<3500'	140	151	204	181	2	678
	>3500'	0	0	0	0	0	0
Cherokee	<3500'	980	839	880	1,395	3	4,097
	>3500'	98	98	65	165	0	426
George Washington	<3500'	269	242	236	360	10	1,117
	>3500'	202	276	226	761	119	1,584
Jefferson	<3500'	566	375	222	157	7	1,327
	>3500'	254	144	245	569	1,257	2,469
Monongahela	<3500'	48	19	77	57	0	201
	>3500'	28	43	45	23	0	139
Total National Forest	<3500'	5,186	4,140	5,141	8,143	249	22,859
	>3500'	1,117	925	953	2,055	1,494	6,544
National Parks	<3500'	527	446	310	696	561	2,540
	>3500'	256	84	68	62	155	625
Cherokee Indian Reservation	<3500'	107	48	31	0	0	186
	>3500'	26	5	66	0	0	97
Other Federal	<3500'	742	675	666	1,246	213	3,542
	>3500'	0	0	0	0	0	0
State	<3500'	2,342	1,918	2,040	2,604	2,474	11,378
	>3500'	51	60	88	90	166	455
Private	<3500'	208,746	200,101	226,517	474,549	351,241	1,461,154
	>3500'	3,906	3,046	3,017	4,984	4,021	18,974
Total	<3500'	217,649	207,328	234,705	487,237	354,737	1,502,656
	>3500'	5,355	4,119	4,192	7,193	5,835	26,694

[1] Elevation classes adjusted for latitudinal variation. See text for further explanation.

acres for these habitats. Areas classified as grass-forb by the imagery data were stratified into high and low elevation using the latitudinal elevation break discussed under general high-elevation model development. The high elevation represented habitats for SG 5 and the low elevation represented habitats for SG 8. Data were further stratified by tract size, ownership, and section group to produce the tabular summaries.

Analysis and Results

There are approximately 1.5 million acres of early successional habitat at lower elevations and 27,000 acres above 3,500 feet (table 3.33). The majority of this habitat is on private lands. National forests provide 2 percent of the low-elevation, early successional habitat and 25 percent of the high-elevation, early successional habitat. For both elevation classes, approximately half of the early successional habitat is in tracts larger than 20 acres. The Southern Cumberland Plateau and Southern Ridge and Valley (section group 5) and Southern Appalachian Piedmont (section group 6) contain much of the low-elevation grass-shrub habitat (table 3.34). Eighty-six percent of the high-elevation early successional habitat is in the Blue Ridge Mountains (section group 2).

Acreage of high elevation bald habitats is expected to remain near or slightly above the current level over the next 15 years. However,

Table 3.34 The acres of suitable habitat for high elevation bald (species group 5) and early successional grass/shrub species (species group 8) by section group for the Southern Appalachian Assessment area.

Section Group[1]	Elevation Class[2]	Tract Size Class <5ac	5–10ac	10–20ac	20–100ac	>100ac	Total
Section Group 1	<3500'	33,019	30,372	33,211	59,759	9,886	166,247
	>3500'	721	677	705	1,365	119	3,587
Section Group 2	<3500'	33,870	25,860	23,110	27,039	6,500	116,379
	>3500'	4,553	3,393	3,437	5,778	5,716	22,877
Section Group 3	<3500'	20,411	17,645	17,832	26,015	17,759	99,692
	>3500'	81	49	49	50	0	229
Section Group 4	<3500'	32,002	27,840	27,786	46,559	13,589	147,776
	>3500'	1	0	0	0	0	1
Section Group 5	<3500'	32,859	36,511	49,054	175,916	251,456	545,796
	>3500'	0	0	0	0	0	0
Section Group 6	<3500'	65,458	69,099	83,712	151,950	55,548	425,767
	>3500'	0	0	0	0	0	0

[1] Section Groups:
1 = Northern Ridge and Valley, Allegheny Mountains, Northern Cumberland Mountains
2 = Blue Ridge Mountains
3 = Northern Cumberland Plateau, Southern Cumberland Mountains
4 = Central Ridge and Valley
5 = Southern Cumberland Plateau, Southern Ridge and Valley
6 = Southern Appalachian Piedmont

[2] Elevation classes adjusted for latitudinal variation. See text for further explanation.

the effects from air pollution could adversely affect quality of the remaining habitat. Populations of the rare species associated with this habitat will continue at low levels.

Acreage of early successional habitat at low elevations will probably remain near the current level. However, habitat quality for some associated species will continue to decrease due to continued loss of agricultural land to improved pasture. Continuing isolation of these habitats will result.

Closed Canopy Deciduous Forest Species (SG 10, 16, & 18)

These groups include species associated with closed-canopy, mid- to late-successional deciduous forests (tables 3.35, 3.36, and 3.37).

Model Development

Primary forest types include mixed mesophytic hardwood, oak, bottomland hardwood, white pine-hemlock-hardwood, northern hardwood, and mixed pine-hardwood forests. Forest stands of these types were selected from the remote sensing data and classified as suitable habitat. The data, adjusted for successional stage distribution as discussed above, were stratified by ownership and section group to produce the tabular summaries.

Analysis and Results

There are approximately 17 million acres of habitat in the assessment area for species requiring closed-canopy, deciduous forests (table 3.38). Approximately 71 percent of this habitat is on private land. National forest and

Table 3.35 The mid to late successional deciduous forest species (species group 10) for the Southern Appalachian Assessment area.

Common Name	Scientific Name
Lobed barren–strawberry	*Waldsteinia lobata*
Appalachian bugbane	*Cimicifuga rubifolia*
Eastern gray squirrel	*Sciurus carolinensis*
Eastern fox squirrel	*Sciurus niger*
Eastern wood–pewee	*Contopus virens*
Downy woodpecker	*Picoides pubescens*

Table 3.36 The bottomland forest species (species group 16) for the Southern Appalachian Assessment area.

Common Name	Scientific Name
Virginia cup–plant	*Silphium connatum*
Prothonotory warbler	*Protonotaria citrea*

Table 3.37 The mixed mesic forest species (species group 18) for the Southern Appalachian Assessment area.

Common Name	Scientific Name
Tiny anemone	*Anemone minima*
Price's potato–bean	*Apios priceana*
Anderson's brachymenium	*Brachymenium andersonii*
Piratebush	*Buckleya distichophylla*
Peaks of Otter salamander	*Plethodon hubrichti*
Cow Knob salamander	*Plethodon punctatus*
A millipede	*Brachoria dentata*
Hungry Mother millipede	*Brachoria ethotela*
Big Ceder Creek millipede	*Brachoria falcifera*
Hoffman's xystodesmid millipede	*Brachoria hoffmani*
A millipede	*Brachoria separanda hamata*
Cedar millipede	*Brachoria cedra*
A millipede	*Buotus carolinus*
Venetia millipede	*Conotyla venetia*
A millipede	*Dixioria coronata*
A millipede	*Dixioria fowleri*
McGraw Gap xystodesmid	*Nannaria ericacea*
Shenandoah Mountain xystodesmid	*Nannaria shenandoah*
A millipede	*Pseudotremia alecto*
A millipede	*Rudiloria trimaculata tortua*
A millipede	*Semionellus placidus*
Diana fritillary butterfly	*Speyeria diana*
Manhart's sedge	*Carex manhartii*
Purple sedge	*Carex purpurifera*
Roan Mountain sedge	*Carex roanensis*
Liverwort	*Cheilolejeunea evansii*
	Collinsonia verticillata
Southern lady's–slipper	*Cypripedium kentuckiense*
White–leaved sunflower	*Helianthus glaucophyllus*
Appalachian little brown jug	*Hexastylis arifolia var. ruthii*
Mountain heartleaf	*Hexastylis contracta*
Dwarfflowered heartleaf	*Hexastylis naniflora*
French Broad heartleaf	*Hexastylis rhombiformis*
Small whorled pogonia	*Isotria medeoloides*
Butternut	*Juglans cinerea*
Fraser's loosestrife	*Lysimachia fraseri*
Broadleaf phlox	*Phlox amplifolia*
Pinkshell azalea	*Rhododendron vaseyi*
Highlands moss	*Schlotheimia lancifolia*
Large–flowered skullcap	*Scutellaria montana*
Short–styled Oconee bells	*Shortia galacifolia var. brevistyla*
Oconee bells	*Shortia galacifolia var. galacifolia*
Lance–leafed goldenrod	*Solidago lancifolia*
Mottled trillium	*Trillium discolor*
Persistent trillium	*Trillium persistens*
Least trillium	*Trillium pusillum*
Trillium	*Trillium pusillum var. monticulum*
Hairy blueberry	*Vaccinium hirsutum*

national park lands provide 23 percent and 4†percent of suitable habitat for these groups of species. This habitat type is found throughout the assessment area (table 3.39).

It is expected that these habitats will remain near or slightly higher than current levels over the next 15 years.

High-Elevation Spruce-Fir/Northern Hardwood Forest Species (SG 15)

This group includes species associated with high-elevation, mid- to late-successional spruce-fir and northern hardwood forests (table 3.40).

Model Development

Suitable habitat was defined as forest stands of the appropriate types occurring at higher elevations as defined in model development discussion for general high elevation species. Suitable habitat was then stratified into three elevational classes. For the latitude representing the northern portion of the Great Smoky Mountains National Park these classes were 3,500 to 4,800, 4,800 to 5,800, and >5,800 feet. These elevational classes, adjusted for latitude, ranged from 2,660 to 3,960; 3,960 to 4,960; and >4,960 feet at the extreme northern end of the assessment area to 3,970 to 5,270; 5,270 to 6,270; and >6,270 feet at the extreme southern end. The data, adjusted for successional stage distribution, were stratified by ownership and section group to produce the tabular summaries.

Analysis and Results

There are approximately 184,000 acres of high-elevation, spruce-fir northern hardwood forest in the assessment area (table 3.41, fig. 3.62). Approximately 47 percent of suitable habitat is on national park land, 32 percent on national forest land, and 20 percent on private land. Only 10,000 acres (6 percent) of this habitat occurs above 5,800 feet. Approximately 54 percent of the habitat above 5,800 feet is on national park land. The majority (73 percent) of the high-elevation, spruce-fir northern hardwood habitat is in the Blue Ridge Mountains (section group 2) (table 3.42).

The outlook for this community and the 23 species associated with these habitats is uncertain due to the negative effects of air pollution and exotic pests. A downward trend for these habitats is expected over the next 15 years.

Table 3.38 The acres of suitable habitat for closed canopy deciduous forest species (species groups 10, 16, and 18) in forest cover by ownership for the Southern Appalachian Assessment Area.

Ownership	Acres
National Forest	
Talladega	170,369
Chattahoochee	633,823
Pisgah/Nantahala	898,716
Sumter	71,921
Cherokee	531,908
George Washington	947,120
Jefferson	613,328
Monongahela	69,719
Total National Forest	3,936,904
National Parks	724,456
Cherokee Indian Reservation	25,552
Other Federal	41,884
State	306,782
Private	12,376,973
Total	17,412,904

Table 3.39 The acres of suitable habitat for closed canopy deciduous forest species (species groups 10, 16, and 18) in forest cover by section group for the Southern Appalachian Assessment Area.

Ownership	Acres
Section Group 1	4,775,736
Section Group 2	7,139,540
Section Group 3	1,213,410
Section Group 4	1,221,333
Section Group 5	2,055,703
Section Group 6	1,006,780

[1] Section Groups:
1 = Northern Ridge and Valley, Allegheny Mountains, Northern Cumberland Mountains
2 = Blue Ridge Mountains
3 = North Cumberland Plateau, Southern Cumberland Mountains
4 = Central Ridge and Valley
5 = Souther Cumberland Plateau, Southern Ridge and Valley
6 = Southern Appalachian Piedmont

Table 3.40 The high elevation spruce–fir/northern hardwood species (species group 15) for the Southern Appalachian Assessment area.

Common Name	Scientific Name
Cheat Mountain salamander	*Plethodon nettingi*
Spruce–fir moss spider	*Microhexura montivaga*
Carolina northern flying squirrel	*Glaucomys sabrinus coloratus*
Virginia northern flying squirrel	*Glaucomys sabrinus fuscus*
Fraser fir	*Abies fraseri*
Trailing wolfsbane	*Aconitum reclinatum*
Liverwort	*Bazzania nudicaulis*
Peak moss	*Brachydontium trichodes*
Rugel's ragwort	*Cacalia rugelia*
Northern goshawk	*Accipter gentilis*
Hoffman's cleidogonid millipede	*Cleidogona hoffmani*
A millipede	*Cleidogona lachesis*
A ghost moth	*Hepialus sciophanes*
Clingman covert	*Mesodon clingmanicus*
Fraser fir geometrid	*Semiothisa fraserata*
Appalachian oak fern	*Gymnocarpium appalachianum*
Mount Leconte moss	*Leptothymenium sharpii*
Liverwort	*Plagiochila corniculata*
Goldenrod	*Solidago glomerata*
Liverwort	*Sphenolobopsis pearsonii*
Clingman's hedgenettle	*Stachys clingmanii*
Purple turtlehead	*Chelone lyonii*
Northern saw–whet owl	*Aegolius acadicus*

Table 3.41 The acres of suitable habitat for high elevation spruce fir/northern hardwood forest species (species group 15) by ownership for the Southern Appalachian Assessment area.

	Elevation Class[1]			
Ownership	3500–4800'	4800–5800'	>5800'	Total
National Forest				
Talladega	0	0	0	0
Chattahoochee	10	0	0	10
Pisgah/Nantahala	5,396	17,441	2,807	25,644
Sumter	0	0	0	0
Cherokee	1,412	907	201	2,520
George Washington	782	0	0	782
Jefferson	396	3,710	51	4,157
Monongahela	22,925	3,311	0	26,236
Total National Forest	30,921	25,369	3,059	59,349
National Parks	34,140	47,190	5,535	86,865
Cherokee Indian Reservation	244	81	0	325
Other Federal	0	0	0	0
State	14	57	347	418
Private	22,744	13,518	1,218	37,480
Total	88,063	86,215	10,159	184,437

[1] Elevation classes adjusted for latitudinal variation. See text for further explanation.

Figure 3.62 Spatial distribution of suitable habitat for high-elevation spruce-fir/northern hardwood forest species (species groups 15) for the SAA area.

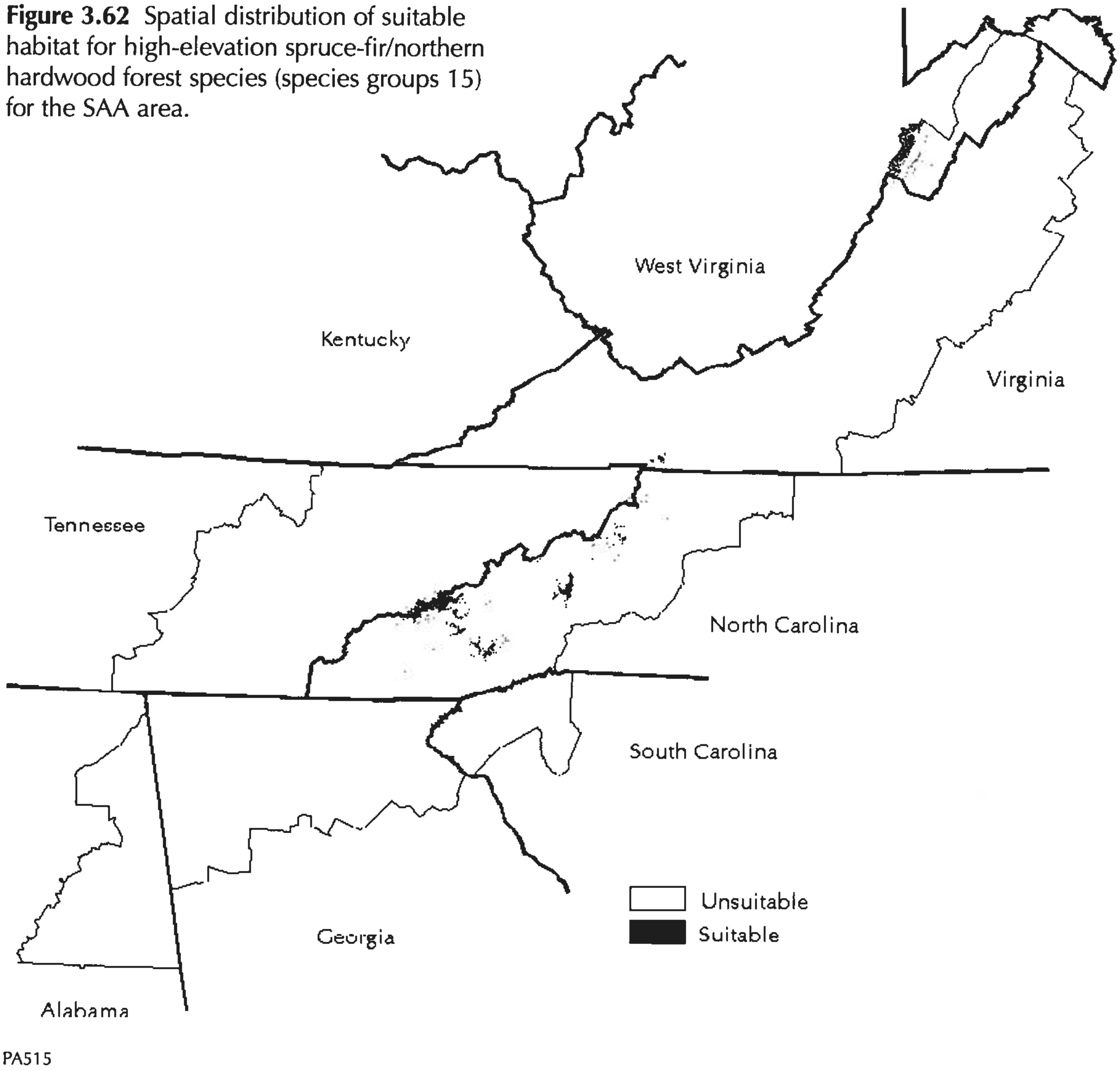

Table 3.42 The acres of suitable habitat for high elevation spruce fir/northern hardwood forest species (species group 15) by section group for the Southern Appalachian Assessment area.

	Elevation Class^2			
Section Group^1	3500–4800'	4800–5800'	>5800'	Total
Section Group 1	42,591	6,856	0	49,447
SectionGroup 2	45,472	79,359	10,159	134,990

[1] Section Groups:
1 = Northern Ridge and Valley, Allegheny Mountains, Northern Cumberland Mountains
2 = Blue Ridge Mountains

[2] Elevation classes adjusted for latitudinal variation. See text for further explanation.

Black Bear

Black bears are associated with a broad range of forest types and successional stages.

Model Development

For analysis, suitable land cover was defined as forest cover of any type, as well as herbaceous and wetland land cover. All areas within one-half mile of major highways (Class 1) or in tracts smaller than 10,000 acres were classified as unsuitable. The remaining tracts (suitable land cover, less than one-half mile from major highways, tracts >10,000 acres) were classified as potential habitat.

Because of the influence of roads on levels of poaching, highway mortality, and disturbance of bears, open-road density greatly affects habitat security (Brody and Pelton 1989, Van Manen 1991). Two approaches were used to evaluate open-road density of potential habitat tracts. First, total open-road density was calculated for each tract by dividing total miles of roads (all road classes) by the area of each tract. Potential habitat tracts were classified by open-road density class (<0.4; 0.4 to 0.8; 0.8 to 1.2; 1.2 to 1.6; 1.6 to 2.0; >2.0 mile per square mile) for spatial display. The second approach examined variability of open-road density within tracts. To do this, a road density surface was developed for the assessment area using a 1-square-mile grid. Then, withintract densities were stratified by density class, ownership, and section group to produce tabular summaries.

Analysis and Results

There are approximately 21 million acres of suitable bear habitat in the assessment area (table 3.43, fig. 3.63). Approximately 28 percent of the suitable habitat has relatively low open-road densities (<0.8 mi/mi^2), 23 percent has moderate open-road densities (0.8 to 1.6 mi/mi^2) and 49 percent has relatively high open-road densities (>1.6 mi/mi^2). Nearly 75 percent of the suitable bear habitat is on private land. However, more than half (57 percent) of the suitable habitat on private land has relatively high open-road densities. Approximately 86 percent of the suitable habitat on other federal lands has open-road densities exceeding 1.6 mi/mi^2. As opposed to this,

Table 3.43 The acres of suitable habitat for black bear by open road density class and ownership for the Southern Appalachian Assessment area.

	Proportion of Suitable Habitat by Road Density Class (mi/mi^2)[1]					
Ownership	<0.4	0.4–0.8	0.8–1.2	1.2–1.6	1.6–2.0	>2.0
National Forest						
Talladega	25	11	16	17	13	18
Chattahoochee	45	10	12	14	8	12
Pisgah/Nantahala	43	11	12	11	7	16
Sumter	29	11	14	10	12	23
Cherokee	44	10	11	13	8	15
George Washington	37	11	14	12	9	17
Jefferson	37	12	15	12	10	13
Monongahela	43	11	13	13	9	15
Total National Forest	40	11	13	13	9	15
National Parks	68	6	9	7	3	7
Cherokee Indian Reservation	25	10	7	16	9	34
Other Federal	2	3	3	6	8	78
State	27	10	15	13	12	22
Private	13	7	10	12	13	44
Total	20	8	11	12	12	37

Ownership	Suitable Acres	% of Ownership in Suitable Habitat
National Forest		
Talladega	217,133	95
Chattahoochee	694,659	93
Pisgah/Nantahala	914,048	89
Sumter	74,077	93
Cherokee	579,526	92
George Washington	991,482	93
Jefferson	625,924	91
Monongahela	68,581	73
Total National Forest	4,165,005	91
National Parks	654,338	78
Cherokee Indian Reservation	19,647	40
Other Federal	81,161	70
State	486,203	84
Private	15,954,521	51
Total	21,360,875	57

[1] Road density based on single placement of 1 square mile sample blocks.

Figure 3.63 Spatial distribution of suitable habitat for black bear and road density class for the SAA area.

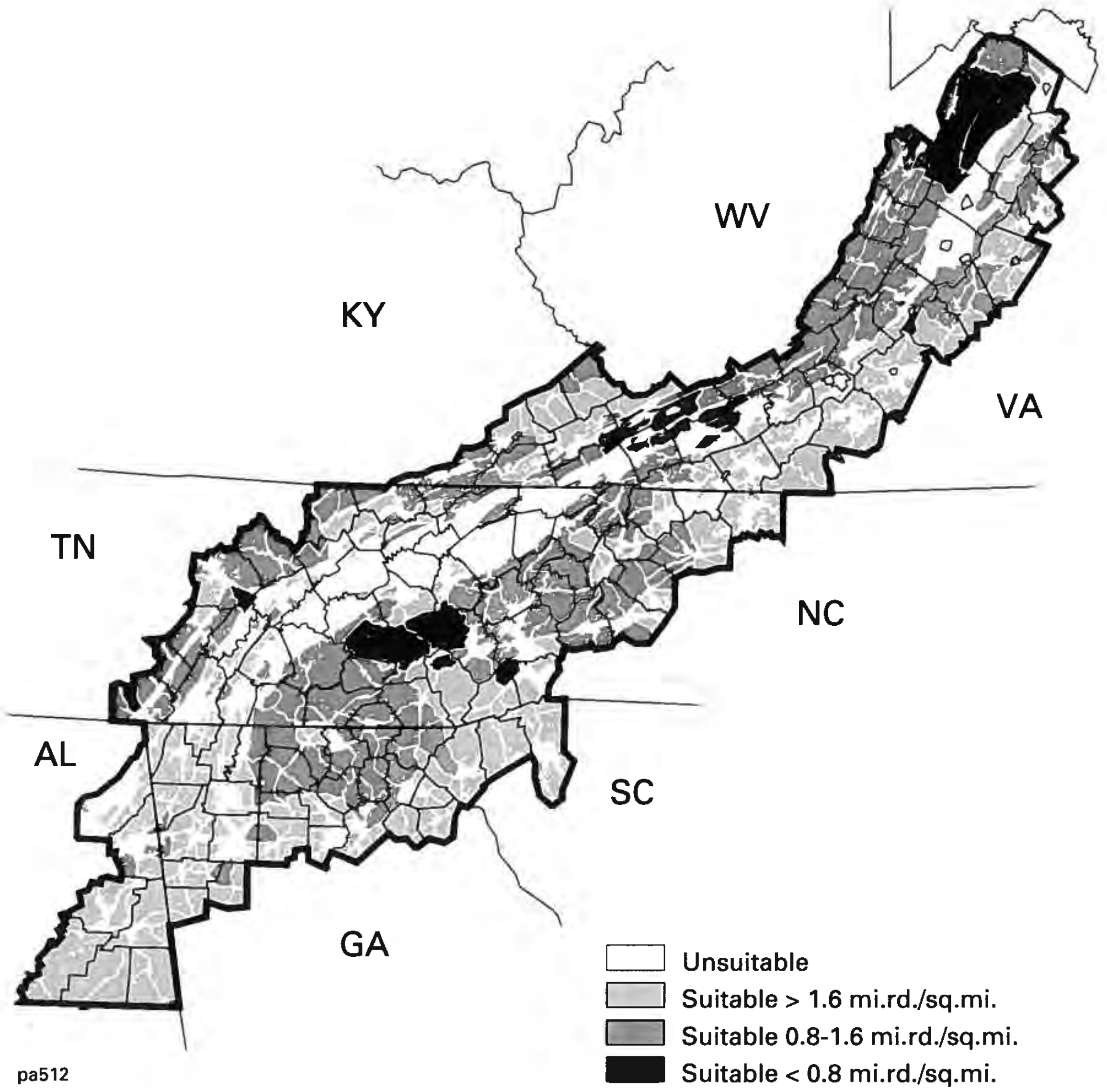

Table 3.44 The acres of suitable habitat for black bear by open road density class and section group for the Southern Appalachian Assessment area.

	Proportion of Suitable Habitat by Road Density Class (mi/mi^2)[1]					
Section Group[1]	<0.4	0.4–0.8	0.8–1.2	1.2–1.6	1.6–2.0	>2.0
Section Group 1	24	10	14	13	12	27
Section Group 2	30	9	11	12	10	29
Section Group 3	23	10	12	12	12	30
Section Group 4	8	7	12	14	14	45
Section Group 5	11	6	10	12	14	47
Section Group 6	6	5	7	11	13	58

Section Group[1]	Suitable Acres	% of Ownership in Suitable Habitat
Section Group 1	4,555,652	54
Section Group 2	7,333,189	69
Section Group 3	1,834,466	70
Section Group 4	1,167,326	25
Section Group 5	2,383,010	55
Section Group 6	4,073,100	59

[1] Section Groups:
1 = Northern Ridge and Valley, Allegheny Mountains, Northern Cumberland Mountains
2 = Blue Ridge Mountains
3 = Northern Cumberland Plateau, Southern Cumberland Mountains
4 = Central Ridge and Valley
5 = Southern Cumberland Plateau, Southern Ridge and Valley
6 = Southern Appalachian Piedmont

[2] Road density based on single placement of 1 square mile sample blocks.

more than 50 percent of the suitable habitat on national park and national forest land has open-road densities of 0.8 mi/mi^2 or less (74 and 51 percent, respectively). Approximately 91 percent of national forest land, 84 percent of state land, and 78 percent of national park land are suitable bear habitat, while only 51 percent of private land is suitable habitat.

Approximately 70 percent of the Northern Cumberland Plateau and Southern Cumberland Mountains (section group 3) and 69 percent of the Blue Ridge Mountains (section group 2) is suitable bear habitat, while only 25 percent of the Central Ridge and Valley (section group 4) is suitable (table 3.44). Open-road densities in the suitable bear habitat generally are higher in the Central Ridge and Valley (section group 4), Southern Cumberland Plateau and Southern Ridge and Valley (section group 5) and Southern Appalachian Piedmont (section group 6) than in the other portions of the SAA area.

A comparison of the bear habitat suitability model (fig. 3.63) with the current county-wide density estimates provided by state agency biologists (fig. 3.52) reveals a relatively strong correlation between these two measures of bear habitat. With some exceptions, high bear densities are associated with areas of low open-road densities (<0.8 mi/mi^2), medium bear densities were found in areas of moderate open-road densities (0.8 to 1.6 mi/mi^2), and areas where bear are present at low densities generally have higher open-road densities (<1.6 mi/mi^2). In areas where bears currently are absent, such as the Southern Cumberland Plateau, Southern Ridge and Valley, and Southern Appalachian Piedmont in Alabama and Georgia, open-road densities generally are high. This result suggests that bear mortality associated with open roads may be one of the factors limiting population expansion. However, these areas also have limited oak mast capability (table 3.18), which also may limit bear occupancy. Bears also are currently absent from much of the Northern Cumberland Plateau and Southern Cumberland Mountains (section group 3) in Tennessee. However, this portion of the assessment area, which is isolated from the Appalachian bear population by a large area of unsuitable habitat in the agriculturally dominated Central Ridge and Valley has moderate open-road densities and relatively high oak mast capability. This suggests that this area warrants further study for possible bear reintroduction.

The forecast is for potential habitat to remain stable on public land. Decreases in the amount of potential habitat are expected on private lands due to continued loss of forested habitats and increased development.

Future Opportunities for Terrestrial Wildlife and Botanical Resources

Question 3:

What habitat types, habitat parameters and management activities are important in providing the distribution and types of habitats to viable populations and/or desired habitat capability for the "short list" of wildlife and plants?

Question 4:

Based on our current knowledge of ecological unit land capabilities for the Southern Appalachians, what are the general habitat mixes and conditions needed to recover threatened and endangered (T&E) species, conserve viability of concern (VC) species; maintain the existing species and community diversity that will not result in the loss of viability for any plant or animal species, and provide sustainable levels of species populations on national forests?

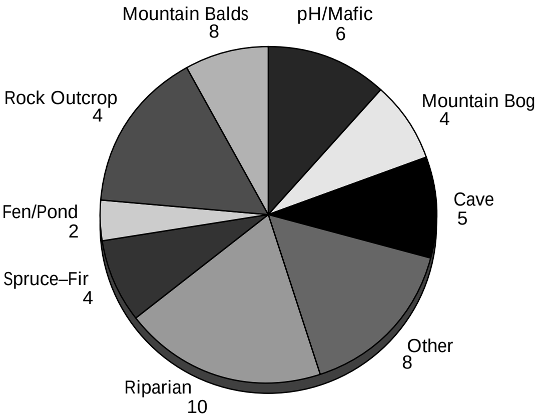

Figure 4.1 The distribution of the 51 terrestrial federally listed threatened, endangered, and proposed species according to community association in the Southern

Due to short time frames and the sensitivity of these topics, the Southern Appalachian Assessment (SAA) did not identify specific actions for sustaining various habitats. This chapter identifies habitats of concern that should be consistently managed throughout the SAA area and discusses the relationships between land ownership and ecological units. Actions for maintaining species groups based on habitat association are presented. The responsibilities and potential roles for private and public lands in maintaining the full diversity of habitats in the SAA are also discussed.

Rare Communities

The conservation of rare communities is the key to conserving the rare plant and animal species in the SAA area. Approximately 84 percent (43 out of 51) of the federally listed T&E plant and animal species associated with rare community and streamside habitats (fig. 4.1), and 74 percent (270 out 376) of the terrestrial viability concern (VC) species is associated with 7 rare community species groups and streamside habitats (fig. 4.2). These habitats occur on less than one percent of the land area

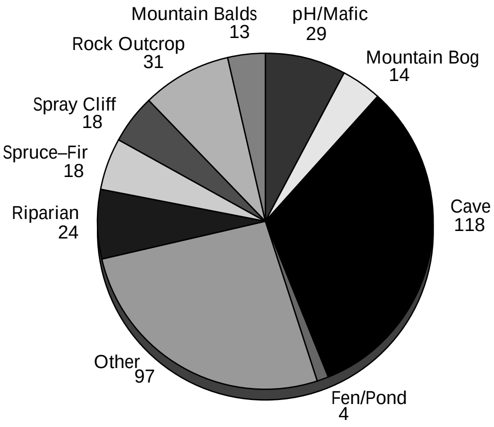

Figure 4.2 The distribution of the 366 terrestrial species with viability according to community association in the Southern

in the SAA area. The following are some considerations for maintaining the species groups based on rare communities developed from information in current recovery plans for some federally listed species.

Cave Habitat

Critical factors in protecting cave resources, including endangered bat species, are: proper gating of cave entrances to exclude human impacts; avoiding the alteration of cave entrances in order to maintain the proper temperature and humidity regimes in the caves; and maintaining the integrity of surface water recharge in the caves (USDI FWS 1976, USDI FWS 1978, USDI FWS 1982a).

Mountain Bog Habitat

Major threats include hydrology alterations, siltation, and encroachment of woody vegetation. Many bogs have been filled or drained for conversion to pasture or other agriculture activities. Restoration and/or maintenance of proper hydrology are primary management needs for these sites. Removal of competing woody vegetation is necessary to preserve some existing sites. Prescribed burning on bog sites would benefit the federally listed green pitcher plant, but the effects of fire on the other federally listed plant species in this habitat are unknown (USDI FWS 1990b, USDI FWS 1991b, USDI FWS 1994a).

Fen or Pond Wetlands

These communities vary from wet meadows, typically pastured, to true ponds. Long-term threats include nearby development that alters the hydrology of the area and changes that allow encroachment of woody vegetation. Siltation and competition from weedy invaders could become serious threats if habitats surrounding ponds are not protected.

High-Elevation Balds

The greatest threat to these communities and their associated species is overuse by human visitors. Air pollution may also be playing a part in the decline of these communities. Adequate protection of these areas from damage by people is essential for the recovery and maintenance of T&E and VC species. Management to control encroaching woody vegetation may be appropriate in some locations (USDI FWS 1987, USDI FWS 1989, USDI FWS 1993b, USDI FWS 1993c).

High pH or Mafic Habitats

Some rare species are affected negatively by disturbance, while some respond positively to disturbances such as fire. Depending on objectives for a particular species and location, management options may range from prescribed burning and timber harvesting to limiting of timber harvesting and road development (USDI FWS 1995c).

Rock Outcrop and Cliff Habitat

Needs for these habitats include protecting from overuse by human visitors, maintaining early successional conditions on talus slopes, burning on sandstone cliff and quartzite ledges and outcrop communities, eliminating threats from rock quarrying, preventing overgrazing by deer and feral goats, and protecting adjacent forest vegetation from timber harvesting and air pollution in high-elevation granitic dome communities (USDI FWS 1979, USDI FWS 1983, USDI FWS 1991a, USDI FWS 1995a).

Montane Spruce-Fir Forest

High-elevation spruce-fir forest communities have been reduced to current levels by the past century of logging, exotic insect infestations, and possibly other factors not yet fully understood. In recent years, Fraser firs (*Abies Fraseri*) in these stands have suffered extensive mortality due to infestations of balsam woolly adelgid (*Adelges piceae*). Current threats to this community and associated species include exotic species infestations, air pollution, and degrading of habitat by opening forest canopies, raising soil temperatures, and decreasing soil moisture (USDI FWS 1990a).

Seeps, Springs, and Streamside Habitats

Management considerations for these habitats include maintaining bald eagle (*Haliaeetus leucocephalus*) nest and roost sites, maintaining canopy openness of sand and gravel bars, and reducing human disturbances to sites. Water flows should be maintained, shading should be reduced where needed to help associated species, and habitat conversion to agricultural land uses should be avoided (USDI FWS 1982b, USDI FWS 1990c, USDI FWS 1995a).

Mountain Longleaf Pine Forests

The greatest opportunities for maintaining mountain longleaf pine woodland appear to be in red-cockaded woodpecker (*Picoides borealis*) management areas in the Southern Ridge and Valley on the Talladega National Forest. Talladega and Shoal Creek Ranger Districts in Alabama have identified a tentative habitat management area totaling approximately 120,000 acres. Management direction has been established for red-cockaded woodpecker recovery (USDA FS 1995).

Mid- and Late-Successional Deciduous Forests (Includes Mixed Pine-Hardwood Forests)

The mid- and late-successional deciduous forests in the Southern Appalachians are an important habitat for 80 species on the special list. Less than 50 percent of this habitat is in tracts larger than 5,000 acres. Priority should be given to maintaining the remaining existing larger tracts that have the potential to support the species associated with mid- and late-successional forests. Currently, national forests and national parks contain the largest portion of these large tracts and most likely will continue to provide the core habitat for source populations of deciduous forest species. Private landowners with large tracts, through their voluntary participation, should be invited to identify their lands as additional habitats, especially for area sensitive species. The majority of mid- to late-successional deciduous forest acreage occurs on private lands. If current levels of this habitat type are to be maintained, private landowner involvement will be necessary.

“Forest interior species” are thought to be negatively affected by increased interactions with predators and nest parasites associated with adjoining nonforest or early successional habitats. These “edge effects” may be related to larger landscape patterns (Robinson and others 1995). When managing for sustainable forest interior habitat, the landscape/forest interior assumptions discussed in Chapter 3 should be used to balance incorporation of early successional habitat.

Maintaining T&E and VC species may require protecting sites from road construction, preventing loss of forests to development, and mitigating measures for some silviculture practices (USDI FWS 1984, USDI FWS 1985, USDI FWS 1994b).

Mid- and late-successional oak forests are primary providers of oak mast for dependent wildlife species. Techniques for estimating oak mast production calculation techniques are discussed in Chapter 3 and Whitehead (1989).

Sufficient late-successional deciduous forest will need to be maintained to provide special habitat features required by some species, such as large cavity trees, large standing snags, perhaps greater than 20 inches in diameter at breast height (d.b.h.), and den trees. Spatial arrangement of these features should be considered. An example for calculating minimum levels of late-successional acreage required to maintain these special features can be found in SAA process records (Hedrick, unpublished).

A sustained flow of vigorous mid- and late-successional deciduous forest habitats can be maintained over the long term by using a silvicultural management system (even-aged, two-aged, or uneven-aged) compatible with a landowner’s overall natural resource objectives.

Early Successional Habitats

Early successional habitats (0- to 10-year-old forest communities and abandoned/idle land) are required by 10 species and are important for several game species and habitat generalist species. These habitats can result from even-aged regeneration harvests, group-selection harvests, disturbance (i.e. insect, disease, fire), and nonintensively managed, cultivated land. These very dynamic habitats are not abundant and succeed rapidly into sapling/pole forest habitats. For this reason, land management strategies should consider the landscape principles of isolation, patch size, and source/sink communities when planning for these habitats.

Little attention has been given to the size of early successional habitats. A patch created by group selection harvest or a natural disturbance may not support all the species identified for early successional habitats. For this reason, the size of early successional patches is a consideration when providing these habitats. In addition, standing snags larger than 12 inches dbh is an important component of these habitats.

It is possible that isolated early successional habitats may not be inhabited by less mobile species. If the areas are inhabited, they may serve only as a sink population source with little opportunity for population expansion due to the short life of this habitat type and isolation from other suitable habitats. Early successional habitats should be provided near current permanent source habitats or future planned early successional habitats in order to lessen possible isolation of these habitats.

To provide early successional habitats on national forests will require strategies that emphasize even-aged harvests in conjunction with group selection harvests. This approach should maintain species dependent on early successional habitat types and will help meet the public demand for game species on national forests.

Black Bear Habitat

Remoteness from human activity is a key habitat parameter for black bears, but determining what constitutes remote habitat is problematic. Road density is a measure of remoteness, but there appears to be no definitive road density threshold at which habitat quality begins to decline. Activities that result in increased human activities during all times of the year decrease the quality of black bear habitat. In the absence of specific threshold levels, national forests with black bear habitat objectives should, as a goal, maintain an open-road density of 0.8 miles or less per square mile through seasonal and permanent road closures (Pelton 1986). Managers of state and private tracts may also want to consider road closures to benefit black bear. Closing roads and seeding them create secure brood range, nesting habitat, and feeding areas during the spring, summer, and fall months for other species associated with these open habitats. Largely because of the security they provide, national parks and national forests will continue to be the core of quality black bear habitat in the SAA.

The Changes in Southern Appalachian Assessment Forest Vegetation from Natural Processes and Human-Caused Disturbances

Question 5:

What changes and/or trends in forest vegetation or soil productivity are occurring in different ecological subsections in the Southern Appalachians in response to human-caused disturbances or natural processes?

Question 6:

What are the potential effects of the presence or absence of fire on forest health?

Ecosystems and their constituents respond to changes in climate, geomorphology, and soil environments. Changes, or disturbances to prevailing conditions, occur continually. There are three major dimensions of disturbance: the size, the time involved, and the magnitude or intensity. The size of a disturbed area may range from the gas formed from the loss of a single tree to tens of thousands of acres. Some changes, such as long-term climate and weathering of rocks into soil, occur slowly over tens to thousands of years. Others, such as the effects of fire, may take less than a day. Intensity of disturbance also varies.

Disturbances can be broadly grouped into those resulting from human influence and those not caused by humans. Human-caused changes, such as introduction of exotic pests, extirpation of species, and utilization of natural resources, raise particular concern because their long term consequences often are unknown.

Natural disturbances may be similar to past disturbances, whereas human disturbances are much greater in magnitude today than at any previous time. Humans have been part of ecosystems in North America, and the Southern Appalachian Assessment (SAA) area, for over 10,000 years. They have shaped the ecosystems in which they live. Prior to European settlement and industrialization, native Americans affected ecosystems through agriculture, hunting, village construction, fire, and dispersal of plants and animals to new areas during their travels. Modern society has dramatically increased disturbances because of industrialization, new technologies, and human population increases.

Recent human-caused disturbances include the exclusion of fire and the impacts of exotic forest pests such as chestnut blight, gypsy moth, Dutch elm disease, balsam and hemlock woolly adelgids, many exotic plants, and feral hogs. The role of fire and vegetation responses to its impact in the pre-European settlement forests across the Southern Appalachian landscape may have been much more pronounced than today. Because of modern human-caused disturbances, the current landscape is probably unlike anything that occurred in the past. Future vegetation is likely to be greatly affected by the direct and indirect impacts of exotic pests. Some factors are: (1) the amount and distribution of older-age forest stands, (2) fire suppression, (3) air pollutants, and (4) introduced pests. Silvicultural activities designed to manage vegetation and regenerate commercially valuable tree species are also major human disturbances. A range of silvicultural techniques will be discussed at the end of this chapter.

Changes resulting from some natural causes, such as earthquakes, storms, and droughts, cannot be controlled and are generally accepted. Changes that result from management or utilization of natural resources can be evaluated and altered as part of management policy. Examination of the impacts of alterable changes, therefore, is an essential part of management planning.

In this chapter, the Terrestrial Team addressed two questions related to disturbance. The first question is, "What changes and/or trends in forest vegetation or soil productivity are occurring in different ecological subsections in the Southern Appalachians in response to human-caused disturbances or natural processes?" Lightning-caused fires can be, and have been, suppressed. Because many forest ecosystems evolved in response to natural fire patterns, fire exclusion can cause subtle, but potentially important, changes in future forest composition, structure, and productivity. The second question, therefore, was, "What are the potential effects of the presence or absence of fire on forest health?" Before addressing these questions, we briefly summarize current knowledge about disturbance in Southern Appalachian ecosystems.

Natural Disturbance

Disturbance dynamics

Plant communities of the Appalachians are characterized by compositional fluctuations, as individual plants grow, die, and are replaced (McGee 1984). Some vegetation changes are driven by characteristics of the individual plant species independent of their environment. Other changes are caused by factors outside the vegetation and independent of its nature. A commonly used term to describe changes in species composition that dominate a given area through time is "succession" (Barbour, Burk, and Pitts 1987).

Gap phase reproduction (patch disturbance) results from single trees or small groups of trees dying. The small openings that result from these perturbations are quickly revegetated by new plants that become established or by existing understory vegetation that is released from overhead competition.

Average rates of canopy gap formation have been estimated in several cases. Studies in the Southern Appalachians have found canopy gaps forming at an average of 0.4 to 2.0 percent of the land area annually (Runkle 1985) with canopy resistance ranging from 50 to 200 years. Lorimer (1980), working in a primary "virgin" cove and hemlock forest at Joyce Kilmer Memorial Forest, estimated that the average canopy mortality in a decade was 5.5 percent, with 3.8 percent in low-disturbance decades and up to 14.0 percent in those decades with major disturbance events. Disturbances of higher than average intensity occurred at about 30- to 40-year intervals. Runkle and Yetter (1987) found that gaps formed at a rate of 1 percent of the land surface per year in their study areas. Runkle (1982) estimated for old-growth mesic forests in general, that recognizable gaps occupied 17.3 percent of the canopy in Joyce Kilmer Memorial Forest and 8.9 to 24.2 percent of the Great Smoky Mountains (Schafale and Weakley 1990). Timber harvests that resemble gap-phase dynamics (e.g. single-tree-selection and group-selection cuttings) might be carried out in appropriate forest types at a rate of 1 percent per year and be within the normal variability of natural processes. This approach has been suggested as a means of hastening the development of old-growth characteristics (Runkle 1991) and is worthy of investigation (Lorimer and Frelich 1994).

Large-Scale Disturbances

A number of climatic, edaphic, and biotic factors can create catastrophic disturbance. Although the causes are external, community attributes often influence the degree of change and gradient of effect. An example of this can be seen with Virginia pine (*Pinus virginiana*) dominated communities. This species tends to occur naturally in even-aged stands of relatively pure composition. The species is relatively short-lived and often found growing on shallow soils. Since it is shallow-rooted, it is prone to windthrow, particularly when crowns are heavy with snow and ice. Thus, wind, ice, and snow can remove a large section of Virginia pine forest, while barely affecting other pine or hardwood types of similar size and age.

Table Mountain pine (*Pinus pungens*) is a fire-dependent species native to the Southern Appalachians. It has serotinous cones that open when exposed to high temperatures resulting from medium- to high-intensity fires. It can begin producing cones with viable seeds at a young age. It typically grows on fire-prone southeast to southwest facing slopes and ridgetops that are often droughty. Table Mountain pine is well adapted to this pyric environment which excludes most tree and shrub species adapted to more mesic conditions. A recent study using tree-ring analysis of fire-scarred trees of Table Mountain pine forests on Brush Mountain in southwestern Virginia indicated that from 1798 to 1944, fires burned approximately every 10 years. After 1935, following acquistion by USDA Forest Service (FS), the study area burned only once. The study concluded that continued fire exclusion would lead to oak-dominated plant communities (Sutherland and others 1993).

Native American Caused Fires

Fire disturbance is the most well researched of all natural disturbances operating in North America (White 1979). Fire is particularly important in conifer-dominated forests and can also be important in drier types of deciduous forests. The frequency and intensity of fire depend on precipitation amounts, fuel accumulation, and seasonal characteristics of the vegetation. Fire may be the common denominator for the development of oak forests on upland sites and their past and present ecological status (Abrams 1992, Barrett 1995).

The pattern of fire during the past 10,000 years by native Americans and early European settlers has affected the current composition of most forests in the SAA area. Periodic burning likely plays a major role in promoting advanced oak regeneration. Early historical accounts describing the impacts of native Americans on the forests and grasslands in the Southern Appalachians are largely anecdotal and sometimes controversial. Unfortunately, there is a lack of empirical evidence documenting the role of fire and the abundance of oak in the Southern Appalachians.

Perhaps the best, and most objective, evidence about the composition of forests before European settlement is the pollen record from pond and bog sediments that have accumulated for thousands of years. Research in eastern Tennessee indicates that during the Early Archaic period, 8000 to 6000 years before present (BP), major wood-charcoal hearth fire constituents were oak. By the Late Archaic period, 4000 to 1500 years BP, disturbance-favored (early successional) species comprised 25 percent of the wood charcoal preserved as ethnobotanical remains (Delcourt and Delcourt 1986). *Quercus* (oak), *Castenea* (chestnut), *Carya* (hickory) and *Pinus* (pine), constituted the majority of total tree pollen during the Woodland (1500 to 1000 years BP), Mississippian (1000 to 500 years BP), and Historic (300 years BP) cultural periods for Tennessee sites.

In the late Holocene Epoch, the forests near Black Pond in the Central Ridge and Valley section of east Tennessee were predominantly oak and pine with subdominants of hickory and chestnut (Delcourt and others 1986). At the time of European settlement, landscapes of the southeastern United States were not covered by extensive unbroken old-growth forests. Instead, vegetation patterns at 500 years BP were the result of continued individualistic responses of plant populations to long-term changes in climate, prevailing disturbance regimes, and native American activities that included the use of fire and development of agriculture (Delcourt and others 1993).

Oak species are apparently well adapted to an environment that includes periodic fire. Relative to other hardwoods, fire favors oaks because of their thick bark, sprouting ability, resistance to rot after scarring, and the suitability of fire-created seedbeds for acorn germination (Lorimer 1985, Abrams 1992). Studies have shown that stands which had been thinned, grazed, or lightly burned during the past two decades generally possessed a greater reservoir of oak regeneration than undisturbed stands (Carvell and Tryon 1961). Periodic fire probably checks succession in oak forests, because most later successional species, such as red maple (*Acer rubrum*), exhibit low resistance to fire. Recent studies have indicated the potential for widespread oak replacement by more shade tolerant species in mature forests (McGee 1986, Fryar 1993).

An oak study that included data from Forest Inventory and Analysis (FIA) plots on the Cherokee National Forest in 1989 showed that 38.0 percent of the total live volume of growing stock on the forest was oak and 14.2 percent of all live stems were oak. However, only 7.9 percent of all live stems in the 1 to 7 inch d.b.h. were oak. In comparison, there were over seven times as many soft maple, white pine, and dogwood stems (collectively) as total oak stems in this diameter range. This study concluded that the future of many oak stands was uncertain (Fryar 1993). The loss of oak dominance may vary with soil and site factors and probably will be slower on dryer sites. Loss of oak dominance in forests where fire has been mostly excluded during the twentieth century, and the lack of such patterns in forests periodically burned, should be considered important indirect evidence that fire played a vital role in maintaining oak dominance before European settlement. If, in the current oak forests, factors antagonistic to oak regeneration (such as a lack of fire) persist into the twenty-first century, a reduction in oak dominance seems inevitable (Abrams 1992).

Lightning-Caused Fires

Data on 1986 to 1993 occurrence of lightning-caused and human-caused fires are available for national forests and national parks (fig. 5.1). On the Cherokee National Forest in Tennessee, during the 16-year interval from 1977 to 1993, 114 fires occurred, with an annual mean of about seven. For the time period spanning 1915 to 1993, 290 fires on the George Washington National Forest in Virginia were attributed to lightning, with a mean of about 4 fires per year. Lightning fires are more frequent on slopes facing southeast to southwest. In the Great Smoky Mountain National Park, lightning fires averaged six per year over an area of approximately one million acres. Data from all sources indicate that approximately 15 percent of fires in the SAA area are attributable to lightning.

A survey was conducted in the SAA area to determine statistics for fire occurrence in general. The following tabulation presents wildfire frequency and size during the period 1988 to 1993 by ownership:

Ownership	Fires (Number)	Area (Acres)
State and private	29,834	212,342
Federal	2,240	241,844
All	32,074	454,186

One percent of these fires was larger than 100 acres when extinguished. Lightning represented a small, but significant, proportion of ignition source for these fires, as shown below:

Ownership	Lightning (%)	Arson (%)	Other (%)
State and Private	3	34	63
Federal	12	48	40

The greater proportion of lightning sources of ignition on federal lands is partly a result of their location in mountainous terrain where almost half of all lightning strikes occurs on ridge tops. For the case study areas, an average of approximately 15 percent (one out of every six fires) was lightning caused.

Annually, an average of six lightning fires per one million acres occurs in the Southern Appalachians. This frequency is greater than that recorded for the Great Plains, Mississippi Basin, or northeastern United States, but less than portions of the western and southeastern United States where up to 20 or more lightning-caused fires per one million acres are recorded (Schroeder and Buck 1970).

Windstorm

Thunderstorms occur primarily in late spring and summer. Some thunderstorms and sustained high winds associated with hurricane tracks occur in the late summer or early fall.

Occasional high winds are associated with coastal winter storms. These storms can be quite severe due to ice or snow loads on trees and other vegetation. Windthrown trees result in pit and mound microrelief, providing an agent of soil mixing and producing different kinds of rooting sites for seedlings (White 1979).

Winds in association with heavy precipitation or snow melt that lead to soil saturation can increase windthrow and landslides, particularly on shallow soils. Fire or insect outbreaks sometimes occur in years after windstorms damage vegetation. The dominance of species adapted to open growing conditions on wind-exposed knolls and steep slopes in forested regions has been noted (White 1979).

Ice and Snow

Some trees are more prone to damage by ice storms than others. Studies have shown that some oaks, hickory, white pine, and hemlock are resistant to extensive glaze-induced damage while black oak, yellow-poplar, chestnut oak, black cherry, northern red oak, black locust, and other pines are not (Whitney and Johnson 1984; Carvell, Tryon, and True 1957; Abell 1934). Ice storms may limit the elevational range for some tree species in the Southern Appalachians. In conjunction with wind, ice and snow loads can cause wind throw. Damage to trees from ice and snow can increase the risk of pest problems and increase fuel loads, resulting in high-intensity fires.

Landslides and Earth Movement

The frequency of landslides and the response of vegetation have been studied in the Southern Appalachians (White 1979). Intense rainstorms, often on previously saturated soil, seem to be the major factor initiating landslides in the Great Smoky Mountains and other portions of the Blue Ridge. Numerous sub-surface geologic faults exist within the SAA area. Minor

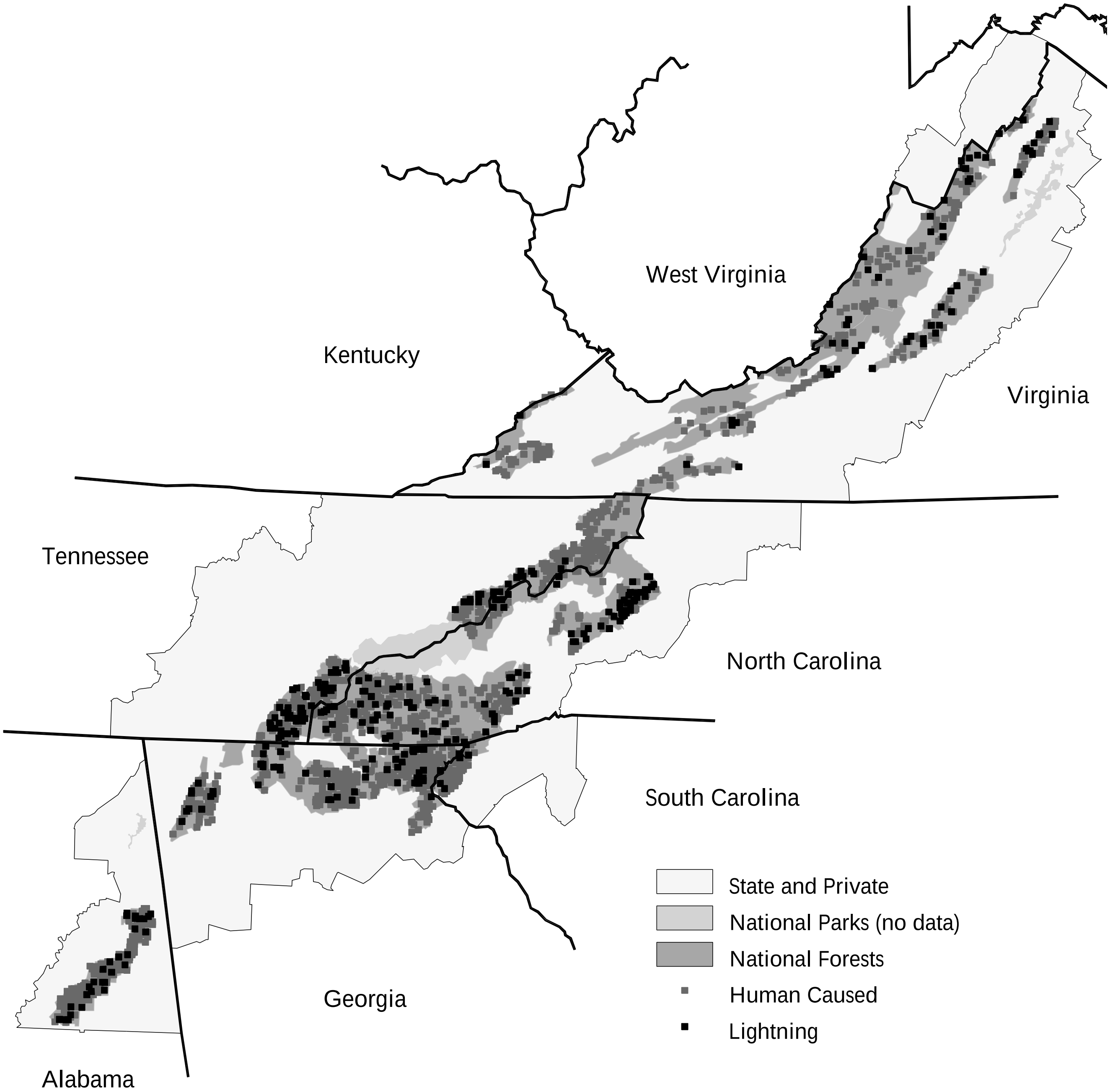

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Figure 5.1 Lightning-caused and human-caused wildfires occurring on national forests and national parks during 1986 to 1993 (Data source: National Interagency Fire Management Integrated Data Base).

earthquakes occur periodically but do not affect vegetation.

Precipitation Variability

Variations in precipitation cause flooding, landslides, water-level fluctuations in ponds and bogs, drought, and increased fire frequency and intensity. Drought periodically reduces the importance of mesic species and causes irregular compositional fluctuations in forests. It also reduces growth rates and affects seedling establishment of some species. Severe droughts kill some trees outright and physiologically weaken others. Drought can trigger or intensify decline and mortality in some tree species. High basal areas can exacerbate the impacts of pest epidemics following droughts.

An extraordinarily severe drought occurred in the Southern Appalachians in the summer of 1925. Over a 4-month period, rainfall near Asheville, NC, totaled 5.11 inches or 32 percent of normal. A follow-up study showed that black oak, red oak, and scarlet oak were particularly susceptible (Hursh and Haasis 1931).

Among plant communities in the Southern Appalachians, the ones most affected by variations in precipitation are on wetlands, and dry-to-xeric sites prone to fire, and on sites vulnerable to insect and disease epidemics. Imbalances in age-class distributions can further increase effects. Currently, a majority of stands on public lands in the Southern Appalachians is relatively even-aged and between 70 and 90 years old. Vulnerability to drought and to subsequent insect and disease outbreaks is high.

Frost Damage

Freezing temperatures just before or during budbreak in the early spring damage plants. Damage is greater when freezing temperatures follow a period of warm weather, which promotes growth and budbreak. Most plants are susceptible to frost damage. Budbreak for oak species normally overlaps late spring freezes and frosts. Shaded oaks tend to break bud earlier than oaks growing in open conditions. Released oaks with extensive recent growth are often damaged by frosts. McGee (1988) suggests that weather and budbreak are often related to regeneration problems with oaks.

Biotic Disturbance

Animals, insects, and diseases alter vegetation continuously or periodically. Natural biotic agents play an important part in ecosystem function. Insect outbreaks, for example, may facilitate nutrient cycling and balance of energy flows. Insect damage can often follow other disturbances such as wind, ice storms, drought, or fire. Some insects, such as bark beetles (*Ips* spp.), attack stressed trees first and provide "natural" thinning regimes in overstocked pine stands.

During droughts, defoliators such as locust leafminers (*Xenochalepus dorsalis*), elm spanworms (*Ennomos subsignarius hbn.*), and fall cankerworms (*Alsophila pometaria*) may become epidemic and defoliate large areas. These processes, however, may help balance nutrient budgets, particularly on sites of low productivity. Disease may function similarly to remove individually stressed trees or stands that have been weakened by other causes.

The effects of mammals and birds on forest vegetation usually are less significant than those of insects, but they can be locally important. Damage from deer browsing on hardwood regeneration is common in some parts of the Southern Appalachians and may limit establishment and growth of oak regeneration. Deer tend to be selective in browsing herbaceous plants and may limit the occurrence and abundance of some lilies and orchids. Beavers (*Castor canadensis*) historically played a very extensive and underestimated role in creating and maintaining an ever-changing mosaic of ponds and wetlands along streams in valleys. They were extirpated from many parts of the SAA area but they are returning and creating conflicts with other land uses. Now-extinct or absent species including elk (*Cercus canadensis*), bison (*Bison bison*), and passenger pigeons (*Ectopistes migratorius*) undoubtedly helped to shape the pre-European vegetational landscape. It has been suggested that large herbivores were partially responsible for the maintenance of high elevation grassy balds (Weigl and Knowles 1995). The small, prairie-like grasslands with endemic grassland plants now found in the Shenandoah Valley of Virginia are remnants of a vegetation type that occurred extensively in the "Great Valleys" of the Appalachians and were undoubtedly maintained in part by large herbivores.

Exotic pests, often introduced by human commerce, have the potential to affect forested ecosystems dramatically. The absence of natural predators and lack of genetic resistance among hosts can result in significant resource losses. Some exotic animals, insects, and disease problems have greatly affected vegetation in the Southern Appalachians. Feral hogs have severely damaged vegetation in the Great Smoky Mountains National Park and threaten to do so elsewhere. Chestnut blight, Dutch elm disease, dogwood anthracnose, butternut canker, balsam (*Adelges picea* Ratz.) and hemlock woolly adelgids (*Adelges tsugae*), and gyspy moth (*Lymantria dispar* L.) are exotics that have already had dramatic effects in Southern Appalachian forests.

Oak decline has affected thousands of forested acres where some oak species (especially scarlet oak and black oak) dominate. This complex phenomenon is caused by a combination of tree age, site factors that induce stress, and normally nonaggressive insects and fungi. As oaks mature, stresses alter tree physiology and render them susceptible to root disease and insects. Susceptible trees dieback and eventually die. Oak decline is a natural process, but its impacts are compounded by past land use, loss of species such as American chestnut (*Cantanea dentata*), replacement with species less adapted to the site, and other forces and conditions.

Silviculture and Prescribed Fire

Types of Silvicultural Activities

Disturbance drives the dynamics of forest communities. Damage or death of plants makes resources available in the ecosystem. Because disturbance is so variable, responses are also variable. Silviculture is based on an understanding of responses to disturbance. Its application might be viewed as a way of increasing predictability in the system by controlling the timing and types of disturbance. Silvicultural systems are planned processes in which a stand is tended, harvested, and re-established, very much as a gardener might plant, tend, and harvest a corn crop. The system name is based on the number of age classes and/or the regeneration method used.

Even-Aged Silvicultural Systems

An even-aged silvicultural system is a planned sequence of treatments designed to maintain and regenerate a stand with one age class. The range of tree ages is usually less than 20 percent of the rotation length. The four basic methods of even-aged silviculture are:

1. Clearcutting: A method of regenerating an even-aged stand in which a new age class develops in a fully exposed micro-environment after removal, in a single cutting, of all trees in the previous stand. Regeneration is from natural seeding, direct seeding, planted seedlings, and/or advance reproduction.
2. Coppice: A method of regenerating an even-aged stand in which all trees in the previous stand are cut and the majority of regeneration is from stump sprouts or root suckers.
3. Seed Tree: A method of regenerating an even-aged stand in which a new age-class develops from seeds that germinate in fully exposed micro-environments after removal of all the previous stand except for a small number of trees left to provide seed. Seed trees are removed after the regeneration is established.
4. Shelterwood: A method of regenerating an even-aged stand in which a new age class develops in the moderated micro-environment provided by residual trees. The sequence of treatments can include three distinct types of cuttings: (1) an optional preparatory cut to enhance conditions for seed production, (2) an establishment cut to prepare the seedbed and create a new age class, and (3) removal cut(s) to release established regeneration from competition with the residual trees (overwood).

When even-aged stands are created using the clearcutting method, successional stages 1 (grass/forb), 2 (shrub/seedling), 3 (sapling/pole), 4 (mid successional), 5 (late successional), and 6 (old forests) develop sequentially as the stand ages.

Conditions created by the seed tree method of regeneration are identical to clearcutting, except that a small number of seed trees scattered throughout the stand is retained in the stand during successional stages 1 and 2 and, sometimes, into successional stage 3.

In a typical shelterwood system, overwood is retained into successional stage 2 or 3. Depending on the amount of overwood retained (which can vary widely in shelterwoods), stages 2 and 3 may be somewhat prolonged due to height growth suppression resulting from reduced light penetration to developing regeneration. After overwood removal, successional stages 3, 4, and 5 occur sequentially.

One or more of the even-aged silvicultural systems can be applied in all of the forest habitat groups. Clearcutting with planting has been widely used to establish stands of loblolly, shortleaf and white pines. Planting of hardwoods has not been successful. Using the clearcutting method to regenerate hardwoods requires that appropriate regeneration sources be present at the time of harvest. The coppice method is only appropriate for sprouting species. The seed tree method has been widely used in loblolly and shortleaf pines. It is not used in hardwood regeneration because hardwood regeneration strategies do not depend on seed dispersal after cutting. Because they can create the wide range of conditions for regeneration, shelterwood methods are applicable in all forest habitat groups. Some shelterwoods are designed specifically to influence species composition of the new stand, e.g. to maintain an oak component in the new stand and, therefore, may have a significant impact on wildlife habitat.

Two-Aged Silvicultural Systems

Two-aged silvicultural systems involve a planned sequence of treatments designed to maintain and regenerate a stand with two age classes. The resulting stand may be two-aged or tend toward an uneven-aged condition as a consequence of both an extended period of regeneration establishment and the retention of reserve trees that may represent one or more age classes. There are a number of variants. One or more of the two-aged silvicultural systems can be applied in all of the forest habitat groups.

1. Clearcutting with Reserves: A clearcutting method in which varying numbers of reserve trees are left standing to attain goals other than regeneration. The regeneration phase of this system creates successional stages 1, 2, and 3, but, in contrast to the clearcutting system, some overstory trees are retained to meet specified objectives. The overstory trees retained, called reserve trees, may be small or large trees, or combinations of small and large trees, retained for: future growth; certain species components; current or future den trees; future sources of snags or coarse woody debris; or some level of visual quality. Due to the retention of a few overstory trees, a somewhat two-storied vertical structure develops during stages 2 and 3. Late in stage 3 or early in stage 4, the younger age class will begin to merge vertically with the older age class, although some vertical structure will remain in stage 4 and, perhaps, increase in stage 5 due to differential species development in mixed species stands. Depending on the kinds of trees initially retained, stages 4 and 5 may contain trees much larger than would normally be found in mid- or late-successional stands. Therefore, at least some of the attributes of much older stands can be provided in stands managed with this system.
2. Coppice with Reserves: A method of regenerating a stand in which the majority of regeneration is from stump sprouts or root suckers, and in which reserve trees are retained to attain goals other than regeneration. The conditions created with coppice with reserves are the same as with clearcutting with reserves or shelterwood with reserves, depending on the number of reserve trees retained.
3. Seed Tree with Reserves: A seed-tree method in which some or all of the seed trees are retained after regeneration is established to attain goals other than regeneration. The conditions created in a seed tree with reserves is identical to that created by clearcutting with reserves. The only difference between the two systems is that in the regeneration period, the trees retained have the specific function of producing seed to regenerate the stand.
4. Shelterwood with Reserves: A variant of the shelterwood method in which some or all of the shelter trees are retained well beyond the period of normal retention to attain goals other than regeneration. Initial conditions created are identical to those for the even-aged variant of this method, i.e., a micro-environment moderated by retention of residual trees. However, retaining overstory

trees beyond 20 percent of the rotation creates a distinct two-storied condition that persists for 20 to 40 years. Stand density or stocking must be reduced enough to allow for the long-term development of the new age-class. Stands develop through all successional stages with some residual trees in place. As in the other two-aged systems, some of the attributes of much older stands can be provided at a younger age in stands managed with this system. The choice of residual trees is dictated by management objectives. Choosing residual trees for cavity trees, mast producers, growth, or future snags or coarse woody debris provides the values associated with those trees. After 40 to 60 years, several silvicultural options are available, depending on management objectives: (1) the older trees can be retained into the future along with the younger age class, (2) the older age class can be removed, leaving the younger age class as an even-aged stand, or (3) the regeneration process can be initiated again by reducing stand density or stocking to an appropriate level.

Uneven-Aged Silvicultural Systems

Uneven-aged silvicultural systems are planned sequences of treatments designed to maintain and regenerate uneven-aged stands, meaning stands with three or more age-classes. There are several variants:

1. Single Tree Selection: A method of creating new age classes in uneven-aged stands in which individual trees of all size classes are removed more or less uniformly throughout the stand to achieve desired stand structural characteristics. In application, cuttings are made to control the frequency distribution of tree diameters using the negative exponential (reverse J-shaped) distribution as a target. For a given application, this target distribution is completely defined by stand basal area, maximum tree diameter, and ‘q,’ the exponential decay parameter. The resulting stand is one that has a continuous canopy cover containing a broad range of tree sizes. Single tree selection is very restricted in its application, due primarily to the ecological characteristics of Southern Appalachian species. The continuous forest canopy characteristic of single tree selection requires, for successful application, species that can regenerate and develop under shaded conditions. Most Southern Appalachian forests are comprised of canopy species that are intolerant or intermediate in their tolerance of shade. The exceptions are forests that contain hemlock, white pine, sugar maple, or beech, all of which are relatively shade tolerant and to which the application of single tree selection should theoretically be possible. Single tree selection has been successful in the beech-birch-maple forests of the northeastern United States, but the distribution of this type in the Southern Appalachians is extremely limited. Trials are currently underway in white pine, but research efforts to use single tree selection in mesic Southern Appalachian hardwoods, and in mesic to somewhat xeric oak stands, have been unsuccessful. Single-tree selection has been successful in loblolly pine stands in the South, but only with the application of herbicides to control hardwood competition.
2. Group Selection: A method of regenerating uneven-aged stands in which trees are removed and new age-classes established, in small groups. The maximum width of groups is approximately twice the height of the mature trees, with small openings providing micro-environments suitable for shade-tolerant regeneration, and with the larger openings providing conditions suitable for more shade-intolerant regeneration. Regeneration cuttings create, through time, a mosaic of patches of different ages. The range in patch sizes in Southern Appalachian conditions is from 0.2 acres up to about 1.5 acres. Within each patch, successional stages 1 through 6 develop sequentially.
3. Group Selection with Reserves: A variant of the group selection method in which some trees within the group are left standing to attain goals other than regeneration. The conditions created are identical to group selection, except for the effects of residual trees.

Successful regeneration can be achieved with both group selection and group selection with reserves with all forest habitat groups due to the variety of opening sizes that can be created using group selection.

Other Silvicultural Treatments

Intermediate Treatments

In addition to regeneration cuttings, silvicultural systems may include a number of other treatments needed to accomplish management objectives. Collectively, these are usually called intermediate treatments, and they include cleanings, liberation cuts, weedings, and thinnings. Cleanings are release treatments made in an age class not past the sapling stage to free the favored trees from less desirable individuals of the same age class which overtop them or are likely to do so. A liberation cut is a release treatment in a stand not past the sapling stage to free favored trees from competition of older, overtopping trees. A weeding is a release treatment in a stand not past the sapling stage that eliminates or suppresses undesirable vegetation regardless of crown position. Thus, cleanings, liberation cuts, and weedings take place during successional stages 1 or 2. One effect of all three treatments is to increase, at least temporarily, the amount of light reaching the forest floor. Herbaceous and woody plant production is increased. These treatments may also influence tree species composition.

Thinnings are silvicultural treatments made to reduce stand density primarily to improve growth of residual trees, to enhance forest health, or to recover potential mortality. Thinnings are classed as crown thinning, free thinning, low thinning, mechanical thinning, or selection thinning depending on the criteria for removing or retaining trees. In every case production of herbaceous and woody vegetation on the forest floor increases due to increased light.

Prescribed Fire

Prescribed fire is used for enhancing biological diversity, vegetative composition, and stand structure. A number of rare communities and the rare plant and animal species that inhabit them, benefit from fire. Examples are mountain bog communities, high elevation balds, and high pH mafic habitats. These communities are described in appendix C.

Forest types and plant communities where fire plays a role in community dynamics include: red spruce/Fraser fir (possibly minor effects); yellow birch boulder field forest; high-elevation red oak forest; montane oak-hickory forest; heath; white pine forest (possibly); chestnut oak forest (possibly); interior upland dry to mesic oak-hickory forest; xeric shortleaf pine; xeric pitch pine-Table Mountain pine ridge forest; xeric Virginia pine ridge forest; heath bald shrub land; grassy bald; Blue Ridge-Piedmont ultramafic barren; Southern Appalachian bog; and longleaf pine.

In the absence of periodic fire, two of the five rare forested communities in the SAA area, mountain longleaf pine woodlands and Table Mountain pine/pitch pine woodlands, are being replaced by hardwoods and loblolly pine. The endangered red-cockaded woodpecker is associated with the mountain longleaf pine woodlands in northeastern Alabama and northwestern Georgia. Table Mountain pine has serotinous cones that open only when exposed to high temperatures from crown fires. Continuing fire exclusion will probably result in continued decline in this ecosystem.

Periodic fire is an important factor in nutrient recycling. Prescribed burning can approximate natural fire regimes and provide a means of restoring fire-dependent and fire-associated vegetation. Some ecological communities such as pine and oak forests may be threatened because of several decades of fire suppression.

Without fire or other vegetative management practices that approximate fire effects, oak dominance may shift dramatically in future years toward shade tolerant and fire intolerant species such as soft maples, white pine, and sourwood. Early successional habitats, which are not abundant in the region and are located primarily on private land, result from even-aged regeneration harvests, group selection harvests, and disturbances such as insects, diseases, and fire.

Prescribed fires are large but infrequent contributors to the total annual amount of particulate matter in localized rural areas. However, in the region as a whole, prescribed fire is a regular, but small, contributor of particulate matter (SAMAB 1996b).

An environmental attitudes survey conducted for the SAA showed that the majority of respondents disagreed with the statement, “Using fire as a management tool in the national forest is a good idea.” (SAMAB 1996c). A significant change in public perception may be needed to gain acceptance of this practice in order for managers to be able to use this tool on national forests.

The Effects to Southern Appalachian Assessment Forest Ecosystems from Native and Exotic Pests

Question 7:

How is the health of the forest ecosystem being affected by native and exotic pests?[1]

In answering this question, impacts of the most damaging diseases, insects, and exotic plants in the Southern Appalachian Assessment (SAA) forests were considered. For each disease or pest, the historical and current status of the forest health problem are presented with a discussion of the host type, vulnerability, biology, expected trends of infestation, mortality or damage potential, and possible ecological implications.

Declines are complex diseases initiated by adverse environmental factors that create biotic and abiotic stress and often culminate in lethal attacks by organisms that are otherwise not harmful. Thus, these diseases differ from those caused by single primary pathogens in that trees suffer from many abiotic and biotic stress factors. In the context of these diseases, predispositional stress refers to environmental pressure sufficient to trigger changes in the physiology, form, or structure of a tree. The stress factors can be abiotic (e.g., extremes of moisture or heat) or biotic (e.g., insect defoliation, infection by fungi, or combination of these). In the absence of such stresses, the organisms of secondary action, often ubiquitous in the ecosystem, occupy various niches ranging from saprophyte to weak pathogen. Without these organisms, trees would most likely recover when the stress abates.

In recent decades, decline diseases have killed or damaged millions of trees in the eastern United States. Because declines are frequently initiated by broad environmental changes, they may suddenly emerge over a wide area, and the types of sites where they develop may appear to be closely related. This assessment examines the impact of oak and red spruce declines on the regional forests.

Several forest tree diseases that are not defined as declines also occur in the Southern Appalachians. In some instances, these diseases have symptom complexes similar to those induced by air pollutants. Causal disease agents range from simple abiotic stress, such as prolonged drought or spring frost, to complexes of fungi, insects, and abiotic stresses. This assessment considers the impacts of dogwood anthracnose, beech bark disease, butternut canker, Dutch elm disease, and the chestnut blight.

Numerous insect species injure trees in the forests of the eastern United States. Insects attack all parts of trees, including foliage, shoots, cones, seeds, stems, and roots. Injury may be negligible, or it may be catastrophic. With the exception of this southern pine beetle, this assessment of forest insect concentrates on exotic species, including the European and Asiatic gypsy moth, hemlock woolly adelgid, balsam woolly adelgid, and the Asiatic oak weevil.

Tree Declines

Oak Decline

Oak decline is not new. Forest workers have reported occurrences since the mid-1800s (Beal 1926, Balch 1927) and in every decade since the 1950s (Millers and others 1990). In fact, oak decline may have become more common and severe since the 1950s due to the

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[1] The original assessment question included air pollution. The SAA Atmospheric Technical Report (1996b) includes a discussion of ozone

predisposing action of an extreme drought early in that decade (Tainter and others 1990, Dwyer and others 1995). An apparent increase in incidence and severity in the early 1980s led to an intensification of survey and monitoring activities (Starkey and others 1989, Starkey and others 1992, Oak and others 1991) and, more recently, to development of risk rating systems for managers (Oak and Croll 1995, Oak and others, in press).

Forest Inventory and Analysis (FIA) surveys have determined that oak types mostly in upland oak and oak-pine stands cover 17.4 million acres in the Southern Appalachians. Oaks, therefore, are extremely important both economically and ecologically. Oak decline is a widely distributed disease that is changing forest composition and structure in this vast resource.

Oak decline is a disease complex involving environmental stress (often drought), root disease (e.g. Armillaria root disease), and insect pests of opportunity (e.g. 2-lined chestnut borer), and physiologically mature trees (Staley 1965, Wargo and others 1983, Wargo 1977). The diagnostic symptoms separating it from other diseases of oak are slow, progressive dieback of overstory trees from the top downward and from the outside inward. It results from disturbed carbohydrate physiology and water relations when mature trees become stressed and subject to root disease (Wargo and others 1983, Manion 1981, Hyink and Zedaker 1987). The introduction of the gypsy moth has exacerbated and accelerated oak decline because oaks are preferred hosts and spring defoliation contributes to the chain of events that increase susceptibility. Susceptible trees die within a few years after dieback exceeds one-third of the crown volume, but not all affected trees reach this point. Trees with lower levels of dieback often recover from visible crown symptoms (Oak, unpublished). Species in the red oak group are most susceptible (particularly black, *Quercus velutina,* and scarlet oaks, *Quercus coccinea).* Hickories are the only non-oak species commonly observed with symptoms in decline areas (Starkey and others 1989).

Like all native diseases and insects, oak decline is a completely natural ecosystem process that has always affected some component trees. The unprecedented amount of oak and changes in stand structure caused by past land use distinguishes the current decline situation from those that have occurred in the past. The decimation of the once-dominant American chestnut by the chestnut blight and land abuse early in the 20th century have resulted in forests with a higher percentage of oak now than at any time in the past.

Methods developed by Starkey and others (1992) permit the classification of oak forests into several categories with respect to oak decline–host type, vulnerable host type, and affected. Stands in which oaks comprise a plurality of stems are considered to be in the host type (fig. 6.1). Fifty-four percent of the host type is considered vulnerable (fig. 6.2). Vulnerable stands are old enough to have attained pole or sawtimber size and have at least 30 sq. ft. of oak basal area per acre–sufficient for potentially serious resource impacts if oak decline develops.

About 1.7 million acres of vulnerable host type were in turn found to be affected by oak decline based on the detection of dieback symptoms in one or more dominant or codominant oaks (fig. 6.3). Thus, 8 percent of the vulnerable host type area and 10 percent of the host type are affected.

Occurrence of oak decline varies by ownership and state. Private owners control nearly 80 percent of the host type area but have the lowest oak decline incidence (18 percent of the host type). By contrast, national forests make up nearly 19 percent of the host type area, but the incidence of affected stands is 2 times greater than that for private owners (17 percent of host type) (fig. 6.4). The reason for the disparity in oak decline incidence is that national forests have a higher frequency of oak-dominated stands of advanced physiological age on sites with average to low site productivity (Oak and others 1991). Among states, North Carolina and Virginia have the highest decline incidences–17 and 14 percent of the vulnerable host type area, respectively.

Oak decline will continue to be a forest health issue in the SAA area, especially on national forests. About 19 percent of national forest land already has oak decline damage, and a nearly identical percentage has no damage but is vulnerable. Among national forests, the George Washington and Jefferson National Forests have the highest incidences (fig. 6.5).

Oaks will not be eliminated from decline-affected areas; their numbers and diversity are being reduced. Oak diversity is reduced because of the greater relative susceptibility of

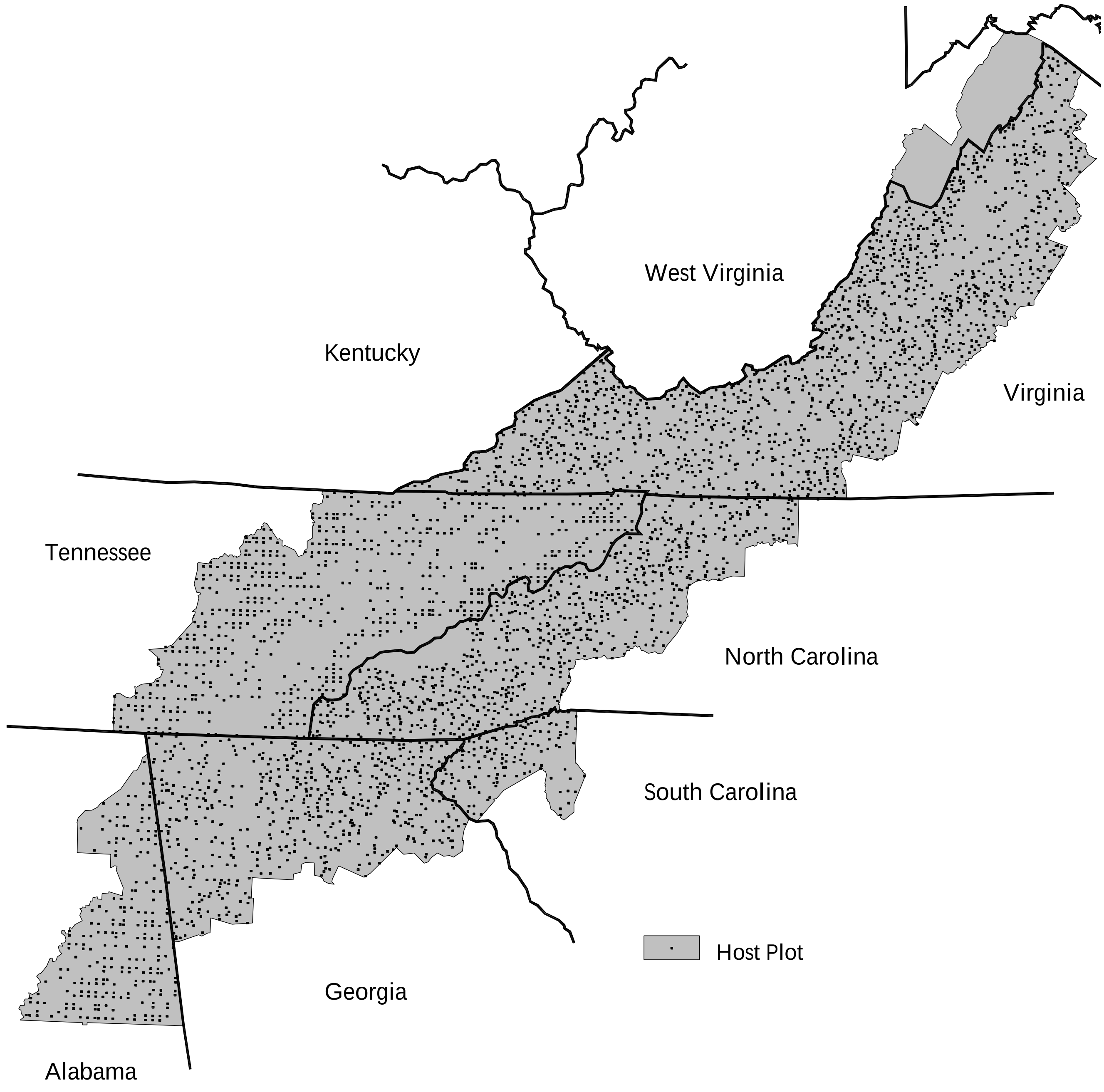

Figure 6.1 Stands classified as host type for oak decline if a plurality of stems are oak (Data source: FIA) in the SAA area.

species in the red oak group, and numbers are being reduced due to the replacement of dead and dying oaks by other species. Red maple *(Acer rubrum),* blackgum *(Nyssa sylvatica),* and other relatively shade-tolerant species are most commonly replacing dead and dying oaks

(Anderson and Cost, in press). This change has several effects on ecosystem structure and function. Structure becomes more complex as canopy density is reduced and the number of small openings increases. The quantity of dead standing trees and down woody debris

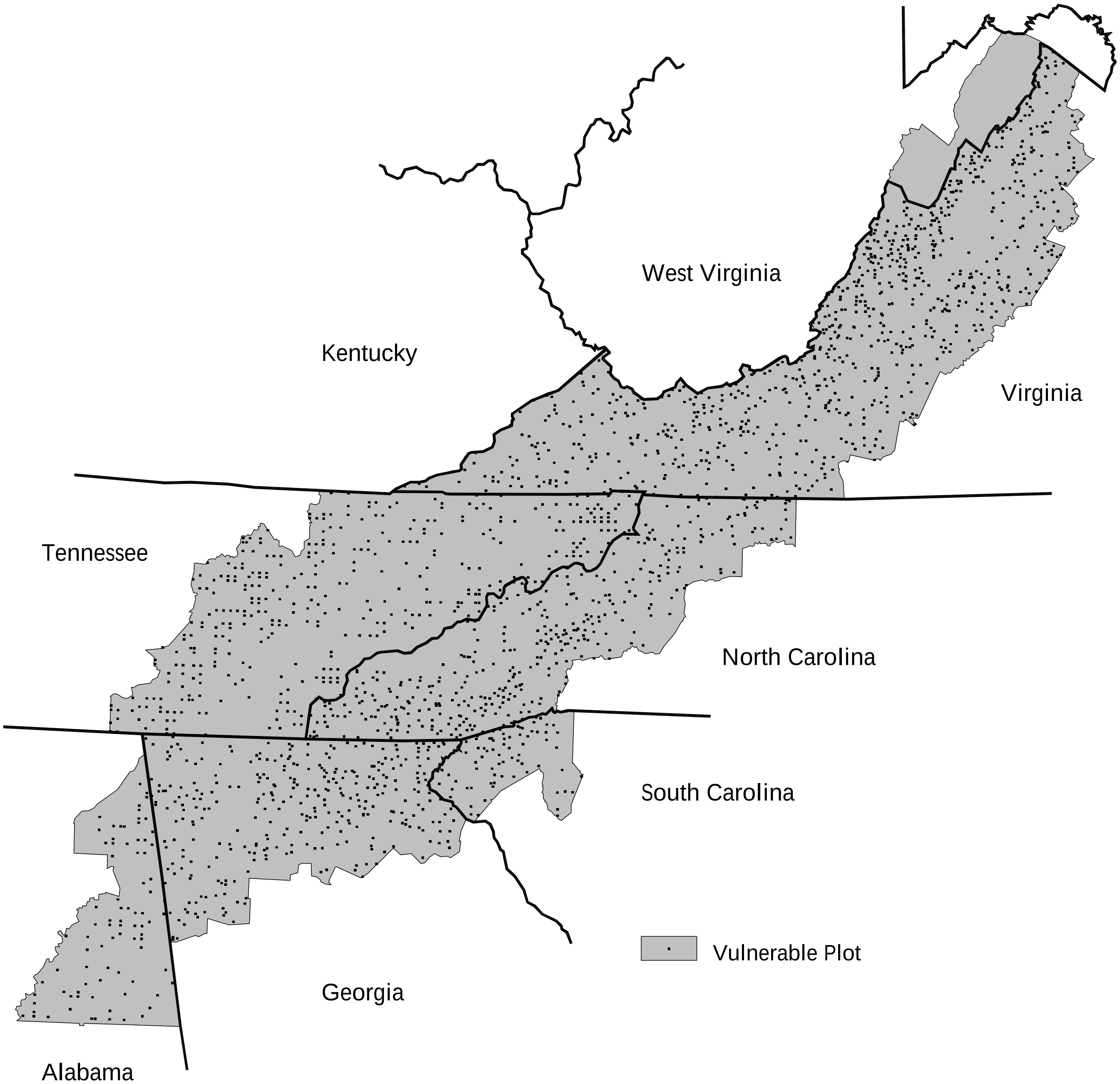

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Figure 6.2 Oak decline vulnerable plots in the SAA area. Vulnerable plots are defined if pole or sawtimber size has at least 30 square feet of oak basal area per acre (Data source: FIA).

increases denning sites for some animals but perhaps more than can be effectively exploited. Overall susceptibility to decline and gypsy moth defoliation is reduced due to a smaller oak component. Hard mast production potential, already severely reduced from historic levels due to loss of the American chestnut to chestnut blight, is further reduced in quantity, quality, and diversity as the number of oak decreases and as species in the red oak group suffer greater impacts than those in the white oak group (Gysel 1957, Oak and others 1988).

FH042

Figure 6.3 Oak decline affected plots in the SAA area. Plots are affected when dieback symptoms are detected in one or more dominant or codominant oaks (Data source: FIA).

The areas of greatest impact will be immediately behind the advancing front of the gypsy moth. Repeated severe defoliation in spring by this insect increases susceptibility to decline (Wargo 1977). Heavy oak mortality has occurred over large areas. Major losses will probably be most common on national forests and in Virginia and North Carolina. Subsequent gypsy moth outbreaks and oak decline events will be less severe due to the reduction in abundance of preferred host species.

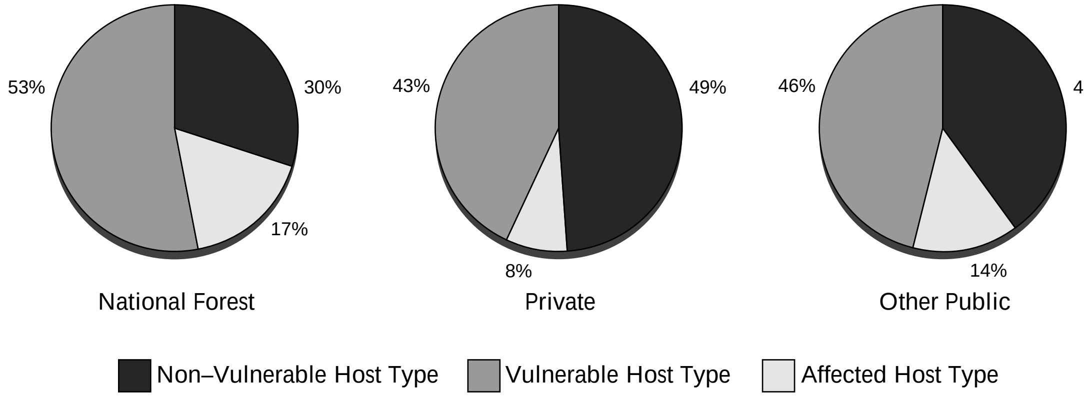


Figure 6.4 Proportion of host type that is non–vulnerable, vulnerable, and affected by oak decline for three ownership categories. (Source: FIA)

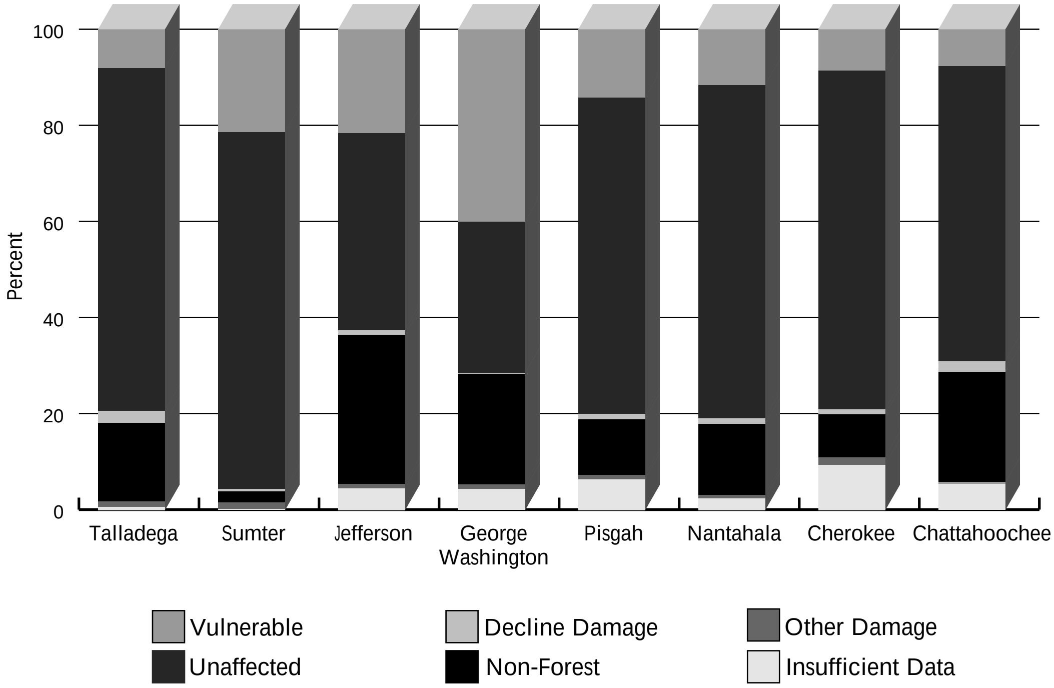


Figure 6.5 Proportion of area within each national forest classified according to oak decline risk. (Source: Continuous Inventory of Stand Conditions)

Management responses to oak decline range from doing nothing to altering forest composition and structure to maintain oak abundance and diversity through silviculture practices. The selection of an option depends on the relative importance placed on oaks in the landscape and the cost of treatment. One option is to maintain oak through timber harvesting or other disturbances (e.g. fire) that encourage oak reproduction. Portions of the landscape will always be vulnerable, but the present relatively uniform, vulnerable condition over large areas could be altered. In weighing the need for action, the value of oaks to wildlife should be added to their value as timber species.

Spruce Decline

Red spruce decline in the northeastern United States has been reported since the early 1980s (Peart and others 1992). Symptoms include high mortality rates, canopy crown deterioration, reduced growth rates, and shifts in forest tree species composition. Research results from the National Acid Precipitation

Assessment Program (NAPAP) suggest that atmospheric deposition may be implicated (NAPAP 1991). Exposure to ambient cloud water can reduce the cold tolerance of red spruce. Increases in winter damage to red spruce in the Northeast have contributed to crown damage and increased mortality in that region. This impact occurs infrequently in the Southern Appalachians, where temperatures seldom approach the cold tolerance limits for red spruce.

Evidence of red spruce decline and pollution involvement in the Southern Appalachians is less substantial. The red spruce-Fraser fir ecosystem occupies approximately 103 square miles in the Southern Appalachian Mountains of southwestern Virginia, eastern Tennessee, and western North Carolina. The trees are generally confined to mountain peaks above 5,000 feet elevation. NAPAP studies (NAPAP 1991) in the Southern Appalachians have documented extensive mortality of Fraser fir and decreases in crown vigor and annual growth in red spruce. Fraser fir (*Abies fraseri*) mortality, frequently pictured in popular publications, was the direct result of an insect, the balsam woolly adelgid.

Although it has been suggested that air pollution may have rendered fir more susceptible to the adelgid, supporting evidence is incomplete. In mixed stands with dying fir, spruce decline can be partially explained by increases in wind damage and soil temperatures (Nicholas and others 1992). Symptoms of decline in spruce-dominated stands, at high elevations with a high frequency of cloud interception, have led scientists to consider impacts of atmospheric deposition. Acid deposition components of sulfate, nitrate, and hydrogen ions at high elevations greatly exceed those at lower elevations. This is primarily due to the increased volume of precipitation and high ion concentrations in cloud water. Exposure to ambient cloud water with concentrated sulfate and nitrate anions (negatively charged ions) has been shown to accelerate foliar leaching of essential cations (positively charged ions). Field surveys and fertilization studies indicate that red spruce in the Southern Appalachians, are experiencing calcium and zinc deficiencies, while those in the Northeast are generally not (Eagar and Adams 1992).

NAPAP research (Barnard and Lucier 1990, Shriner and others 1990), as well as ongoing studies (Nodvin and others 1995) have demonstrated that the high elevation forests appear to be nitrogen saturated. Nitrogen inputs from rain, snow, and cloud water combined with inputs from natural biological process exceed the capacity of soils and vegetation to immobilize nitrogen. The leaching of excess nitrogen depletes essential base cations from the soils and acidifies soil water. In addition, there is evidence that aluminum is being mobilized into soil water at levels that interfere with plant uptake of calcium, magnesium, and zinc. Soils in the Southern Appalachians generally have a large capacity to absorb sulfate, but current sulfate loading rates will likely exceed soil sulfate absorption capacity within a few decades (Johnson and Lindbert 1992).

Detection of a spruce decline in the Southern Appalachians is difficult since forest structure in most areas has deteriorated since the early 20th century due to logging and infestation of Fraser fir by balsam woolly adelgid. Species composition and site quality changes after logging have been documented and current work indicates the ongoing adelgid infestation is causing dramatic changes in forest structure and composition. Most information about southern spruce-fir forests is based on pre-adelgid old-growth stands, but future assessments must include the realities of disturbed, second-growth forests when determining if stand condition is normal or if other stressors are also present.

Exotic Diseases

Dogwood Anthracnose

Caused by *Discula destructiva,* Redlin, dogwood anthracnose was first observed in the United States in Washington state in 1976 and in New York 2 years later. The disease has spread rapidly down the Appalachians, primarily on *Cornus florida,* the eastern flowering dogwood. This species is the most common in the Eastern United States and is most affected by the disease, but other dogwood species are susceptible. By 1988, dogwood anthracnose had been reported in more than 60 counties in eight northeastern states, including West Virginia and Virginia. By 1995, the disease had been confirmed in northern Georgia (1987), western North Carolina (1988) and as far south as northern Alabama (fig. 6.6). This disease is now

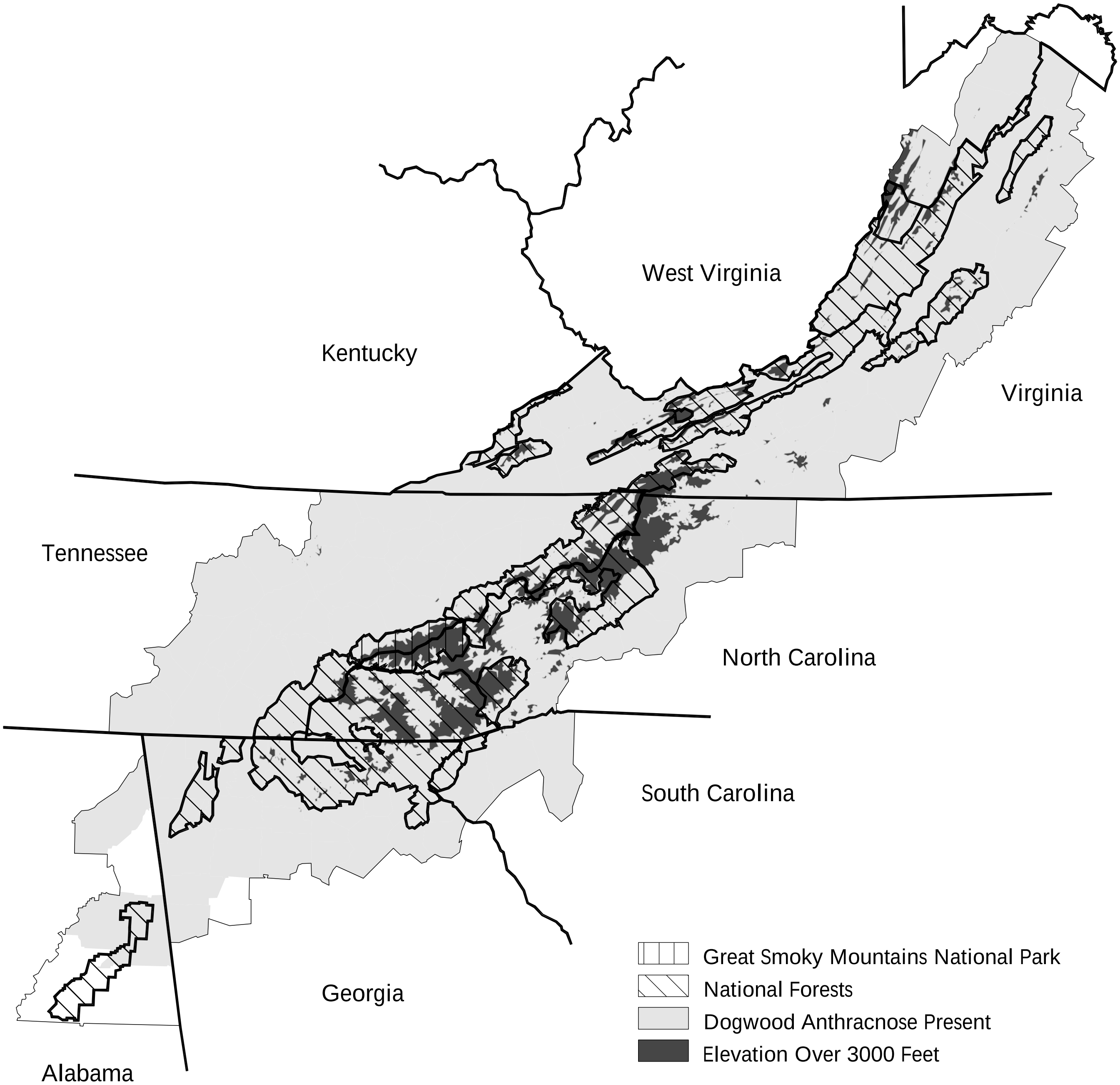


Figure 6.6 The distribution of dogwood anthracnose in the SAA area.

found in over 12 million acres in 180 counties (Anderson and others 1994).

Infection begins as leaf spots that may enlarge to kill the entire leaf. The fungus also infects twigs and spreads to the main stem.

cankers and epicormic shoots along its entire length. The stem cankers are capable of killing dogwoods, however, larger dogwoods often die 2 to 3 years after the first symptoms are observed due to the stress of repeated defoliation.

Dogwood is an important understory and midstory species in many ecosystems throughout the southern United States and its loss from any of these systems would have significant ecological consequences.

It may be too soon for reliable projections about the future of flowering dogwood in the many forest types in which it grows throughout the SAA area. Rate and severity of infection vary with several factors. In the South, infection is most likely at high elevations and on moist to wet sites. Shade increases the risk of infection and mortality. Denser stands of dogwoods seem to have less severe infection however. Dogwood stands on a southern or western aspect also have less severe infection, possibly because these stands are drier and get more sunlight.

Research continues to find potentially resistant trees in woodlands where dogwood anthracnose has been present for more than a decade. Potentially resistant survivors have been identified from a population of flowering dogwoods devastated by anthracnose in the late 1970s in southeastern New York. *Cornus kousa* is a known host of *D. destructiva* but seldom shows the severe disease symptoms that *C. florida* develops. The first generation hybrids of *C. florida* x *C. kousa,* introduced as the Stellar series, possess increased genetic resistance to anthracnose.

High-value landscaping trees can generally be protected by mulching, pruning, watering during droughts, and application of a fungicide, but no practical controls are available for dogwoods in forest environments.

Beech Bark Disease

Beech bark disease (BBD) is a complex of two causal agents, the beech scale insect, *Cryptococcus fagisuga,* and a fungus, *Nectria coccinea faginata.* Beech scale insects are, and have long been, a common pest of beech and other trees throughout most of North America. The disease is easily identified by the white woolly material, secreted by the female, which can be seen on the trunks of infested beech. By itself, the scale insect does not fatally injure beech. However, when the insect joins forces with *Nectria,* the two of them together become a symbiotic and fatal combination (Houston 1975). Simply stated, the scale insect penetrates the bark, allowing the fungus to invade.

American beech *(Fagus grandifolia)* grows from Maine to Florida, west to Wisconsin and Texas, and in most counties in the Southern Appalachians. It is very shade tolerant and is often found growing in association with maples and birch (Houston and O'Brien 1983). In the Southern Appalachians, it is an important component of the cove hardwood forests as well as others. At high elevations it may form dense clonal stands known as beech gaps. Clonal refers to stands originating from sprouts of a single or small number of mother trees; hence has very low genetic diversity. Because of their lack of value to early loggers, many old beeches have survived and are frequently some of the oldest trees still existing in the SAA area. On the whole, American beech had no life-threatening diseases for many years. That began to change in 1890 with the arrival of beech bark disease to Nova Scotia.

Accounts from Europe indicate that the disease was killing European beech *(Fagus sylvatica)* before 1849, but it was not until 1914 that the disease complex was discovered and the *Nectria* fungus identified. By 1932, the scale-fungal complex had spread from Nova Scotia into the United States and had been identified in both Maine and Massachusetts (Houston 1975). By the 1980s, reports of the disease came from the Monongahela National Forest in West Virginia (Houston and O'Brien 1983) and, in 1993, it was found in the Great Smoky Mountains National Park in both North Carolina and Tennessee (Johnson 1995).

Declines in the beech scale population occasionally occur over large areas suggesting that environmental factors may affect the insect. More research is needed on biological control of BBD. The ladybird beetle, *Chilocorus stigma*, feeds on the scale; and a fungus, *Nematogonum ferrugineum (Gonatorrhodiella highlei),* has been reported to parasitize *Nectria* fungi. Scales on high-value ornamental trees can be controlled with insecticides. Some trees free of the disease have been found in affected areas, indicating some resistance to the scale insect. Breeding programs to increase resistance in the beech population and programs to discover the roles of biocontrol agents should be investigated (USDA FS 1993).

Butternut Canker

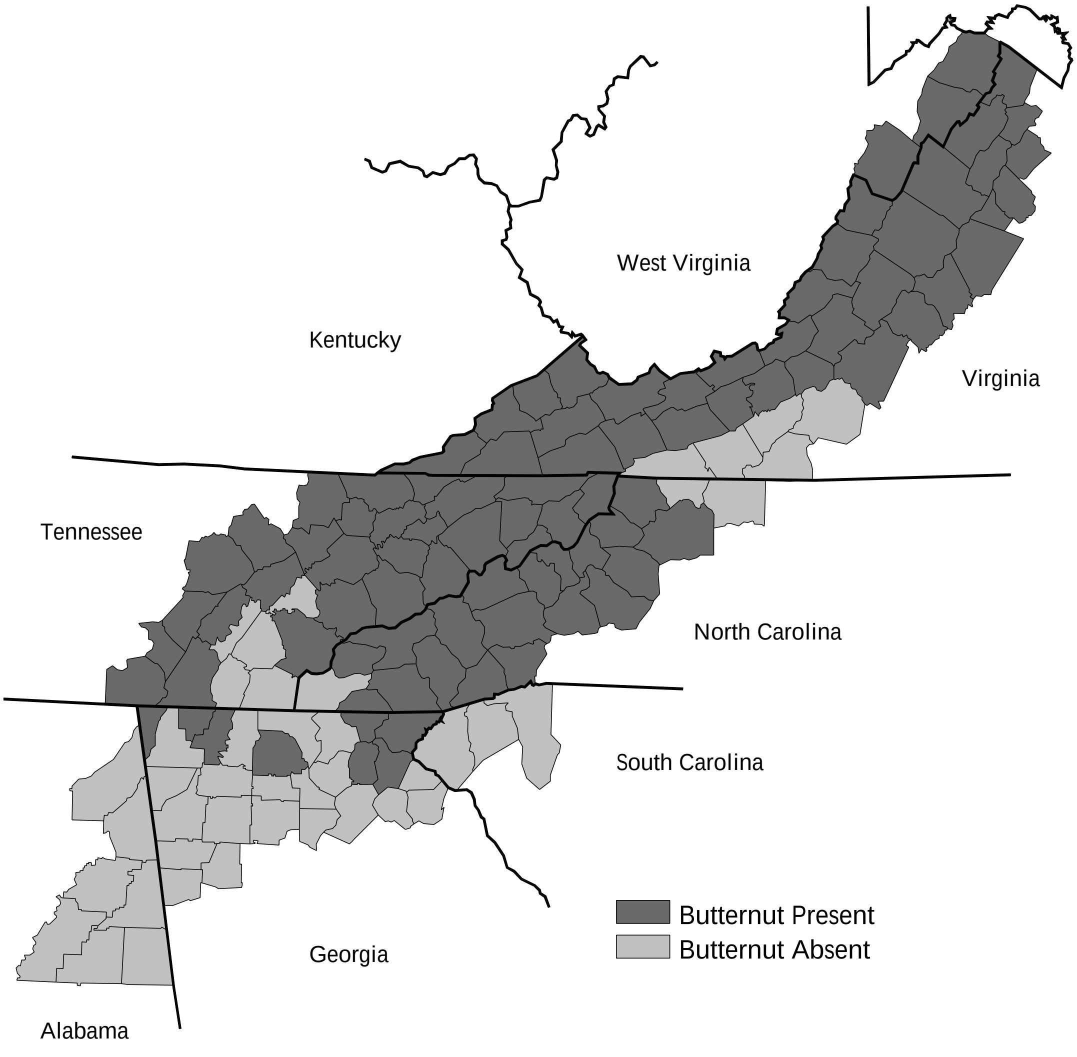
Butternut canker disease was first identified in 1967 (Anderson 1988). It is caused by the fungus, *Sirococcus clavigignenti-juglandacerum* (USDA FS 1994). During the past three decades the disease has killed nine-tenths of the butternut *(Juglans cinera)* trees in the Southern Appalachians. Unfortunately, the fungus went largely unnoticed because butternut trees are generally scattered and death from the disease is slow. Nuts from infected trees generally are not viable, therefore, declining trees do not reproduce.

Butternut normally does not occur in pure stands, but is scattered through cove and upland hardwood stands throughout its range (fig. 6.7). Its wood is highly valued and its nuts provide food for humans and wildlife.

Genetic resistance to the disease appears to exist–there are still scattered uninfected butternut trees throughout most of its range–but surviving trees are often being cut by landowners who fear that the disease will eventually infect and kill the trees, resulting in economic loss. This harvest of uninfected trees threatens to severely reduce the remaining genetic pool of resistant butternut. The identification and protection of surviving uninfected butternut trees on federal lands (Ostry and others 1994, USDA FS 1994) may be warranted. Private landowners should be informed of the genetic value of resistant or uninfected trees and encouraged to conserve such trees.

Dutch Elm Disease

Dutch elm disease, caused by the insect-carried fungus *Ceratocystis ulmi,* was introduced into the United States in 1930. It has been considered primarily an urban problem, as elms have been planted extensively as shade trees in cities and towns. This disease is spread



by two species of elm bark beetles and also by root grafts between trees in urban settings (Hanisch and others 1983).

American elm *(Ulmus americana)* is native to most of the United States, including the entire SAA region. It is most common on flats and bottomlands below 2,000 feet in elevation (Little 1971). American elm is a scattered component in mixed mesic hardwood stands throughout the SAA area, except at high elevations, but does not generally occur in pure stands. Dutch elm disease affects the species throughout its range. The disease also affects other elm species growing in the Southern Appalachians.

American elm is declining slowly in forest stands. Unlike urban elm populations, forest trees are relatively isolated from one another, and spread of the disease is slow and sporadic. Loss of American elm is of concern, but the disease is not an immediate threat to the species. Protection of individual elms in urban settings can be successful, but the cost is high. Treatment in forest settings is impractical. Additional research into both the ecological role of American elm and the health of wild American elms seems warranted.

Chestnut Blight

Chestnut blight (*Cryphonectria parasitica*) was first recorded in the United States in 1904 at the New York Zoological Park. The fungus probably arrived on nursery stock from Asia several years before. The disease spread rapidly because microscopic fungus spores can be transported by wind or on the feet of migrating birds and insects.

American chestnut had not co-evolved with the disease and had no resistance to it. Trees were quickly infected and began to die almost at once. Before the chestnut blight, American chestnut flourished on suitable sites between 1,200 feet and nearly 6,000 feet in elevation on southerly slopes and up to 4,800 feet on northerly ones. Preferring moist, but well-drained, upland soils derived from sandstone, shale, granite, or gneiss, American chestnut often made up 25 to 50 percent of hardwood stands. In many places, the proportion of chestnut in stands approached 100 percent. It did not grow well on limestone sites and was infrequent in valleys or other lowland sites with clay soils and poor internal drainage.

By 1929, nearly all counties in the SAA area were infested; and by about 1940, most of the standing chestnut trees were dead. Today, American chestnut persists throughout its former range as root sprouts growing in the understory, only occasionally attaining nutbearing age. Chestnut sprouts are numerous and will continue to survive as understory plants throughout the SAA area, though the number is probably decreasing. American chestnut is intolerant of shade and suitable disturbance is infrequent in most areas. A gradual loss of the genetic resources is expected over time without action. Sprouts generally live for 5 to 10 years before being top-killed by the blight, which girdles the stem. Often chestnuts reach heights of 25 feet or more, but they rarely flower and bear fruit before dieback.

If the species is to survive, areas with extensive chestnut root stocks should be identified and silvicultural practices should be employed in those areas to protect or enhance chestnut survival. Research should be continued into both genetic engineering for blight resistance and development of hypovirulence in the blight fungus. Planting of so-called “blight-free” chestnut has been widely publicized, but this practice is ineffective. Some seedlings advertised as “blight-free” are merely uninfected or, at best, less susceptible than chestnuts surviving in the woods as sprouts of the former population. This practice raises false hopes among the public and may discourage research funding. It should be publicly exposed.

Insect Pests

Southern Pine Beetle

Southern pine beetle (SPB) (*Dendroctonus frontalis*), infestations have occurred cyclically throughout recorded history in the South. An outbreak of SPB in a county is defined as a condition where one or more active SPB spots occur per 1,000 acres of susceptible host type. SPB outbreaks move from low levels of infestation to high levels over several years. The cycles may be localized or regional and depend upon weather and other stress factors as well as the interrelationship between the populations of SPB and its predators.

The SPB adult is 2 to 4 millimeters in length and brownish to black in color. The female SPB kills conifers by boring under the bark and destroying the cambium layer of the tree. They construct winding egg galleries while feeding and laying eggs. During outbreaks, trees are usually mass-attacked by thousands of beetles.

SPB outbreaks were reported in the late 1700s and early 1800s, but outbreaks were not systematically surveyed and recorded until the 1960s. The worst outbreak in the Southern Appalachians since the 1960s occurred between 1973 and 1976. Between 1960 and 1990, SPB outbreaks killed over $901 million worth of timber. Risk of attack by the southern pine beetle (SPB) is one factor in deciding whether to thin or regenerate southern yellow pine stands and mixed stands of yellow pine and hardwood.

The crowns of trees attacked by SPB during warm, dry weather may fade in color within 2 weeks. Dying trees are first light greenish-yellow, then yellow, and finally reddish-brown. Females often enter trees in bark crevices, and pitch flowing to the outside usually forms whitish pitch tubes. In conjunction with fading crowns and pitch tubes, reddish boring particles of chewed bark will accumulate in bark crevices.

SPB outbreaks in the SAA area are generally less dramatic than those on the Piedmont and Coastal Plain of the south because yellow pine forests types are less common in the Appalachian Mountains. SPB outbreaks have significant ecological implications, not only because of the loss of relatively scarce habitat, but because at least one yellow pine species, Table Mountain pine, cannot reproduce in the absence of fire. Table Mountain pine stands killed by SPB do not regenerate, and are permanently lost. To help land managers reduce stand susceptibility, hazard rating systems have been developed throughout the Southeastern United States. In the Southern Appalachians, the Mountain Risk System is recommended by most entomologists (Price 1994).

European Gypsy Moth

The European gypsy moth, *Lymantria dispar* (L.), is a major defoliator of hardwood trees in both forest and urban landscapes. It was introduced from Europe into Massachusetts sometime between 1867 and 1869, and because the favored host, oak, is widespread in the eastern deciduous forests, it thrived and continues to expand its range west and south each year. By the 1980s, gypsy moth was established throughout the Northeast. Today the quarantined area considered generally infested is in all or part of 16 states, including parts of West Virginia and Virginia which are in the SAA area.

The adult female gypsy moth cannot fly, so natural spread of this pest is limited to the distance that the young larvae can disperse on wind currents in a process known as ballooning. Occasionally, however, humans transport gypsy moth life stages over very long distances on vehicles, outdoor household articles, and nursery products.

The gypsy moth has a single generation per year. The egg masses, which contain from 75 to more than 1,000 eggs each, hatch in the spring at approximately the same time that budbreak occurs in the oaks. The young caterpillars climb upward, disperse via ballooning, then settle down to feed. Over the next six weeks, the caterpillars continue to feed and grow, going through six molts or growth stages, before pupating for two weeks, then emerging as adults. The adult stage is very short-lived (2 to 4 days) and does not feed at all. In fact, adult gypsy moths do not have the mouthparts necessary for feeding. The sole purpose of the adults is to locate a mate. The adult female gypsy moth cannot fly, but a chemical that she emits (pheromone) allows the males to locate her for mating. After mating, the eggs are laid in a single mass for overwintering (McManus and others 1992). Gypsy moth populations are subject to a number of natural controls that can limit their growth potential. Cool, wet weather during hatch can result in high levels of mortality in the young caterpillars. Epizootics of a naturally occurring virus and fungus can cause widespread collapses in gypsy moth populations. Despite these factors, gypsy moth populations periodically increase to outbreak levels and cause widespread defoliation (McManus and others 1992).

The gypsy moth has defoliated trees across nearly 72 million acres since 1924. About a half of that total, approximately 36 million acres, was defoliated between 1982 and 1992. This coincides with the advance of gypsy moths into the oak forest of Pennsylvania, Maryland, Virginia, and West Virginia. The gypsy moth arrived in the Southern Appalachians about 10 years ago. The first noticeable defoliation was reported in 1984. During the past 10 years, gypsy moths have defoliated more than 4 million acres in Virginia and more than 1 million acres in West Virginia (USDA FS 1994). Tree mortality after defoliation depends on the number of successive defoliations and the condition of the tree at the time of defoliation. The most severe losses occur in oak stands growing on poor sites in which trees have been under recent stress and are prone to oak decline.

Currently, only a portion of the SAA area is permanently infested by the gypsy moth. Isolated infestations have been detected and eradicated in the following counties in the SAA area: Clay, Buncombe, Ashe, Watauga, and Yancey counties in North Carolina; Giles, Floyd, and Carroll counties in Virginia; Rhea, Washington, Grainger, Johnson, Sequatchie, and Unicoi counties in Tennessee; and White and Fannin counties in Georgia. However, all of the area is at risk as the gypsy moth continues to spread. Oaks are a major component of the forests in the SAA area and a preferred food of gypsy moth larvae (Liebhold 1995).

Despite existing management strategies, losses are expected to continue as the moth migrates down the Appalachians. However, the rate at which spread occurs is affected by the strategies implemented.

Predictions based on the current rate of spread (fig. 6.8) are built on the assumption that eradication projects will continue to be implemented when isolated infestations are detected. Rates of spread would be expected to increase drastically if isolated infestations are not eradicated, with more than 90 percent of the SAA area becoming generally infested by the year 2010 (USDA FS & APHIS 1995, Liebhold and others 1995). Suppression programs do not have any effect on gypsy moth spread rates, but they may be used to mitigate losses in selected areas in the generally infested regions.

Although species vary in their ability to recover from gypsy moth defoliation, most will succumb after a few years of repeated attack. In some stands, trees die after several years of defoliation while in others one defoliation may kill trees depending on other site variables. Species composition and tree vigor are major factors in tree mortality caused by gypsy moth defoliation.

Vulnerability ratings of stands can be used to estimate the possible damage from gypsy moth attack. Vulnerability is defined as the probability of mortality that might result from defoliation.

Domestic quarantines are maintained to regulate the human-aided, long distance transport of gypsy moths from the infested to uninfested areas. Detection programs outside of the infested area pinpoint sites where gypsy moths have been introduced through inadvertent violations of the quarantine. When isolated reproducing populations are detected, eradication programs are implemented to eliminate them. Where gypsy moth is permanently established, suppression programs are carried out to reduce gypsy moth damages (USDA FS 1990).

In response to concerns that the U.S. Department of Agriculture (USDA) was not adequately addressing the apparent increase in spread rates over the past three decades (Liebhold and others 1992), the USDA Forest Service (FS) in cooperation with Animal and Plant Health Inspection Service (APHIS); the states of Michigan, West Virginia, Virginia, and North Carolina; and the National Park Service, has embarked on a pilot project called “Slow the Spread” (STS). The STS goal is to determine the feasibility of reducing the rate at which gypsy moth is currently spreading, by comprehensively implementing integrated pest management strategies over large geographic areas in the transition zone. The transition zone is located between the infested and uninfested areas. If the strategy proves successful, it could delay the impact and cost associated with gypsy moth outbreaks and suppression as gypsy moths spread through the SAA area. The STS project evaluation is expected to be complete by 1999.

The role of APHIS in STS is to administer the quarantine and conduct surveys to detect isolated infestations that are remote from the area that is generally infested. The role of the Forest Service is in gypsy moth survey and suppression in the generally infested area, either directly on federal lands or cooperatively with the states on nonfederal land. Both APHIS and the Forest Service assist states with projects to eradicate isolated infestations on nonfederal land, while the Forest Service alone is responsible for eradication on federal land (USDA FS 1990).

Specific management strategies for the gypsy moth are covered in detail in the Draft Environmental Impact Statement for Gypsy Moth Management in the United States, 1995 (DEIS). The preferred alternative includes USDA participation in suppression, eradication, and STS strategies. The DEIS is expected to be

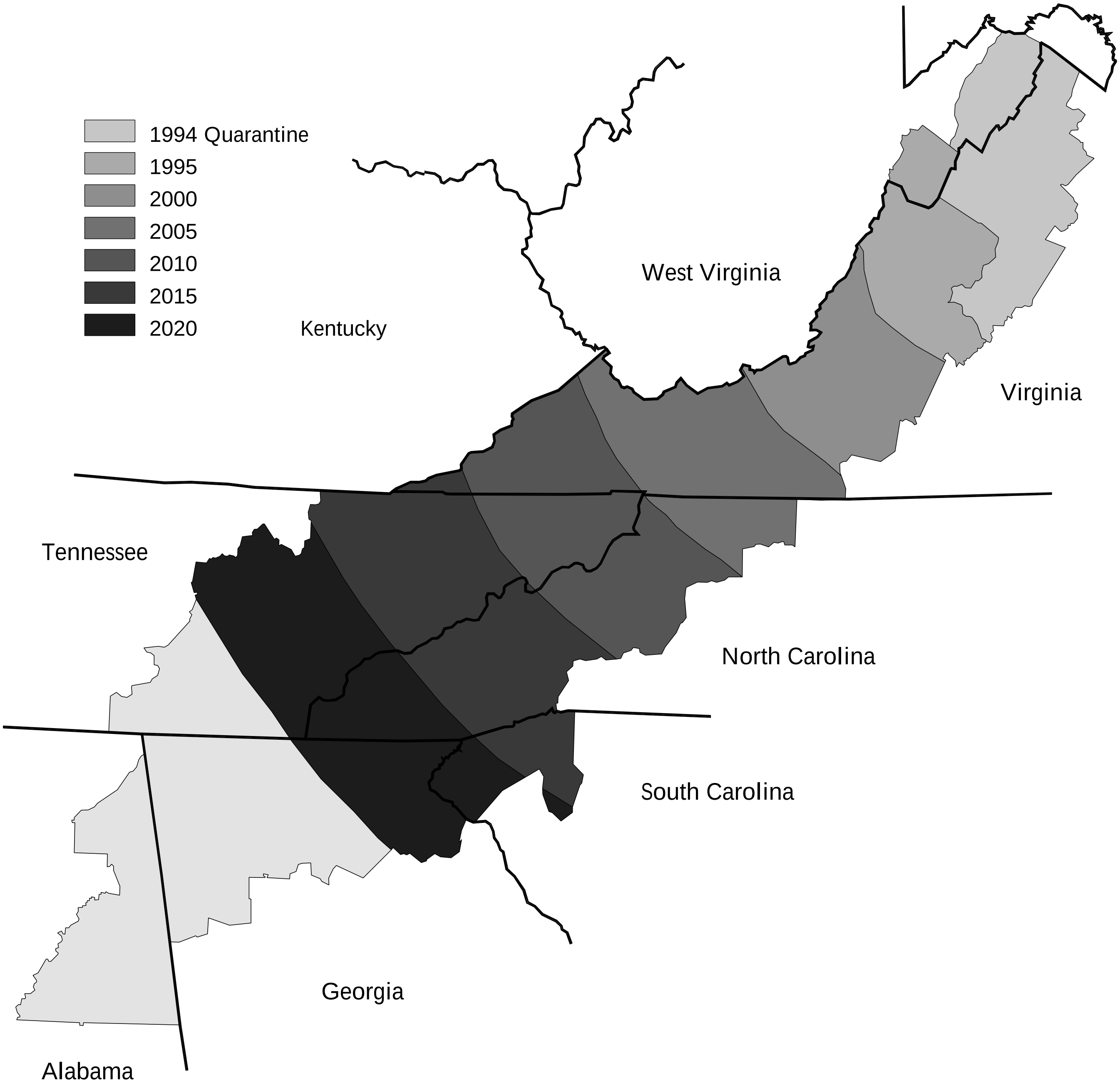


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Figure 6.8 The current infestation and predicted spread of gypsy moth in the SAA area.

finalized by the end of 1996. The final document will supersede the existing 1985 FEIS and will provide the programmatic framework for gypsy moth control over the next 5 to 10 years.

Possible responses to gypsy moth range from doing nothing, to aggressively implementing one of the management strategies documented in the 1995 FEIS for Gypsy Moth Management in the United States. The selection of a management strategy appropriate to a specific area depends on the location of that area relative to the advancing front of gypsy moth populations. On sites where impacts from gypsy moth populations are expected to interfere with management objectives, such as recreation or timber, an array of control tactics is available to suppress or eradicate the infestation. Specific control tactics are discussed in detail in the 1995 FEIS and are briefly outlined in table 6.1.

Continued location, delineation, and elimination of isolated gypsy moth populations will be important to maintain gypsy moth spread at rates no faster than predicted. Further evaluation of the STS project is needed to determine if spread rates can be reduced from those predicted in Figure 6.8. If the STS strategy is demonstrated to be biologically sound and economically efficient, it may be integrated into the national strategy for management of the gypsy moth.

Silvicultural practices, in combination with programs such as STS, need to be implemented to control the damage from gypsy moth. Such practices can modify susceptibility and vulnerability of stands before the gypsy moth affects them.

It may be appropriate to develop plans to: (1) provide more information to the public about gypsy moth, (2) suggest control options, (3) develop and implement an integrated plan for altering the forest composition in high-risk areas on state and federal land, and (4) and assess high-risk areas on private land and assist landowners.

Hemlock Woolly Adelgid

Hemlock woolly adelgid, *Adelges tsugae,* an insect species native to Asia, was first identified in the eastern United States in 1924 in Richmond, VA, but it has recently expanded into the Southern Appalachians and threatens to spread throughout the ranges of eastern and Carolina hemlock. It is currently established along the mountainous regions around the Shenandoah Valley, and it is spreading southward along the Blue Ridge, and northward into New England. The adelgid may be spread by wind, birds, or mammals (McClure 1990). Long range movement of the adelgid by migrating songbirds in the spring could explain why northward spread has been faster than southward spread. All of the SAA area in Virginia, except for seven counties in the extreme western part of the commonwealth, are now infested.

There are two species of hemlock in the SAA area, eastern hemlock *(Tsuga canadensis)* and Carolina hemlock *(Tsuga caroliniana).* The former is an important component of riparian ecosystems, providing cooling shade for streams, contributing nutrients for streams through litterfall, and providing winter shelter for wildlife. It may also be important as a feeding and nesting niche for neotropical migrant birds (Rhea and Watson 1994). Carolina hemlock, on the other hand, is less understood ecologically. It generally occupies more xeric sites on ridges and rock outcrops, but it also probably provides cover and nesting sites for birds and small mammals. Both eastern hemlock and Carolina hemlock are threatened by the adelgid (figs. 6.9 and 6.10).

Once infested by the adelgid, hemlocks are weakened, gradually lose their foliage, and are unable to refoliate or produce cones. Mortality occurs after complete defoliation, generally within 5 years of initial infestation (McClure 1987). There is no known genetic resistance to adelgids in either of the native Appalachian hemlock species, but resistance is known to

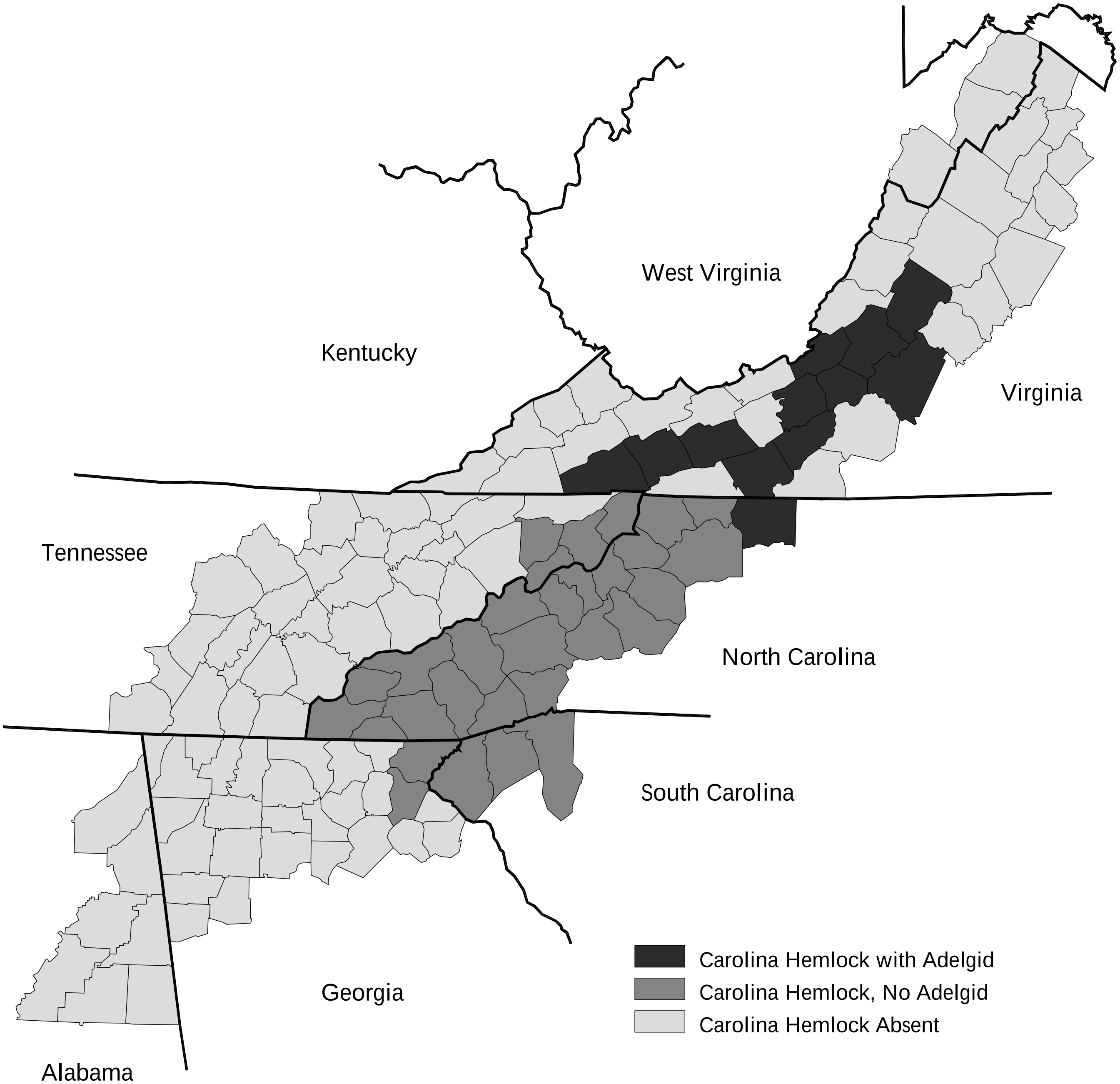
Table 6.1 Gypsy moth monitoring and treatment options available with suppression, eradication, and "slow the spread" strategies.

	Activity		
		Eradication	
	Suppression	Monitoring Methods	Slow the Spread
Treatment Options[1]	Defoliation survey	Pheromone traps	Pheromone traps
Bacillus thuringiensis	x	x	x
Diflubenzuron	x	x	x
Virus[2]	x	x	x
Mass Trapping[2]		x	x
Mating Disruption[2]		x	x
Sterile Insects[2]		x	x

[1] No treatment is an option in all strategies

occur in hemlocks native to Asia and in the two species native to the Western United States. Individual hemlock trees can be protected by spraying or soil treatments, but such treatment is impractical for forest trees (Rhea 1996). It appears that all untreated hemlocks, with the possible exception of small geographically-isolated populations, could eventually be killed by the adelgid. Loss of hemlock will negatively impact riparian ecosystems and may result in a substantial decline in habitat quality for birds and other wildlife (Rhea 1996).

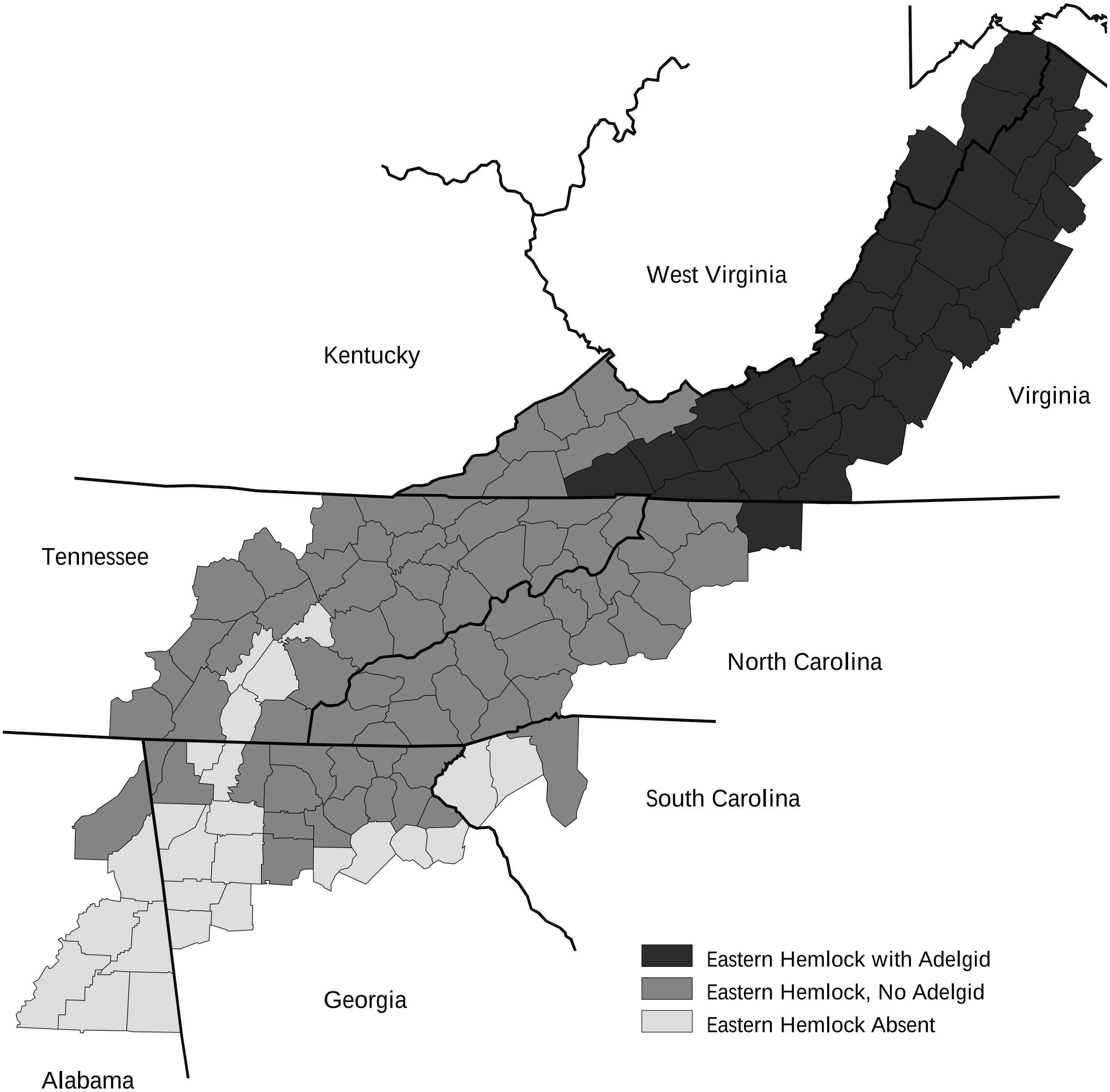
If the two species are to be preserved, efforts to treat and protect selected hemlocks in key areas should be continued and expanded. Research should be initiated into possible genetic engineering to transfer adelgid resistance from other hemlock species into eastern



and Carolina hemlocks. As soon as possible, a collection of seed and scion material should be made from throughout the ranges of both hemlock species in the Southern Appalachians. This material would then be used to establish a hemlock nursery in an area where it can readily be protected to preserve as much of the genetic bases of both species as possible.

Balsam Woolly Adelgid

The balsam woolly adelgid is one of the most significant disturbance factors to high-elevation Southern Appalachian spruce-fir forests. The balsam woolly adelgid was first detected in the Southern Appalachians on Mount Mitchell in the Black Mountains of



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North Carolina in 1957, but it is suspected to have arrived in the southern mountains in the 1930s via reforestation experiments. When mature, Fraser fir, a Southern Appalachian endemic, is highly susceptible to adelgid attack. Death occurs within 5 years after first attacks. Adelgid infestations spread throughout the Black Mountains within a few years after initial detection (Speers 1958). The insect then spread to the Fraser fir communities throughout the Southern Appalachians. Fraser fir is the only fir species found in the southeastern United States and only has natural populations in western North Carolina, eastern Tennessee, and southwestern Virginia. Since the detection of the insect in the Southern Appalachians, the insect spread to all natural fir populations by the early 1980s.

The balsam woolly adelgid is a small, wingless insect whose North American populations are entirely female and reproduce from unfertilized eggs. An adelgid may lay as many as 100 eggs. The balsam woolly adelgid produces at least two generations per year in North America, and may produce up to four generations in the South. The adelgid is primarily disseminated by wind, but also by gravity, humans, nursery stock, and animals.

During feeding, the adelgid injects salivary compounds into the Fraser fir bole, stimulating the cambium to produce abnormal xylem. The xylem forms wider-than-normal annual rings, called *rotholz,* that are a dark red in color. *Rotholz* causes an increasing and significant reduction in sapwood conductance; thus, the balsam woolly adelgid causes severe water stress in infested Fraser firs (Speers 1958).

While most fir species have a wound response to adelgid infestation, this mechanism seems to be incomplete in most Fraser fir. Other fir species, especially those that have co-evolved with the insect, respond vigorously to adelgid damage and often recover. In fact, even a few stands of Fraser fir seem to have some resistance. The infested Fraser fir on Mount Rogers, Virginia, for example, often produce more outer bark at a higher rate than infested fir in the rest of the Southern Appalachians. This response may explain what appears to be a limited resistance of the Mount Rogers populations.

Human control efforts to reduce the spread of the adelgid have failed. The first infested trees detected in the Great Smoky Mountains were cut to slow the spread of infestation. Preventative cuts were soon discontinued, however, when it was discovered that eggs and young adelgids are detached during felling, literally creating a cloud of infestation sources that can be carried a considerable distance by wind. Various insecticides have proven effective. Unfortunately, most are also highly toxic to other insects. In addition, since the adelgid is a stem-feeder, aerial application techniques do not work, and each infested bole must be sprayed by hand. A less toxic, but less effective, alternative (potassium oleate soap) is applied annually to stands around the parking lot and observation tower trail at Clingman's Dome, but even these stands are beginning to show significant impact from the adelgid (Eager 1984).

The balsam woolly adelgid is extremely resistant to climate-caused mortality. Native and introduced predators of the insect have had little effect. The result has been that the adelgid has dramatically changed the Southern Appalachian spruce-fir ecosystem (Nicholas and others 1992).

The biology of the balsam woolly adelgid has been studied for more than 30 years, but the probability of Fraser fir extinction has not yet been answered satisfactorily. This uncertainty is reflected by the U.S. Fish and Wildlife Service's 1993 review of Fraser fir for possible listing as a threatened or endangered species. Its listing was deemed "possibly appropriate." Some scientists predict that it will survive, based on observations of successful regeneration and cone-bearing trees. There may be a cycle of adelgid infestation followed by fir regeneration that survives to produce viable seeds before death.

Asiatic Gypsy Moth

In 1990, U.S. and Canadian regulatory officials documented the introduction of the Asiatic gypsy moth (AGM) into various ports in the Pacific Northwest. Ports in Washington, Oregon, and British Columbia first reported the AGM in 1991. Ships carrying egg masses from Russian ports most likely introduced the pest while visiting West Coast ports. The moths were reported to have entered North Carolina in July 1993, arriving on a munitions ship docked near Wilmington. North Carolina has since begun a $9.4 million project to eliminate AGM from the two counties apparently affected. Female Asiatic gypsy moths are capable of strong directed flight and have a host range broader than that of the European gypsy moth strain currently established in North America. Studies have demonstrated that the AGM feeds more voraciously than the European gypsy moth, and grows faster and larger, feeding on similar tree species. In the former Soviet Union, the AGM browses on an estimated 600 tree species.

The flying ability of the female AGM means that the species could spread at a rate of three times as fast as its European relative. It is virtually impossible to tell the difference between the two gypsy moth strains based on appearances. To identify the Asian strain, scientists must capture a female moth in flight or genetic analysis of mitochondrial DNA markers.

Asiatic Oak Weevil

The Asiatic oak weevil, *Cyrtepistomus castaneuous*, is an accidentally introduced pest that has spread throughout eastern North American forests. It feeds on many hardwood tree species in the eastern United States. The insect has one generation per year, and overwinters primarily as larvae in the soil. Adults are most commonly found from July to October (Campbell and Schlarbaum 1994).

The weevil has not yet been reported to be causing economic damage to timber. Probably the most critical damage is to the root systems in the dormant season through midsummer by the larvae. The insect usually does not cause enough visible damage to be noticed, but defoliation of seedlings, under controlled conditions can be severe (Schlarbaum and others 1993).

Future prognosis is uncertain. The Asiatic oak weevil may become a problem in seed orchards or in areas with high concentrations of oak (Triplehorn 1955). There have been few studies monitoring the populations or the damage to oak. If this pest is to be understood, it must be monitored for population increases and damage to forests. Recommendations for changes in management practices require sufficient data on susceptibility and vulnerability.

Exotic Plants

When exotic species are introduced into a favorable new environment without their normal complement of limiting factors such as pathogens, predators, and competition, they often expand aggressively. Introduced plants that can grow, reproduce, and spread rapidly tend to produce major disturbances in their new plant communities. The effects of exotic plants depend on the specific character of the plants themselves, and the intended use of the land they occupy.

Exotic plant species have been introduced into the Southern Appalachians since the beginning of European settlement of the region. Some plants were brought intentionally as agricultural crops and domestic plants. Others were introduced accidentally when seeds were carried into the region by wind, water, humans, or animals. Many of these introductions have posed no problems, remaining essentially within the boundaries of human cultivation. Some, however, have escaped and spread, displacing native vegetation, and causing ecological disturbance and, in some cases, economic loss or impaired land use.

Both privet (*Lingustrum* spp.) and Japanese honeysuckle *(Lonicera japonica)* are shade-tolerant and form a dense layer of low vegetation, sometimes altering forest regeneration patterns. Asiatic bittersweet *(Celatrus orbiculatis)* another pervasive shade-tolerant plant, is not known to hamper stand regeneration. Nepalgrass *(Microstegium vimineum)* carpets moist forest understories, changing the composition of the herbaceous layer.

Some introduced shade-tolerant species, such as autumn olive *(Elaeagnus umbellata),* multiflora rose *(Rosa multiflora),* and kudzu *(Pueraria lobata)* can cause local problems. Canada thistle *(Cirsium arvense),* is a large, fast growing, spiny plant that aggressively colonizes roadsides, fields, lawns, and other relatively open areas. It causes losses on cropland, obstructs rights-of-way, impairs use of residential and recreation areas, and displaces native flora on sites it colonizes.

Sometimes introduced plants produce positive effects. While Japanese honeysuckle *(Lonicera japonica)* can displace native vegetation, it produces valuable browse for deer, fruit for songbirds, and nesting and escape cover for a variety of birds and small mammals. It also bears masses of fragrant blossoms, which probably account for its original introduction. Honeysuckle, might be considered desirable in some residential areas, and in many forestry and wildlife management areas, but it is undesirable as a competitor with sensitive plants, or in areas such as national parks, where maintenance of native vegetation is a management objective.

National forests in the Southern Appalachians have generally not attempted to control exotic plants except for kudzu, which has serious localized impacts on forestry. Other exotics, such as introduced privet threaten to become problems in spots on national forests. Non-native plants such as crown vetch, lespedezas, white dutch clover, and tall fescue have commonly been planted for erosion control after timber harvests and road construction, or as food for wildlife.

National parks, however, generally have programs to control exotic plants. Parks in the SAA region list approximately 40 species varying by park requiring control. Other exotic plants currently in the U.S. have the potential to invade forests and parklands. Where national parks adjoin national forests and other federal and state ownerships, uncontrolled infestations of exotic plants often cross boundaries and create continuing management problems for the parks.

Four basic strategies are available for solving exotic plant problems: prevention, eradication, suppression, and biological control.

- Prevention is the identification and interdiction of exotic plants, plant parts, or plant propagules before they enter the United States.
- Eradication is the complete elimination of a population of an introduced exotic. It is effective against relatively small, localized infestations but requires intense effort and may be relatively expensive. Extensive use of herbicides is usually necessary, and some injury to desirable plants or the surrounding environment may be unavoidable. Eradication of large, well-established populations usually is not feasible.
- Suppression is the periodic control or elimination of a population of exotics within a generally infested area, such as the seasonal treatment of thistles within a campground. Suppression offers only a temporary solution to the exotic plant problem, and generally must be repeated at regular intervals. It generally becomes a permanent maintenance project unless biological control can be established.
- Biological control involves the identification and introduction of an exotic plant's natural control agents, usually insects or fungi, from its native environment. This is an expensive and time-consuming process because extensive research must be conducted to ensure that the proposed control agent will not cause further problems in its new environment. Biological control, if successful, brings the exotic plant species into balance with its environment so that it continues to be a component of the plant community but will not dominate it. However, biological control is not always possible or practical.

The Effects of Management Practices on the Health of Forest Vegetation

Question 8:

How are current and past management affecting the health and integrity of forest vegetation in the Southern Appalachians?

The assessment focused primarily on forest management activities that involve manipulation of vegetation. Unquestionably, other activities such as land use conversion, mining, grazing, and agriculture can have significant impacts on the structure and composition of the forest. Other reports prepared for the assessment included discussions of some of these impacts.

Four major topic areas were examined: (1) past management, primarily from the early 1900s throughout the 20th century; (2) recent forest management on national forests and private land; (3) current timber markets, growth, and inventory on public and private lands; and (4) three forms of active management that can have significant effect on future forest health–integrated pest management, genetic resource conservation programs, and improved monitoring systems.

History of Forest Management

Timber harvesting in the early 1900s dramatically affected the Southern Appalachian Assessment (SAA) area landscapes. Between 1900 and 1920, roughly 60 percent of the Southern Appalachian forest was cut over. In 1908, the Secretary of State's report estimated 86 percent of the acreage in the Southern Appalachians was cleared; in various stages of regrowth; or in young, secondary forest. According to the report, practically the entire Southern Appalachian forest had been burned. The land-use practices of the late 19th century and early 20th century resulted in large expanses of even-aged forests. Multiple-use management and fire control were instituted on public land after the Great Depression. Tourism and recreational use skyrocketed between 1945 and 1960. In 1963, even-aged management became national forest policy and common practice in the Southern Appalachian national forests (Yarnell 1995). Since that time, even-aged harvesting on national forests has been done on about 0.5 percent of the national forest acreage annually. Thus, about 5 percent has been cut per decade.

Recent Trends in Forest Management

The health of forest vegetation is affected by both past and current management. Past impacts have greatly influenced stand composition and structure. Some current pest problems and predictable future problems may be directly or indirectly the result of past events. An example is found in the current dominance of oak types in the SAA area. Oak has always been an important component of the ecosystem, but probably became more important with the loss of chestnut to chestnut blight in the early 20th century. Other activities of that time, including abusive logging practices, grazing, and wildfire, may have created conditions more favorable for oak regeneration than for pioneer tree species. These young oak stands gave rise to current oak overstory dominance. Many oak stands today, however, are vulnerable to oak decline, a stress-mediated disease complex of mature oak. Now and in the near future, another agent of change, the gypsy moth, threatens to further impact oak forest types because oaks are the preferred food source of gypsy moth. Oak defoliation coupled with decline is likely to cause high mortality rates in oak forest types. As a result, the current trend in vegetation for much of the SAA area is toward a reduction in stocking of oaks, and toward a forest dominated by maple, yellow-poplar, ash, blackgum, and perhaps white pine.

Table 7.1 A summary of the acres of silvicultural activities for private and state owned land by states within the Southern Appalachian Assessment area between October 1, 1989 and September 30, 1994.

	Treatment Acres[1]			
	Tree Planting	Natural Regeneration	Timber Stand Improvement	Prescribed Burning[2]
Alabama	27,689	844	20,935	21,462
Georgia	359,924	780,000	3,500	826,000
North Carolina	10,455	11,114	1,063	2,030
South Carolina	9,810	10,500	1,465	3,197
Tennessee	68,149	226	601	65,064
Virginia	52,691	20,008	30,345	7,854
Total	518,718	822,692	57,909	925,607

[1] Acres derived from state reports collected at the district (multi-county) level. Since state districts do not coincide precisely with counties in the Southern Appalachian Assessment (SAA) area, acres include activities in some counties outside the SAA area.

[2] Includes burning for fuel reduction, hardwood control, wildlife habitat, Threatened and Endangered species and site preparation.

(Source: State Foresters)

Forest Management on Public and Private Land

The possible effects on vegetative structure and composition, and consequent effects on forest health were assessed by compiling information on private land and a sample of SAA national forests. To assess the amount of various forestry activities on private and state land in the SAA area, questionnaires were sent to state foresters. Case studies were done on three national forests with land primarily in the SAA area. Herbicide use from 1991 to 1994 was assessed. Kinds of products, rates of application, and acres treated were determined for the case study on national forests.

Over 3 million acres of public and private land in the SAA area have received some form of vegetation management treatment during the past 6 years. Table 7.1 shows 1989 to 1994 for each state, based on information provided by state foresters: the amount of tree planting, natural regeneration, timber stand improvement, and prescribed burning on private and state land. Over the whole region, 38 percent of the regeneration was accomplished naturally and 62 percent by tree planting. Table 7.2 shows a trend toward natural regeneration since 1988 (Lantz 1994). The implications of this trend for forest health are probably mixed.

Natural regeneration, which is generally associated with less intensive site preparation, will usually result in more vegetatively diverse mixed pine-hardwood stands which should be more resistant to some pests.

Even-age regeneration harvesting (clearcutting, seedtree, and shelterwood systems) on national forests is declining (table 7.3). For the case study forests, only about half as much regeneration harvesting occurred in 1994 as in 1991. Site preparation for artificial regeneration, tree planting, and timber stand improvement acres have declined over the last five years.

Acres treated with herbicides on the three case forests declined dramatically from 1990 through 1994. Methods of herbicide application are shifting from broadcast toward individual stem treatments on public lands.

Table 7.2 The trends of harvested versus planted acres within the Southern Appalachian Assessment area for the years between 1988–1994.

Season	Acres Harvested (M)	Acres Planted (M)	Percent Planted
1988-89	3,675	2,290	62
1989-90	3,660	1,912	52
1990-1991	2,667	1,709	64
1991-1992	3,038	1,721	56
1992-1993	3,392	1,691	50
1993-1994	4,066 (estimated)	1,696	42

Table 7.3 An acreage summary of some vegetation management activities for the Cherokee, George Washington, and Jefferson National forests case study from 1990 to 1994.

	1990	1991	1992	1993	1994
Cherokee National Forest					
Regeneration Cutting (Even–aged method)	N/A	2,928	2,219	1,084	1,036
Thinning	N/A	10	71	220	298
Tree Planting	1,540	1,444	1,488	1,194	1,000
Site Preparation for Natural Regeneration	834	1,096	950	1,288	1,109
Natural Regeneration without Site Preparation	0	0	613	82	108
Site Preparation for Artificial Regeneration	1,810	1,811	1,722	1,150	841
Timber Stand Improvement	3,233	1,390	1,798	1,441	1,219
George Washington National Forest					
Regeneration Cutting (Even–aged method)	N/A	1,950	1,754	1,369	971
Thinning	N/A	304	268	286	294
Tree Planting	736	513	534	340	90
Site Preparation for Natural Regeneration	2,363	2,149	2,373	2,058	1,535
Natural Regeneration without Site Preparation	0	0	0	0	0
Site Preparation for Artificial Regeneration	464	452	328	163	42
Timber Stand Improvement	862	1,429	678	1,010	575
Jefferson National Forest					
Regeneration Cutting (Even–aged method)	N/A	876	694	1,214	489
Thinning	N/A	379	71	72	0
Tree Planting	438	358	259	396	383
Site Preparation for Natural Regeneration	2,087	1,666	1,657	787	597
Natural Regeneration without Site Preparation	0	0	68	614	406
Site Preparation for Artificial Regeneration	459	278	255	244	199
Timber Stand Improvement	1,071	975	969	1,707	907

For national forests in the SAA area, essentially no prescribed burning to control understory species was accomplished between 1990 and 1994. Little or no controlled burning has been done on National Park Service land in recent years.

Changes in Land Use

Changes in land use with regard to the utilization of timber products have a great impact on stand structure and composition which in turn affect forest health. These land use changes are often a result of many factors: “The supply of timber is more complex than the supply of most commodities, because timber is produced by dynamic forests and controlled by a variety of owners. The inventory of timber growing stock can be altered by timber harvests, natural forces, or investments in regeneration and stand improvements. Harvest and investment decisions in turn are influenced by competing demands for forestland and landowner preferences.” (SAMAB 1996C).

Forest acreage has decreased by 2 percent since the mid-1970s. This decrease in forested acres is expected to continue at the same pace through the year 2010. This loss of forest acres is occurring primarily on private lands. Clearing is for development and conversion to agricultural use. See Chapter 3 for additional discussions of changing land use patterns.

Existing Timber Inventory and Markets

A number of key findings included in the timber economy chapter of the SAA Social/Cultural/Economic Technical Report is relevant to this issue of current management and its effect on forest health (SAMAB 1996c):

1. National forests, on average, produce less timber than private lands in the region. As a result, national forests have more timber inventory per acre, less removal, less growth, and slightly higher mortality than private land in the area.
2. While holding 17 percent of the timberland in the SAA area, the national forests hold a disproportionately high share of the highest-valued sawtimber. It is likely that national forests will continue to have a dominant influence over the production, and therefore the prices, of high-quality oak sawtimber in the Southern Appalachians.

3. Timber production from the national forests of the region expanded from the late 1970s through the mid-1980s. After peaking in 1985, timber sale levels have declined in the region, especially in 1991. (See table 7.4 for acres of harvest for the SAA national forests from 1991 to 1994.) Current sale levels are now roughly comparable to those of the 1970s.

Forest Health and Timber Supply

Ultimately, increased mortality and reductions in growth resulting from forest health problems could have important effects on forest management and timber supply in the Southern Appalachians. Three forest health issues are particularly relevant to timber supply: (1) gypsy moths in the northernmost part of the region, (2) oak decline from southern Virginia to northern Georgia, and (3) southern pine beetles in the southern quarter of the SAA. Mortality and forest growth rates across the timber subregions were examined for evidence of these impacts, but none was found. There may be a substantial lag between pest incidence and growth/mortality effects measurable in regional surveys. Continued monitoring and further research of pest impacts on timber supplies are warranted.

The assessment of timber markets in the SAA indicates that markets for high-quality oak species are especially strong. In addition, it indicates that markets for low-quality material for pulp and composite board manufacture are also expanding. Taken in combination, these findings suggest that more intermediate treatments of oak stands could become economically viable in the future. Intermediate treatments could also improve stand vigor, thereby mitigating the effects of oak decline in these stands. Evolving markets may therefore provide an opportunity to improve forest health.

Integrated Pest Management

Native insects and pathogens are normal parts of functioning forest ecosystems and can profoundly influence forest structure, species composition, and diversity. Some of these functions include regulating populations of woody and herbaceous plants and, hence, regulating forest succession, carbon, and nutrient cycling; serving as a food source for vertebrates and invertebrates; creating wildlife habitat; pollinating; and acting as mycorrhizal symbionts. It is neither desirable nor possible to eradicate them on a broad scale.

By contrast, introduced insects, pathogens, animals, and weeds are not normal parts of the invaded ecosystems. For the most part, their effects are similar to natives, but the magnitudes of the changes they cause are more extreme. This is due to the lack of co-evolved resistance mechanisms in their new hosts and the absence of the parasites, predators, and diseases that served to regulate their populations in their native ecosystems. A few beneficial parasites have been introduced to control other introduced insect pests.

Some of the insects and pathogens introduced into the SAA area include the chestnut blight fungus, the European gypsy moth, the beech bark disease insect-pathogen complex, the hemlock woolly adelgid, the balsam woolly adelgid, the dogwood anthracnose fungus, the butternut canker fungus, the Dutch elm disease fungus, and the Asiatic oak weevil.

Insect and pathogen populations fluctuate over time. Examples of extreme population sizes from the SAA are an outbreak of elm spanworm (a native insect defoliator) that occurred between 1954 and 1964 in north Georgia, western North Carolina, and eastern Tennessee (Ciesla and others 1963, Ciesla and others 1965) and the chestnut blight epidemic that covered the SAA during the 1920s and 1930s. In the former case, the outbreak collapsed due primarily to a native wasp parasite of elm spanworm eggs. A previous outbreak of this insect was recorded between 1878 and 1881, when about 1.5 million acres were defoliated. Chestnut blight had no prior history in the SAA area before being detected in 1908. The blight did not abate until virtually all American chestnut trees in the SAA were killed. The tree persists today as small stump sprouts in the understory, growing for a few years until it is killed back to the ground.

In an ecological context of ecosystems, the term “pest” is meaningless. Only when human values are introduced does “pest” acquire meaning: an insect or pathogen that reduces natural resources that are valued by humans. Pest management is the application of techniques to protect human values against impacts that are in conflict with human values. Integrated pest management (IPM) “is an

Table 7.4 A summary of acres by cutting method for national forests within the Southern Appalachian Assessment area for 1991 to 1994.

	Acres Sold by Cutting Method				
	Regeneration Cutting[1]		Thinning		Total
Forest	(Acres)	(%)	(Acres)	(%)	(Acres)
FY 1991					
Talladega	1,305	53	1,167	47	2,472
Chattahoochee	3,686	66	1,879	34	5,565
Cherokee	2,928	78	10	22	2,938
George Washington	1,950	87	304	13	2,254
Nantahala/Pisgah	1,639	81	396	19	2,035
Andrew Pickens	0	-	0	-	0
Jefferson	876	70	379	30	1,255
Total	12,384	75	4,135	25	16,519
FY 1992					
Talladega	1,134	37	1,928	63	3,062
Chattahoochee	2,855	66	1,469	34	4,324
Cherokee	2,219	97	71	3	2,290
George Washington	1,754	87	268	13	2,022
Nantahala/Pisgah	1,936	72	747	28	2,683
Andrew Pickens	17	18	79	82	96
Jefferson	694	91	71	9	765
Total	10,609	70	4,633	30	15,242
FY 1993					
Talladega	243	17	1,198	83	1,441
Chattahoochee	1,718	42	2,353	58	4,071
Cherokee	1,084	83	220	17	1,304
George Washington	1,369	83	286	17	1,655
Nantahala/Pisgah	1,512	68	712	32	2,224
Andrew Pickens	0	0	339	100	339
Jefferson	1,214	94	72	6	1,286
Total	7,140	58	5,180	42	12,320
FY 1994					
Talladega	668	12	4,708	88	5,376
Chattahoochee	1,557	44	1,997	56	3,554
Cherokee	1,036	78	298	22	1,334
George Washington	971	77	294	23	1,265
Nantahala/Pisgah	1,353	64	776	36	2,129
Andrew Pickens	16	11	129	89	145
Jefferson	489	100	0	0	489
Total	6,090	43	8,202	57	14,292

[1] Includes clearcut, seedtree, and shelterwood methods

ecological approach to pest management where all available necessary techniques are consolidated into a unified program, so that populations can be managed in such a manner that economic damage is avoided and adverse side effects are minimized." (National Academy of Sciences 1969). IPM arose out of concern over widespread use of non-selective pesticides in the 1960s with little regard for ecosystem impacts. It has evolved from a simplistic blending of biological control agents with more traditional chemical insecticide treatments and acknowledges the many interactions that exist between insects, plant diseases, and the environment.

In the above definition, the word "economic" could be replaced by "scenic, biologic (as in biodiversity), wildlife habitat, human health and safety," or any other management objective (i.e. social value) alone or in combination. However, the actual or perceived economies of these social values determine whether an IPM program and implemented. Social values without easily quantified economies will support IPM programs only when a high level of difficult-to-obtain social consensus exists. The vast majority of epidemics and outbreaks of forest insects and pathogens is not managed either before the fact as prevention, or after the fact as suppression or attempted eradication.

As defined by the National Academy of Sciences (1969), the basic principles of IPM are:

- consideration of ecosystem functions;
- utilization of indigenous natural control agents;
- maintenance (or enhancement) of ecosystem complexity;
- avoidance of ecologically disruptive actions;
- application of minimum selective hazards;
- exclusion from new areas;
- host plant adaptability to ecosystems;
- prediction of population trends; and
- maintenance of sub-economic (or other social value) thresholds.

IPM methods can be classified into four categories: prevention, silvicultural, biological, and chemical.

Preventative methods include such activities as risk rating of landscapes prior to infestation, training personnel, detection, diagnosis, and evaluation of those threats, and exclusion of threats from areas of interest where they do not yet exist.

Silvicultural methods involve maintaining or enhancing resistance to and resilience after stress. These can include the improving of tree and/or stand vigor by thinning, salvage of individual trees or stands that pose threats to surrounding forests, proper selection of harvest method and scheduling, and the use of prescribed fire. Applied genetic methods of silviculture include: matching tree species to the sites that they are best adapted, selecting the most competitive individuals, and using genetically improved stock.

Biological methods include the use of behavioral chemicals such as sex or aggregation pheromones; the use of viruses, bacteria, or fungal pathogens; and the use of parasitic or predatory insects. Since behavioral compounds are synthetic, sufficient quantities are available for large scale detection surveys and eradication projects.

Chemical methods involve the application of direct chemical control agents such as insecticides, fungicides, or in some cases, herbicides.

IPM approaches are rarely applied in the SAA area due primarily to economic and political considerations. The public generally has incomplete knowledge of, and/or lacks consensus on the threats to economic or social values of most forest insects and pathogens, although millions of acres are affected each year. Where sufficient perception, knowledge, and a degree of consensus exist (such as for gypsy moth and southern pine beetle management) IPM programs are employed. These programs are detailed in Environmental Impact Statements (EIS) that guide federal cost sharing for detection, evaluation, and treatment of infestations of these two pests (USDA FS 1987, USDA FS 1995). Several steps are common to both programs. These are:

- survey and detection;
- evaluation of resources at risk;
- economic analysis;
- project proposal;
- National Environmental Policy Act (NEPA) analysis (environmental analysis, public involvement, evaluation of alternatives, selection of preferred alternative);
- project implementation; and
- post-eradication or post-suppression evaluation.
- Monitoring of project results is used to guide and inform research and development of new technologies.

Appendix G includes IPM techniques for Gypsy moth and SPB.

Genetic Conservation Programs

Several tree species in the Southern Appalachians are at risk of extinction or significant genetic loss because of exotic pests. These include American chestnut, chinkapin, butternut, eastern and Carolina hemlock, Fraser fir, flowering dogwood, and American beech. Gene conservation strategies and adequate support are needed to address both short-term and long-term concerns. A small amount of genetic material is conserved in national seed conservation facilities and arboretums, but there is no coordinated, funded strategy to address the gene conservation for most of the imperiled or potentially imperiled woody plants in the Southern Appalachian area.

There is an obvious dilemma in considering what species should be chosen for protection when there are many in need. One criterion must be rarity. Some taxa are naturally rare, whereas others are artificially rare as a result of human actions. Species that have not evolved under situations of rarity may be biologically less stable than those that are naturally rare. Off-site protection may be the best approach for conserving hemlock, butternut, chestnut, chinkapin, Fraser fir, high-elevation samples of flowering dogwood, and American beech. A combination of seed banking germplasm and the outplanting of samples in operational seed orchards would be necessary to conserve genetic material.

The threats from exotic pests for species of most concern in the SAA are particularly menacing. These species are showing little to no host resistance and many (if not all) may be lost as ecosystem components within the next two decades. Due to the severity of pest effects and a low probability of natural resistance, adequate onsite protection is not feasible. Because many of the species are not commercially important, they are not included in typical federal, state, or private genetic resource programs. Two of the species, Fraser fir and American beech, are important components of unique ecological communities–Fraser fir as a component of high-elevation spruce-fir and pure Fraser fir types and American beech as a component of high-elevation beech/birch/maple types, beech gaps, and beech boulderfields. Weakened or nonexistent populations of the above species will have great ecological ramifications.

Historically, tree breeding programs are fairly young. In 1958, the USDA Forest Service (FS), Region 8 Tree Improvement Program was begun. Currently, the Southern Region has readily available, high-quality tree seeds. Established seed orchards are capable of producing most of the seeds needed for reforestation.

Oaks

Northern red oak and white oak are the two most valuable hardwood species found growing in the southern Appalachians and the Piedmont. Both of these species occur widely, and both are very valuable for timber and for wildlife habitat. Neither species is adequately regenerating, either naturally or artificially. A considerable amount of effort and funds is being expended on silvicultural methods to regenerate these species of oak naturally. These methods have not been developed to the point that they can be easily applied. In stands impacted by disease and insects and in stands that have been harvested or will be harvested in the near future, no known methods exist to regenerate oak consistently. Oak seedlings can be planted and generally have adequate survival probabilities. Problems with initiating height growth occur in plantings. In addition, oak seedlings that are being produced in state and private nurseries are extremely variable in quality, and many times seed source and genetic quality are not known. There are currently no standards for acceptable oak seedlings.

Butternut

Butternut is being eliminated from our ecosystems by *Sirococcus clavigigenti-juglan-dacearum,* an exotic fungus that causes a lethal canker. Harvest of all butternut is restricted on federal lands.

There is one ongoing butternut project in the SAA area. The University of Tennessee (UT), FS - Region 8 (Genetic Resources Program and Forest Health), National Park Service–Great Smoky Mountains National Park (GSMNP), and the Tennessee Division of Forestry have been cooperating on butternut conservation. In 1994, butternut genetic conservation/disease screening plantations were established at the Beech Creek Experiment Station, Bent Creek Experimental Forest; North Central Forest Experiment Station experimental farm at Carbondale, Illinois; Francis Marion Seed Orchard; and UT. Another butternut genetic conservation test (nursery phase) performed at the East Tennessee State Nursery (Tennessee Division of Forestry) will be outplanted in the winter of 1995 to 1996. A butternut breeding orchard has been established at UT also. The test and grafted clones contain susceptible butternut, putative resistant butternut, and heartnut (Japanese walnut cultivar), which has resistance to butternut canker.

Future plans are to continue to survey for resistant and immune butternut and butternut x heartnut hybrids (buartnut or butterjap). Nuts will be collected and materials placed in genetic conservation/disease screening tests. A research group in the FS in the Lake States is actively working on a screening program for disease resistance. Any materials collected and propagated in orchards would be made available for their testing and breeding.

Hemlock

The hemlock woolly adelgid is an exotic pest that is destroying eastern hemlock over a considerable portion of its range. Hemlock is being killed by the adelgid in the George Washington National Forest, Jefferson National Forest, Shenandoah National Park, and along the Blue Ridge Parkway in northern Virginia. Its range increases along the Blue Ridge Mountain chain each year. It is anticipated that the Carolina hemlock will be similarly impacted. Due to the restricted habitat of the Carolina hemlock, it is highly probable that it will soon achieve endangered status.

Eastern hemlock shows no observable levels of resistance and there are no known biological controls for this pest. The adelgid appears to have the potential to eliminate eastern hemlock from major portions of its range. The Carolina hemlock, a close relative of eastern hemlock, could be in danger of extinction if the pest moves into western North Carolina and east Tennessee.

A passive conservation approach would be to collect samples of the native hemlock, either seed or cuttings for grafting, and establish the material in genetic conservation areas that can be protected from the insect with IPM practices. The area would need to be established where chemical pesticides could be used for protection from the insect.

A more active conservation approach would be to establish a selection and breeding program. Some work has been done and some information has been gathered by the National Arboretum. The insect, imported from China has a variable effect on native species of hemlock in China and Japan. In the U.S., western hemlock (*Tsuga heterophylla*) may also be resistant to this pest. It is possible that resistant hybrid hemlocks can be produced. Hemlock also produces steady cone crops at reasonably young ages which facilitates testing.

Dogwood

Flowering dogwood is a small tree occurring in the understory of eastern North American forests. The species is an important

valuable. It is also widely planted into landscapes. In the late 1980s, a fungal disease, dogwood anthracnose, began killing individual trees in the northern United States.

The FS annually looks for potentially resistant trees throughout the National Forest System to contribute to the Resistance Screening Center at Asheville, North Carolina.

Despite 5 years of work and testing of 300 seed lots (each seed lot represents one parent tree), very few potentially resistant seed lots have been identified (Young 1995). Nevertheless, some closely related species, both North American and Asian, and some individuals of the native flowering dogwood have been found to be resistant to the fungus. If breeding and screening procedures prove successful, it may be feasible for a full-scale program to be developed to restore dogwood to the landscape. In a program of this type, existing personnel, equipment, and available land at the Genetic Resource Management facilities could be utilized as a breeding/genetic conservation area for the production of resistant dogwood seed.

American Chestnut and Allegheny Chinkapin

The chestnut blight fungus devastated American chestnut and Allegheny chinkapin populations in the 1920s and 1930s. Although the above-ground portion of the trees were killed, the chestnut blight fungus does not affect the root systems. American chestnut now exists as a relatively short-lived sprout and Allegheny chinkapin forms small bushes. Observations of chestnut sprouts and chinkapin bushes over time indicate that there is a continuing population decrease.

No methods of controlling the blight are known. The abundance of species is rapidly declining. Both chestnut and chinkapin depend on disturbance to replenish the root reserves and/or stimulate abundant fruiting. If active gene conservation of these species is not undertaken soon, both will probably become extinct. The most feasible means would be to collect specimens of both species and propagate them for genetic conservation until a solution to the blight arises. A crossing program with Asian species may be developed, biotechnology might provide relief in the form of a resistant tree or an altered disease organism, or virulent form.

Some active breeding work in various locations and research into altering the disease to one less virulent that could displace the present strain is being performed. Genetically engineered resistance has been accomplished for an extremely limited number of crop species. No adequate tissue culture system, however, is now available for these species and any alleles that may provide resistance are unknown. From a technical perspective, funding molecular research at this time appears to be a poor choice over traditional research. Regardless of research approaches, it is imperative that some genetic material be preserved now for the future opportunities to work with these species.

Table Mountain Pine

Table Mountain pine has relatively no commercial value as a timber species due to its poor form. It is relatively rare to find stands of this fire-dependent, serotinous-coned species today in the Southern Appalachians. The species is currently being lost to bark beetles, stand decadence, and the marked absence of stand replacement fires. Without intervention and/or direct management of the species, much of the remaining genetic diversity in the species could be lost over the next decade.

American Beech

American beech is currently threatened by beech bark disease, an exotic pest problem. Because beech occurs over a very large geographic area, the disease isn't a problem throughout its range. Where affected, beech stands have been greatly impacted. In the Great Smoky Mountains National Park, a few stands have been affected.

Although there is a greater likelihood for some natural genetic resistance because the species is widespread throughout the southern United States; so far, natural resistance has not been documented, and the threat posed by the disease could be as great as was the threat from chestnut blight. (Chestnut also had a large geographic range.) The species is as ecologically important as a hard mast producer; as a den tree; as a component of beech, birch, and maple communities; and as a keystone species in high elevation beech gaps and beech boulderfield plant communities. Resistance screen-

undoubtedly be needed for this species in the future.

American Elm

The loss of American elm from the exotic Dutch elm disease is well documented for urban and historical settings. Little is documented regarding the role of elm as an ecosystem component. Some efforts have been made to cross Siberian and other Asian elms with American elm to produce a disease resistant variety. Further research could be done to determine the feasibility of reintroduction of American elm into its historical range where it is now absent.

Improved Monitoring Systems

Forest monitoring systems should be able to provide information to landowners and managers on the ecological status of forests; what changes are occurring; what the causal agents of the change are; if changes indicate a trend; what the expected outcome is if trends continue; and what effect management decisions might have on existing conditions. To enable land owners and managers to manage forest ecosystems in a sustainable manner, both spatially and temporally, intensive and extensive monitoring systems are needed. In addition, sustainable management of forests needs to consider the socioeconomic benefits of healthy forests and the legal, institutional, and economic infrastructure that will be necessary.

A forest monitoring system should provide annual reports on the condition of forests. Forest ecosystems are dynamic, and forces acting upon those dynamics can change quickly.

The Forest Health Monitoring (FHM) program is a multi-agency program led by the FS. This program has four main components: Detection Monitoring, Evaluation Monitoring, Intensive Site Ecosystem Monitoring, and Research on Monitoring Techniques. The focus of FHM is to evaluate the condition, changes, and trends in indicators of U.S. forest ecosystem health; monitor indicators of pollutant exposure and habitat condition; seek associations between human-induced stresses and the ecological condition of the forests; and provide annual reports and periodic interpretive assessments on the ecological status and trends to

Summary of Key Findings

The assessment of terrestrial ecosystems focused on forest health and on terrestrial plant and animal resources. Assessment topics included broad landscape habitat and landcover patterns, federally listed threatened and endangered species, rare species and communities, popular game species, possible national forest old-growth forest, oak decline, exotic pests and diseases, biological diversity, fragmentation, black bear, genetic conservation programs, and neotropical migrant birds.

The information provides a framework for land managers to develop natural resource management objectives that can contribute to sustaining wildlife and plant habitats in the Southern Appalachian Assessment (SAA) area. The information and opportunities identified in the SAA expand the perspective of landowners beyond their own administrative boundaries. For example, most national forests are preparing to begin the first regular periodic revision of their forestland management plans. Decisions on the amounts of various habitats on national forests, and management direction to sustain those habitat levels, will be made during the revision process. The SAA information should help to directly feed that process, and SAA resource elements and parameters should be considered in making forest plan decisions. Private land is vital to the future of some wildlife and botanical resources.

The terrestrial report was designed to answer eight questions, four pertaining to wildlife and botanical resources and four pertaining to forest health. This chapter lists the questions and provides a summary of findings that helps to answer each question. More detailed discussions of these findings can be found in the previous chapters.

Identification of Wildlife and Plant Species and Important Habitats in the SAA

Question 1:

Based on available information and referenced material, what plant or animal species occur within the SAA area, and what are their habitat associations?

More than 20,000 species of plants and animals may occur in the SAA area. A complete list was not prepared; instead, the focus was on species of biological and social importance. Important broad classes of vegetation and landcover, as well as rare communities, were included in the assessment to provide a comprehensive look at habitats.

A “short list” of 472 plant and animal species was identified for focus in the SAA. This list includes 225 plants, 155 invertebrates, 47 birds, 23 amphibians and reptiles, and 22 mammals. The total includes 51 federally listed T&E species, 366 species whose viability is of concern (VC species), 38 species of high interest to natural resource managers and the public, 10 game species, and 7 other species with demanding habitat requirements.

Sixteen land cover types were analyzed: nine forest cover types, plus agricultural pasture, agricultural cropland, grass/forb early successional, developed, barren, wetland, and water. For each of the forested land cover types, four successional classes were recognized.

Thirty-one rare community types occur in the SAA area.

Habitat associations were determined for 442 of the 472 species on the short list and documented in a species habitat matrix.

Information from this work resulted in the grouping of species into 19 species groups based on habitat associations and the development of broad-scale spatial habitat suitability models for selected species groups. The assessment focused on these 19 species groups.

The Status, Trends, and Spatial Distribution of Terrestrial Habitats and Wildlife and Plant Populations

Question 2:

What are the status, trends, and spatial distributions of populations and habitats in the SAA area for:

Federal T&E species?

VC species (regionally sensitive)?

Unique or underrepresented communities (including areas with potential to become old growth)?

Wildlife species that are hunted, viewed, or photographed?

Species for which there is high management/public interest?

Species having special or demanding habitat needs?

Species considered to be true ecological indicators?

Status and trends of SAA terrestrial ecosystems

Distributions of the 26 million acres of forest in the Southern Appalachians are:

Broad Forest Type	Percent
Deciduous	67.3
Mixed	15.4
Evergreen	17.3

Forest type group	Million Acres	Percent of SAA total
Oak	17.6	47.1
Southern yellow pine	3.8	10.1
Mixed pine-hardwood	3.2	8.6
Mixed mesophytic hardwood	3.1	8.4
W. pine-hemlock-hardwood	0.8	2.2
W. pine-hemlock	0.7	1.8
Northern hardwood	0.6	1.6
Bottomland hardwood	0.4	1.2
Montane spruce-fir	0.09	0.2

Land distribution by ownership:

Forested Land Ownership	Million Acres	Percent of SAA Forest
Private	20.2	77
National forest	4.5	17
National park	0.82	3
State	0.531	2
Other federal/Indian	—	1

Total forest acres have decreased by 2 percent since the mid-1970s, and based on past land use trends, this decrease in forest acres is expected to continue at the same pace through the year 2010. This loss is occurring primarily in private forest for development and conversion to other agricultural land uses.

Land distributions by successional class:

Successional Class	Percent of Forest Area
Early	8
Sapling/pole	22
Middle	52
Late	18

Class Percent Change Since Mid-1970s:

Successional Class	Total SAA	NF Land	Nonindustrial Private Land
Early	+26	-4	+28
Sapling/pole	-27	+12	-27
Middle	+3	-6	+5
Late	+42	+34	+50

National forests contain approximately 1.1 million acres that could become old-growth forest. Decisions on which of these acres will be targeted for management as old-growth communities will be made during the forest planning process.

Acreages occupied by nonforest cover types are:

Cover Type	Million Acres
Pasture land	6.5
Early successional	1.5
Cropland	1.3
Developed	1.2
Water, barren, & wetlands	0.7

Since the early 1980s, large urban areas have grown by 35 percent, and small urban areas by 53 percent. Cultivated croplands have diminished by 25 percent, while noncultivated croplands (orchards, etc.) have increased by 9 percent. Grass pasture has diminished by 3 percent, while legume pasture has increased by 38 percent.

Status of rare communities

Thirty-one rare community types were identified in the SAA area. These types are important for sustaining current populations of federally listed species and VC species. Almost 75 percent of the terrestrial rare plant and animal species and their associated habitats are found in one or more of the 31 rare communities, which occur on less than 1 percent of the SAA land area.

A total of five rare forest communities was identified. About 90,100 acres of montane spruce-fir forest exist in the SAA area. About 62,600 acres (69 percent) are in national parks, and additional acreage is in national forests. More than 80 percent of known beech gap forests is on public land. These communities, therefore, can be adequately managed by public agencies. However, approximately 60 percent of the occurrences of mountain longleaf pine woodlands, Table Mountain/pitch pine woodlands, and Carolina hemlock forests is on private lands.

Ten rare, nonforest communities (calcareous cliffs, calcareous woodlands and glades, caves, granitic flatrocks, mafic and calcareous fens, mafic cliffs, mafic woodlands and glades, mountain lakes, sinkholes and karstlands, and wet prairie) occupy less than 1 percent of the total SAA area. About 95 percent of the occurrences for these communities is on private lands. Public land contains 75 percent of the occurrences of 12 rare communities (beaver ponds and wetland complex, boulderfields, granitic domes, grassy balds, heath balds, high-elevation rocky summits, mountain ponds, river gravel and cobble bars, sandstone cliffs, spray cliffs, swamp forestbog complex, and talus slopes). Four rare, nonforest communities (seasonally dry sinkhole ponds, serpentine woodlands and glades, shale barrens, and sphagnum and shrub bogs) are equally divided between public and private ownerships.

Summary of occurrence data for federally listed and VC species

The determination of the status of rare species was an important part of the assessment. The list of 51 federally listed species and 366 VC species was compiled from information from the U.S. Fish and Wildlife Service, the state natural heritage programs, and peer review of the initial species lists. Habitat relationships were determined for the species in this category, with the exception of 30 species. These species-habitat associations received peer review, but much information about them is intuitive.

About 75 percent of these species is associated with small microhabitats. These species, therefore, are not suited for broadscale analysis of habitat suitability. For these species, the analysis of current status focused primarily on their spatial occurrences, based on records from state natural heritage programs.

Species occurrences in the SAA area are:

Type	Number
T&E animal	251
T&E plant	537
VC animal	908
VC plant	2,335

Eleven of the 19 species groups contain T&E species, and 17 of the 19 include VC species.

The distribution of occurrences for T&E and VC species by ownership class is:

Ownership Class	T&E Species	VC Species
Private	493	1,802
National forest	154	952
National park	90	315
State	47	113
Other federal	4	53
Total	788	3,243

Private land contains the largest number of occurrences of federally listed species associated with five communities:

- Caves (101 of 129 occurrences)
- Mountain bogs (54 of 88 occurrences)
- Fen or pond wetlands (6 of 8 occurrences)
- High pH or mafic habitats (60 of 79 occurrences)
- Mixed mesic habitats (55 of 90 occurrences)

National forests contain the largest number of federally listed species associated with two communities:

- Rock outcrop and cliff habitats
- Southern yellow pine (active red-cockaded woodpecker colonies)

Nonindustrial private land contains the largest number of occurrences for VC species in five communities:

- Caves (318 of 360 occurrences)
- Mountain bogs (213 of 310 occurrences)
- Fen or pond wetlands (40 of 46 occurrences)
- High pH or mafic habitats (222 of 371 occurrences)
- Rock outcrop and cliff habitats (275 of 513 occurrences)

National forests contain the largest number of occurrences for VC species associated with spray cliffs (45 of 88 occurrences).

Landscape habitat suitability analysis

To identify broadscale habitat patterns in the assessment area, spatial analysis of habitat suitability was conducted for 10 of the 19 species groups. These species groups were selected because their habitat associations lend themselves to broad, landscape-level analysis using remote sensing data. Suitability analysis was not attempted for species groups with either highly specific habitat requirements (e.g. spray cliff species, high pH, or mafic species) or very general requirements (e.g. habitat generalist species). Six habitat suitability products were developed:

- Area-sensitive, mid- to late-successional deciduous forest species
- General high-elevation forest species
- Seep, spring, and streamside species
- High-elevation bald/early successional species/early successional grass-shrub species
- Closed canopy deciduous forest species
- High elevation spruce-fir/northern hardwood forest species

Habitat suitability also was modeled for black bears.

These landscape-level models represent only gross habitat suitability based on general habitat requirements. Results of the suitability models provide a regional picture of habitat potential.

- Spruce-fir/Northern Hardwood Habitats (estimated 184,000 acres)

Potential habitat for 23 associated species (4 T&E and 18 VC). About 47 percent of this habitat is in national parks, and 32 percent in national forest. Of 41 occurrences of T&E species associated with this habitat, 15 are on national parks, 13 are on nonindustrial private land, and 11 are on national forests. Of 102 occurrences of associated VC species, 73 are on national parks or national forests. Outlook: uncertain, due to air pollution and exotic pests. A downward trend is expected over the next 15 years.

- High Elevation Balds (estimated 27,000 acres)

Habitat for 18 associated species (4 T&E and 13 VC). About 73 percent of this habitat is on private ownership; 25 percent is in national forests. About one half of these sites is larger than 20 acres. Of 58 occurrences of T&E species associated with this habitat, 37 are on private land, and 14 on national forest land. Of 297 occurrences of VC species, 119 are on national forests, 37 are on national parks, and 129 are on nonindustrial private land. Outlook: stable in extent, but possibly declining in quality due to air pollution.

- General High Elevation Forest Habitats (estimated 350,000 acres)

About 150,000 acres (42 percent) are in tracts larger than 5,000 acres, with the potential to support all seven of the associated T&E and VC species. Of these large tracts, 74 percent of the acreage is in national parks, and 17 percent in national forests. Outlook: uncertain, due to the effects of air pollution and exotic pests; downward trend expected over the next 15 years.

- Early Successional Habitats (estimated 1.5 million acres)

Ten T&E and VC species are associated with this habitat. Approximately half of the occurrences of this habitat is in tracts 20 acres or larger in size; 97 percent of the total acreage is private land, while 2 percent is national forest.

- Riparian Habitats (estimated 2.3 million acres, of which 1.5 million acres are in forest riparian habitat)

A total of 49 species are associated with this habitat, of which 10 are T&E. National forests contain 37 percent of the occurrences for 12 of these species, national parks contain 16 percent of the occurrences for 8 species, and nonindustrial private lands contain 42 percent of the occurrences for 16 species.

- Mid- to Late-Successional Deciduous Forest Habitats (estimated 17 million acres)

There are 66 species associated with these habitats, not including species identified in other species groups. Approximately 71 percent of these habitats occur on private land, while 23 percent are in national forests. Five T&E species are associated with these habitats; 61 percent of the occurrences of these species are on nonindustrial private lands, while 23 percent are on national forest.

A total of 58 VC species in four species groups are associated with these habitats. These include 44 occurrences of three species in mid- to late-successional deciduous forest species group (66 percent of which are in national forests), one occurrence of a single species in the bottomland species group (on state land), 452 occurrences of 37 species in the mixed mesic forest species group, and 235 occurrences of 12 species in the mixed xeric forest species group.

- Habitats for Area-Sensitive Species Associated with Mid- to Late-Successional Deciduous Forests (estimated 15.8 million acres)

Slightly more than half of this habitat, 8.2 million acres, is in tracts larger than 5,000 acres in size; these larger tracts are thought to have the potential to support all 16 of the bird species in this species group. Approximately 51 percent of these larger tracts is on private land, while 39 percent is on national forests. Approximately 66 percent of this habitat type is considered to be forest interior habitat; the relative proportion of interior by ownership is 97 percent on national parks, 90 percent on national forests, 58 percent on private land, and 49 percent on other federal. Outlook: overall habitat acres in large tracts, and associated forest interior habitat, will continue to decrease due to loss of forestland to other uses. This decrease will occur primarily on private lands.

- Black Bear Habitat (estimated 21 million acres)

Fifty-one percent of this acreage has a total road density less than 1.6 miles per square mile. Approximately 75 percent of the total habitat acreage is on nonindustrial private land, while 19 percent is in national forests. Suitable bear habitat is found on 91 percent of national forestland, 84 percent of state land, 78 percent of national park land, and 51 percent of private land. Outlook: bear habitat will remain stable on public land, but will decrease on private land due to continued loss of forested habitats and increased development.

Status of game species

Estimates of current (1995) and historical (1970) population densities for 10 major game species were provided by state wildlife agencies included in the assessment area. Density estimates were derived from harvest and survey data where available, as well as from professional judgement by appropriate state agency biologists.

White-tailed deer and eastern wild turkey densities were generally low to medium for most of the SAA area, with higher densities in a few counties. Populations of both species have increased greatly across the entire SAA area since 1970. Densities for these species are highest on nonindustrial private land, national forests, and state land. The outlook is for the population increase to level off and become stable.

Black bear population densities have generally increased since 1970. Bears are present at low to medium densities in parts of the SAA area, particularly on national forest and national park land. The species is absent in many areas.

Ruffed grouse population densities are generally at medium to low in areas where the species occurs (generally in areas with moderate or higher elevations). National forests and national parks contain the highest densities. Populations have declined since 1970, possibly due to a decreased proportion of acres in the sapling/pole successional class which grouse favor. National forests will continue to provide the major source of grouse habitat and hunting opportunity. However, both grouse populations and the quality of their habitat are expected to decline over the next 15 years.

The population density of bobwhite quail has decreased markedly in the SAA since 1970. Densities are generally lower on national forests and national parks than on other ownerships. The higher densities for quail are associated with greater proportions of the landscape in agriculture and grass/shrub habitats. Quail populations will continue to decrease due to shifts in agricultural practices and continuing isolation of suitable habitat.

Future Needs and Management Opportunities

Question 3:

What habitat types, habitat parameters, and management activities are important in providing the distribution and types of habitats to maintain viable populations and/or desired habitat capability for the "short list" of wildlife and plants?

And

Question 4:

Based on our current knowledge of ecological unit land capabilities for the Southern Appalachians, what are the general habitat mixes/conditions needed to:

Recover T&E species?

Conserve populations of VC species?

Maintain the existing species and community diversity that will not result in the loss of viability for any plant or animal species (in the context of the entire SAA region)?

Provide sustainable populations of species at desired levels on national forests?

Rare Communities

The rare communities are the key to conserving rare plant and animal species in the SAA area. About 84 percent (43 of 51) of the terrestrial T&E species is associated with rare species community groups and streamside habitats. These habitats occur in less than 1 percent

included in the body of the report. The rare communities are:

- Cave communities
- Mountain bog communities
- Fen or pond wetlands
- High-elevation balds
- High pH or mafic habitats
- Rock outcrop and cliff habitats
- Montane spruce-fir forest
- Seeps, springs, and streamside habitats
- Mountain longleaf forests

Broad-scale Habitat Types

In addition to conservation of rare communities, management strategies should continue to provide:

- Mid- and late-successional deciduous forests (including mixed pine-hardwood forests), particularly in tracts larger than 5,000 acres
- Early successional habitats, with appropriate sizes and distribution
- Black bear habitat
- Oak hard mast capability

The Changes in SAA Forest Vegetation from Natural Processes and Human-Caused Disturbances

Question 5:

What changes or trends in forest vegetation or soil productivity are occurring in response to human-caused disturbances or natural processes?

Currently, 70 percent of the land in the Southern Appalachians is forested. Over three-fourths of that forest is privately owned. About 17 percent of the forest is in national forests and 3 percent is in national parks. Oaks in combination with other species dominate the stands on almost half of the forestland. Mixtures of pine and hardwoods dominate on 12 percent, and southern yellow pines dominate on 4 percent of the forestland.

Forest acreage has decreased by about 2 percent since the mid-1970s. A slow rate of decrease in forest acreage is expected to continue through the year 2010. Losses of forestland for more intensive human uses such as road and home construction are partially offset by natural reversion of pasture and cropland to forest. Clearing of forest for development or agriculture occurs primarily on private land.

Oak is becoming increasingly susceptible to a decline brought on by the combined effects of maturity, drought stress, gypsy moth defoliation, and root disease. Fir, hemlock, beech, and dogwood are being lost to exotic insects and diseases, Table Mountain pine is failing to regenerate after bark beetle attacks because of the absence of fire, and spruce-fir stands appear to be in decline. Timber harvests and prescribed burning on some public land have resulted in the regeneration of shade-intolerant pines and hardwoods. Lack of active management in other stands has led to the development of dense understories, and to the senescence of overstory trees of some species.

Past land uses and atmospheric deposition have reduced soil productivity in some places. Abusive logging practices and cycles of forest clearing, crop cultivation, abandonment, and reforestation caused soil erosion and reduced soil productivity in the 19th and early 20th centuries. Effects of atmospheric deposition are complex and difficult to measure with precision. Nitrate deposition has a fertilizing effect, but it also can acidify soils with low buffering capacities, and excessive amounts can adversely affect plant health. Reductions in soil productivity attributable to atmospheric deposition have not been fully demonstrated in the Southern Appalachians.

The biggest vegetative trend in the study area is toward a reduction in stocking of oaks and increases in stocking of maples, yellow-poplar, blackgum, and eastern white pine. The composition of future stands will be strongly influenced by timber harvesting practices and the presence or absence of prescribed fire. Current rates of ecosystem disturbance appear to be low when compared to rates estimated for regimes that existed prior to settlement of the area by Europeans and for regimes in the late 19th and early 20th centuries.

Question 6:

What are the potential effects of the presence or absence of fire on forest health?

Fire is perhaps the most common form of major natural disturbance in most of the ecosystems of the Southern Appalachians. Fire is particularly important in systems dominated by southern yellow pine, and its ecological effects in those systems are well understood. Effects on xeric deciduous forests also are important but are less well understood. Fire may be a major factor in the development of oak forests on upland sites.

A role for fire in the development of oak regeneration has been demonstrated in the Coastal Plain, the Piedmont and Cumberland Plateaus, and the Interior Highlands, and one would expect a similar role for the Southern Appalachians. Thinning, grazing, or light burning appears to increase the amount of oak regeneration beneath maturing stands of mixed hardwoods. Periodic fire probably also checks plant succession in oak forests, because later successional species, such as red maple, have low resistance to fire damage. Thus, fire may be useful in slowing or stopping the current ecological trend from oak domination to domination by more shade-tolerant species.

In the absence of fire, two rare forest communities in the Southern Appalachians–mountain longleaf pine woodlands and Table Mountain pine-pitch pine woodlands–are being replaced by hardwoods and loblolly pine. The endangered red-cockaded woodpecker is associated with mountain longleaf pine woodlands in northeastern Alabama and northwestern Georgia. Table Mountain pine has cones that open only when exposed to high temperatures from fires. Fire exclusion will lead to the continued decline of this community.

Other forest types and plant communities in which fire is important for regeneration and maintenance are: red spruce-Fraser fir, yellow birch boulder fields, high-elevation red oak, montane oakhickory, white pine, chestnut oak, dry to mesic oak-hickory, xeric shortleaf pine, xeric Virginia pine, heath balds, grassy balds, ultramafic barrens, and bogs.

Thus, prescribed forest burning appears to promise many potential benefits for ecosystems in the Southern Appalachians. Additional information is needed on its precise effects in the mountains, and on the risks associated with its use. Prescribed burning is considerably less common in the Southern Appalachians than on the Piedmont Plateau and the Coastal Plain of the South.

The Effects to SAA Forest Ecosystems from Native and Exotic Pests

Question 7:

How is the health of the forest ecosystem being affected by native and exotic pests?

Many important tree species in the Southern Appalachians are being severely affected by attacks from native and exotic pests. Effects of air pollution are less certain than those of pests, but they are potentially quite serious.

Flowering dogwoods are imperiled by dogwood anthracnose. In tests of 300 seedlots, little resistance to the disease was identified. Dogwood anthracnose has been found in every county in the Southern Appalachians, and all the flower dogwoods in some stands have already been killed. Likelihood of infection increases with elevation and amount of overhead shade. The prognosis for the species is not good.

Similarly, the futures of Carolina hemlock and eastern hemlock are clouded by the hemlock woolly adelgid. Individual trees can be protected with insecticides, but survival prospects for unprotected trees are not good. Loss of hemlocks could have severe ecological effects in riparian zones, where they are now common.

Since its presence was first reported in the Southern Appalachians in 1957, the balsam woolly adelgid has killed large numbers of Fraser firs. The adelgid is now found throughout the range of Fraser fir, and is resistant to climate-caused mortality as well as native and introduced predators. Thus, the long-term prognosis for Fraser fir is uncertain. A spruce-fir decline has also been reported in the Southern Appalachians, but it has not been well documented.

Butternut is under attack by the butternut canker. Trees infected with the canker eventually are killed, and very limited resistance has been found. Butternut trees on national forests are being protected from logging, but many private landowners have cut their merchantable butternuts to get some income before the disease strikes.

The loss of the American chestnut to chestnut blight is a well-known story. The ecological effects of the loss of this species were large and may still be occurring. The disease also reduced Allegheny chinquapin to a brush species.

American elms in the forest are killed by Dutch elm disease, but the effects are less serious than in urban shadetrees. The importance of American elm in forest ecosystems is not known.

Table Mountain pine is disappearing from the Southern Appalachians. Death is often caused by bark beetles, but the species is not reproducing because fire is being excluded.

Southern pine beetle outbreaks occur periodically in the Southern Appalachians. The outbreaks kill Table Mountain and other southern yellow pines.

Oaks make up the most common species group in the study area. A combination of factors has made them more important than in the past. Oak decline and gypsy moths are likely to decrease the importance of oaks.

Oak decline is caused by many factors, including diseases, advancing tree age, and insect damage. Oak decline has been reported by forest workers for more than a century, but the damage appears to be accelerating. The vulnerability of a stand to oak decline appears to increase with tree size, tree age, and oak basal area in the stand. Incidence of oak decline is only about half as frequent on private as on public land. Among national forests, those in North Carolina and Virginia have highest incidence.

Introduced to North America around 1869, the European gypsy moth has moved southward through the Appalachians. It is now common in northern Virginia. Control efforts have produced mixed results. Oak leaves are a favored food, and defoliation of oaks by this flightless insect makes the trees more susceptible to oak decline.

The Asiatic gypsy moth poses an even greater threat because adult females can fly and because this species attacks a much wider range of plant hosts. In 1995, Asiatic gypsy moths were found in two counties in North Carolina. Both these infestations were massively treated at great cost. Eradication of this species while populations are small and their range is limited is paramount to control.

Introductions of exotic plant species have caused significant disruption of some parts of the Southern Appalachian ecosystems. Extensive programs may be needed to manage, control, or eradicate these species. Symptoms of ozone damage are common on the foliage of trees in the Southern Appalachians. At a minimum, ozone exposure stresses forest communities. In combination with other stress factors such as drought and insect attacks, its effects may be magnified. There is some evidence that ozone damage has caused some growth loss to trees in northern Virginia and northern Alabama and Georgia. Some plant species appear to be more sensitive to ozone exposure at high than at low elevations. There is little evidence, however, that ozone has a strong effect on spruce or fir at high elevations in the Southern Appalachians.

Sulfate and nitrate deposition appear to be greatest in the northern tip of the study area, and at the highest elevations. Heavy deposition of these materials has the potential to acidify soils at high elevations, reducing their productivity and altering stream chemistry.

The Effects of Current and Past Management Practices on the Health and Integrity of Forest Vegetation

Question 8:

How are current and past management practices affecting the health and integrity of forest vegetation in the Southern Appalachians?

Management of the area's national forests in the first half of the century concentrated on reforestation of cutover land, watershed improvement, erosion control, and fire protection. Vigorous regrowth, restoration of watersheds, and expansion of wildlife populations were obvious and satisfying results. As timber inventories increased, selective logging occurred across the region (Yarnell 1995).

Selective logging failed to regenerate the desired tree species, so the Forest Service began to rely upon even-aged management, primarily with clearcutting, in the 1960s. This practice created a mosaic of relatively small even-aged stands across the landscape. Other management practices included favoring yellow pine over hardwoods in some places through site preparation and planting, and a limited amount of prescribed burning. The general policy of extinguishing wildfires was continued.

In response to public objections, the Forest Service has severely curtailed its use of clearcutting, and it adopted a general policy of ecosystem management in 1992. Today, prescribed burning is used to retain rare communities, enhance wildlife habitat, and reduce fuel loadings that could lead to catastrophic wildfires. Nevertheless, prescribed burning is not common in hardwood stands in the Southern Appalachians. Current management approaches have not been in place long enough to evaluate the results objectively.

The Chapter 3 of the SAA Social, Cultural, and Economic Technical Report (1996) has three key findings related to management practices:

1. On average, national forestland is at higher elevations and is less productive than private land in the region. National forest stands are logged less frequently, so they have higher average timber inventory per acre, less removals, less growth, and slightly higher mortality than private land in the area.
2. While they contain only 17 percent of the timberland in the Southern Appalachians, national forests hold much larger proportions of the highest quality sawtimber.
3. Timber harvesting from the national forests expanded in the 1970s through the mid-1980s. It peaked in 1985 and has declined rapidly since then. Current levels are comparable to those in the 1970s.

From the standpoint of timber production, the biggest forest health problems in the Southern Appalachians are gypsy moths in northern Virginia, oak decline from southern Virginia to northern Georgia, and southern pine beetles in the southern quarter of the region. These agents increase tree mortality and reduce growth.

Treatments could be imposed to improve the vigor of individual trees and mitigate the effects of oak decline. Evolving markets for low-quality trees and strong markets for high-quality oak timber could provide profitable opportunities to improve forest health.

Gypsy moth impacts could be reduced through: (1) risk rating to identify vulnerable stands and thinnings and salvage cuttings, (2) quarantine to prevent introduction into uninfested areas, (3) careful monitoring of the spread of the insect. Biological controls of gypsy moths include mass trapping of males, mating disruption through pheromone releases, release of sterile insects, and the use of viruses. Chemical controls include diflubensuron and acephate.

Impacts of southern pine beetles can be reduced by rating risks in individual stands and treating the stands where risks are high. Existing infestations can be stopped by cutting and leaving infested trees, cutting and removing them, or cutting and burning them. Biological control methods include enhancement of habitat for parasites and predators of the beetles. Dursban and lindane are insecticides used against southern pine beetles.

Genetic conservation seems desirable for tree species that might be destroyed by exotic pests. Species at risk include American chestnut, chinquapin, butternut, Fraser fir, flowering dogwood, and eastern and Carolina hemlock. Backcrossing to create resistant hybrids may be feasible for American chestnut, butternut, and hemlock.

Research and Information Needs

The following are items identified as research needs by the Terrestrial Team to help to validate assumptions made during the SAA, to provide answers to deal with current forest health threats to forest ecosystems, and to provide information for broad-scale monitoring of landcover changes, rare communities, and selected plants and animals. The research and information needs include:

- Improve the accuracy of satellite remote sensing technology for use with expanded landcover classes. Accomplish this by completing field checks for accuracy assessment and incorporate needed changes to improve the accuracy of the existing LANDSAT remote sensing data. Also incorporate other existing land cover data, such as exists for TVA lands.
- Develop definitions and operational instructions for identifying old-growth forest types in the SAA.
- Increase baseline data for occurrences of rare communities in the SAA.
- Develop management guidelines for the 31 rare communities in the SAA.
- Develop conservation strategies for the federally listed species and viability concern species based on their association with rare communities and broad habitat types.
- Establish corporate database and procedures for monitoring the trends of selected terrestrial resource elements of both biological and social significance that were identified during the SAA.
- Establish corporate database for occurrences of federally listed species and viability concern species that is secure, yet can be made readily accessible for future management/ planning efforts.
- Validate habitat relationships for federally listed species and globally imperiled (G1) species.
- Conduct searches for spruce-fir, moss spider habitat using “smart” technology (use the GIS databases assembled for the SAA, and develop a habitat model to search for suitable habitat).
- Relate broad landscape patterns (i.e., at the section level) and local land uses to forest landbird relative abundance and productivity.
- Develop information for early successional habitat and associated species related to patch size, patch isolation, and relationship to adjacent habitats for upland game species and forest early successional landbirds.
- Develop techniques for translocating selected priority rare plant and animal species.
- Begin looking at genetic conservation programs for selected priority rare plant and animal species.
- Continue refining the current knowledge for habitat requirements related to black bear, with emphasis on remote habitat needs and road density/road use relationships.
- Continue periodical monitoring of spruce-fir populations across the region.
- Study frequency and variability of Fraser fir seed crops.

- Survey the Smoky Mountains and elsewhere in the region for individual Fraser firs which show signs of adelgid resistance.
- Conduct basic taxonomic and autecological research on spruce-fir bryophytes, especially obligate epiphytes of fir, and determine how they are affected by the loss of fir.
- Initiate genetic engineering to transfer adelgid resistance from other hemlock species into eastern and Carolina hemlocks.
- Identify surviving uninfected butternut trees on federal lands.
- Continue research on resistance in butternut and development of resistant planting stock.
- Monitor wild populations of American elm to track species health.
- Standardize native seed mixtures for use by SAA forests based on local testing.
- Conduct an assessment of the extent and ecological effects of exotic plant infestations on national forest lands in the SAA area, including cost/benefit analysis of eradication/control projects on a species-by-species basis.
- Continue research on genetic engineering both to transfer blight resistance genes from Chinese chestnut into American chestnut, and to develop successful hypovirulent strains of the blight fungus for innoculating native chestnut root sprouts.
- Initiate a breeding program in an area geographically isolated from the chestnut blight in order to assure survival of an array of chestnut genetic material.
- Identify areas with extensive chestnut rootstock populations, and employ silvicultural practices in those areas which will protect or enhance chestnut survival.
- Develop strategies for regenerating yellow pine, particularly Table Mountain pine, in areas affected by southern pine beetle (SPB) in order to avoid loss of these types. Prescribed burning in Table Mountain pine sites infested by SPB should be specifically addressed.
- Further develop models for predicting susceptibility of pines to SPB attack in the mountains, including shortleaf, pitch, Virginia, and Table Mountain pines.
- Investigate the role of fire in regeneration of oak species.
- Develop an understanding of oak reproduction in the absence of advance regeneration.
- Develop a better understanding of the overall history and role of fire in the Southern Appalachian forests, including effects on hardwood species other than oaks.
- Determine what role fire played in the protohistoric period (1600s to 1700s).
- Develop methods for using prescribed fire to enhance biological diversity, vegetative composition, and stand structure as related to maintenance of ecosystem components.
- Develop gene conservation strategies to protect declining tree species.
- Develop silvicultural practices to reduce losses to forest pests.

Appendix A

The Data Sources for the Assessment of Terrestrial Resources in the Southern Appalachians

Introduction

Identification of medium and fine scales data sources to address current status and past trends for broad land cover/vegetation types, communities, habitats, populations, and components of forest health was accomplished early in the process. Primary sources of data included: LANDSAT remotely sensed data; Forest Inventory and Analysis (FIA), Continuous Inventory of Stand Condition (CISC), and other inventories; Species Element of Occurrence (EOR) data; county density estimates for game species; 1:100,000 DLG ownership coverage; 1:100,000 water/stream reaches; 1:100,000 road coverage; 1:100,000 Digital Elevation Models; and the Southern Forest Health Atlas. Data analysis and interpretation processes relied heavily upon Geographical Information System (GIS) spatial and quantitative capabilities for data storage, retrieval, analysis, and display. Scientists and experts reviewed selected analyses and narratives throughout the assessment.

The Southern Forest Health Atlas

The Southern Forest Health Atlas is a GIS database of the 13 southern states designed to show point-in-time status of several forest conditions and help in evaluation of the effects of combinations of conditions on forest health. It was originally developed as the Southern Forest Atlas Project, funded by the National Survey Program in the mid-1980s (Marx 1988). The original purpose was to test correlations of atmospheric pollutant concentrations with poor forest health conditions, but it has since evolved into a more comprehensive database including major pest conditions, weather, soils, and forest resources in addition to atmospheric deposition. It is maintained at the USDA Forest Service, Southern Region, Forest Health Field Office, Asheville, NC, and is updated annually. Data layers and sources include:

1. Forest Types: county distribution for 22 Society of American Foresters (SAF) cover types with acreage and volume. Source: USDA Forest Service, Southern Research Station.
2. Soils: state soil associations (combinations of associated phases of soil series) with attributes (e.g. texture, pH, water-holding capacity, internal drainage). Source: State Soil Geographic (STATSGO) Database (1994).
3. Weather: monthly averages, monthly deviations from the mean and 0.5 degree grids for precipitation, maximum and minimum daily temperatures, relative humidity, and wind speed for the period 1951 to 1990. Source: National Climatic Data Center.
4. Ozone Concentration: point and girded coverages of 7-hour averages and the number of hourly occurrences above certain ppb levels since 1973. Source: EPA monitoring stations.
5. Pest Stressors: data may include one or more of the following: incidence, severity, risk rating, host range. Stressors included are annosum root disease, balsam woolly adelgid, beech bark disease, butternut canker, dogwood anthracnose, fusiform rust, gypsy moth, hemlock woolly adelgid, littleleaf disease, oak decline, and southern pine beetle. Source: Most data supplied by the USDA Forest Service, Southern Region Forest Health, from field survey. Other contributors include pest management specialists from state forestry agencies, USDA Forest Service Research (Forest Inventory and Analysis and Forest Insect and Disease Research Work Units) and the Animal and Plant Health Inspection Service.

Forest Inventory and Analysis

The FIA provides information to public and private sectors on the status, trends and uses of forests in the US. Information contained in FIA comes from a series of permanent forest sample plots. There are 7,160 plots in the Southern Appalachian Assessment (SAA) area. The approximate densities of these plots range from one plot per 3,500 acres to one plot per 5,000 acres. The FIA information is administered by three research project leaders with different sampling and estimation procedures. The FIA is designed to assess large sampling areas. The inventories are commonly designed to meet sampling errors at the state level at the 67 percent confidence limit, with a 3 percent error per 1 million acres of timberland being the maximum allowable sampling error for area. As the sampling areas are subdivided into smaller sizes, sampling errors increase and reliability of estimates decrease (Hansen and others 1992).

For the SAA, FIA information was used primarily to determine successional class percentages of the identified forest cover classes and 20-year trends for forest cover classes and successional classes (Chapter 3). This information was stratified according to total area, ecological sections and section groups and broad ownership categories. FIA information was also used in many of the forest health hazard rating prediction models (Chapters 6 and 7).

Continuous Inventory of Stand Condition

The National Forest System, Southern Region, maintains CISC, a database designed to continually reflect current forest description of every stand mapped. It also tracks planned management activities within a stand. Information in CISC is based on field examinations and aerial photographic interpretation. CISC has GIS capabilities with related tabular attribute data for each stand. These data were used to characterize forest cover successional classes on national forests and display and analyze initial inventory of possible old-growth on national forests (Chapter 3). CISC was also used in some forest health hazard rating prediction models (Chapter 6).

Satellite Imagery

LANDSAT satellite Thematic Mapper™ digital imagery was the primary data source used to produce a spatial land cover data theme for the SAA. The analysis of current land cover conditions (Chapter 3) and habitat suitability (Chapter 3) relied on the satellite imagery. Imagery acquired between May 1992 and August 1994 from 13 scenes was required to provide coverage of the SAA. Both leaf-on and leaf-off images were available to the contractor, Pacific Meridian Resources (PMR), to perform the classification. Recent leaf-on satellite imagery was the primary source of spectral data. Ancillary data, including digital elevation models and National Wetlands Inventory (NWI) maps, were also used in developing the classification. A16 class hierarchical land cover classification and associated decision rules were defined to support assessment activities. Classification, review, and editing produced a final raster cover classification that labeled each 30- by 30-meter (approximately 1/4 of an acre) image resolution element (pixel). A polygon land cover layer generalized to a 2-acre minimum mapping was derived from the raster classification. The final polygon classification was rasterized to provide the classification in an alternative format for analysis activities.

A multi-phase assessment of the accuracy of the land cover classification is being planned. More than two hundred primary sampling units distributed across the SAA at the nominal locations of Forest Health Monitoring sample points are the basis for the analysis. At each sample point a stereo triplicate of 1:12,000 color infrared aerial photography has been acquired and a circular sampling unit approximately 8,100 feet in diameter has been defined. The aerial photo interpretation phase of the assessment involves examination of a sample of between 1,500 and 2,000 land cover polygons located within, or intersected by, the sampling unit boundaries. A portion of the sample polygons will be visited during the ground phase of the accuracy assessment.

Because the accuracy assessment for the LANDSAT data will not be completed prior to the printing of the Terrestrial Report, caution in the use of this data is necessary. The validation and correction (if needed) of this data cannot be completed until the field portion of the accuracy assessment is conducted. For this reason, these data were only used in cases when describing acres of combined forestland classes summaries, nonforest summaries, and landscape habitat suitability analysis (that utilized combined classes in most cases).

Biological Conservation Database

Selected biological conservation database (BCD) EOR data fields were obtained in March 1995 from natural heritage programs of the Alabama Department of Conservation and Natural Resources; Georgia Department of Natural Resources; North Carolina Department of Environment, Health and Natural Resources; South Carolina Wildlife and Marine Resources Department; Tennessee Department of Environment and Conservation; Virginia Department of Conservation and Recreation; and West Virginia Department of Natural Resources.

Use of EOR data centered on occurrence of federally listed T&E species and species with viability concern (federal category 1 or 2, globally ranked 1, 2, or 3). Any occurrences of rare communities were also utilized. The key records utilized in the SAA were Element Code (ELCODE), latitude, and longitude. The reliability of the individual records was considered, but could not be validated due to inconsistency in data entry for the first observation and last observation for an EOR. For this reason, all records were considered in the analysis. Users of the derived SAA data themes for T&E and viability concern species should be aware that all records are included. The raw data obtained from states in the SAA area will be destroyed or returned by January, 1996. Only derived themes will remain in the SAA data set and will contain no EOR locational information. The EOR data were the primary data used in determining findings shown in Chapter 3.

Game Species County Density Estimates

County population density estimates were obtained from biologists with the Alabama Department of Conservation and Natural Resources; Georgia Department of Natural Resources; North Carolina Wildlife Resources Commission; South Carolina Wildlife and Marine Resources Department; Tennessee Wildlife Resources Agency; Virginia Department of Game and Inland Fisheries; and West Virginia Division of Natural Resources. These estimates are based on a combination of field inventories and knowledge of local state biologists. This was the primary data used in developing the findings shown in Chapter 3.

Ownership

Ownership was generated from USGS 1:100,000 Digital Line Graph (DLG) files. The 1:100,000 DLG data contained only the proclamation boundaries for national forest lands. To include actual national forest ownership, the 1:24,000 stand cover layers were used to derive actual national forest ownership. Most of the analysis results were stratified to ownership based on this data set.

Water and Stream Reaches

Water body were generated from EPA Reach File Version 3.0 (RF3) digitized from 1:100,000 scale USGS 30- by 60-minute quadrangle maps. The RF3 database contains primarily 4th order streams. Most 2nd and 3rd order streams, and nearly all 1st order streams, are not included in the database. The SAA Aquatics Resources Team estimated that about 30 percent of the total length of headwater reaches on upper slopes is represented. Riparian habitat was estimated using these data elements (Chapter 3).

Roads

Road data were developed from USGS 1:100,000 Digital Line Graph (DLG) files. The roads within this database are identified based on road size and use. Class 1 includes all primary highways, both federal- and state-numbered routes. Class 2 is secondary routes such as major county roads. Class 3 is minor paved county roads and major gravel-surfaced roads. Class 4 includes paved streets in cities and towns and lesser rural gravel roads. These data were used to analyze black bear habitat and habitat for area-sensitive species in Chapter 3.

Elevations

Elevation information was derived using the 30 arc-second Digital Elevation Model (DEM). The general scale for this data set is 1:250,000. The data layer contains elevation values on a 3,050 foot grid. These data were developed through a series of procedures conducted by the Defense Mapping Agency Topographic Center and the National Telecommunications and Information Administration. Elevation data were used to distinguish suitable habitat for species and communities related to elevational changes (Chapter 3).

Natural Resources Inventory Database – Trends for Non-Forest Lands

The 1982 and 1992 Natural Resources Inventory (NRI) database was used to determine trends for nonforest land cover types since the early 1980s (Chapter 3). This data set was obtained from the Natural Resources Conservation Service. The NRI has more than 300,000 primary sampling units and approximately 800,000 sample sites nationwide (USDA NRCS 1994).

Ecological Mapping Unit

Spatial information-related ecological mapping units for provinces, sections, and subsections were obtained from 1:2,000,000 DLG data layers. These boundaries were developed from R.G. Bailey's (1995) work for provinces and sections. The subsections were developed by a USDA Forest Service interdisciplinary team in the Southern Region. Most of the analysis results were stratified to ecological sections or subsections based on this data set.

National Interagency Fire Management Integrated Data Base

Information used to map and report wildfire occurrence on public lands administered by the USDA Forest Service was obtained from individual fire reports within the SAA area. Fire reports provide timely statistical data and information for both administrative purposes and managers to use in making land and resource management decisions. The report is a record of occurrence, related fire behavior conditions, and the suppression actions taken by management. Data collected from a fire report enable the manager to monitor program performance and plan the most cost-effective fire management organization.

Individual Fire Reports for the USDA Forest Service are stored in the National Interagency Fire Management Integrated Data Base (NIFMED) at the Kansas City Computer Center. By the use of computer runstreams, data archived in this base were retrieved to provide wildfire information for public lands administered by the USDA Forest Service. Information for wildfires on lands administered by state, private, and the National Park Service was provided by representatives from those agencies. For the SAA, fire report information was used to display fire locations, fire size, fire causes, number of fires, and number of acres of private, state, and public lands burned by wildfire.

Appendix B

The List of Special Plant and Animal Species for the Southern Appalachian Assessment

Appendix B contains the complete list of special species identified for emphasis in the Southern Appalachian Assessment.

Table B-1 provides information for each species that includes scientific name, common name, taxa, federal status, global rank, criteria used to select a species, and assigned grouping based on habitat association.

Table B–1 The list of 472 terrestrial plant and animal special species found in the Southern Appalachian Assessment (SAA) area according to taxa as determined using the assessment screening criteria.

Scientific Name	Common Name	Taxa	Federal Status	Global Rank	SAA Criteria[1]	Species Group[2]
Aneides aeneus	Green Salamander	Amphibian	2		2	7
Desmognathus aeneus	Seepage Salamander	Amphibian	2		2	11
Desmognathus imitator	Imitator Salamander	Amphibian			5	11
Desmognathus quadramaculatus	Blackbelly Salamander	Amphibian			5	11
Desmognathus santeetlah	Santeetlah Dusky Salamander	Amphibian			5	11
Desmognathus welteri	Black Mountain Salamander	Amphibian			5	11
Desmognathus wrighti	Pigmy Salamander	Amphibian			5	11
Eurycea aquatica	Dark-sided (Brownback) Salamander	Amphibian	2		2	11
Eurycea junaluska	Junaluska Salamander	Amphibian	2	2	2	11
Eurycea wilderae	Blue Ridge Two-lined Salamander	Amphibian			5	11
Gyrinophilus palleucus	Tennessee Cave Salamander	Amphibian	2		2	1
Gyrinophilus subterraneus	West Virginia Spring Salamander	Amphibian	2		2	1
Leurognathus marmoratus	Shovelnose Salamander	Amphibian			5	11
Plethodon hubrichti	Peaks of Otter Salamander	Amphibian	2	2	2	10
Plethodon jordani	Jordan's Salamander	Amphibian			5	11
Plethodon kentucki	Cumberland Plateau Salamander	Amphibian			5	11
Plethodon nettingi	Cheat Mountain Salamander	Amphibian	T	3	1	15
Plethodon petraeus	Pigeon Mountain Salamander	Amphibian		1	2	7
Plethodon punctatus	Cow Knob Salamander	Amphibian	2	3	2	10
Plethodon shenandoah	Shenandoah Salamander	Amphibian	E	1	1	7
Plethodon yonahlossee	Yonahlossee Salamander	Amphibian			5	11
Accipiter gentilis	Northern Goshawk	Bird	2		2	15
Aegolius acadicus	Northern Saw-whet Owl	Bird			5	15
Aimophila aestivalis	Bachman's Sparrow	Bird	2	3	2	8
Ammodramus henslowii	Henslow's Sparrow	Bird	2		2	8
Bonasa umbellus	Ruffed Grouse	Bird			4	12
Coccyzus erythropthalmus	Black-billed Cuckoo	Bird			5	13
Colaptes auratus	Northern Flicker	Bird			6	12
Colinus virginianus	Northern Bobwhite	Bird			4	8
Contopus virens	Eastern Wood-Pewee	Bird			5	10
Dendroica caerulescens	Black-throated Blue Warbler	Bird			5	14
Dendroica cerulea	Cerulean Warbler	Bird	2		2	13
Dendroica discolor	Prairie Warbler	Bird			5	8
Dendroica fusca	Blackburnian Warbler	Bird			5	14
Dendroica pensylvanica	Chestnut-sided Warbler	Bird			5	5
Dendroica virens	Black-throated Green Warbler	Bird			5	13
Dryocopus pileatus	Pileated Woodpecker	Bird			6	13
Dumetella carolinensis	Gray Catbird	Bird			5	12
Empidonax virescens	Acadian Flycatcher	Bird			5	11
Falco peregrinus anatum	American Peregrine Falcon	Bird	E		1	7
Haliaeetus leucocephalus	Bald Eagle	Bird	T		1	11
	Worm-eating Warbler	Bird			5	13
Hylocichla mustelina	Wood Thrush	Bird			5	13
Lanius ludovicianus	Loggerhead Shrike	Bird	2		2	8
Limnothlypis swainsonii	Swainson's Warbler	Bird			5	13
Loxia curvirostra	Red Crossbill	Bird			5	14
Melanerpes carolinus	Red-bellied woodpecker	Bird			7	13
Melanerpes erythrocephalus	Red-headed Woodpecker	Bird			6	12
Meleagris gallopavo	Eastern Wild Turkey	Bird			4	12
Oporornis formosus	Kentucky Warbler	Bird			5	13
Parula americana	Northern Parula	Bird			5	13
Picoides borealis	Red Cockaded Woodpecker	Bird	E		1	17
Picoides pubescens	Downy woodpecker	Bird			7	10
Picoides villosus	Hairy Woodpecker	Bird			7	13
Piranga olivacea	Scarlet Tanager	Bird			5	13
Piranga rubra	Summer Tanager	Bird			5	13
Protonotaria citrea	Prothonotory Warbler	Bird			5	16
Scolopax minor	American Woodcock	Bird			4	11
Seiurus aurocapillus	Ovenbird	Bird			5	13
Seiurus motacilla	Louisiana Waterthrush	Bird			5	11

Table B–1 (cont.) The list of 472 terrestrial plant and animal special species found in the Southern Appalachian Assessment (SAA) area according to taxa as determined using the assessment screening criteria.

Scientific Name	Common Name	Taxa	Federal Status	Global Rank	SAA Criteria[1]	Species Group[2]
Sitta pusilla	Brown-headed Nuthatch	Bird			5	17
Spizella pusilla	Field Sparrow	Bird			5	8
Thryomanes bewickii altus	Appalachian Bewick's Wren	Bird	2		2	5
Vermivora chrysoptera	Golden-winged Warbler	Bird			5	8
Vermivora pinus	Blue-winged Warbler	Bird			5	8
Vireo flavifrons	Yellow-throated Vireo	Bird			5	13
Wilsonia canadensis	Canada Warbler	Bird			5	14
Wilsonia citrina	Hooded Warbler	Bird			5	13
Amerigoniscus henroti	Powell Valley Terrestrial Cave Isopod	Invertebrate		1	2	1
Antrolana lira	Madison Cave isopod	Invertebrate	T	1	1	1
Apochthonius coecus	A cave pseudoscorpion	Invertebrate		1	2	1
Apochthonius holsingeri	A cave pseudoscorpion	Invertebrate		1	2	1
Arianops jeanneli	A cave pselaphid beetle	Invertebrate		1	2	1
Arrhopalites clarus	A cave springtail	Invertebrate		1	2	1
Atheta annexa	A rove beetle	Invertebrate		2	2	
Atheta troglophila	A rove beetle	Invertebrate		1	2	
Brachoria cedra	Cedar millipede	Invertebrate		1	2	18
Brachoria dentata	A millipede	Invertebrate		1	2	18
Brachoria ethotela	Hungry Mother millipede	Invertebrate		2	2	18
Brachoria falcifera	Big Cedar Creek millipede	Invertebrate		1	2	18
Brachoria hoffmani	Hoffman's xystodesmid millipede	Invertebrate		2	2	18
Brachoria separanda hamata	A millipede	Invertebrate		2	2	18
Buotus carolinus	A millipede	Invertebrate		1	2	18
Caecidotea henroti	Henrot's cave isopod	Invertebrate		2	2	1
Caecidotea holsingeri	Greenbriar Valley cave isopod	Invertebrate		3	2	1
Caecidotea incurva	Incurved cave isopod	Invertebrate		2	2	1
Caecidotea pricei	Price's cave isopod	Invertebrate		3	2	1
Caecidotea sinuncus	An isopod	Invertebrate		1	2	1
Caecidotea vandeli	Vandel's cave isopod	Invertebrate		2	2	1
Catocala herodias gerhardi	Herodias underwing	Invertebrate		3	2	
Chitrella superba	A cave pseudoscorpion	Invertebrate		1	2	1
Cicindela ancocisconensis	A tiger beetle	Invertebrate		3	2	
Cicindela patruela	Barrens Tiger beetle	Invertebrate		3	2	
Cleidogona hoffmani	Hoffman's cleidogonid millipede	Invertebrate		2	2	15
Cleidogona lachesis	*A millipede*	*Invertebrate*		*2*	*2*	*15*
Conotyla venetia	Venetia millipede	Invertebrate		2	2	18
Dixioria coronata	A millipede	Invertebrate		2	2	18
Dixioria fowleri	A millipede	Invertebrate		2	2	18
Euchlaena milnei	Looper moth	Invertebrate	2		2	6
Foveacheles paralleloseta	A cave pseudoscorpion	Invertebrate		1	2	1
Glyphyalinia clingmani	Fragile supercoil	Invertebrate	2		2	14
Helicodiscus hexodon	Toothy coil	Invertebrate	2		2	
Hepialus sciophanes	A ghost moth	Invertebrate	2		2	15
Islandiana speophila	Cavern sheetweb spider	Invertebrate		1	2	1
Kleptochthonius lutzi	A cave pseudoscorpion	Invertebrate		1	2	1
Kleptochthonius proximosetus	A cave pseudoscorpion	Invertebrate		1	2	1
Kleptochthonius regulus	A cave pseudoscorpion	Invertebrate		1	2	1
Kleptochthonius similis	A cave pseudoscorpion	Invertebrate		1	2	1
Kleptochthonius species 1	A cave pseudoscorpion	Invertebrate		1	2	1
Lirceus culveri	Rye cove isopod	Invertebrate	2		2	1
Lirceus usdagalun	Lee County Cave isopod	Invertebrate	E		1	1
Litocampa barringerorum	A cave dipluran	Invertebrate		1	2	1
Litocampa bifurcata	A cave dipluran	Invertebrate			2	1
Litocampa cookei	A cave dipluran	Invertebrate			2	1
Litocampa holsingeri	A cave dipluran	Invertebrate		2	2	1
Macrocotyla hoffmasteri	Hoffmaster's cave flatworm	Invertebrate		3	2	1
Mesodon clingmanicus	Clingman Covert	Invertebrate	2		2	15
Microcreagris valentinei	A cave pseudoscorpion	Invertebrate		1	2	1
Microhexura montivaga	Spruce-Fir Moss Spider	Invertebrate	E		1	15

Table B–1(cont.) The list of 472 terrestrial plant and animal special species found in the Southern Appalachian Assessment (SAA) area according to taxa as determined using the assessment screening criteria.

Scientific Name	Common Name	Taxa	Federal Status	Global Rank	SAA Criteria^1	Species Group^2
Miktoniscus racovitzae	Racovitza's Terrestrial Cave Isopod	Invertebrate		2	2	1
Mundochthonius holsingeri	A cave pseudoscorpion	Invertebrate		1	2	1
Nampabius turbator	A cave centipede	Invertebrate			2	1
Nannaria ericacea	McGraw Gap Xystodesmid	Invertebrate		2	2	18
Nannaria shenandoah	Shenandoah Mountain Xystodesmid	Invertebrate		1	2	18
Nesticus carolinensis	Linville Cavern spider	Invertebrate		1	2	1
Nesticus cooperi	Lost Nantahala Cave Spider	Invertebrate	2		2	1
Nesticus crosbyi	A nesticid spider	Invertebrate		1	2	1
Nesticus holsingeri	Holsinger's Cave spider	Invertebrate		2	2	1
Nesticus mimus	A cave spider	Invertebrate		2	2	1
Nesticus paynei	A cave spider	Invertebrate		2	2	1
Nesticus sheari	A nesticid spider	Invertebrate		2	2	1
Nesticus silvanus	A nesticid spider	Invertebrate		3	2	1
Nesticus tennesseensis	A cave spider	Invertebrate		2	2	1
Paravitrea ternaria	Sculptured supercoil	Invertebrate	2		2	
Paravitrea varidens	Roan supercoil	Invertebrate	2		2	14
Patera clarki nantahala	Noonday globe snail	Invertebrate	T		1	6
Phanetta subterranea	A spider	Invertebrate		3	2	1
Phyciodes batesii	Tawny crescentspot butterfly	Invertebrate	2		2	19
Poecilophysis extraneostella	A cave mite	Invertebrate		2	2	1
Poecilophysis weyerensis	A cave mite	Invertebrate		2	2	1
Polygyriscus virginicus	Virginia Fringed Mountain Snail	Invertebrate	E		1	6
Pseudanophthalmus avernus	Avernus Cave beetle	Invertebrate	2	1	2	1
Pseudanophthalmus cordicollis	Little Kennedy Cave beetle	Invertebrate	2	1	2	1
Pseudanophthalmus deceptivus	Deceptive Cave beetle	Invertebrate	2	1	2	1
Pseudanophthalmus delicatus	A cave beetle	Invertebrate		2	2	1
Pseudanophthalmus egberti	New River Valley Cave beetle	Invertebrate	2	1	2	1
Pseudanophthalmus gracilis	A cave beetle	Invertebrate		1	2	1
Pseudanophthalmus hadenoecus	Timber ridge cave beetle	Invertebrate	2	1	2	1
Pseudanophthalmus hirsutus	Lee County Cave beetle	Invertebrate	2	1	2	1
Pseudanophthalmus hoffmani	A cave beetle	Invertebrate		1	2	1
Pseudanophthalmus holsingeri	Holsinger's Cave beetle	Invertebrate	1	1	2	1
Pseudanophthalmus hubbardi	Hubbard's Cave beetle	Invertebrate	2	1	2	1
Pseudanophthalmus hubrichti	Hubricht's Cave beetle	Invertebrate	2	1	2	1
Pseudanophthalmus intersectus	Crossroads Cave beetle	Invertebrate	2	1	2	1
Pseudanophthalmus limicola	Mud-dwelling cave beetle	Invertebrate	2	1	2	1
Pseudanophthalmus longiceps	Long-headed cave beetle	Invertebrate	2	1	2	1
Pseudanophthalmus nelsoni	Nelson's Cave Beetle	Invertebrate	2	1	2	1
Pseudanophthalmus nickajackensis	Nickajackensis cave beetle	Invertebrate			2	1
Pseudanophthalmus parvicollis	Thin-neck cave beetle	Invertebrate	2	1	2	1
Pseudanophthalmus paulus	Nobletts Cave beetle	Invertebrate	2		2	1
Pseudanophthalmus paynei	Paynes Cave beetle	Invertebrate	2		2	1
Pseudanophthalmus petrunkevitchi	Petrunkevitch's cave beetle	Invertebrate	2	1	2	1
Pseudanophthalmus pontis	Natural Bridge Cave beetle	Invertebrate	2	1	2	1
Pseudanophthalmus potomaca potomaca	South Branch Valley cave beetle	Invertebrate	2	1	2	1

Table B–1(cont.) The list of 472 terrestrial plant and animal special species found in the Southern Appalachian Assessment (SAA) area according to taxa as determined using the assessment screening criteria.

Scientific Name	Common Name	Taxa	Federal Status	Global Rank	SAA Criteria^1	Species Group^2
Pseudanophthalmus potomaca senecae	Seneca cave beetle	Invertebrate	2	1	2	1
Pseudanophthalmus praetermissus	Overlooked Cave beetle	Invertebrate	2	1	2	1
Pseudanophthalmus punctatus	Spotted Cave beetle	Invertebrate	2	1	2	1
Pseudanophthalmus pusio	A cave beetle	Invertebrate		1	2	1
Pseudanophthalmus quadratus	Straley's Cave beetle	Invertebrate	2	1	2	1
Pseudanophthalmus rotundatus	A cave beetle	Invertebrate		1	2	1
Pseudanophthalmus sanctipauli	Saint Paul Cave beetle	Invertebrate	2	1	2	1
Pseudanophthalmus sericus	Silken cave beetle	Invertebrate	2	1	2	1
Pseudanophthalmus sidus	Meredith Cave beetle	Invertebrate	2		2	1
Pseudanophthalmus species 10	A cave beetle	Invertebrate		1	2	1
Pseudanophthalmus species 11	A cave beetle	Invertebrate		1	2	1
Pseudanophthalmus species 4	A cave beetle	Invertebrate		1	2	1
Pseudanophthalmus species 5	A cave beetle	Invertebrate		1	2	1
Pseudanophthalmus species 6	A cave beetle	Invertebrate		1	2	1
Pseudanophthalmus species 7	A cave beetle	Invertebrate		1	2	1
Pseudanophthalmus species 8	A cave beetle	Invertebrate		1	2	1
Pseudanophthalmus species 9	A cave beetle	Invertebrate		1	2	1
Pseudanophthalmus thomasi	Thomas' Cave beetle	Invertebrate	2	1	2	1
Pseudanophthalmus vicarius	A cave beetle	Invertebrate	2	2	2	1
Pseudanophthalmus virginicus	Maiden Spring Cave beetle	Invertebrate	2	1	2	1
Pseudosinella hirsuta	A cave springtail	Invertebrate		1	2	1
Pseudotremia alecto	A millipede	Invertebrate		1	2	18
Pseudotremia armesi	A millipede	Invertebrate		2	2	1
Pseudotremia lusciosa	Germany Valley cave millipede	Invertebrate		1	2	1
Pseudotremia momus	A millipede	Invertebrate		2	2	1
Pseudotremia princeps	South Branch Valley cave millipede	Invertebrate		1	2	1
Pseudotremia tuberculata	A millipede	Invertebrate		2	2	1
Rhagidia varia	A cave mite	Invertebrate		3	2	1
Rudiloria trimaculata tortua	A millipede	Invertebrate		2	2	18
Semionellus placidus	A millipede	Invertebrate		3	2	18
Semiothisa fraserata	Fraser Fir geometrid	Invertebrate	2		2	15
Sigmoria whiteheadi	A millipede	Invertebrate		1	2	
Speyeria diana	Diana Fritillary Butterfly	Invertebrate	2	3	2	18
Speyeria idalia	Regal Fritillary Butterfly	Invertebrate	2	3	2	
Sphalloplana chandleri	Chandler's planarian	Invertebrate		1	2	1
Sphalloplana consimilis	Powell Valley planarian	Invertebrate		1	2	1
Sphalloplana virginiana	Rockbridge County Cave planarian	Invertebrate	2	1	2	1
Striaria columbiana	A millipede	Invertebrate		2	2	
Striaria species 1	A millipede	Invertebrate		1	2	
Stygobromus abditus	James cave amphipod	Invertebrate		2	2	1
Stygobromus baroodyi	Rockbridge County cave amphipod	Invertebrate		2	2	1
Stygobromus biggersi	Bigger's Cave amphipod	Invertebrate	2	1	2	1
Stygobromus conradi	Burnsville cove cave amphipod	Invertebrate	2	1	2	1
Stygobromus cumberlandus	Cumberland cave amphipod	Invertebrate		2	2	1
Stygobromus ephemerus	Ephemeral cave amphipod	Invertebrate		1	2	1
Stygobromus estesi	Craig County cave amphipod	Invertebrate		1	2	1
Stygobromus fergusoni	Montgomery County cave amphipod	Invertebrate		1	2	1
Stygobromus gracilipes	Shenandoah Valley cave amphipod	Invertebrate		2	2	1
Stygobromus hoffmani	Alleghany County cave amphipod	Invertebrate		1	2	1
Stygobromus interitus	New Castle Murder Hole amphipod	Invertebrate		1	2	1
Stygobromus leensis	Lee County cave amphipod	Invertebrate		1	2	1
Stygobromus morrisoni	Morrison's cave amphipod	Invertebrate	2	2	2	1
Stygobromus mundus	Bath County cave amphipod	Invertebrate	2	1	2	1
Stygobromus pseudospinosus	Luray Caverns amphipod	Invertebrate		1	2	1
Stygobromus species 7	Sherando Spinosoid amphipod	Invertebrate		1	2	1
Stygobromus spinosus	Blue Ridge Mountain amphipod	Invertebrate		2	2	1
Stygobromus stegerorum	Madison Cave amphipod	Invertebrate		1	2	1
Stylodrilus beattiei	A cave lumbriculid worm	Invertebrate		1	2	1
Trichopetalum krekeleri	West Virginia Blind cave millipede	Invertebrate		1	2	1
Canis rufus	Red Wolf	Mammal	E		1	9

Table B–1(cont.) The list of 472 terrestrial plant and animal special species found in the Southern Appalachian Assessment (SAA) area according to taxa as determined using the assessment screening criteria.

Scientific Name	Common Name	Taxa	Federal Status	Global Rank	SAA Criteria[1]	Species Group[2]
Castor canadensis	Beaver	Mammal			7	11
Corynorhinus rafinesquii	Rafinesque's big-eared bat	Mammal	2	3	2	11
Corynorhinus townsendii virginianus	Virginia Big-eared Bat	Mammal	E		1	1
Felis concolor cougar	Eastern Cougar	Mammal	E		1	9
Glaucomys sabrinus coloratus	Carolina Northern Flying Squirrel	Mammal	E		1	15
Glaucomys sabrinus fuscus	Virginia Northern Flying Squirrel	Mammal	E		1	15
Microtus chrotorrhinus carolinensis	Southern rock vole	Mammal	2	3	2	7
Myotis austroriparius	Southeastern bat	Mammal	2		2	1
Myotis grisescens	Gray Bat	Mammal	E		1	1
Myotis leibii	Eastern small-footed bat	Mammal	2		2	1
Myotis sodalis	Indiana Bat	Mammal	E		1	1
Neotoma floridana haematoreia	Southern Appalachian Eastern woodrat	Mammal	2		2	7
Neotoma magister	Allegheny woodrat	Mammal	2		2	7
Odocoileus virginianus	White-tailed deer	Mammal			4	12
Procyon lotor	Raccoon	Mammal			4	11
Sciurus carolinensis	Eastern Gray Squirrel	Mammal			4	10
Sciurus niger	Eastern Fox Squirrel	Mammal			4	10
Sorex palustris punctulatus	Southern Water shrew	Mammal	2	3	2	11
Sylvilagus floridanus	Eastern Cottontail	Mammal			4	8
Sylvilagus obscurus	Appalachian cottontail	Mammal	2		2	5
Ursus americanus	Black Bear	Mammal			4	9
Abies fraseri	Fraser fir	Plant	2	2	2	15
Aconitum reclinatum	Trailing wolfsbane	Plant		3	2	15
Ageratina luciae-brauniae	Lucy Braun's white snakeroot	Plant	2		2	7
*Allium allegheniense*s	Allegheny onion	Plant		3	2	5
Allium cuthbertii	Striped garlic	Plant		3	2	7
Allium speculae	Little river canyon onion	Plant	2		2	7
Amorpha glabra	Appalachian indigo bush	Plant		3	2	7
Amphianthus pusillus	Pool Sprite	Plant	T		1	7
Anemone minima	Tiny anemone	Plant		3	2	18
Apios priceana	Price's potato-bean	Plant	T		1	18
Arabis georgiana	Georgia rockcress	Plant	2	2	2	6
Arabis serotina	Shale barren rock cress	Plant	E	2	1	7
Arenaria cumberlandensis	Cumberland sandwort	Plant	E		1	7
Arenaria godfreyi	Godfrey's stitchwort	Plant	2	1	2	6
Aspiromitus appalachianus	A hornwort	Plant		1	2	11
Asplenium scolopendrium var american	Hart's tongue fern	Plant	T	1	1	6
Aster avitus	Alexander's rock aster	Plant	2	1	2	7
Aster georgianus	Georgia aster	Plant	2		2	6
Aster surculosus	Creeping aster	Plant		3	2	7
Astragalus neglectus	Cooper's milkvetch	Plant	2	3	2	6
Aureolaria patula	Spreading false-foxglove	Plant	2	2	2	6
Bazzania nudicaulis	Liverwort	Plant	2	2	2	15
Betula uber	Virginia round-leaf birch	Plant	T	1	1	11
Bigelowia nuttallii	Nuttall's rayless goldenrod	Plant		2	2	
Brachydontium trichodes	Peak moss	Plant		2	2	15
Brachymenium andersonii	Anderson's brachymenium	Plant	2		2	18
Bryocrumia vivicolor	Gorge moss	Plant	2	1	2	3
Buckleya distichophylla	Piratebush	Plant	2	2	2	18
Cacalia rugelia	Rugel's ragwort	Plant	2	3	2	15
Calamagrostis cainii	Cain's reedgrass	Plant	2	2	2	7
Calamovilfa arcuata	Cumberland sandgrass	Plant	2	2	2	11
Calystegia catesbiana ssp. sericata	Blue Ridge bindweed	Plant		3	2	8
Cardamine clematitis	Mountain bitter cress	Plant	2	2	2	11
Carex manhartii	Manhart's sedge	Plant	2	2	2	18
Carex misera	Wretched sedge	Plant		3	2	7
Carex polymorpha	Variable sedge	Plant	2	2	2	19

Table B–1(cont.) The list of 472 terrestrial plant and animal special species found in the Southern Appalachian Assessment (SAA) area according to taxa as determined using the assessment screening criteria.

Scientific Name	Common Name	Taxa	Federal Status	Global Rank	SAA Criteria[1]	Species Group[2]
Carex purpurifera	Purple sedge	Plant	2	3	2	18
Carex radfordii (=C. species 3)	Radford's sedge	Plant		1	2	19
Carex roanensis	Roan Mtn. sedge	Plant	2	1	2	18
Carex ruthii	Ruth's sedge	Plant		3	2	11
Carex schweinitzii	Schweinitz's sedge	Plant		3	2	2
Cheilolejeunea evansii	Liverwort	Plant	2	H	2	18
Chelone cuthbertii	Cuthbert's turtlehead	Plant		3	2	2
Chiloscyphus appalachianus	Liverwort	Plant	2	1	2	3
Clematis addisonii	Addison's leatherflower	Plant	2	2	2	6
Clematis coactilis	Virginia white-haired leatherflower	Plant		3	2	7
Clematis socialis	Alabama leather-flower	Plant	E	1	1	11
Clematis viticaulis	Millboro leatherflower	Plant	2	2	2	7
Collinsonia verticillata		Plant		2	2	18
Conradina verticillata	Cumberland rosemary	Plant	T		1	11
Coreopsis latifolia	Broadleaf coreopsis	Plant		3	2	6
Coreopsis pulchra	Woodland tickseed	Plant		2	2	
Crataegus harbisonii	Harbison's hawthorn	Plant	2		2	
Cuscuta harperi	Harper's dodder	Plant	2		2	
Cyperus granitophilus	Granite-loving flatseed	Plant		3	2	7
Cypripedium kentuckiense	Southern lady's-slipper	Plant	2	3	2	18
Delphinium exaltatum	Tall larkspur	Plant	2	3	2	6
Diervilla rivularis	Mountain bush honeysuckle	Plant		3	2	
Diphylleia cymosa	Umbrella leaf	Plant		3	2	11
Draba aprica	Whitlow grass	Plant		3	2	7
Echinacea laevigata	Smooth Coneflower	Plant	E	3	1	6
Elymus svensonii	Svenson's wild-rye	Plant	2	2	2	6
Euphorbia purpurea	Darlington's spurge	Plant	2	3	2	6
Eurhynchium pringlei	Pringle's eurhynchium	Plant	2	2	2	3
Fothergilla major	Witch alder	Plant		3	2	19
Gaylussacia brachycera	Box huckleberry	Plant		3	2	19
Gentiana austromontana	Appalachian gentian	Plant		3	2	5
Geum geniculatum	Bent avens	Plant	2	1	2	5
Geum radiatum	Spreading avens	Plant	E	1	1	5
Glyceria nubigena	Smoky Mountain manna grass	Plant	2	2	2	11
Grammitis nimbata	Dwarf polypody fern	Plant	2	3	2	3
Gymnocarpium appalachianum	Appalachian oak fern	Plant	2	3	2	15
Gymnoderma lineare	Rock gnome lichen	Plant	E	2	1	7
Hasteola suaveolens	Sweet Indian plantain	Plant		3	2	11
Hedyotis purpurea var. montana	Roan mountain bluet	Plant	E	2	1	5
Helenium brevifolium	Shortleaf sneezeweed	Plant		3	2	2
Helenium virginicum	Virginia sneezeweed	Plant	1	2	2	4
Helianthus glaucophyllus	White-leaved sunflower	Plant		3	2	18
Helianthus longifolius	Longleaf sunflower	Plant		3	2	
Helonias bullata	Swamp pink	Plant	T	3	1	2
Heuchera alba	White alumroot	Plant		2	2	7
Heuchera longiflora	Long-flowered alumroot	Plant		3	2	6
Hexastylis arifolia var. ruthii	Appalachian little brown jug	Plant		3	2	18
Hexastylis contracta	Mountain heartleaf	Plant	2	3	2	18
Hexastylis naniflora	Dwarf-flowered heartleaf	Plant	T	2	1	18
Hexastylis rhombiformis	French Broad heartleaf	Plant	2	2	2	18
Hudsonia montana	Mountain golden heather	Plant	T	1	1	7
Hydrothyria venosa	An aquatic lichen	Plant		3	2	11
Hymenophyllum tayloriae	Gorge filmy fern	Plant		1	2	3
Hymenophyllum tunbridgense	Tunbridge fern	Plant	2		2	3
Hypericum adpressum	Creeping St. John's-wort	Plant	2	2	2	2
Hypericum buckleyi	Blue Ridge St. John's-wort	Plant		3	2	5
Cardamine flagellifera	Bittercress	Plant		3	2	11
Cardamine micranthera	Small anthered bittercress	Plant	E	1	1	11
Carex amplisquama	Fort mountain sedge	Plant	2	2	2	19
Carex austrocaroliana	South Carolina sedge	Plant		3	2	11

Table B–1(cont.) The list of 472 terrestrial plant and animal special species found in the Southern Appalachian Assessment (SAA) area according to taxa as determined using the assessment screening criteria.

Scientific Name	Common Name	Taxa	Federal Status	Global Rank	SAA Criteria[1]	Species Group[2]
Carex barrattii	Barratt's sedge	Plant		3	2	2
Carex biltmoreana	Biltmore sedge	Plant		3	2	6
Hypericum dolabriforme	Straggling St. John's wort	Plant		3	2	6
Hypericum graveolens	Mountain St. John's-wort	Plant		3	2	5
Hypericum mitchellianum	Mitchell's St. John's-wort	Plant		3	2	5
Ilex collina	Long-stalked holly	Plant		3	2	2
Iliamna corei	Peter's mountain mallow	Plant	E	1	1	7
Iliamna remota	Kankakee globe-mallow	Plant	2	1	2	11
Isoetes virginica	Quillwort	Plant	2	1	2	4
Isotria medeoloides	Small whorled pogonia	Plant	E	3	1	18
Jamesianthus alabamensis	Jamesianthus	Plant	2	3	2	
Juglans cinerea	Butternut	Plant	2	3	2	10
Juncus caesariensis	New Jersey rush	Plant	2	2	2	2
Juncus gymnocarpus	Coville's rush	Plant		3	2	2
Krigia montana	False dandelion	Plant		3	2	7
Leavenworthia exigua var. exigua	Glade cress	Plant		3	2	6
Lejeunea blomquistii	*Liverwort*	*Plant*	*2*	*1*	*2*	*3*
Leptothymenium sharpii	Mt. Leconte moss	Plant	2	1	2	15
Liatris helleri	Heller's blazing star	Plant	T	1	1	5
Liatris turgida	Shale-barren blazing star	Plant		3	2	7
Lilium grayi	Gray's lily	Plant	2	2	2	5
Lysimachia fraseri	Fraser's loosestrife	Plant	2	2	2	18
Lysimachia graminea	Grass-leaved loosestrife	Plant		2	2	
Marshallia grandiflora	Large-flowered barbara's-buttons	Plant	2	2	2	2
Marshallia morhii	Morh's Barbara's buttons	Plant	T		1	11
Marshallia trinervia	Broadleaf Barbara's buttons	Plant		3	2	11
Megaceros aenigmaticus	A hornwort	Plant		2	2	11
Minuartia fontinalis	Water stitchwort	Plant	2		2	
Monotropsis odorata	Sweet pinesap	Plant	2	3	2	19
Nestronia umbellula	Nestronia	Plant		3	2	19
Neviusia alabamensis	Alabama snow wreath	Plant	2	2	2	6
Orthotrichum keeverae	Keever's bristle-moss	Plant	2	1	2	6
Parnassia grandifolia	Large-leaved grass-of-parnassus	Plant		2	2	2
Paronychia virginica var. virginica	Yellow nailwort	Plant	2	1	2	7
Paxistima canbyi	Canby's mountain-lover	Plant	2	2	2	6
Phacelia fimbriata	Fringed scorpion-weed	Plant		3	2	14
Phlox amplifolia	Broadleaf phlox	Plant		3	2	18
Phlox bifida ssp. stellaria	Cleft phlox	Plant	2		2	6
Phlox buckleyi	Sword leaved phlox	Plant		2	2	19
Pityopsis ruthii	Ruth's golden aster	Plant	E	1	1	11
Plagiochila austinii	Liverwort	Plant		3	2	3
Plagiochila caduciloba	Liverwort	Plant	2	2	2	3
Plagiochila corniculata	Liverwort	Plant		3	2	15
Plagiochila echinata	Liverwort	Plant	2	1	2	3
Plagiochila sharpii	Liverwort	Plant	2	2	2	3
Plagiochila sullivantii var. spinigera	Liverwort	Plant	2	2	2	3
Plagiochila sullivantii var. sullivantii	Liverwort	Plant	2	2	2	3
Plagiochila virginica var. caroliniana	Liverwort	Plant	2	2	2	3
Plagiochila virginica var. euryphylla	Liverwort	Plant	2	1	2	3
Plagiochila virginica var. virginica	Liverwort	Plant		2	2	6
Plantago cordata	Heart-leaf plantain	Plant		3	2	11
Plantanthera integrilabia	White fringeless orchid	Plant	2	2	2	2
Platanthera leucophaea	Eastern prairie fringed orchid	Plant	T	2	1	4
Poa paludigena	Bog blue grass	Plant	2	3	2	2
Polymnia laevigata	Tennessee leafcup	Plant		3	2	

Table B–1(cont.) The list of 472 terrestrial plant and animal special species found in the Southern Appalachian Assessment (SAA) area according to taxa as determined using the assessment screening criteria.

Scientific Name	Common Name	Taxa	Federal Status	Global Rank	SAA Criteria[1]	Species Group[2]
Porella appalachiana	Liverwort	Plant	2	1	2	3
Porella wataugensis	Liverwort	Plant		1	2	3
Potamogeton tennesseensis	Tennessee pondweed	Plant		3	2	4
Prenanthes barbata	Bearded rattlesnake-root	Plant	2	2	2	
Prenanthes roanensis	Roan rattlesnakeroot	Plant		3	2	5
Prunus alleghaniensis	Alleghany plum	Plant	2	3	2	19
Ptilimnium nodosum	Harperella	Plant	E	2	1	11
Pycnanthemum curvipes	Tennessee mountain mint	Plant		3	2	6
Pycnanthemum torrei	Torrey mountain-mint	Plant		2	2	6
Radula voluta	Liverwort	Plant		2	2	3
Rhododendron carolinianum	Carolina Rhododendron	Plant		3	2	5
Rhododendron cumberlandense	Cumberland azalea	Plant		2	2	5
Rhododendron vaseyi	Pink-shell azalea	Plant		3	2	18
Robinia viscosa var. hartwegii	Hartwig's locust	Plant		1	2	7
Robinia viscosa var. viscosa	Clammy locust	Plant		3	2	5
Rubus whartoniae	Wharton's dewberry	Plant	2		2	
Rudbeckia heliopsidis	Sun-facing coneflower	Plant	2	2	2	
Rudbeckia triloba var. pinnatiloba	Pinnately-lobed brown-eyed sunflower	Plant	2	3	2	7
Sabatia capitata	Rose pink	Plant		2	2	
Sagittaria fasciculata	Bunched arrowhead	Plant	E	1	1	11
Sagittaria secundifolia	Kral's water-plantain	Plant	T		1	2
Sarracenia jonesii	Mountain sweet pitcherplant	Plant	E	1	1	2
Sarracenia oreophila	Green pitcher plant	Plant	E	2	1	2
Saxifraga careyana	Golden-eye saxifrage	Plant		3	2	7
Saxifraga caroliniana	Carolina saxifrage	Plant	2	2	2	7
Schlotheimia lancifolia	Highlands moss	Plant	2	2	2	18
Scirpus ancistrochaetus	Northeastern bullrush=Barbed bullrush	Plant	E	2	1	4
Scutellaria montana	Large- flowered skullcap	Plant	E	2	1	18
Scutellaria saxatilis	Rock skullcap	Plant		2	2	7
Sedum nevii	Nevius' stonecrop	Plant	2	2	2	7
Senecio millefolium	Divided-leaf ragwort	Plant	2	2	2	6
Shortia galacifolia var. brevistyla	Short-styled oconee bells	Plant	2	1	2	18
Shortia galacifolia var. galacifolia	Oconee bells	Plant	2	2	2	18
Sida hermaphrodita	Virginia mallow	Plant		3	2	11
Silene ovata	Mountain catchfly	Plant	2	3	2	6
Silene regia	Royal catchfly	Plant		3	2	
Silphium brachiatum	Cumberland rosinweed	Plant	2		2	
Silphium connatum	Virginia cup-plant	Plant		3	2	16
Sisyrinchium dichotomum	White irisette	Plant	E	1	1	6
Smilax biltmoreana	Biltmore carrion-flower	Plant		2	2	12
Solidago glomerata	Goldenrod	Plant		3	2	15
Solidago lancifolia	Lance leafed goldenrod	Plant		3	2	18
Solidago rupestris	Rock goldenrod	Plant		2	2	11
Solidago simulans	Granite dome goldenrod	Plant		1	2	7
Solidago spithamaea	Blue Ridge goldenrod	Plant	T	1	1	5
Sphenolobopsis pearsonii	Liverwort	Plant	2	2	2	15
Spiraea virginiana	Virginia spiraea	Plant	T	1	1	11
Splachnum pennsylvanicum	Southern dungmoss	Plant		2	2	2
Stachys clingmanii	Clingman's hedgenettle	Plant		3	2	15
Stellaria corei	Core's starwort	Plant		3	2	11
Talinum mengesii	Menge's flame-flower	Plant		3	2	7
Thalictrum subrotundum	Reclined meadowrue	Plant	2		2	
Tomanthera auriculata	Auriculate false-foxglove	Plant	2	2	2	6
Tomanthera pseudophyllum	Shiner's false-foxglove	Plant	2	2	2	
Tortula ammonsiana	Ammons' tortula	Plant	2	1	2	6
Trichomanes petersii	Dwarf filmy fern	Plant		3	2	7
Trifolium calcaricum	Running glade clover	Plant	2	1	2	6
Trillium discolor	Mottled trillium	Plant		3	2	18

Table B–1(cont.) The list of 472 terrestrial plant and animal special species found in the Southern Appalachian Assessment (SAA) area according to taxa as determined using the assessment screening criteria.

Scientific Name	Common Name	Taxa	Federal Status	Global Rank	SAA Criteria[1]	Species Group[2]
Trillium lancifolium	Narrow-leaved trillium	Plant		3	2	
Trillium persistens	Persistent trillium	Plant	E		1	18
Trillium pusillum (T.p. var 1)	Least trillium	Plant	2	3	2	18
Trillium pusillum var. monticulum	Trillium	Plant	2	3	2	18
Trillium rugelii	Southern nodding trillium	Plant		3	2	6
Trillium simile	Sweet white trillium	Plant		3	2	6
Vaccinium hirsutum	Hairy blueberry	Plant	2	3	2	18
Viburnum bracteatum	Arrowwood	Plant	2		2	11
Vitis rupestris	Sand grape	Plant		3	2	11
Waldsteinia lobata	Lobed barren-strawberry	Plant		2	2	18
Xanthoparmelia monticola	A foliose lichen	Plant		2	2	7
Xerophyllum asphodeloides	Eastern turkey beard	Plant		3	2	19
Xyris tennesseensis	Tennessee yellow-eyed grass	Plant	E		1	6
Chelone lyonii	Purple turtlehead	Plant		3	2	15
Cimicifuga rubifolia	Appalachian bugbane	Plant	2	3	2	18
Cladonia psoromica	Bluff mountain reindeer lichen	Plant	2	1	2	4
Clemmys muhlenbergii	Bog Turtle	Reptile	2		2	2
Pituophis m. melanoleucus	Northern Pine Snake	Reptile	2		2	19

[1] SAA Criteria Code
1 = Federally Threatened or Endangered
2 = Viability Concern Species
4 = Game Species
5 = High Management/Public Interest
6 = Demanding Habitat Requirements
7 = Keystone Species

[2] Species Group Codes
1 = Cave Habitats
2 = Mountain Bogs
3 = Spray Cliffs
4 = Fen or Pond Wetlands
5 = High Elevation Balds
6 = High pH or Mafic Habitats
7 = Rock Outcrop and Cliffs
8 = Early Successional Habitats
9 = Wide Ranging Area Sensitive Species
10 = Mid– to Late–Successional Forest Species
11 = Seep, Spring, and Streamside Habitat
12 = Habitat Generalist
13 = Area Sensitive Deciduous Forest
14 - General High Elevation Habitats
15 = High Elevation Spruce–Fir Forest
16 = Bottomland Forests
17 = Southern Yellow Pine Habitats
18 = Mixed Mesic Habitats
19 = Mixed Xeric Habitats

Appendix C

Descriptions and Summaries of the Broad Vegetation Classes, Rare Communities, and a Display of the National Forest's Initial Inventory of Possible Old Growth

Appendix C provides a brief description of the 16 broad vegetation classes and the 31 rare community types identified for the Southern Appalachian Assessment (SAA) area. Also included in this appendix is additional information for these classes. The national forests in the SAA area developed an initial inventory of possible old growth for consideration in future forest planning efforts. This initial inventory is shown spatially for the 36 national forest ranger districts located in the SAA area.

Broad Vegetation Class Descriptions

White Pine/Hemlock/ Hardwood Forest

This habitat group (includes two of the broad vegetation classes) occurs on mesic to somewhat xeric sites over a broad range of topographic conditions including ravines, valley flats, sheltered low ridges, open north-facing slopes at high elevations, and steep exposed slopes. For the purposes of describing this type, no distinction is being made between pure white pine/hemlock forests and mixed white pine/hemlock/hardwood forests.

This category includes forest dominated by hemlock *(Tsuga canadensis)* and white pine *(Pinus strobus),* singly or in mixtures with each other, and associated hardwood species. Hemlock may dominate forests of ravines and flats along streams at low to intermediate elevations, and at higher elevations, on open north-facing slopes. White pine may share dominance in the low- to intermediate-elevation forests, or hemlock may be associated with mesophytic hardwoods, particularly yellow-poplar *(Liriodendron tulipifera).* Shrub layers are typically ericaceious, with Rhododendron maximum, and doghobble *(Leucothoe fontaesiana),* and laurel *(Kalmia latifolia)* being very common. The herb layer may include *Mitchella repens, Viola rotundifolia, Tiarella cordifolia, Polystichum acrostichoides, Dryopteris intermedia,* and *Thelypteris noveboracensis.* White pine forests are particularly common along the Blue Ridge escarpment of North and South Carolina and Georgia. White Pine sometimes forms pure stands, but is often is mixed with hemlock along streams and with oaks *(Q. rubra, Q. alba, Q. montana, Q. velutina, and Q. coccinia)* upland slopes. The shrub layer may be dense, dominated by *Rhododendron* spp., *Vaccinium* spp., and *Gayussacia* spp. Herbaceous cover is usually sparse or absent.

This type is common in Georgia, the Carolinas, and Tennessee, and is somewhat less common in Virginia and West Virginia. It grades to mixed mesophytic hardwoods, northern hardwoods, mesic oak, and xeric oak forests. The soils in the forest are usually quite acid. Species diversity is low. White pine dominance in some areas is thought to be the result of various disturbances in predominantly oak forests. But, white pine also shows the ability to increase in the understory of oak-dominated stands in the absence of disturbance.

Mixed Mesophytic Hardwood Forest

This habitat group occurs at low to moderate elevation on mesic sites, generally on concave landforms, in ravines, and on north- and east-facing slopes.

The typically tall forest canopy is occupied by a broad range of mesophytic tree species. On acidic soils *Liriodendron tulipifera, Betula lenta, Acer rubrum,* and *Tsuga canadensis* are the primary canopy species, with *Rhododendron maximum* and *Leucothoe fontanesiana* dominating the shrub layer. The herb layer may be sparse and contain a relatively few species such as *Galax urceolalta, Mitchella repens, Epigaea repens,* and *Thelypteris noveboracensis.* On less acidic, sometimes circumneutral soils, a large number of mesophytic tree species may be found. *Liriodendron tulipifera, Tilia americana var. heterophylla, Betula lenta, Magnolia accuminata, Prunus serotina, Fraxinus americana, Fagus grandifolia, Quercus rubra, Carya cordiformis,* and *Halesia tetraptera* occur in various mixtures. On the "richest" sites, *Acer saccharum* and *Aesculus flava* are usually present. The shrub and small-tree layer is also diverse and may include *Cornus florida, Carpinus caroliniana, Magnolia tripital, M. fraseri, Ostrya virginiana, Acer spicatum, A. pennsylvnicum, Hydrangea arborescens, Lindera benzoin, Calycanthus floridus,* and *Cornus alternifolia.* The herb layer is typically extremely diverse and would include *Cimicifuga racemosa, Trillium erectum, Caulophylum thalictroides, Impatiens pallida, Laportea canadensis, Adiantum pedatum, Hepatica acutiloba, Asarum canadense, Tiarella cordifolia, Actea pachypoda, Dryopteris intermdeia, Arisaema triphylum, Podophyllum peltatum, Dicentra canadensis, D. cucullaria,* and many other mesic herbs.

Collectively, the variations of the mesophytic cove hardwood type are widespread throughout the Southern Appalachian region. They are not as frequent, however, in the drier, northern part of the region. The richest of the variations is limited in areal extent. The mesophytic cove hardwood type primarily grades into northern hardwood, mesic oak, and white pine/hemlock/hardwood types. Mesophytic cove hardwoods have been much studied because of their botanical significance and their economic importance. The variations in this habitat group are well-documented, but causal factors for the variations are the subject of continuing investigation. Response to disturbance is reasonably well understood for some variants, but non-anthropogenic disturbance regimes are not well established.

Oak Forests

The SAA assessed both mesic and xeric oak types collectively. Descriptions are provided for both these types here.

Mesic Oak Forests

This habitat group occurs from low to high elevations on dry (sub-) mesic sites, frequently on linear or convex landforms on north- and east-facing slopes or at high elevations, and sometimes on concave landforms on southerly and westerly aspects.

At low to moderate elevations, *Quercus rubra* and *Q. alba* share dominance with other oaks *(Q. velutina* or *Q. montana),* hickories *(Carya* spp.*),* and *Acer rubrum,* as well as with some mesophytic species, particularly *Liriodendron tulipifera.* At high elevations *Q. rubra var. borealis* forms pure or nearly pure stands. Accessory tree and shrub species include *Cornus florida, Hammamilis virginiana, Oxydendron arborea, Amelanchier arborea,* and *Halesia tetraptera.* Herb layers vary from sparse to dense, some with ericaceous cover, and some with mesophytic herbs.

Mesic oak forests are common and occur thoughout the Southern Appalachians. Mesic oak forests grade into mesophytic cove hardwoods, white pine/hemlock/hardwoods, and xeric oak forests at low to moderate elevations, and to northern hardwood forests and spruce-fir forests at higher elevations. The forests classified in this category occupy a large area of forest in the Southern Appalachians. For wildlife species utilizing acorns as a food source, this habitat group, along with xeric oak forests, are extremely important. The large oak component in this category (particularly at low to intermediate elevations), as well as the oak component in some mixed mesophytic forests may result from disturbance regimes that differ from those of the present.

Xeric Oak Forests

This habitat group occurs on south- and west-facing slopes, and on broad and narrow convex landforms, over a broad range of elevations.

Dominance of oaks is often nearly complete, but hickories *(Carya glabra* and *C. ovalis),* sourwood *(Oxydendron arborea),* blackgum *(Nyssa sylvatica),* and red maple *(Acer rubrum)* are common associates. At low elevations on broad convex landforms, scarlet oak *(Quercus coccinea)* in mixtures with black oak *(Q. velutina),* southern red oak *(Quercus falcata)* and white oak *(Q. alba)* is common on xeric sites, usualy with shrub layers of mountain laurel *(Kalmia latifolia),* blueberry *(Vaccinium* spp.)

and other ericaceous species. Herbaceous cover is generally sparse or absent. The most xeric sites may include post oak *(Q. stellata)* and blackjack oak *(Q. marilandica).* At intermediate elevations xeric oak forests are dominated by chestnut oak *(Q. montana),* scarlet oak, and mixtures of these two species, again frequently with ericaceous understories and sparse herbaceous cover. At intermediate to high elevations, white oak is found in mixtures with other oaks, hickories, and red maple on fairly dry, exposed sites.

Xeric oak forests are very common throughout the Southern Appalachians and at a broad range of elevations. They are particularly common in the Ridge and Valley section and in intermountain valleys.

Xeric oak forests grade primarily into mesic oak forests, oak-pine communities and pine communities. These forests seem to be rather stable compositionally. Regeneration following disturbance tends to be oak-dominated, but many of the sites on which xeric oak communities can be expected to occur contain a pine component resulting from various disturbances.

Mixed Pine-Hardwood Forests

The SAA assesses mesic and xeric types collectively. A description of both these types is provided here.

Mesic Mixed Pine-Hardwood Forest

This habitat group occurs over a broad range of topographic positions including well-drained creek bottoms, concave land surfaces on all slope directions and on linear slopes on all slope directions. Mesic yellow pine/hardwood communities are restricted to low elevations, but white pine/hardwood mixtures occur at intermediate elevations.

Pine species include loblolly pine *(Pinus taeda)* in the Piedmont-Mountain transition, and shortleaf pine *(P. echinata)* and white pine *(P. strobus)* in the Piedmont-Mountain transition and in the mountains. Hardwoods include white oak *(Quercus alba),* black oak *(Q. velutina),* chestnut oak *(Q. montana),* northern red oak *(Q. rubra),* red maple *(Acer rubrum),* yellow poplar *(Liriodendron tulipifera),* dogwood *(Cornus florida),* hickories *(Carya* spp.*),* and, at low elevations in the Piedmont-Mountain transition, sweetgum *(Liquidambar styraciflua).* Shrubs include *Vaccinium* spp., *Euonymus americana, Vitis* spp. and *Toxicodendron radicans,* and ericaceous species in white pine/hardwood mixtures. Typical herbs include G*oodyeara pubescens, Desmodium nudiflorum,* and *Hexastylis* spp., but coverage is generally sparse.

Mesic mixed pine/hardwood communities grade into mesic oak, southern yellow pine communities, xeric pine/harwood communities, white pine/hemlock communities and occasionally into mixed mesophytic hardwood communities. Commonly, the yellow pine component in these stands originated after the abandonment of agricultural activity, although fire may also have been a factor in some cases. These same disturbance regimes may have also been important in the case of white pine/hardwood mixtures, but the accumulation of the shade tolerant white pine regeneration in long-undisturbed hardwood stands and the ascension of white pine to the canopy after mortality of canopy hardwoods suggests other successional pathways may be operative.

Xeric Mixed Pine-Hardwood Forest

This habitat group occurs at low to intermediate elevations, on both broadly and sharply convex landforms, usually with a southerly or westerly exposure.

The canopy is dominated by a mixture of oaks *(Quercus* spp.*)* and pines *(Pinus* spp.*).* Oaks include scarlet *(Q. coccinea),* black *(Q. velutina),* and chestnut *(Q. montana)* at both low and intermediate elevations, and post *(Q. stellata),* blackjack *(Q. marilandica),* and southern red *(Q. falcata)* at low elevations. Pines include shortleaf *(P. echinata),* Virginia *(P. virginiana),* Pitch *(P. rigida)* and Table Mountain *(P. pungens).* Other canopy species frequently found include sourwood *(Oxydendron arboreum),* red maple *(Acer rubrum),* sassafras *(Sassafras albidum)* and blackgum *(Nyssa sylvatica).* The shrub layer is typically ericaceous, with *Kalmia latifolia, Gaylussacia* spp., and *Vaccinium* spp. among the most common species found. Typical herbs include *Epigea repens, Galax aphylla,* and *Pteridium aquilinum.*

This habitat group is found throughout the Southern Appalachians, frequently on sandstones or associated with granitic domes. It grades into xeric oak communities, mesic oak communities, pine communities and heath balds. Xeric mixed pine hardwood communities most likely resulted from disturbances, e.g. fire, that promoted the regeneration of pines, and either contained a hardwood component at the time of disturbance or have since been invaded by hardwoods. Through time, these communities will increasingly be dominated by hardwoods unless disturbed in a way that is similar to the disturbance from which they originated.

Montane Spruce-Fir Forest

This habitat group occurs at very high elevations, generally above 5500', in all topographic positions.

The forest is dominated by red spruce *(Picea rubens)* and Fraser fir *(Abies fraseri).* Red spruce occurs in forests as low as 4500' in mixtures with northern hardwoods. It may dominate stands in the 5000' to 5500' elevation range. Fraser fir begins to appear around 5500' in mixture with red spruce, and above 6000' may form pure stands. Yellow birch *(Betula lutea)* and mountain maple *(Acer spicatum)* are common associates. Shrubs include *Rhododendron catawbiense, Vaccinium erythrocarpum, V. constablaei, Rubus canadensis,* and *Viburnum alnifolium.* The herb layer may be dense and include *Oxalis montana, Dryopteris campyloptera, Aster divaricatus, Clintonia borealis, Solidago glomerata, Carex pensylvanica, Maianthemum canadense,* and others.

The southern limit is Richland Balsam Mountain in North Carolina and the central Smoky Mountains along the North Carolina-Tennessee border. The montane spruce-fir forest also occurs in Virginia and West Virginia. It grades to northern hardwoods and may be adjacent to heath balds and grassy balds. Large trees of Fraser fir have been eliminated from this forest by the balsam wooly adelgid during the last 30 years. Although fir reproduction is often abundant, the character of the forest has been drastically changed.

Northern Hardwood Forest

This habitat group occurs on high-elevation, concave landforms and north-facing slopes.

Canopy dominance is shared by mixtures of mesophytic tree species including beech *(Fagus grandifolia),* buckeye *(Aesculus flava),* sugar maple *(Acer saccharum),* and yellow birch *(Betula lutea).* Yellow birch is sometimes considered the most characteristic species. Other canopy species may include basswood *(Tilia americana var. heterophylla),* white ash *(Fraxinus americana),* and black cherry *(Prunus serotina).* Common mid-story species include striped maple *(Acer pensylvanicum),* mountain maple *(A. spicatum),* hophornbeam *(Ostrya virginiana),* mountain ash *(Sorbus americana),* and *(Amelanchier arborea).* Shrubs include moosewood *(Viburnum alnifolium), Rhododendron catabiense, Hydrangea arborescens,* and dogwood *(Cornus alternifolia).* The herb layer is well developed and diverse including *Monarda didyma, Claytonia caroliniana, Caulophylum thalyctroides, Viola canadensis, Impatiens pallida, Actea pachypoda, Collinsonia canadensis,* and many others.

This habitat group is common in the high mountain areas of North Carolina, Tennessee, Virginia and West Virginia. The northern hardwood forest grades into the mixed mesophytic hardwood forest, the high-elevation red oak forest, the spruce-fir forest, and is often adjacent to grassy balds and heath balds. The canopy of this forest is sometimes dominated by one or two species. The beech gap variant, located at very high elevations, is an example of single-species dominance.

Bottomland Hardwood Forests

These forest communities occur in river bottoms and floodplains that originate in the piedmont and mountains, and continue into the coastal plains in the southeast United States. This community is not common in the SAA area. The bottomland soils are well-drained loams and silt loams. Tree species occurring in these forests typically include red maple, river birch, water hickory, green ash, sweet gum, sycamore, willow oak, laurel oak, overcup oak, water oak, and elms. Tree species on the adjacent higher elevation second bottoms where flooding is less frequent, include cherrybark oak, swamp chestnut oak, hickories, American beech, and yellow poplar.

The primary disturbance regimes include flooding and natural tree mortality resulting in small gaps in the forest canopy. Infrequent fire could also play a role in these forests during dry years. Because annual flood events have been altered and due to fire suppression, American beech and red maple may become more prominent in this community.

Southern Yellow Pine Forests

Southern yellow pines were assessed as a whole in the SAA. Descriptions are provided below for a collective group called other yellow pine, and for longleaf pine.

Other Yellow Pine

This habitat group occurs on all topographic positions at low to intermediate elevations.

Canopies are dominated by loblolly pine *(Pinus taeda),* shortleaf pine *(P. echinata),* Virginia pine *(P. virginiana),* pitch pine *(P. rigida),* or Table Mountain pine *(P. pungens).* Sometimes mixtures of the above occur. A host of mesic and xeric hardwood species (oaks, hickories, yellow-poplar, sweetgum, dogwood, sourwood, blackgum, etc.) occur as minor components (see mixed pine/hardwood descriptions). The shrub layer may be almost totally ericaceous on xeric sites to totally non-ericaceous on more mesic sites. Herb layers are generally sparse.

This habitat group occurs throughout the Southern Appalachians, but is perhaps most common in the mountain-piedmont transition zone, and represented by shortleaf pine and loblolly pine. Abandonment of agricultural activity may be the most important factor overall in the occurrence of loblolly, Virginia, and shortleaf pine communities, but loblolly pine has been extensively planted. Fire is more closely linked to pitch and Table Mountain pine communities.

Mountain Longleaf Pine Forest

This habitat group occurs on xeric ridge sites and on south- and west-facing slopes at the southern end of the Appalachians in Georgia and Alabama, at elevations up to 1960 feet (600 m).

The canopy is dominated by longleaf pine *(Pinus palustris),* but may also contain other pines *(P. echinata* and *P. taeda),* and oaks *(Quercus stellata, Q. prinus, Q. marilandica, Q. coccinea,* and *Q. falcata).* The shrub layer is ericaceous and includes *Vaccinium* spp. and *Gaylussacia* spp. Typical herbs include *Pteridium aquilinum, Andropogon gyrans, Aster dumosus, Coreopsis major,* and *Eupatorium album.*

This type occurs only in the mountains of Alabama and adjacent areas in Georgia. It grades into xeric oak and xeric oakpine mixtures as well as mesic oak forest on north-facing slopes. Periodic fire is presumed to have played a role in the development of this type, and in the absence of fire, particularly on more mesic sites, species composition is shifting toward hardwood dominance.

Cedar Woodlands (over limestone and dolomite)

This habitat group occurs on level to gently rolling valley topography over limestone or dolomite parent material at low elevations in the western part of the Southern Appalachian region.

The canopy is dominated by eastern red cedar *(Juniperus virginiana)* or by eastern red cedar and a mixture of hardwoods. Hardwoods include hackberry *(Celtis laevigata),* hickory *(Carya glabra),* chestnut oak *(Quercus montana),* black oak *(Q. velutina),* and post oak *(Q. stellata).* The shrub-small tree layer includes redbud *(Cercis canadensis),* winged elm *(Ulmus alata),* dogwood *(Cornus florida),* blue ash *(Fraxinus quadrangulata),* privet *(Forestiera ligusstrina),* sumac *(Rhus aromatica),* buckthorn *(Rhamnus caroliniana),* and coral-berry *(Symphoricarpus orbiculatus).* Typical herbs are *Aristida longispica, Sporobolus* spp., *Erigeron ramosus, Rudbeckia triloba, Arenaria patula, Hypericum* spp., *Euphorbia dentata, Galium virgatum,* and *G. pilosum.*

This habitat group is restricted to the zones of sedimentary rock within the region, i.e. the Appalachian Valley and beyond. It grades into xeric oak and mesic oak communities.

Developed

These are areas of intensive use with much of the land covered by structures or impervious paved surface. Included in this category are cities, towns, and areas occupied by mills, shopping malls, and industrial complexes. The general definition is areas with at least 50 percent impervious surface and less than 25 percent vegetation cover.

Barren (rock outcrops and barren soil)

Land of limited ability to support vegetation. In general this includes areas of thin soil, sand, or rock. These conditions may be natural, such as granite domes, or human caused, such as strip mining. It may also include transition areas from which vegetation has been removed, as in clearcutting or in preparation for commercial development. Vegetation, if present, is sparse and occupies less than 25 percent of the area.

Agricultural Pasture

These are areas with more than 25 percent vegetative cover where the existing vegetation is predominately perennial grasses, grasslike plants, and forbs. It is usually fenced and maintained for livestock grazing. If tree canopy is present, it represents less than 25 percent land cover.

Agricultural Cropland

These are areas with more than 25 percent vegetative cover that are intensively managed for the production of crops that are removed on an annual or periodic basis. The cover class includes land planted in grain, vegetables, or similar crops. It also includes vineyards, orchards, and christmas tree plantations. With the exception of orchards and christmas tree plantations, tree crowns occupy less than 25 percent of the area.

Early Successional Herbaceous-Shrub Habitats

These are non-cultivated areas with a predominant vegetative cover of herbaceous plants and shrubs covering at least 25 percent of the area. The predominant vegetation may be herbaceous, consisting of grasslike plants, shrubs, or a mixture of these. Shrubs are woody plants usually less than 20 feet tall. Mountain balds and rhododendron slicks are examples. Abandoned agricultural fields and areas of forest in regeneration may be classified as herbaceous-shrub. If trees are present, the crowns occupy less than 25 percent of the area.

Water

Areas of permanent surface water, either free-flowing streams or rivers, or nonflowing lakes and reservoirs. Emergent wetlands with less than 25 percent vegetative cover are included in this cover type.

Wetlands

These are areas of significant non-tidal emergent wetland with more than 25 percent vegetative cover. The vegetation is dominated by persistent emergents, emergent mosses and lichens, along with shrubs and trees. If tree canopy is present, it represents less than 25 percent of the land cover.

Status Summaries for the Broad Vegetation Classes

Included are detailed summaries developed during the analysis of status and trends for forest and nonforest ecosystems. Included are tables C-1 to C-17 referenced in Chapter 3.

Table C-1 The current acres by successional class and forest type group for all ownerships in the Southern Appalachian Assessment area based upon FIA, CISC, and LANDSAT data.

	Current Timberland Acreage and Percents					
	Grass/Seedling/Shrub Stage		Sapling/Pole Stage		Mid Successional Stage	
FIA Forest Type Group	Acres	%	Acres	%	Acres	%
Maple–Beech–Birch Forests	7,445	0.4	95,671	1.8	356,503	2.8
Oak–Hickory Forests	814,009	43.0	3,187,729	58.6	8,395,027	66.1
Elm–Ash–Cottonwood Forests	15,937	0.8	22,529	0.4	129,023	1.0
White Pine–Hemlock Forests	67,107	3.5	245,249	4.5	280,491	2.2
Spruce–Fir Forests	0	0	896	0	11,481	0.1
Southern Yellow Pine Forests	572,418	30.3	602,435	11.1	1,879,563	14.8
Longleaf Pine Forests	7,725	0.4	1,060	0	19,385	0.2
Oak–Pine Forests	406,623	21.5	1,281,636	23.6	1,635,265	12.9
Totals	1,891,264	8	5,437,205	22	12,706,738	52

	Late Successional Stage		All Stages Totals	
FIA Forest Type Group	Acres	%	Acres	%
Maple–Beech–Birch Forests	66,154	1.5	525,773	2
Oak–Hickory Forests	3,174,064	70.9	15,570,829	64
Elm–Ash–Cottonwood Forests	19,579	0.4	187,068	1
White Pine–Hemlock Forests	24,840	0.6	617,687	2.5
Spruce–Fir Forests	67,208	1.5	79,585	0.3
Southern Yellow Pine Forests	387,507	8.7	3,441,923	14.0
Longleaf Pine Forests	27,485	0.6	55,655	0.2
Oak–Pine Forests	711,309	15.9	4,034,833	16.5
Totals	4,478,146	18	24,513,353	

(Source: USDA, Forest Service, Southern Research Station, Forest Inventory and Analysis Unit USDA Forest Service, Southern Region, Continuous Inventory of Stand Conditions data)

SAA derived from remotely sensed data for National Park lands

able C-2 The distribution of current acres for forest and non–forest land cover types for each of the 7 states in the Southern Appalachian Assessment area based pon LANDSAT data.

nd Classes Derived from emotely Sensed Data	State Acres and Percents							
	Alabama	%	Georgia	%	North Carolina	%	South Carolina	%
ch State's Forest Acres	1,825,200		3,981,975		4,934,367		821,152	
Percent of Total SAA		7		15		19		3
Percent of State's Land Base		63		71		82		65
ch State's Deciduous Forest Acres	1,077,042		2,327,864		3,875,995		269,801	
Percent of Total SAA		6		12		20		1
Percent of State's Land Base		59		58		79		33
ch State's Evergreen Forest Acres	291,551		460,192		174,682		82,294	
Percent of Total SAA		17		26		10		5
Percent of State's Land Base		16		12		4		10
ch State's Mixed Forest Acres	456,607		1,193,919		883,690		469,057	
Percent of Total SAA		9		23		17		9
Percent of State's Land Base		25		30		18		57
rest Total % within State		100		100		100		100
btotal Nonforest	1,052,445	37	1,612,457	29	1,117,437	18	447,303	35
determinate	3863	.13	0	0	0	0	0	0
tals	2,881,508	8	5,594,432	15	6,051,804	16	1,268,455	3

nd Classes Derived from emotely Sensed Data	Tennessee	%	Virginia	%	West Virginia	%	Total All States Within SAA	%
ch State's Forest Acres	5,249,755		8,351,197		1,008,777		26,172,423	
Percent of Total SAA		20		32		4		100
Percent of State's Land Base		61		71		82		
ch State's Deciduous Forest Acres	3,606,213		7,263,017		866,287		19,286,219	
Percent of Total SAA		19		38		4		100
Percent of State's Land Base		69		87		86		
ch State's Evergreen Forest Acre	562,261		152,727		19,486		1,743,193	
Percent of Total SAA		32		9		1		100
Percent of State's Land Base		11		2		2		
ch State's Mixed Forest Acres	1,081,281		935,453		123,004		5,143,011	
Percent of Total SAA		21		18		2		100
Percent of State's Land Base		21		11		12		
rest Total % within State		100		100		100		
btotal Nonforest	3,341,613	39	3,437,306	29	224,677	18	11,233,238	30
determinate	540	.01	9,336	.08	0	0	13,739	.04
tals	8,591,908	23	11,797,839	32	1,233,454	3	37,419,400	

urce: Derived from remotely sensed data for the Southern Appalachian Assessment area)

able C–3 The distribution of the current acres for forest and non–forest habitat groups according to ecological section group units for the Southern Appalachian ssessment area based on FIA and LANDSAT data.

	Section Group 1[1]		Section Group 2		Section Group 3		Section Group 4		Section Group 5		Section Group 6	
andcover Classes	Acres	%	Acres	%	Acres	%	Acres	%	Acres	%	Acres	%
orest Cover Types	5,661,585	67	2,076,867	79	2,508,960	58	4,746,861	68	2,214,576	49	8,963,572	84
Deciduous Types	4,511,717	53.6	1,599,395	61	1,209,319	28.1	2,508,241	36.1	1,509,898	33.4	6,404,472	60.3
Northern Hardwood	241,828	2.8	38,004	1.5	0	0	0	0	48,619	1.1	306,134	2.9
Mixed Mesophytic Hardwood	559,904	6.7	246,470	9.4	61,675	1.4	724,882	10.4	149,329	3.3	1,474,309	13.9
Oak Forests	3,650,430	43.4	1,277,960	48.7	1,090,806	25.4	1,631,360	23.5	1,245,968	27.6	4,554,861	42.9
Bottomland Hardwood	59,555	0.7	36,961	1.4	56,838	1.3	151,999	2.2	65,982	1.4	69,168	0.6
vergreen Types	448,964	5.3	218,279	8.3	832,975	19.4	1,322,001	19.0	332,629	7.4	1,264,760	11.9
White Pine-Hemlock	107,392	1.3	17,621	0.7	0	0	106,421	1.5	0	0	474,032	4.5
Montane Spruce–Fir	1,029	0	0	0	0	0	0	0	0	0	89,072	0.8
Southern Yellow Pine	340,543	4.0	200,658	7.6	832,975	19.4	1,215,580	17.5	332,629	7.4	701,656	6.6
ixed Types	700,904	8.3	259,193	9.9	466,666	10.9	916,619	13.2	372,049	8.2	1,294,340	12.2
White Pine–Hemlock–Hardwood	204,734	2.4	24,812	1.0	0	0	113,661	1.6	8,669	0.2	520,066	4.9
Mixed Pine–Hardwood	496,170	5.9	234,381	8.9	466,666	10.9	802,958	11.6	363,380	8.0	774,274	7.3
onforest Cover Types	2,741,692	33	545,270	21	1,783,251	42	2,195,266	32	2,305,238	51	1,662,517	16
Grass-Shrub, Old Fields	169,834	2.0	99,921	3.8	545,795	12.7	425,767	6.1	147,777	3.3	139,256	0.2
Agricultural Cropland	426,394	5.1	54,467	2.1	109,506	2.4	378,905	5.5	169,117	3.7	132,833	1.3
Agricultural Pasture	1,802,133	21.4	327,242	12.5	856,911	20.0	955,582	13.8	1,482,256	32.8	1,098,310	10.3
Developed	282,164	3.4	33,692	1.3	145,845	3.4	230,238	3.3	280,173	6.2	197,666	1.9
Barren	16,331	0.2	14,526	0.6	15,294	0.4	39,796	0.6	15,700	0.3	10,882	0.1
Water	41,671	0.5	10,665	0.4	77,774	1.8	145,836	2.1	201,154	4.4	79,147	0.7
Wetlands	3,165	0.0	4,767	0.2	32,126	0.7	19,142	0.3	9,061	0.2	4,403	0.0
Nonclassified	8,387	0.1	1,060	0.0	2,902	0.1	1,026	0.0	127	0.0	243	0.0
rand Totals	8,411,664		2,623,197		4,295,113		6,943,153		4,519,941		10,626,332	

ection Groups:

= Northern Ridge and Valley, Allegheny Mountains, Northern Cumberland Mountains

= Blue Ridge Mountains

= Northern Cumberland Plateau, Southern Cumberland Mountains

= Central Ridge and Valley

= Southern Cumberland Plateau, Southern Ridge and Valley

= Southern Appalachian Piedmont

able C–4 The distribution of current acres for 4 successional classes by forest habitats on National Forest System land in the Southern Appalachian Assessment area ased on Continuous Inventory of Stand Conditions (CISC) data.

	Current National Forest System Acreage and Percents									
	Grass/Seedling/Shrub Stage		Sapling/Pole Stage		Mid Successional Stage		Late Successional Stage		All Stages Totals	
A Forest Type Group	Acres	%	Acres	%	Acres	%	Acres	%	Acres	%
laple–Beech–Birch Forests	986	0.7	7,076	2.1	67,382	3.7	15,869	0.8	91,313	2.2
ak–Hickory Forests	58,285	43.2	156,281	46.9	1,121,616	61.5	1,209,026	63.9	2,545,208	60.8
m–Ash–Cottonwood Forests	58	0	84	0	419	0	289	0	850	0
/hite Pine–Hemlock Forests	23,581	17.5	65,462	19.6	56,932	3.1	9,469	0.5	155,444	3.7
ɔruce–Fir Forests	0	0	448	0.1	6,674	0.4	4,578	0.2	11,700	0.3
ɔuthern Yellow Pine Forests	29,576	21.9	44,655	13.4	108,765	6.0	212,335	11.2	395,331	9.4
ɔngleaf Pine Forests	7,725	5.7	530	0.2	8,943	0.5	27,485	1.5	44,683	1.1
ak–Pine Forests	14,802	11.0	58,762	17.6	453,699	24.9	412,281	21.8	939,544	22.5
ɔtals	135,013	3	333,298	8	1,824,430	44	1,891,332	45	4,184,073	

ɔurce: USDA, Forest Service, Southern Region, Continuous Inventory of Stand Conditions data)

able C–5 The distribution of current acres for 4 successional classes by forest habitats on other public land in the Southern Appalachian Assessment area based on A and LANDSAT data.

	Current Timberland Acreage and Percents									
	Grass/Seedling/Shrub Stage		Sapling/Pole Stage		Mid Successional Stage		Late Successional Stage		All Stages Totals	
A Forest Type Group	Acres	%	Acres	%	Acres	%	Acres	%	Acres	%
laple–Beech–Birch Forests	0	0	11,062	5	5,065	1	30,922	3.4	47,079	2.8
ak–Hickory Forests	34,286	54.6	133,771	60.3	346,481	69	664,159	73	1,178,697	69.5
m–Ash–Cottonwood Forests	0	0	0	0	3,156	0.6	148	0	3,304	0.2
/hite Pine–Hemlock Forests	0	0	5,824	2.6	7041	1.4	1,084	0.1	13,949	0.8
ɔruce–Fir Forests	0	0	0	0	0	0	62,684	6.9	62,684	3.7
ɔuthern Yellow Pine Forests	22,404	35.7	5,221	2.4	64,803	12.9	54,547	6.0	146,975	8.7
ɔngleaf Pine Forests	0	0	0	0	0	0	0	0	0	0
ak–Pine Forests	6,112	9.7	65,866	29.7	75,887	15.1	95,817	10.5	243,682	14.4
ɔtals	62,802	4	221,744	13	502,433	30	909,361	54	1,696,340	

ɔurce: USDA, Forest Service, Southern Research Station, Forest Inventory and Analysis Unit; Southern Appalachian Assessment remote sensing imagery for Natonal Park lands)

able C–6 The distribution of current acres for 4 successional classes by forest habitats on private forest industrial and leased lands in the Southern Appalachian ssessment area based on FIA data.

	Current Timberland Acreage and Percents									
	Grass/Seedling/Shrub Stage		Sapling/Pole Stage		Mid Successional Stage		Late Successional Stage		All Stages Totals	
A Forest Type Group	Acres	%	Acres	%	Acres	%	Acres	%	Acres	%
laple–Beech–Birch Forests	1,256	0.4	0	0	556	0.1	0	0	1,812	0.1
ak–Hickory Forests	42,195	12.8	193,249	28.7	239,134	51.7	54,892	97.8	529,470	34.8
m–Ash–Cottonwood Forests	0	0	133,771	19.9	7,367	1.6	0	0	141,138	9.3
/hite Pine–Hemlock Forests	1,966	0.6	9,613	1.4	4,773	1.0	0	0	16,352	1.1
pruce–Fir Forests	0	0	5,824	0.9	0	0	0	0	5,824	0.4
outhern Yellow Pine Forests	204,307	61.9	175,674	26.1	168,929	36.5	1,256	2.2	550,166	36.1
ongleaf Pine Forests	0	0	5,221	0.8	0	0	0	0	5,221	0.3
ak–Pine Forests	80,495	24.4	149,724	22.2	41,765	9.0	0	0	271,984	17.9
otals	330,219	22	673,076	44	462,524	30	56,148	4	1,521,967	

ource: USDA, Forest Service, Southern Research Station, Forest Inventory and Analysis Unit)

able C–7 The distribution of current acres for 4 successional classes by forest habitats on private forest non–industrial lands in the Southern Appalachian Assessment rea based on FIA data.

	Current Timberland Acreage and Percents									
	Grass/Seedling/Shrub Stage		Sapling/Pole Stage		Mid Successional Stage		Late Successional Stage		All Stages Totals	
A Forest Type Group	Acres	%	Acres	%	Acres	%	Acres	%	Acres	%
laple–Beech–Birch Forests	5,203	0.4	77,533	1.8	283,500	2.9	19,334	1.2	385,570	2.2
ak–Hickory Forests	679,243	49.8	2,704,428	62.1	6,687,796	67.4	1,219,651	75.5	11,291,118	65.5
m–Ash–Cottonwood Forests	15,879	1.2	22,361	0.5	118,081	1.2	19,155	1.2	175,476	1.0
/hite Pine–Hemlock Forests	41,560	3.0	164,350	3.8	211,745	2.1	14,243	0.9	431,898	2.5
pruce–Fir Forests	0	0	0	0	4,807	0.0	0	0	4,807	0
outhern Yellow Pine Forests	316,131	23.1	376,885	8.7	1,537,066	15.5	131,683	8.1	2,361,765	13.7
ongleaf Pine Forests	0	0	0	0	10,442	0.1	0	0	10,442	0.1
ak–Pine Forests	305,214	22.4	1,007,284	23.1	1,063,914	10.7	211,914	13.1	2,588,326	15
otals	1,363,230	8	4,352,841	25	9,917,351	57	1,615,980	9	17,249,402	

ource: USDA, Forest Service, Southern Research Station, Forest Inventory and Analysis Unit)

able C-8 The distribution of the current acres by the non–forest land cover types by ownerships within the Southern Appalachian Assessment area based upon ANDSAT data.

	Ownership Acres and Percents							
ınd Classes Derived from Remotely Sensed Data	National Forests	%	National Parks	%	State	%	Department of Defense and Department of Energy	%
ıbtotal Forest	4,468,835	98	820,127	96	531,144	92	83,426	78
ıbtotal Non–Forest	82,896	2	20,560	2	43,050	7	22,762	21
Grass, Forb, Cedar Woodlands, and Early Successional with < 25% woody vegetation	29,401	35	3,165	2	11,831	27	3,542	16
Rock Outcrops, Bare Soil	2,165	3	344	2	1,003	2	875	4
Croplands	3,963	5	682	3	2,811	7	675	3
Pastures	31,329	38	8,709	42	14,538	34	5,607	25
Wetlands	1,300	2	217	1	294	1	881	4
Developed	3,084	4	1,668	8	2,699	6	8,954	39
Water	11,654	14	5,775	28	9,874	23	2,228	10
on–Forest Percent Total		100		100		100		100
determinate	1,906	0	0	0	428	0	506	0
ıtals	4,553,637	12	840,687	2	574,622	2	106,694	0

ınd Classes Derived from Remotely Sensed Data	Cherokee Indian Reservation	%	Private	%	Total All Owners Within SAA	%
ıbtotal Forest	42,033	93	20,226,860	65	26,172,425	70
ıbtotal Non–Forest	3,404	7	11,060,564	35	11,233,236	30
Grass, Forb, Cedar Woodlands, and Early Successional with < 25% woody vegetation	283	8	1,480,129	13	1,528,351	14
Rock Outcrops, Bare Soil	20	1	108,122	1	112,529	1
Croplands	132	4	1,262,959	11	1,271,222	11
Pastures	1,936	57	6,460,314	58	6,522,433	58
Wetlands	0	0	69,972	1	72,664	1
Developed	1,025	30	1,152,369	10	1,169,799	10
Water	8	0	526,699	5	556,238	5
on–Forest Percent Total		100		100		100
determinate	0	0	10899	0	1,3739	0
ıtals	45,437	0	31,298,323	84	37,419,400	

ource: Southern Appalachian Assessment Satellite Remote Sensing Imagery)

able C–9 The 10 year trends (1982 to 1992) for non-forest land cover types shown by states within the Southern Appalachian Assessment area based on NRCSI ata.

	Thousands of Acres and Percent Change											
	Alabama			Georgia			North Carolina			South Carolina		
andCover Use	1982 Acres	1992 Acres	% Change	1982 Acres	1992 Acres	% Change	1982 Acres	1992 Acres	% Change	1982 Acres	1992 Acres	% Change
ropland–Row Crops	250.5	211.2	–16	207.7	155.5	–25	268.4	175.1	–35	89.2	55.4	–38
ropland, Non–Cultivated	5	13.1	162	39.8	43	8	146.9	151.7	3	26.5	29.8	12
orest Land	1723.2	1704.5	–1	3247.7	3154.4	–3	3073.1	2972.7	–3	666.3	664.4	–0
arge Urban & Built Up	68.6	81.3	19	245.8	345.5	41	289.2	401.4	39	124.2	160	29
nall Urban & Built Up	15	23.4	56	37.4	56.4	51	40.9	74.9	83	10.5	15.1	44
rass & Pasture Land	332.1	347.6	5	660	666.9	1	548.1	547.3	–0	129.8	117.9	–9
rass, Forbs, Legumes Mix	21.5	15.6	–27	55.2	40.9	–26	100.3	82.9	–17	20.6	15.5	–25
egume Pasture Land	2.3	1.2	–48	0	0	0	0	0	0	1.2	0	100
ther NRI land Classes	463.5	483.8	4	1098.6	1129.6	3	1582.7	1643.6	4	201.7	211.9	5
otal	2881.7	2881.7		5592.2	5592.2		6049.6	6049.6		1270	1270	

	Thousands of Acres and Percent Change								
	Tennessee			Virginia			All Combined States		
and Cover Use	1982 Acres	1992 Acres	% Change	1982 Acres	1992 Acres	% Change	1982 Acres	1992 Acres	% Change
ropland–Row Crops	348.7	262.9	–25	364.4	282.9	–22	1528.9	1143	–25
ropland, Non–Cultivated	282.6	344	22	610.6	632.9	4	1111.4	1214.5	9
orest Land	4071.4	4012.9	–1	5672.2	5660.3	–0	18453.9	18169.2	–2
arge Urban & Built Up	426.2	569.9	34	0	0	0	1154	1558.1	35
nall Urban & Built Up	52.3	69.1	32	0	0	0	156.1	238.9	53
rass & Pasture Land	1158.4	1071.8	–7	0	0	0	2828.4	2751.5	–3
rass, Forbs, Legumes Mix	597.1	611.1	2	0	0	0	794.7	766	–4
egume Pasture Land	3.6	8.6	139	0	0	0	7.1	9.8	38
ther NRI land Classes	1656	1646	–1	5151.3	5222.4	1	10153.8	10337.3	0
otal	8596.3	8596.3		11798.5	11798.5		36188.3	36188.3	

ource: Natural Resource Conservation Service, National Resource Inventory)

able C–10 The 20 year trends (mid 1970s to 1995) for forest habitats according to ownerships in the Southern Appalachian Assessment area based on FIA data.

	Acres of Timberland											
	National Forests			Other Public			Private Industry & Leased			Non–Industrial Private		
A Forest Type Group	Mid '70s Acres	1995 Acres	% Change	Mid '70s Acres	1995 Acres	% Change	Mid '70s Acres	1995 Acres	% Change	Mid '70s Acres	1995 Acres	% Change
aple–Beech–Birch Forests	114,738	148,643	30	10,200	16,127	58	16,897	1,812	–89	381,598	385,570	1
ak–Hickory Forests	2,720,713	2,694,884	–1	536,053	585,332	9	688,110	529,470	–23	11,339,109	11,291,118	–0
m–Ash–Cottonwood Forests	0	0	0	4,977	3,156	–37	9,247	13,191	43	191,238	175,476	–8
hite Pine–Hemlock Forests	117,140	137,780	18	2,114	12,899	510	12,506	16,352	31	300,433	431,898	44
ruce–Fir Forests	4,811	8,323	73	0	0	0	0	5,221	+100	7,903	4,807	–39
uthern Yellow Pine Forests	367,352	387,132	5	105,357	113,914	8	569,062	550,166	–3	3,035,577	2,361,765	–22
ngleaf Pine Forests	5,868	30,474	119	0	0	0	0	65,866	0	27,253	160,166	0
ak–Pine Forests	505,704	616,064	22	157,099	150,110	–4	238,482	271,984	14	2,525,278	2,588,326	2
tals	3,836,326	4,023,300	5	815,800	881,538	8	1,534,304	1,454,062	–5	17,808,389	17399126	–2

urce: USDA, Forest Service, Southern Research Station, Forest Inventory and Analysis Unit)

able C–11 The 20 year trends (mid 1970s to 1995) for forest successional classes by forest habitats for the total Southern Appalachian Assessment area based on A data.

	Acres of Timberland											
	Grass/Seedling/Shrub Stage			Sapling/Pole Stage			Mid Successional Stage			Late Successional Stage		
A Forest Type Group	Mid '70s Acres	1995 Acres	% Change	Mid '70s Acres	1995 Acres	% Change	Mid '70s Acres	1995 Acres	% Change	Mid '70s Acres	1995 Acres	% Change
aple–Beech–Birch Forests	5,179	10,547	104	124,197	107,060	–14	319,321	391,995	23	60,164	42,550	–29
ak–Hickory Forests	716,729	892,467	25	4,924,449	3,275,932	-33	8,075,223	8,722,118	8	1,567,584	2,210,287	41
m–Ash–Cottonwood Forests	27,020	15,879	-41	31,742	22,361	–29	118,636	128,604	8	28,064	19,155	–32
hite Pine–Hemlock Forests	90,497	48,392	–47	158,816	216,911	37	168,914	305,517	81	13,966	28,109	101
ruce–Fir Forests	0	0	0	4,811	0	–100	0	13,130	+100	7,903	0	–100
uthern Yellow Pine Forests	507,395	581,869	15	789,173	591,723	–25	2,553,439	1,951,408	–24	227,341	287,977	27
ngleaf Pine Forests	5,390	0	–100	5,579	0	–100	22,152	35,865	62	0	5,051	+100
ak–Pine Forests	226,748	434,841	92	1,685,869	1,423,344	-16	1,378,900	1,460,806	6	135,046	307,493	128
tals	1,578,958	1,983,995	26	7,724,636	5,637,331	–27	12,636,585	13,009,443	3	2,040,068	2,900,622	42

e above type groups were collapsed into 3 broad forest type groups , then the percentage change was computed for those groups to compensate for limits in data interpretation.

rest Group Type	% Change	% Change	% Change	% Change
ardwood	23	–33	9	37
ne	4	–16	–16	29
ixed	92	–16	6	144

urce: USDA, Forest Service, Southern Research Station, Forest Inventory and Analysis Unit)

able C–12 The 20 year trends (mid 1970s to 1995) for forest successional classes by forest habitats for National Forest System lands in the Southern Appalachian ssessment area based on FIA data.

	Acres of Timberland											
	Grass/Seedling/Shrub Stage			Sapling/Pole Stage			Mid Successional Stage			Late Successional Stage		
A Forest Type Group	Mid '70s Acres	1995 Acres	% Change	Mid '70s Acres	1995 Acres	% Change	Mid '70s Acres	1995 Acres	% Change	Mid '70s Acres	1995 Acres	% Change
laple–Beech–Birch Forests	0	4,088	100+	14,397	18,465	28	83,832	102,874	23	16,509	23,216	41
ak–Hickory Forests	156,168	136,743	–12	304,860	244,484	–20	1,608,001	1,448,707	–10	651,684	864,950	33
m–Ash–Cottonwood Forests	0	0	0	0	0	0	0	0	0	0	0	0
/hite Pine–Hemlock Forests	18,882	4,866	–74	28,499	37,124	30	64,949	81,958	26	4,810	13,832	188
pruce–Fir Forests	0	0	0	4,811	0	–100	0	8,323	100	0	0	0
outhern Yellow Pine Forests	13,740	39,027	184	24,441	33,943	39	218,255	180,610	–17	110,916	133,552	20
ongleaf Pine Forests	0	0	0	0	0	0	5,868	25,423	333	0	5,051	100
ak–Pine Forests	48,509	43,020	–11	102,005	200,470	97	292,030	279,240	–4	63,160	93,334	48
otals	237,299	227,744	–4	479,013	534,486	12	2,272,935	2,127,135	–6	847,079	1,133,935	34

ne above type groups were collapsed into 3 broad forest type groups , then the percentage change was computed for those groups to compensate for limits in data interpretation.

orest Group Type	% Change	% Change	% Change	% Change
ardwood	–10	–18	–8	33
ne	35	23	3	32
lixed	–11	97	–4	48

ource: USDA, Forest Service, Southern Research Station, Forest Inventory and Analysis Unit)

able C-13 The 20 year trends (mid 1970s to 1995) for forest successional classes by forest habitats for other public lands in the Southern Appalachian Assessment rea based on FIA data.

	Acres of Timberland											
	Grass/Seedling/Shrub Stage			Sapling/Pole Stage			Mid Successional Stage			Late Successional Stage		
A Forest Type Group	Mid '70s Acres	1995 Acres	% Change	Mid '70s Acres	1995 Acres	% Change	Mid '70s Acres	1995 Acres	% Change	Mid '70s Acres	1995 Acres	% Change
laple–Beech–Birch Forests	0	0	0	0	11,062	100	5,837	5065	–13	4,363	0	–100
ak–Hickory Forests	14,572	34,286	135	163,242	133,771	–18	304,384	346481	14	53,855	70,794	31
m–Ash–Cottonwood Forests	0	0	0	1,823	0	–100	3,154	3156	0	0	0	0
/hite Pine–Hemlock Forests	62	0	–100	2,052	5,824	184	0	7041	100	0	34	100
pruce–Fir Forests	0	0	0	0	0	0	0	0	0	0	0	0
outhern Yellow Pine Forests	403	22,404	5459	12,402	5,221	–58	72,941	64803	–11	19,611	21,486	10
ongleaf Pine Forests	0	0	0	0	0	0	0	0	0	0	0	0
ak–Pine Forests	6,987	6,112	–13	86,764	65,866	–24	63,348	75887	20	0	2,245	100
otals	22,024	62,802	185	266,283	221,744	–17	449,664	502433	12	77,829	94,559	21

ne above type groups were collapsed into 3 broad forest type groups , then the percentage change was computed for those groups to compensate for limits in data interpretation.

orest Group Type	% Change	% Change	% Change	% Change
ardwood	135	–12	13	22
ne	+200	–24	–2	10
lixed	–13	–24	20	100

ource: USDA, Forest Service, Southern Research Station, Forest Inventory and Analysis Unit)

able C–14 The 20 year trends (mid 1970s to 1995) for forest successional classes by forest habitats for private forest industrial and leased lands in the Southern ppalachian Assessment area based on FIA data.

	Acres of Timberland											
	Grass/Seedling/Shrub Stage			Sapling/Pole Stage			Mid Successional Stage			Late Successional Stage		
A Forest Type Group	Mid '70s Acres	1995 Acres	% Change	Mid '70s Acres	1995 Acres	% Change	Mid '70s Acres	1995 Acres	% Change	Mid '70s Acres	1995 Acres	% Change
aple–Beech–Birch Forests	0	1,256	+100	0	0	0	0	556	+100	2,325	0	–100
ak–Hickory Forests	69,683	42,195	–39	292,191	193,249	–34	283,077	239,134	–16	43,159	54,892	27
m–Ash–Cottonwood Forests	3,425	0	–100	0	5,824	+100	0	7,367	+100	5,822	0	–100
hite Pine–Hemlock Forests	0	1,966	+100	5,402	9,613	78	7,104	4,773	–33	0	0	0
ruce–Fir Forests	0	0	0	0	5,221	+100	0	0	0	0	0	0
uthern Yellow Pine Forests	197,136	204,307	4	133,913	175,674	31	229,715	168,929	–26	8,298	1,256	–85
ngleaf Pine Forests	0	0	0	0	65,866	+100	0	0	0	0	0	0
ak–Pine Forests	27,008	80,495	198	118,817	149,724	26	92,657	41,765	–55	0	0	0
tals	297,252	330,219	11	550,323	605,171	10	612,553	462,524	–24	59,604	56,148	–6

e above type groups were collapsed into 3 broad forest type groups , then the percentage change was computed for those groups to compensate for limits in data interpretation.

rest Group Type	% Change	% Change	% Change	% Change
ardwood	–41	–32	–13	7
ne	5	84	–27	–85
ixed	196	26	–55	0

urce: USDA, Forest Service, Southern Research Station, Forest Inventory and Analysis Unit)

able C–15 The 20 year trends (mid 1970s to 1995) for forest successional classes by forest habitats for non–industrial forest lands in the Southern Appalachian ssessment area based on FIA data.

	Acres of Timberland											
	Grass/Seedling/Shrub Stage			Sapling/Pole Stage			Mid Successional Stage			Late Successional Stage		
A Forest Type Group	Mid '70s Acres	1995 Acres	% Change	Mid '70s Acres	1995 Acres	% Change	Mid '70s Acres	1995 Acres	% Change	Mid '70s Acres	1995 Acres	% Change
laple–Beech–Birch Forests	5,179	5,203	0	109,800	77,533	–29	229,652	283,500	23	36,967	19,334	–48
ak–Hickory Forests	476,306	679,243	43	4,164,156	2,704,428	–35	5,879,761	6,687,796	14	818,886	1,219,651	49
m–Ash–Cottonwood Forests	23,595	15,879	–33	29,919	22,361	–25	115,482	118,081	2	22,242	19,155	–14
/hite Pine–Hemlock Forests	71,553	41,560	–42	122,863	164,350	34	96,861	211,745	119	9,156	14,243	56
pruce–Fir Forests	0	0	0	0	0	0	0	4,807	+100	7,903	0	–100
outhern Yellow Pine Forests	296,116	316,131	7	618,417	376,885	–39	2,032,528	1,537,066	–24	88,516	131,683	49
ongleaf Pine Forests	5,390	0	–100	5,579	149,724	+100	16,284	10,442	35	0	0	0
ak–Pine Forests	144,244	305,214	112	1,378,283	1,007,284	–27	930,865	1,063,914	14	71,886	211,914	195
otals	1,022,383	1,363,230	33	6,429,017	4,502,565	–30	9,301,433	9,917,351	7	1,055,556	1,615,980	53

he above type groups were collapsed into 3 broad forest type groups , then the percentage change was computed for those groups to compensate for limits in data interpretation.

orest Group Type	% Change	% Change	% Change	% Change
ardwood	39	–35	14	43
ne	–4	–7	–18	38
lixed	112	–27	14	195

ource: USDA, Forest Service, Southern Research Station, Forest Inventory and Analysis Unit)

able C–16 An acreage summary by old growth forest type groups for National Forest land inventoried for analysis for meeting revised forest plan old growth forest oals.

ode	Old Growth Forest Type Group	Acres	%	# of Areas	%	Avg. Size	Min. Size	Max. Size
	Northern Hardwood Forests	22478	2.05	174	1.33	129	3	3284
	Conifer–Northern Hardwood Forests	19218	1.75	427	3.27	45	1	1059
	Mixed Mesophytic forests	88988	8.10	1350	10.34	66	1	3233
0	Hardwood (Elm–Ash–Maple) Wetland Forests	0	0	0	0			
3	River Floodplain Hardwood forests	633	.06	6	.05	106	12	415
1	Dry–Mesic Oak Forests	520910	47.42	6311	48.35	83	1	6814
2	Dry and Xeric Oak Forests	76135	6.93	1309	10.03	58	2	1118
4	Xeric Pine & Pine–Oak Forests	54873	6	908	6.96	60	1	1775
5	Dry and Dry–Mesic Oak–Pine Forests	87832	8	1544	11.83	57	3	2112
6	Upland Longleaf Pine Forests	1913	.17	70	.54	27	1	155
3	Eastern Riverfront Forests	99	.01	3	.02	33	5	52
1	Montane and Allied Spruce–Fir forests	2660	.24	20	.15	133	7	650
7	Rocky, Thin–Soiled, Excessively Drained Cedar Woodlands	0	0	0	0			
ild.	Wilderness Areas, but Not Classed with Old Growth Group	178081	16.21	151	1.16	1179	8	12885
ocls.	Age 100 or Greater, but Not Classed with Old Growth Group	44671	4.07	781	5.98	57	1	2526
	Totals	1098491		13054		84		

ource: R8 Continuous Inventory of Stand Conditions GIS data)

able C–17 An acreage summary of suitable and unsuitable lands for timber production by old growth forest type groups for National Forest land inventories for nalysis meeting revised forest plan old grwoth forest goals.

ode	Old Growth Forest Type Group	Unsuitable Acres	Suitable Acres	Unsuitable # Areas	Suitable # Areas	Unsuitable Avg. Size	Suitable Avg. Size	Unsuitable Min. Size	Suitable Min. Size	Unsuitable Max. Size	Suitable Max. Size
	Northern Hardwood Forests	19,363	3,115	126	49	155	64	3	4	3,284	1,088
	Conifer–Northern Hardwood Forests	10,579	8,639	186	242	57	36	1	1	1,059	267
	Mixed Mesophytic Forests	54,858	34,130	601	749	91	46	1	1	3,233	785
0	Hardwood (Elm–Ash–Maple) Wetland Forests	0	0	0	0						
3	River Floodplain Hardwood Forests	24	609	2	4	12	162	12	18	12	415
1	Dry–Mesic Oak Forests	253,408	267,502	2,443	3,868	104	69	1	1	6,814	2,114
2	Dry and Xeric Oak Forests	41,697	34,438	614	695	86	60	2	3	1,118	628
4	Xeric Pine & Pine–Oak Forests	36,995	17,878	490	418	76	43	1	1	1,775	442
5	Dry and Dry–Mesic Oak–Pine Forests	47,668	40,164	631	913	76	44	3	6	2,112	427
6	Upland Longleaf Pine Forests	1,006	907	41	29	25	31	1	6	155	136
3	Eastern Riverfront Forests	5	94	1	2		47	5	52	6	52
1	Montane and Allied Spruce–Fir Forests	2,593	67	18	2	144	34	16	7	650	40
7	Rocky, Thin–Soiled, Excessively Drained Cedar Woodlands	0	0	0	0						
Vild.	Wilderness Areas, but not classed with Old Growth Group	178,081	0	151	0	1,179		8		12,885	
ocls.	Age 100 or Greater, but not classed with Old Growth Group	24,448	20,223	399	382	61	53	1	1	2,562	558
	Totals	670,725	427,766	5,701	7,353	118	58				

ource: R8 Continuous Inventory of Stand Conditions GIS data)

Rare Community Descriptions

Beaver Pond and Wetland Complex

Found on gently sloping floodplains, valley bottoms, and in headwaters at moderately high elevations, typically on low-gradient streams. Vegetation varies widely by location, water depth, age of impoundment, and disturbance history. Typically, however, it is a mosaic of herbaceous and shrub wetlands with areas of open water which grades into the surrounding vegetation. They are distinguished from other wetland types by having semi-permanent to permanent flooding caused by impoundment by beavers.

Beech Gap Forest

A broad-leafed, deciduous forest with canopy dominated by American beech. They generally occur on steep, upper slopes on the north and northeast side of gaps above 4,500 feet. They can occur on dry-mesic, exposed, south-facing slopes above 4,500 feet but the trees there are stunted and gnarled, the understory and shrub strata sparse, and the herbaceous stratum dense. Strong winds and ice storms periodically damage these forests, creating canopy gaps and contributing to their stunted appearance. They often occur as small patches surrounded by other forest types, montane grasslands, and/or shrublands. It is mostly found in the mountains of North Carolina, Tennessee, and Virginia. Beech gap canopies also include buckeye and yellow birch with a sub-canopy that might include mountain or striped maples, serviceberry, and mountain ash. Typically, there is little shrub development (2 to 10 percent). Herbaceous cover is moderately to very dense (40 to 100 percent cover) and dominated by several species of sedges or by large herbs and patches of ferns, with lesser amounts of sedge. There are significant differences in the physiognomy and species composition within beech gap forests due to topographic position, aspect and elevation. Some canopy trees may be quite old. Beech-nuts may be produced, but reproduction appears to be almost entirely from sprouts. Small canopy gaps are commonly invaded by blackberries.

Boulderfields

Characterized by a somewhat variable canopy, typically dominated by yellow birch, occurring over angular rocks up to 3 feet in diameter covered by thin soil, lichens, mosses or vines. In some cases, the rocks are totally covered by moss. It occurs on steep (20 to 80 percent), north-facing, middle to upper concave slopes, or in saddles between ridges, at elevations between 3,500 and 5,300 feet. Shallow sandy or clay loam with an acidity of pH 4.5 to 5.2, may accumulate on and among the boulders. Some areas may develop a fairly deep humus layer. Seepage above and below the rock surface is common. In addition to yellow birch, Fraser fir, basswood, and other species comprise the canopy. Minor components include basswood, buckeye, black birch, red spruce and red oak. The shrub layer is generally diverse and dense, the herb layer generally sparse. Boulderfields sometimes contain seepage areas which produce wet microhabitats. Boulderfields occur on steep, rocky, north-facing, middle to upper concave slopes or in saddles at high elevations. They are found from Virginia to northern Georgia and westward into the Ridge and Valley and Appalachian Plateau provinces. This group is scattered throughout the high mountains, but is fairly uncommon. Historically, it has not been threatened by logging or other human-caused disturbances due to inaccessibility and stunted trees with little commercial value.

Calcareous Cliffs

This sparsely vegetated group is characterized by significant areas of bare rock, usually limestone, dolomite or marble, with open, scattered vegetation. The cliffs generally occur above medium to large rivers and result from river undercutting and meander formation. Some occurrences are more than 300 feet tall. They are typically xeric, but may contain seepage zones. Thin, rocky soils accumulate in crevices, on ledges, and along rock margins. Occurrences on low slopes are generally more sheltered, less steep, mesic, and accumulate more soil. Vegetation is sparse over patches of rock and becomes more dense in soil accumulations. Trees and shrubs are possible, but a closed canopy never develops. Dominant vegetation is moss and lichens, ferns and calciphilic herbs. They are distinguished from rocky summits by occurring at lower, more sheltered slope positions. Calcareous cliffs are distinguished from other cliffs by type of substrate and floristic differences.

Calcareous Woodlands and Glades

These occur on flat ridgetops and gentle to steep slopes underlain by upper ordovician limestone or on limestone outcrops. Soils are generally dry, thin and rocky on glades, and deeper in woodlands. Sloping occurrences are generally well drained and remain xeric throughout the year, while flat occurrences may be wet during the winter and spring months. Acidity of 7.7 to 8.0 is reported from occurrences over dolomite in the Ridge and Valley of Virginia. This group consists of physiognomic complexes of open rock, grasslands, and woodlands. In the SAA area this group occurs in northern Georgia, northeast Alabama, western Tennessee, western Virginia, and northeastern West Virginia.

Calcareous woodlands and glades may grade into mesic oak-hickory forest (white oak, white oak-red oak-hickory, post oak-black oak), mesic mixed pine-hardwood (oak-eastern redcedar), and xeric oak-hickory forests (chestnut oak, scarlet oak, chestnut oak-scarlet oak, scrub oak). Calcareous woodlands are distinguished from other woodlands dominated by *Juniperus virginiana* (such as mafic woodlands and shale barrens) by occurring over limestone and by supporting a suite of calciphilic herbaceous vegetation.

Carolina Hemlock Forest

This group includes dry to dry-mesic coniferous forests dominated by Carolina hemlock generally occurring on exposed cliffs, rocky slopes and ridges, sometimes extending onto adjacent gentle slopes and valleys, scattered throughout the southern and central Blue Ridge in North Carolina, Tennessee, and Virginia. It is found between 4,380 and 4,460 feet on very acidic, thin, loamy soils rich in organic material. Canopy ranges from open to dense and is dominated by stunted, gnarled Carolina hemlock. Other canopy species may include chestnut oak, red oak, pitch pine, Table Mountain pine, and occasionally eastern hemlock. Carolina hemlock in valley occurrences are tall, not gnarled, and occur with the above species as well as more mesic species, such as white oak, black birch, sugar maple, and American beech. The subcanopy may be absent or contain the canopy trees as well as red maple, striped maple, flowering dogwood, buckeye, and witch hazel. The shrub layer is dense and commonly dominated by ericaceous species. The herbaceous layer is sparse below the shrub layer, although thick patches of galax along with various bryophytes and lichens occur in some sites. This group is uncommon, scattered in a few sites in the Blue Ridge in North Carolina, Tennessee, and Virginia. It is less common in the Piedmont, but occasionally occurs on steep, north-facing river bluffs.

Caves

In the Southern Appalachians, most caves are found in carbonate valleys of the Ridge and Valley province and the Cumberland Plateau. Fissure caves, formed between large rocks, are found in the Blue Ridge province. Many are associated with flowing stream-spring systems and are undergoing continual development, while others are dry. This results in variations of cave life. Cave systems contain unique living communities, strongly influenced by lack of light, a stable and high relative humidity, a limited distribution of nutrients and energy, and moderated temperatures. Underground aquatic systems contain their own community of organisms not found in caves without abundant water. Transients such as bats also use caves. Caves may contain a variety of microhabitats including streams, pools, wet stone, mud flows, dry rock, and mud banks. Cave communities vary greatly between and within occurrences. Physical conditions vary within and between caves both spatially and temporally. Air temperatures are normally steady, but will vary nearer to surface openings. They reflect the local mean annual air temperature on the surface, varying only about 1 degree F in the constant temperature-dark zone. Water temperatures fluctuate more, as much as 20 degrees F.

Granitic Dome

Includes heterogenous occurrences on steep to gently, usually south-facing, sloping outcrops of granite or granite gneiss at low to middle elevations ranging from 690 to 5,000 feet in the Blue Ridge. They are typically dominated by areas of bare rock with vegetation mats scattered throughout. The vegetation varies with soil depth and mat age. Vegetation develops in stages beginning with crustose and foliose lichens and progressing to include herbaceous species, then shrubs and possibly some tree species. Hydrology varies both temporally and spatially. In the higher elevations the outcrops are wetted by frequent rain and heavy fog; seepage zones on the outcrops are common throughout the elevation range. Variance among sites depends on elevation, steepness, exposure, and amount of seepage. There are some differences in species composition due to elevation. The mosaic of vegetation on most granitic domes seems stable. Occurrences of granitic domes are known from western North Carolina, northwestern South Carolina, and northern Georgia. They are distinguished from other cliff types and from Rocky summits by occurring on smooth, exfoliating rock, and by the lack of crevices and deep soil accumulations.

Granitic Flatrock

Flat to gently sloping outcrops of exfoliating granite, granitic gneisses, adamellite and syenite or related rocks occurring at about 1,000 feet. Most are xeric, consisting of bare bedrock or shallow soils with very low water-holding capacity. Depressions in the rock and seepage zones provide areas with more mesic soils. Most of the dry rock surface is covered by lichens. Where more complex growth occurs, the rock surface is covered by vegetation mats of mosses, lichens and herbaceous species. Small, wet depressions and seepages are common on flatrocks and may contain wetland species. The soils are commonly organic or mineral matter caught in the vegetation mats or may be shallow rocky or sandy soils over bedrock. Woody species rarely become established. There is some floristic and vegetational variation among occurrences. The flora of the flatrocks exists as a very old, highly specialized vegetational unit which persists in a balanced ecological equilibrium. Disturbances such as exfoliation of the rock surface, windthrow of trees, and drought prevent development of continuous soil and limit encroachment by woody species. Granitic flatrocks are most common in the Piedmont west and north of the fall line in the Piedmont from Virginia south to Georgia and Alabama. They are distinguished from granitic domes by their flatness, lack of crevices, and species composition. Granitic flatrocks are more typical of the Piedmont physiographic province and are only in the SAA area in the Appalachian Mountain/Piedmont transition zone.

Grassy Balds

Commonly occur on south- to southwest-facing ridgetops, domes, and gentle slopes at elevations above 5,000 feet. Conditions are characterized by strong winds, high rainfall, frequent fog, and extremes of temperature and moisture. Soils are variable, but often less acidic than in surrounding forests. Soils may be somewhat moist and relatively deep. Where the balds grade into rock outcrops, soils are generally dry, shallow and rocky. This group is dominated by grasses and herbaceous species with patches of shrubs and small trees. Dominant species are variable depending on the environmental conditions, land use history, and topographic positions of the grassy balds. The most common herbaceous stratum is mountain oat grass. Species composition of grassy balds is variable, often due to different types of disturbances. Other variation is due to differences in soil moisture and exposure; sedges tend to dominate on moist soils, while mountain oat grass tends to dominate on drier soils. Grassy balds are scattered throughout higher elevations of the Southern Appalachians, primarily from the Great Smoky Mountains northward. This group is frequently surrounded by other high-elevation types including heath balds, montane spruce-fir and northern hardwoods. High-elevation rocky summits often occur within grassy balds. Grassy balds are distinguished from all other high-elevation community types by having extensive areas dominated by herbaceous vegetation. High-elevation rocky summits and granitic domes may contain patches of herbaceous vegetation, but they are small and occur within a complex of bare rock and vegetation of mixed physiognomy.

Heath Balds

Typically dominated by ericaceous shrubs on steep, exposed slopes and ridges, occasionally on rock outcrops, at elevations ranging from 2,000 to 6,500 feet. They are found in the southern Appalachian Mountains of western North Carolina, eastern Tennessee, southwestern Virginia, northeastern Georgia, and northwestern South Carolina. The soils are generally acidic, nutrient-poor, and organic. Extreme cold, high precipitation, frequent fog, and desiccating winds, in combination with the shallow, nutrient-poor soils are key environmental factors. Shrub cover is usually dense, but can be open and garden-like. Dominant shrubs vary with elevation and geographic location, but common dominants are mountain laurel, rhododendron, and blueberries, occurring singly or in various combinations. Herbaceous cover is generally sparse due to the dense cover of shrubs or the presence of exposed rock. Some occurrences of this shrubland type are open with fairly dense herbaceous strata. Composition of herbaceous layer depends on elevation, shrub cover, soil type, soil moisture, and availability of nutrients. Occurs at higher elevations of western North Carolina, eastern Tennessee, southwestern Virginia, northeastern Georgia, and northwestern South Carolina. Heath balds are distinguished by having only scattered, stunted tree species and by being dominated by a generally continuous, ericaceous shrub stratum over a typically sparse herbaceous layer.

High-Elevation Rocky Summits

Found above 4,000 feet on vertical and horizontal rock outcrops of metamorphic, fractured, irregular rock on predominantly north-facing portions of peaks, ridges, and upper slopes. The soils vary from relatively deep mineral or organic material in cracks, to shallow soil over bedrock. Large areas of bare rock are typical. Frequent rainfall, fog deposition, and seepage areas with high winds and shallow soils limit vegetation growth. Vegetation is generally a physiognomic complex dominated by scarcely vegetated rock surfaces and herb-dominated areas on shallow soils and shrub-dominated areas with scattered trees on deeper soils in crevices. Species composition within this group varies depending on soil moisture and depth. This group is found in western North Carolina, eastern Tennessee, and southwestern Virginia. They are distinguished from surrounding vegetation by having extensive bare rock and herb-dominated areas, and by lacking closed-canopy or shrub layers. They are distinguished from cliff communities by occurring on high, exposed sites such as upper slopes and summits. While granitic domes are characterized by smooth exfoliating rock, high-elevation rocky summits have irregular, fractured rock surface.

Mafic and Calcareous Fens

Found on flat to gently sloping areas, on shallow, organic-rich mineral soils over mafic, ultramafic, or calcareous bedrock from 2,400 to 4,200 feet. The semi-permanently to permanently saturated hydrology is maintained by mineral-rich, circumneutral waters from upslope seepages. This group is dominated by wetland graminoid species and bryophytes, with occasional scattered shrub thickets and trees. Very small areas of calcareous seepage in the Ridge and Valley province of Virginia and possibly Tennessee and Alabama are included. Vegetation is dependent upon water supply from upslope seepages. Even slight alterations in this water supply and the drainage of water may cause dramatic changes in fen vegetation. Occurrences of this group are susceptible to damage by trampling and to encroachment by woody species. There are only a few known occurrences and these are small.

Mafic Cliffs

Occur on very steep rocky slopes of mafic igneous or metamorphic rocks, often associated with north-facing river bluffs and may contain cool, moist seepage zones as well as significant areas of dry, bare rock, and shallow pockets of soil. Vegetation is variable within and among sites. Mafic cliffs are characterized by large areas of bare rock with open vegetation. Soil pockets may develop which allow the occurrence of scattered trees and shrubs. Seepage areas may occur and may support more mesophytic species than those that occur on the surrounding rock and dry, thin soil. Mosses and lichens are common on rocks and in seepage areas, while ferns and basophilic herbs grow in cracks and on small soil accumulations. Occurrences are scattered along the Blue Ridge province. This group has few known occurrences with little acreage. They are susceptible to invasion by exotic species and damage by trampling.

Mafic Woodlands and Glades

Occur in the mountains of North Carolina and Virginia on flats or gentle to steep, south- to east-facing upper slopes at 2,500 to 4,400 feet. Soils are generally thin, droughty, or seasonally wet and friable. Vegetation varies from predominantly herb-dominated glades to red-cedar dominated woodlands. Mafic woodlands and glades have a restricted range and most occurrences are small.

Mountain Lakes

The only example of a natural lake system in the SAA area is Mountain Lake, located at 3,870 feet in the Northern Ridge and Valley section, Giles county, Virginia. The lake was formed by the damming of a mountain stream by a rock slide that blocked a narrow valley, perhaps several thousand years ago. Maximum depth is about 100 feet and has varied over the last several centuries due to occasional breaches in the natural dam.

Mountain Longleaf Pine Woodlands

Dominated by longleaf pine and occur on xeric ridges and moderately steep (30 to 70 percent) upper slopes below 1,900 feet at the southern terminus of the Appalachians in Alabama and Georgia. Longleaf pine and other species in these woodlands depend on periodic fire. These are virtually gone due to fire suppression. Most variation in species composition depends on the length of time since a fire. In more recently burned sites, longleaf pine still strongly dominates a relatively open canopy and the herbaceous layer is dense. On fire-suppressed sites, oaks and other hardwoods have more coverage than longleaf pine and, due to increased canopy density, the herbaceous layer is sparse. This group is distinguished from other montane ecological groups by being dominated by longleaf pine. It is distinguished from other longleaf pine woodlands and forests by geographic location; all other longleaf pine types occur in the Coastal Plain or, rarely, in the upper Piedmont.

Mountain Ponds

These are shallow pools found in small upland depressions. They are seasonally to semi-permanently flooded montane wetlands fed by rainfall and shallow-soil ground water movement. They often do not receive significant or constant ground water seepage and are generally less than 1/4 acre. They are characterized by shallow, open water surrounded by vegetation dominated by various wetland shrubs and herbs. Herbaceous vegetation typically occurs in monospecific clumps, with sphagnum mats and scattered shrubs and trees around the margins. They are important breeding and rearing sites for a number of insects and salamanders. They are also of biogeographical importance in the Southern Appalachians because they may harbor disjunct populations of northern or Coastal Plain species. However, they exhibit relatively low vascular plant species diversity compared with other montane, non-alluvial wetlands. This group is not common, with 50 to 80 known occurrences.

River Gravel-Cobble Bars

Occur along moderate- to high-radiant large streams and rivers in areas periodically scoured of woody vegetation. Soils are absent or represented by recently deposited silts and sands in pockets among the various sized gravels, cobbles, boulders, and in-place outcroppings which make up most of the area of the bars. Riverwards, or in locations exposed to stronger or more frequent scouring, these are typically bare rock and grass-dominated vegetation, while tree and shrub thickets dominate in more protected situations.

Sandstone Cliffs

Typically quartzitic sandstone escarpments occurring above streams and rivers and near mountain crests in the Ridge and Valley, Cumberland Plateau, and Cumberland Mountain provinces. They occur at all ridge and slope topographic positions and range from very exposed, xeric cliffs to more sheltered, mesic slope rockhouses. Soils are generally acidic and consist of organic pockets or coarse mineral matter which has accumulated within mats of pioneer vegetation, on ledges, and in crevices. Vegetation is typically sparse and consists of lichens and mosses over the vertical rock surface, with grasses, sedges, and other vascular plants in deeper soils of ledges, crevices, or vegetation mats along the top of the outcrop. This group is widespread in the western SAA area.

Seasonally Dry Sinkhole Ponds

These are a specialized ecological group occurring in Augusta and Rockingham counties, Virginia, on flat valley floors in acidic colluvial material, along the base of the western slope of the northern Blue Ridge. Seasonally dry sinkhole ponds have a much greater diversity of vascular plant species than do mountain ponds.

Serpentine Woodlands and Glades

Found at moderate elevations, about 3,000 feet, on gentle to steep, concave slopes with variable aspects. Vegetation is a physiognomic complex with woodland, grassland, and forest components, varying with soil depth, geology, and fire history. The open woodland is dominated by stunted individuals of pitch pine, Virginia pine, and white oak. Woodland openings have dense coverage of grasses. Forested areas have closed canopies which are dominated by white oak or pitch pine, and a herbaceous stratum dominated by ragwort. Seepage areas with royal fern, Canada burnet, large-flowered parnassia, fringed gentian, and cinnamon fern occur as small inclusions in the forest and woodland. Within the SAA area, serpentine woodlands and glades are limited to a few scattered sites in the southern Blue Ridge of North Carolina and the west-central Piedmont of Virginia.

Shale Barrens

These occur primarily on steep, south-facing slopes and bluffs on outcrops of various shale formations in the Ridge and Valley province. This group usually occurs as open, stunted woodlands, interspersed with grasslands, scattered forbs, shrubs, and areas of exposed bedrock and shale scree. Occurrences are small, <3 acres, and characterized by relatively sparse vegetation over shale, with fragmented rocks strewn over the surface of only skeletal soils. Common canopy dominants are chestnut oak, red oak, scarlet oak, pitch pine, Table Mountain pine, and red-cedar. The herbaceous layer is more diverse than the shrub or canopy layers and contains many species which are generally endemic to shale barrens. This group is found on steep, predominantly south-facing slopes and bluffs. In the SAA area they are restricted to a zone from Frederick county, Virginia, south to Montgomery county, Virginia, and west into Pendleton, Hardy, and Hampshire counties, West Virginia, with a core concentration in the upper watershed of the James River in Allegheny and Bath counties, Virginia. Shale barrens have a limited range in the Southern Appalachians and most occurrences are small in size, generally less than 5 acres. Based on recent aerial photograph analysis there may be 1,000 to 1,500 shale barrens in western Virginia and eastern West Virginia.

Sinkholes and Karstlands

These are areas of karst regions where solution of bedrock has created a subterranean zone that receives some degree of direct or reflected light from the surface. These habitats are often closely associated with caves, and they reflect a transitional gradient from surface communities to the dark subterranean communities of caves. They are often moist, shaded habitats, but sometimes examples or zones within a particular site are very dry. Species inhabiting sinkholes and karstlands are typically adapted to the low light and prevailing moisture regimes found there. Sinkholes and karstlands are characterized by the presence of ferns and bryophytes. Large examples of this ecological group are rare in the SAA area, with only scattered examples. Sinkholes and karstlands are most common in the karst areas of the Northern and Central Ridge and Valley province, as well as in the Cumberland Plateau.

Sphagnum and Shrub Bogs

Contains a heterogeneous grouping of non-alluvial, southern Appalachian wetlands. There are two general types of wetlands included in this diverse group: sphagnum-herb dominated bogs and shrub-dominated bogs. Sphagnum and shrub bogs occur in variable topographic positions (from flat areas to fairly steep slopes) at 1,500 to 5,800 feet in Georgia, North Carolina, Tennessee, Virginia, and perhaps in Alabama and South Carolina. The soil saturation is maintained primarily by by a high water table and frequent rainfall when they occur on flatter areas. The soils are very acidic, wet organic or mucky mineral. They are characterized by a mosaic pattern of shrub thickets and herb-dominated areas underlain by sphagnum mats. Red spruce, white pine, pitch pine, eastern hemlock, black gum, yellow-poplar and red maple plus others may be scattered throughout or may dominate in patches or on the edges. Cotton grass and large cranberry are found at or near the southern limit of their distribution in this ecological group. This group is distributed in western North Carolina, western Virginia, eastern West Virginia, eastern Tennessee, northern Georgia and possibly in South Carolina and Alabama. There are few existing examples of this group and many are in degraded condition.

Spray Cliffs

Includes herbaceous vegetation on rock substrates associated with waterfalls in the southern Blue Ridge escarpment region. It is characterized by a variable but unique assemblage of vascular herbs, algae, and bryophytes, many of which are endemic to this community. It is found on nearly vertical rock surfaces and ledges, slopes, and crevices with shallow soils which are constantly saturated. The hydrology of this community is supplied by constant spray from waterfalls. This community is a variable collection of mosses, liverworts, algae, vascular herbs, and occasional shrubs and trees, most of them requiring constantly moist substrate and very high relative humidity. This group is very limited, known only from a few dozen occurrences, most of which are less than 1 acre in size, and no larger than 2 acres in size.

Spruce-Fir Forests

Includes coniferous forests occurring as discontinuous, irregularly shaped islands above 5,000 feet in western Virginia, eastern Tennessee, and western North Carolina. Within the SAA area, this group also includes areas of red spruce and red spruce-balsam fir in eastern West Virginia and in Shenandoah National Park. Spruce-fir forests have canopies dominated by red spruce or Fraser fir (or balsam fir in one instance in Shenandoah National Park) or a mixture of these species. Density and composition of shrub and herbaceous strata are

occurrence of a mixture of species endemic to the Southern Appalachians and species disjunct from northern boreal forests. Bryophyte cover can be conspicuous, especially in undisturbed, old growth occurrences. The bryophyte flora is diverse, with many endemic or northern disjunct species. The environment is characterized by high moisture levels, low temperatures, strong winds, and acidic low nutrient soils.

Swamp Forest-Bog Complex

Includes palustrine forests and woodlands which are known from the Ridge and Valley and southern Blue Ridge of Virginia and North Carolina, and are likely in adjacent South Carolina, Georgia, and Tennessee below 4,000 feet in poorly-drained bottomlands which are rarely to occasionally flooded. The soils are seasonally to semi-permanently saturated due to a high water table or seepage from adjacent slopes. Canopy composition varies from red spruce dominated woodlands to forests dominated by mixtures of evergreen and deciduous species such as eastern hemlock and red maple, or yellow-poplar, blackgum, and white pine or pitch pine. The dominant shrubs are usually mountain laurel and rhododendron. The herbaceous layer is patchy with small, sphagnum-dominated depressions. Typical herbs are various wetland sedges and ferns.

Table Mountain Pine-Pitch Pine Woodlands

A heterogenous grouping of montane xeric pine and pine-oak dominated vegetation, which generally occurs on sharp ridges and steep slopes with southerly aspects, knobs, and low-elevation peaks on well-drained soils from southeastern West Virginia to northwestern Georgia. Canopy composition of this group varies primarily along an elevational gradient. Below 2,400 feet, on slopes, ridges, and knobs, occurrences are dominated by shortleaf pine. From 2,400 to 2,800 feet on the driest ridges, pitch pine dominates. Above 2,800 feet on slopes and ridges, Table Mountain pine dominates. These forests grade into one another so that some occurrences contain mixtures of these species. Virginia pine and scarlet oak are also common co-dominants. Composition of the shrub and herbaceous strata vary with elevation, exposure, and geographic location. This group is found ern South Carolina, western Tennessee, western North Carolina, central Virginia, and farther north on ridges and steep slopes.

Talus Slopes

Are flat to steep non-vegetated to sparsely-vegetated rock accumulations at 2,500 to 4,600 feet. Soils are absent or consist of slight accumulations of organic material among the rocks. Highly acidic, almost pure sand may accumulate at the base of the talus slopes. There is very little available moisture due to lack of a structured soil layer and to high insolation. These exposed sites are also subject to heavy rains and the influence of ice, snow, and harsh winds, which further limit vegetation establishment. Vegetation very limited. Lichens thrive on the bare rock surfaces, and mosses and other lichens sometimes occur in the organic matter in crevices between rocks. Other vegetation is limited to very scattered individuals which occur most frequently around the periphery of the area of bare rock.

Wet Prairie

Known only from the Shenandoah Valley of Virginia near headwaters of the South Fork of the Shenandoah River in a broad, flat valley over limestone. They are dominated by herbaceous vegetation with scattered trees, with low herbs being dominate.

Status Summaries of the Rare Community Classes

Included are status summaries for rare communities referenced in Chapter 3 (tables C-18 to C-21).

Table C–18 The Southern Appalachian Assessment area list of rare communities and the associated indicator species used in helping to determine community occurrences.

	Indicator Species (Criteria 1 and 2 from Matrix List)	
Rare Community	Scientific Name	Common Name
Beaver Ponds and Wetland Complex	None	
Beech Gap Forest	None	
Boulderfields (Forested)	*Microtus chrotorrhinus carolinensis*	southern rock vole
	Aconitum reclinatus	trailing wolfsbane
	Geum geniculatum	bent avens
	Scutellaria saxatilis	rock skullcap
	Stachys clingmanil	Clingman's hedgenettle
Calcareous Cliffs	*Elymus svensonii*	Svenson's wild-rye
	Heuchera longiflora	Long-flowered alumroot
	Paxistima canybi	Canby's mountain-lover
	Astragalus neglectus	Cooper's milkvetch
Calcareous Woodlands and Glades	*Hypericum dolabriforme*	straggling St. John's wort
	Trifolium calcarium	running glade clover
Carolina Hemlock Forest	None	
Caves	*Myotis grisescens*	gray bat
	Myotis sodalis	Indiana bat
Granitic Domes	*Solidago simulans*	granite dome goldenrod
	Senecio millefolium	divided-leaf ragwort
	(except Lee and Scott County, VA)	
Granitic Flatrocks	*Amphianthus pusillus*	pool sprite
	Cyperus granitophilus	granite-loving flatseed
Grassy Balds	None	
Heath Balds	None	
High Elevation Rocky Summits	*Solidago spithamaea*	Blue Ridge goldenrod
	Bazzania nudicaulis	a liverwort
	Calamgrostis cainii	Cain's reedgrass
	Hudsonia montana	mountain golden heather
	Carex misera	wretched sedge
	Geum radiatum	speading avens
Mafic and Calcareous Fens	*Parnassia grandifolia*	grass of parnassus
Mafic Cliffs	None	
Mafic Woodlands and Glades	*Sisyrinchium dichotomum*	white irisette
	Orthotrichum keeverae	Keever's bristle-moss
Mountain Lakes	None	
Mountain Longleaf Pine Woodlands	None	
Mountain Ponds	*Scirpus ancistrochaetus*	Northeastern bulrush
River Gravel/Cobble Bar	*Pityopsis ruthii*	Ruth's golden aster
	Conradina verticillata	Cumberland rosemary
	Spiraea virginiana	Virginia spiraea
	Calamovilfa arcuata	Cumberland sandgrass
	Solidago rupestris	rock goldenrod
Seasonally Dry Sinkhole Ponds	*Helenium virginicum*	Virginia sneezeweed
Sandstone Cliffs	*Arenaria cumberlandense's*	Cumberland sandwort
	Ageratina Luciae-brauniae	Lucy Braun's white snakeroot
	Allium speculae	Little River canyon onion
Serpentine Woodlands and Glades	None	
Shale Barrens	*Arabis serotina*	Shale barren rockcress
Sinkholes and Karstlands	*Asplenium scolopendrium var.*	Hart's tongue fern
Sphagnum and Shrub Bogs	*Clemmys muhlenbergii*	Bog turtle
	Sarracenia jonesil	Mountain sweet pitcher plant
	Sarracenia oreophila	Green pitcher plant
	Juncus caesariensis	New Jersey rush
	Helonias bullata	Swamp pink
	Poa paludigena	Bog bluegrass
Spray Cliffs	*Bryocrumia vivicolor*	Gorge moss
	Grammitis nimbata	Dwarf polypody fern
	Hymenophyllum turnbridgense	Turnbridge fern
	Lejevenea blomquistii	A liverwort
	Plagiochila caduciloba	A liverwort
Spruce/Fir Forests	*Glaucomys sabrinus coloratus*	Carolina northern flying squirrel
	Glaucomys sabrinus fuscus	Virginia northern flying squirrel
	Semiothisa fraserata	Fraser fir geometrid
	Microhexura montivaga	Spruce-fir moss spider
	Abies fraseri	Fraser fir
Swamp Forest–Bog Complex	None	
Table Mountain Pine/Pitch	*Gaylussacia brachycera*	Box huckleberry
	Pine Woodlands	
Talus Slopes (Non-Forested)	*Plethodon shenandoah*	Shenandoah salamander
Wet Prairie	*Platanthera leucophaea*	Eastern prairie fringed orchid

Table C–19 The distribution of rare communities by state within the Southern Appalachian Assessment area.

Rare/Special Ecological Group		AL	GA	NC	SC	TN	VA	WV
Beaver Ponds and Wetland Complex		x	x	1	x	x	8	x
Beech Gap Forest				10		x	2	
Boulderfields (Forested)			x	96	x	37	5	6
Calcareous Cliffs		2	x	3		2	41	10
Calcareous Woodlands and Glades		1	1	4		2	38	x
Carolina Hemlock Forest			x	11	x	x	1	
Caves		4	11	18	17	56	344	57
Granitic Domes			x	53	18			
Granitic Flatrocks		5	x		x			
Grassy Balds			x	11		x	1	
Heath Balds			4	24	x	x	2	1
High Elevation Rocky Summits			1	135		19	11	
Mafic and Calcareous Fens				12	9	1	41	x
Mafic Cliffs				4			3	
Mafic Woodlands and Glades				11	7		34	
Mountain Lakes								1
Mountain Longleaf Pine Woodlands		x	1					
Mountain Ponds		x	4	2		x	20	2
River Gravel/Cobble Bar		x	3	18	x	78	11	x
Sandstone Cliffs		6	6			2	5	
Seasonally Dry Sinkhole Ponds								63
Serpentine Woodlands and Glades				1	1		1	
Shale Barrens		x		x		x	64	8
Sinkholes and Karstlands		x	x		2	4	x	
Sphagnum and Shrub Bogs		26	12	171	4	2	110	x
Spray Cliffs				40	4	x		
Spruce/Fir Forests				53		14	27	5
Swamp Forest - Bog Complex			1	41	x	x	17	x
Table Mountain Pine/Pitch Pine Woodlands			x	33	4	x	9	1
Talus Slopes (Non-Forested)							9	1
Wet Prairie							3	
Total	2087	44	44	752	66	215	875	91
Percent	100%	2%	2%	36%	3%	10%	42%	5%

x = Occurrences of this group are likely within these ownerships based on known distribution of abiotic and biotic factors (such as geology–rock types, associated species, landforms, etc.) which cause the group to occur at a given location on the landscape.

Table C–20 The distribution of rare communities by ecological unit (section[1]) within the Southern Appalachian Assessment area.

Rare/Special Ecological Group		231A	M221D	M221A	221J	231D	M221B	M221C	221I	221H	231C
Beaver Ponds and Wetland Complex		x	1	4	x	x	2	x	2	x	x
Beech Gap Forest			12								
Boulderfields (Forested)		2	126	12	1	x	3	x	x	2	x
Calcareous Cliffs			7	46	3	2	x	x	x	x	x
Calcareous Woodlands and Glades			4	27	11	1	x	x	1	1	1
Carolina Hemlock Forest		1	11	x							
Caves			50	293	103	6	9	x	24	9	13
Granitic domes		10	61								
Granitic Flatrocks		4				1					
Grassy Balds			12								
Heath Balds			29	2			x				
High Elevation Rocky Summits		4	162								
Mafic and Calcareous Fens			43	19	1						
Mafic Cliffs			6	1							
Mafic Woodlands and Glades		5	46	1							
Mountain Lakes				1							
Mountain Longleaf Pine Woodlands						1	x				x
Mountain Ponds			4	20		4					
River Gravel/Cobble Bar		x	33	2	x	2	x	1	3	68	1
Sandstone Cliffs			x	2	x	x	x	1	2	2	12
Seasonally Dry Sinkhole Ponds			4	59							
Serpentine Woodlands and Glades		1	2								
Shale Barrens			7	64			1				
Sinkholes and Karstlands				3	2	x	x	x	x	1	x
Sphagnum and Shrub Bogs		7	272	19	x	5	x	x	x	x	22
Spray Cliffs		2	42								
Spruce/Fir Forests			82	6			11				
Swamp Forest - Bog Complex		4	48	5			2				
Table Mountain Pine/Pitch Pine Woodlands		3	35	8	x	x	x	x	1	x	x
Talus Slopes (Non–Forested)			9	1			x				
Wet Prairie				3							
Total	2087	43	1108	598	121	22	28	2	33	83	60
Percent	100%	2%	53%	29%	6%	1%	1%	-	2%	4%	2%

x = Occurrences of this group are likely within these ownerships based on known distribution of abiotic and biotic factors (such as geology–rock types, associated species, landforms, etc.) which cause the group to occur at a given location on the landscape.

[1] Sections:

231A	Southern Appalachian Piedmont
M221D	Blue Ridge Mountains
M221A	Northern Ridge and Valley
221J	Central Ridge and Valley
M221B	Allegheny Mountains
M221C	Northern Cumberland Mountains
221I	Southern Cumberland Mountains
221H	Northern Cumberland Plateau
231C	Southern Cumberland Plateau
231D	Southern Ridge and Valley

Table C–21 The distribution of rare communities by ownership within the Southern Appalachian Assessment area.

Rare/Special Ecological Group		National Forests	National Parks	Other Federal	State	Private
Beaver Ponds and Wetland Complex		5	x	x	1	3
Beech Gap Forest		5	3		x	4
Boulderfields (Forested)		63	42		1	40
Calcareous Cliffs		3	2		1	52
Calcareous Woodlands and Glades		2	x	1	x	43
Carolina Hemlock Forest		5	1		x	6
Caves		41	13	x	10	443
Granitic Domes		33	1		7	30
Granitic Flatrocks						5
Grassy Balds		8	3		x	1
Heath Balds		14	7		x	10
High Elevation Rocky Summits		48	43		7	68
Mafic and Calcareous Fens		5	4		3	51
Mafic Cliffs		1	1		x	5
Mafic Woodlands and Glades		6	15		1	30
Mountain Lakes						1
Mountain Longleaf Pine Woodlands		1			x	x
Mountain Ponds		14	x		x	14
River Gravel/Cobble Bar		15	1	x	15	79
Sandstone Cliffs		1	4	x	4	10
Seasonally Dry Sinkhole Ponds		9				54
Serpentine Woodlands and Glades		1				2
Shale Barrens		39	x	x	x	33
Sinkholes and Karstlands		x	x		x	6
Sphagnum and Shrub Bogs		62	43		3	217
Spray Cliffs		23	2		1	18
Spruce/Fir Forests		41	34		7	17
Swamp Forest–Bog Complex		19	12		1	27
Table Mountain Pine/Pitch Pine Woodlands		23	6	x	6	12
Talus Slopes (Non-Forested)		2	7		x	1
Wet Prairie						3
Total	2087	489	244	1	68	1285
Percent	100%	23%	12%	–-	35%	62%

x = Occurrences of this group are likely within these ownerships based on known distribution of abiotic and biotic factors (such as geology–-rock types, associated species, landforms, etc.) which cause the group to occur at a given location on the landscape.

National Forest Initial Inventory of Possible Old Growth Forest

Included within this section is a “closer look” spatial distribution of the stands identified by national forests as an initial inventory (figs. C-1 to C-37). This inventory is shown for each Southern Region ranger district located in the SAA area.

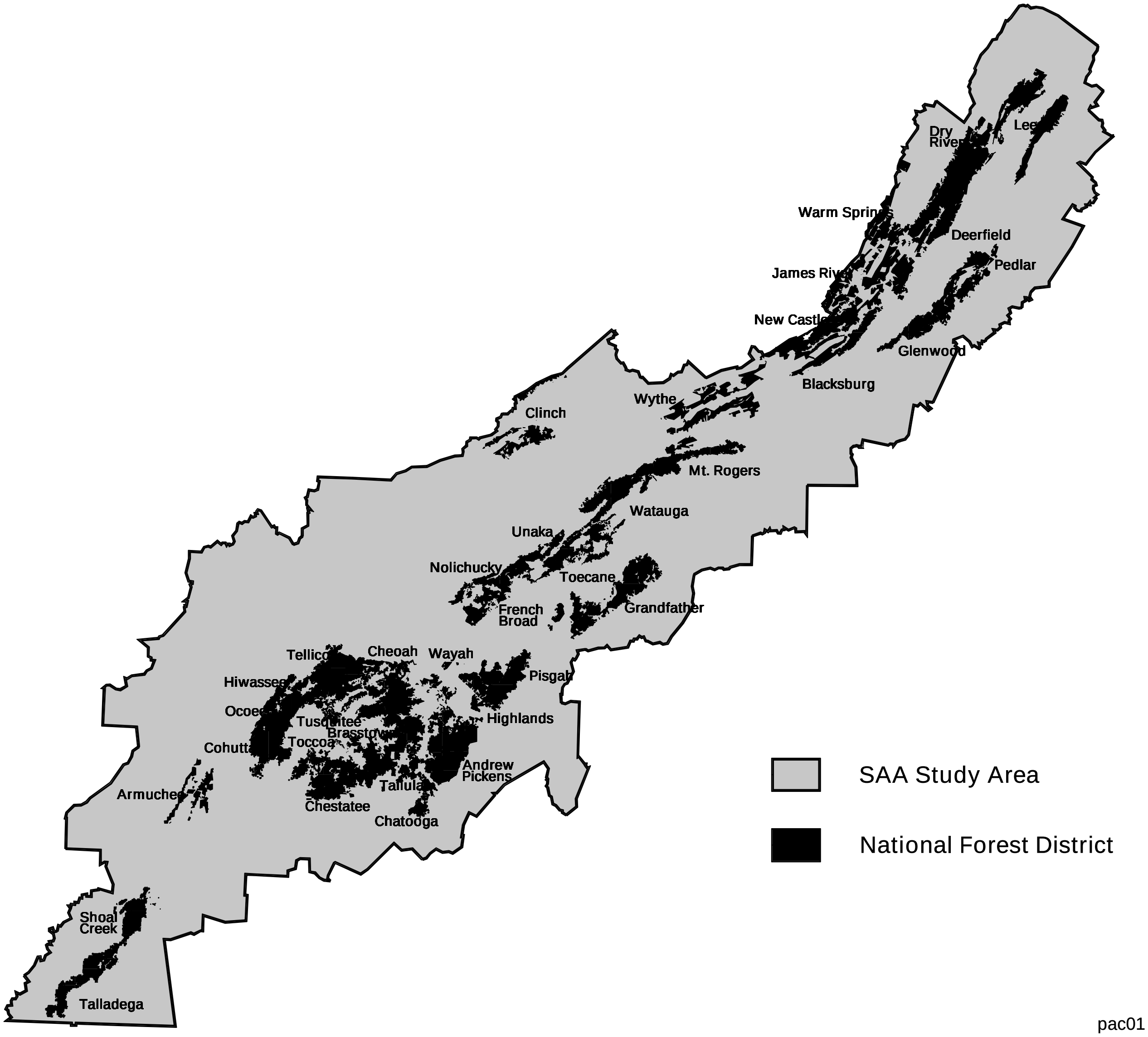

Figure C-1 The national forest ranger districts located in the SAA area.

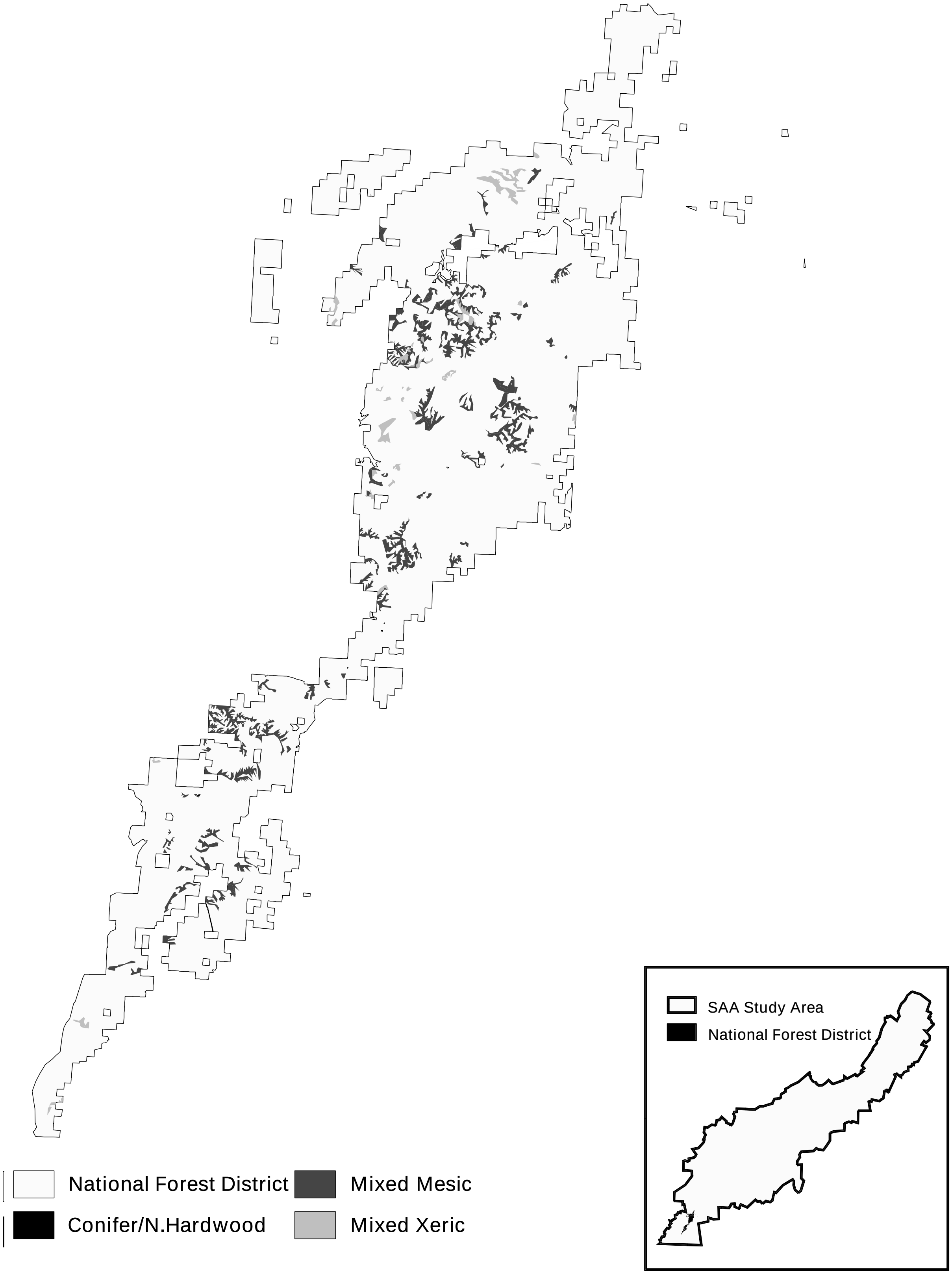

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Figure C-2 The spatial distribution of the initial inventory of forest stands selected as possible old growth on national forests in the SAA area for the Shoal Creek Ranger District.

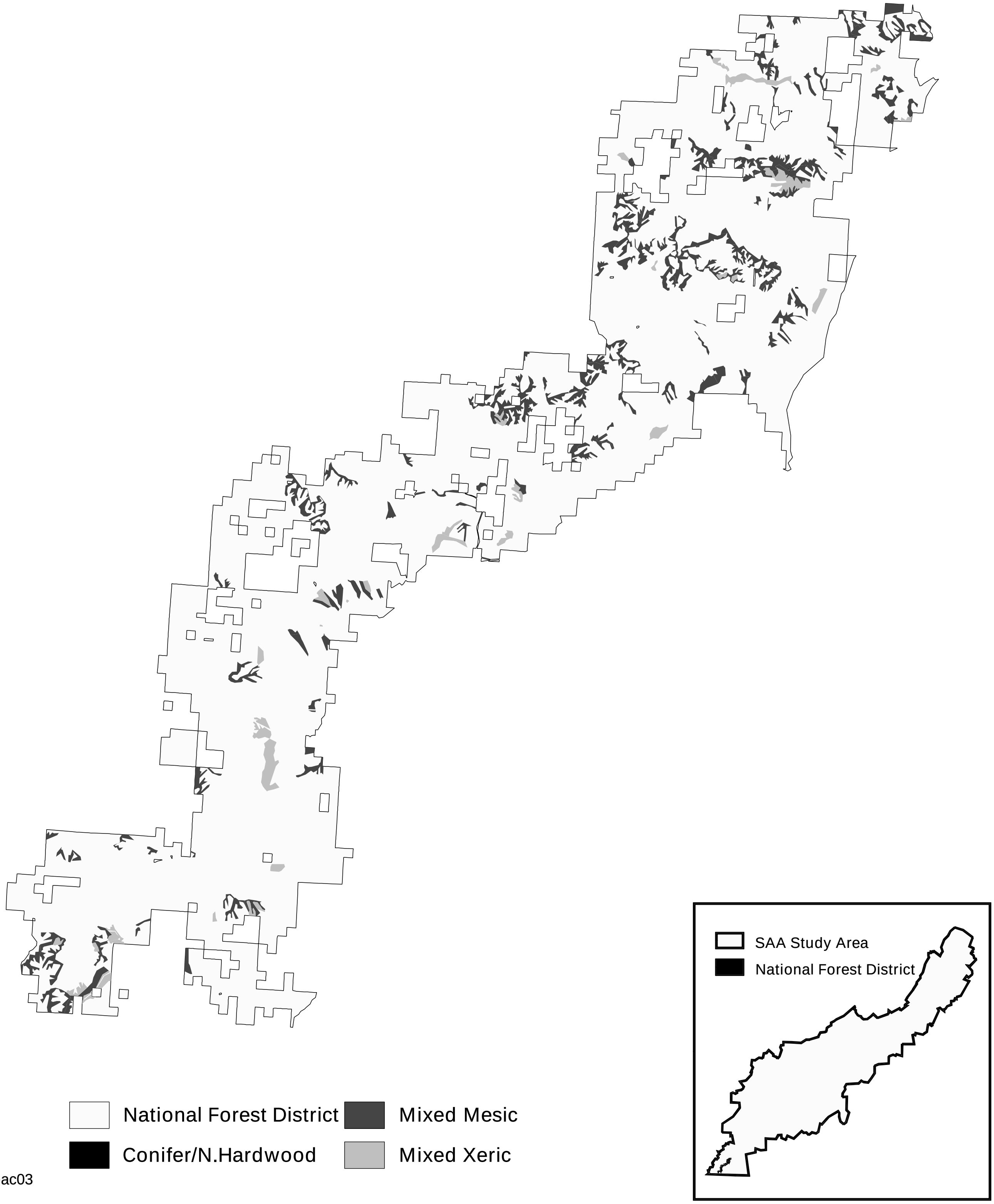

Figure C-3 The spatial distribution of the initial inventory of forest stands selected as possible old growth on national forests in the SAA area for the Talladega Ranger District.

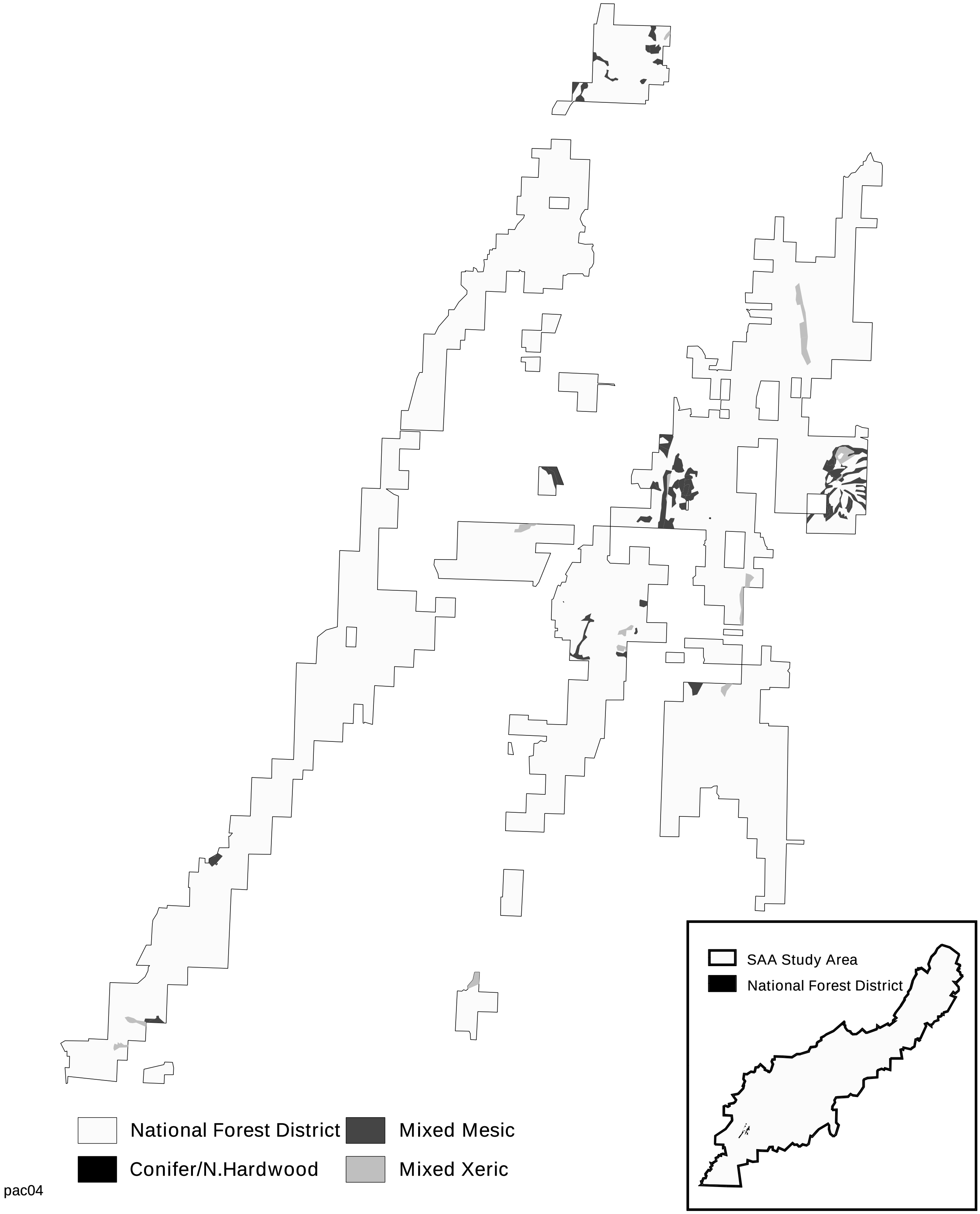

Figure C-4 The spatial distribution of the initial inventory of forest stands selected as possible old growth on national forests in the SAA area for the Armuchee Ranger District.

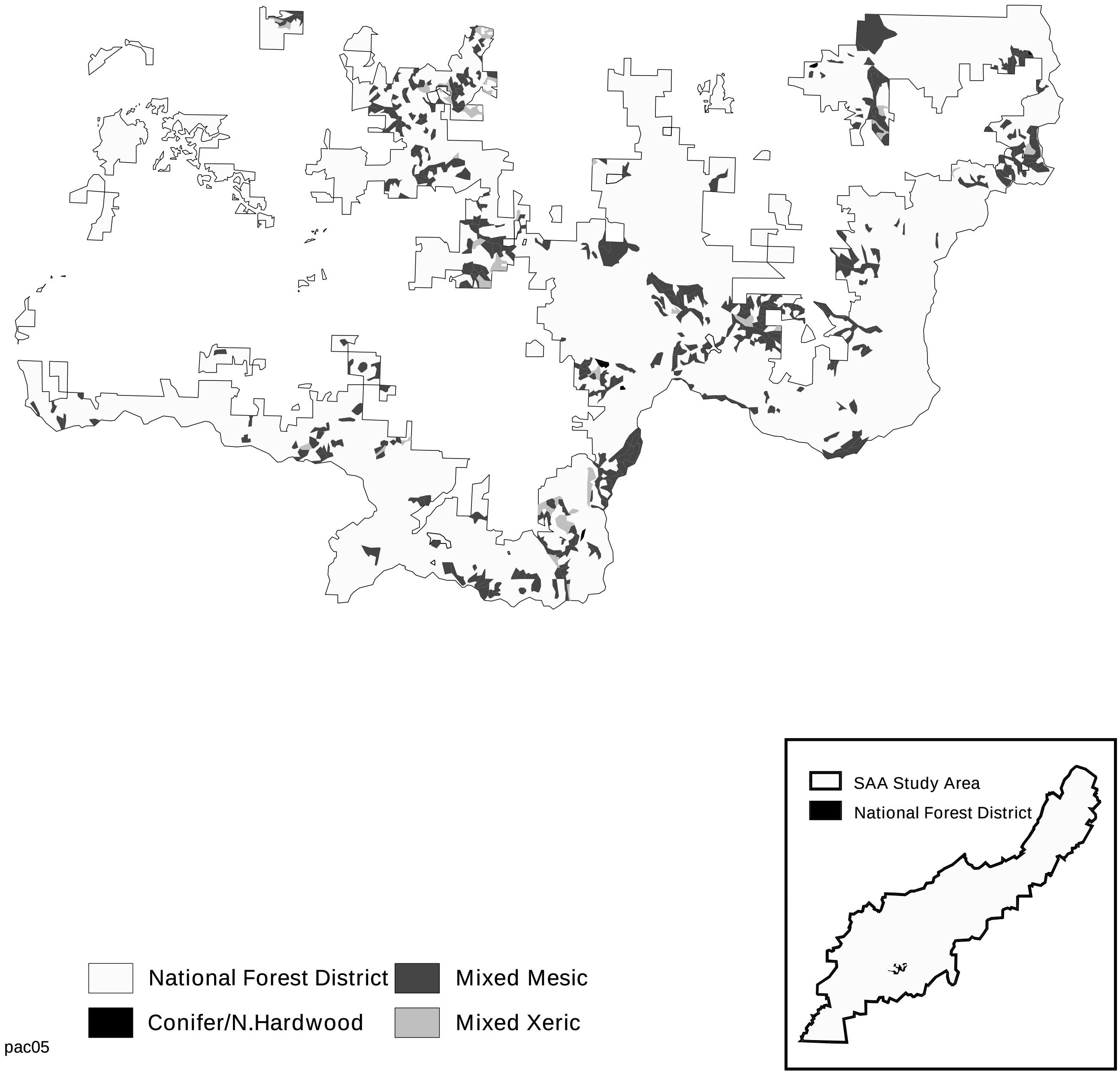

Figure C-5 The spatial distribution of the initial inventory of forest stands selected as possible old growth on national forests in the SAA area for the Brasstown Ranger District.

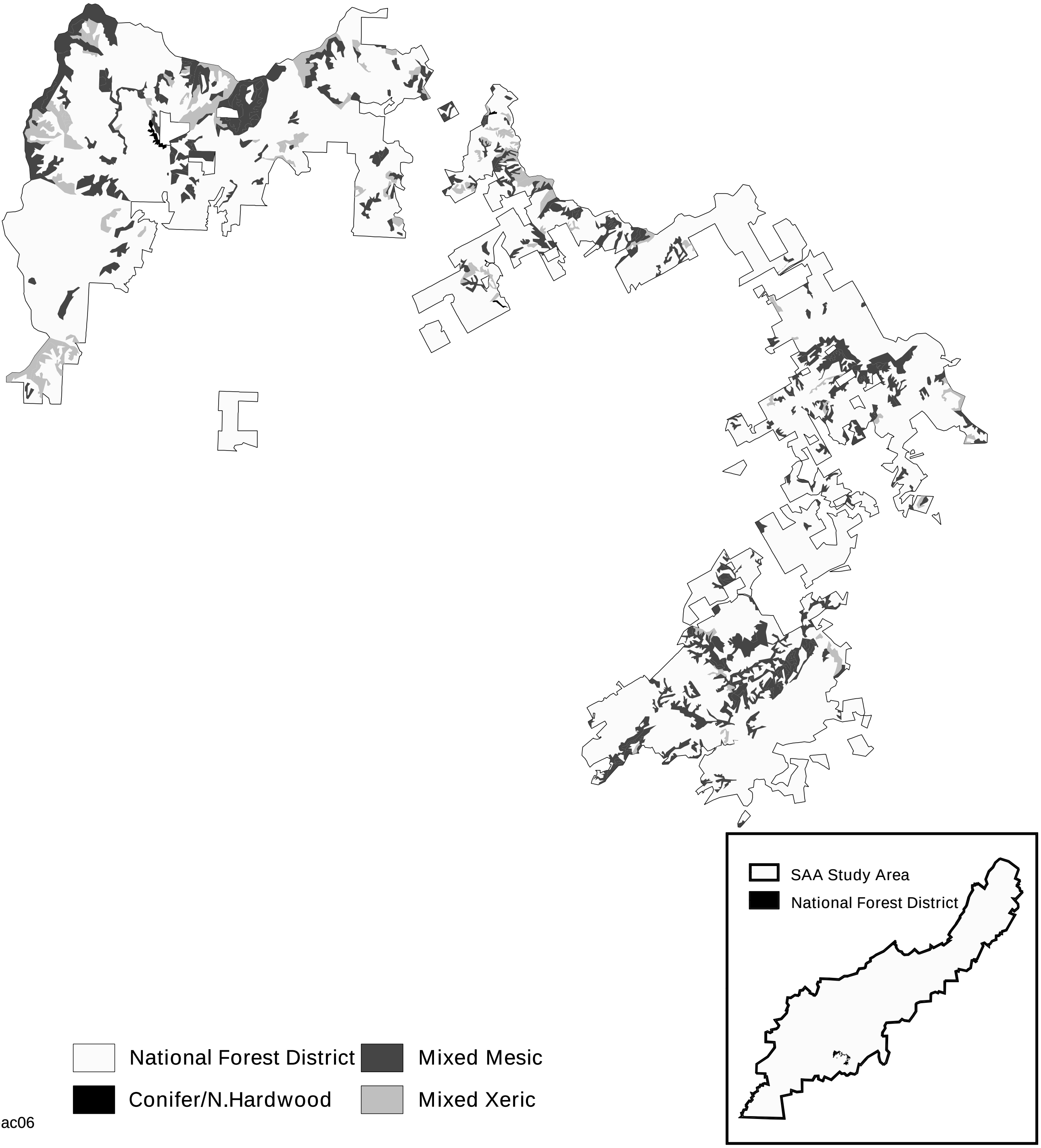

Figure C-6 The spatial distribution of the initial inventory of forest stands selected as possible old growth on national forests in the SAA area for the Chattooga Ranger District.

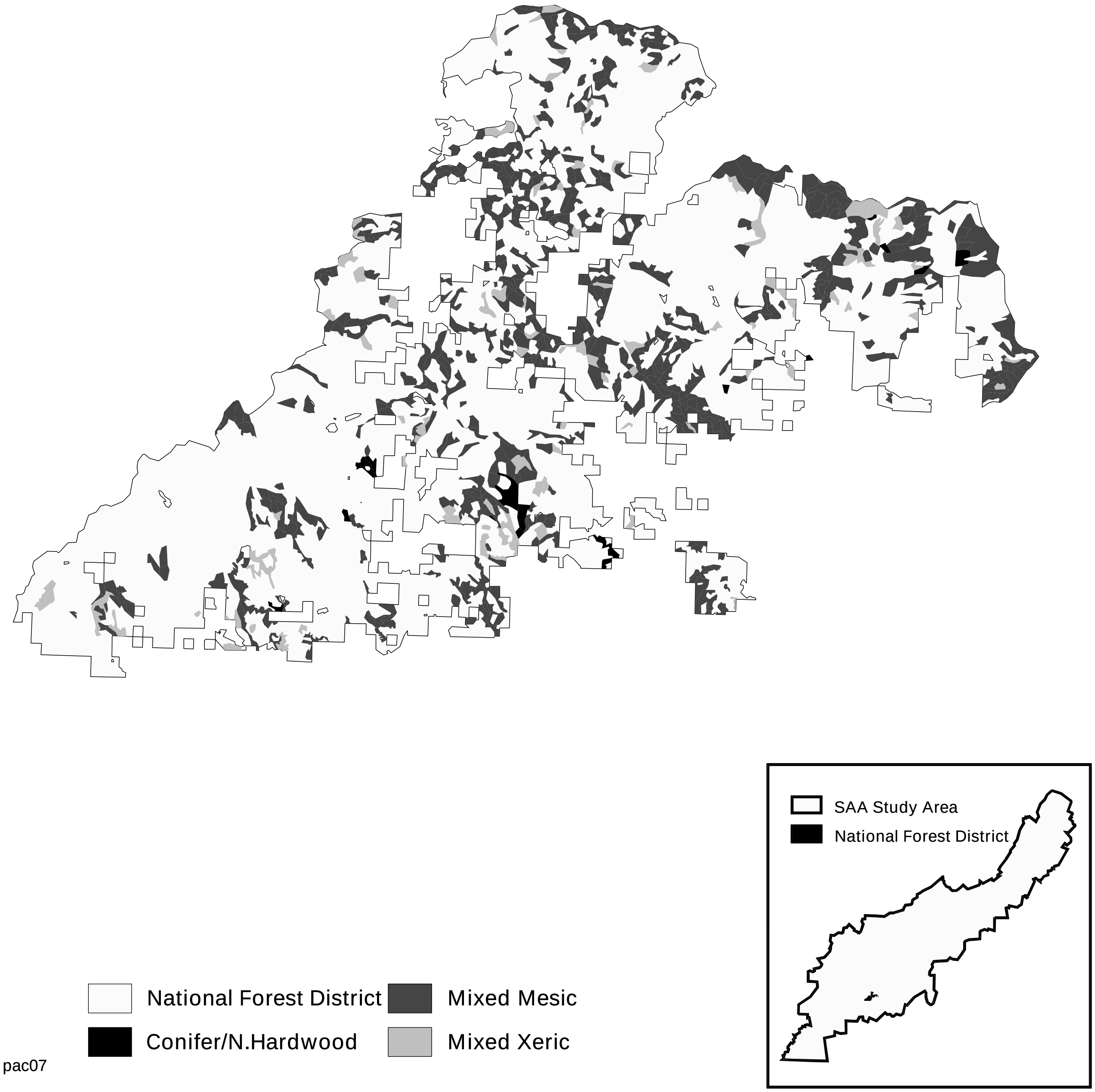

Figure C-7 The spatial distribution of the initial inventory of forest stands selected as possible old growth on national forests in the SAA area for the Chestatee Ranger District.

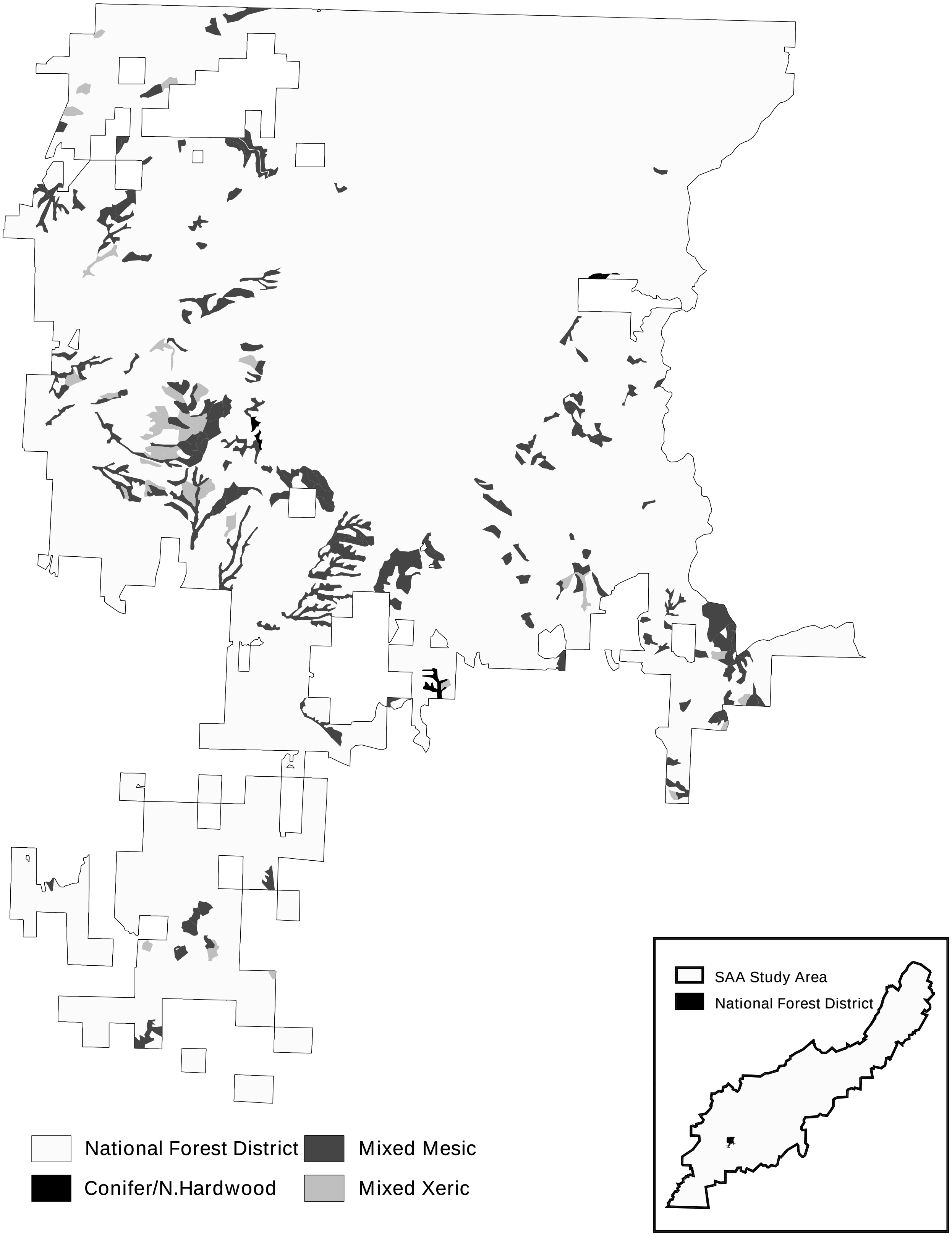

Figure C-8 The spatial distribution of the initial inventory of forest stands selected as possible old growth on national forests in the SAA area for the Cohutta Ranger District.

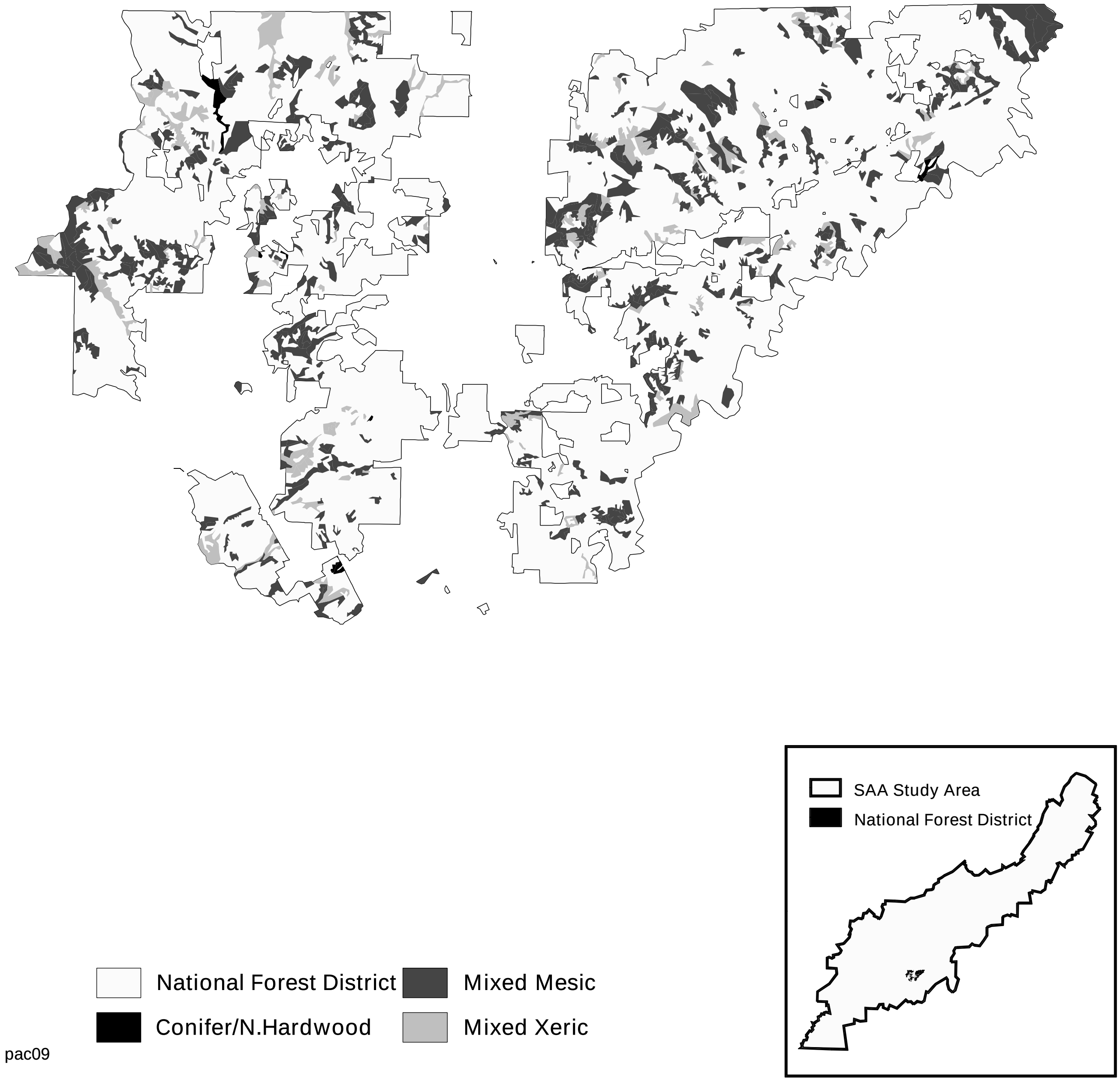

Figure C-9 The spatial distribution of the initial inventory of forest stands selected as possible old growth on national forests in the SAA area for the Tallulah Ranger District.

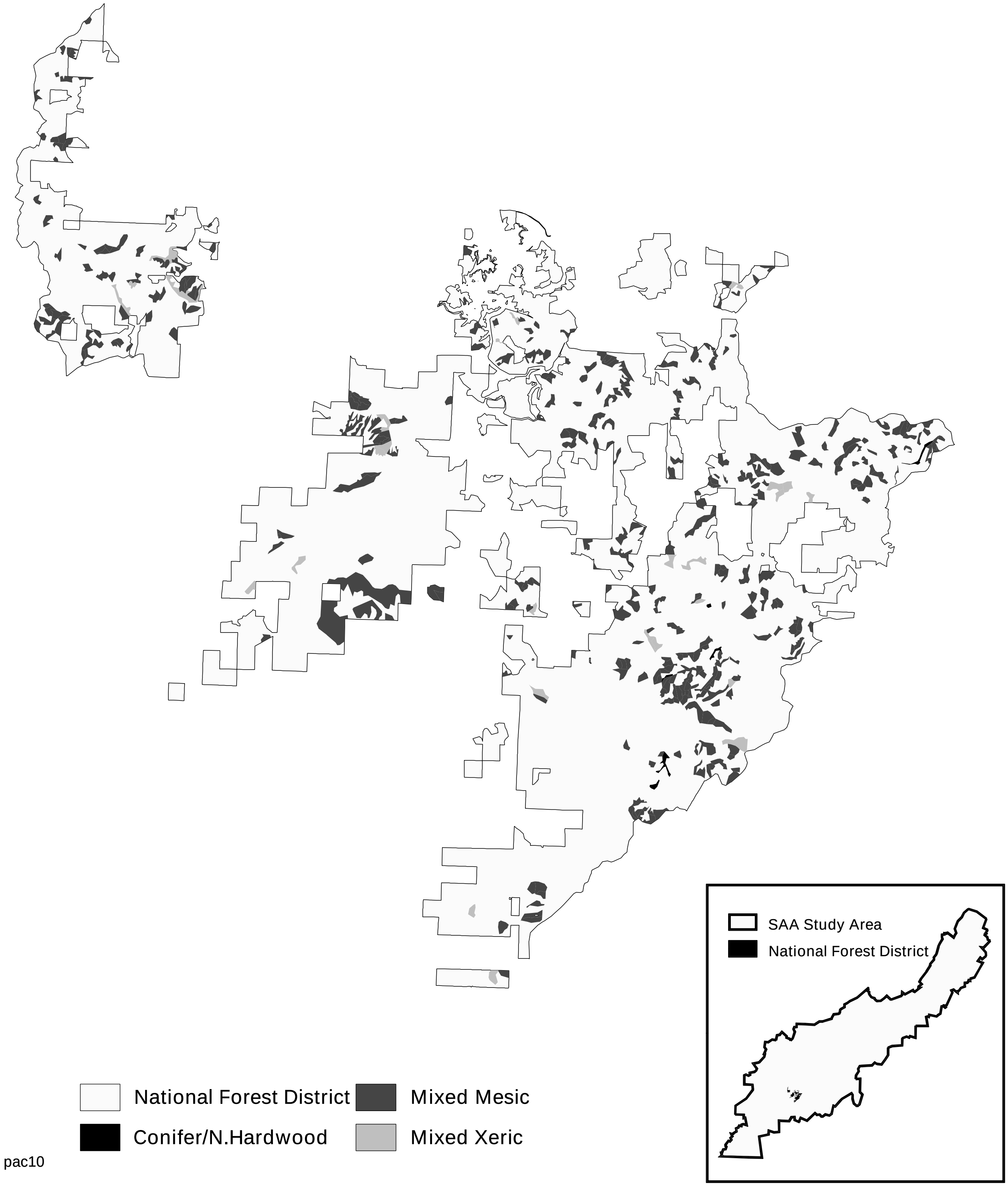

Figure C-10 The spatial distribution of the initial inventory of forest stands selected as possible old growth on national forests in the SAA area for the Toccoa Ranger District.

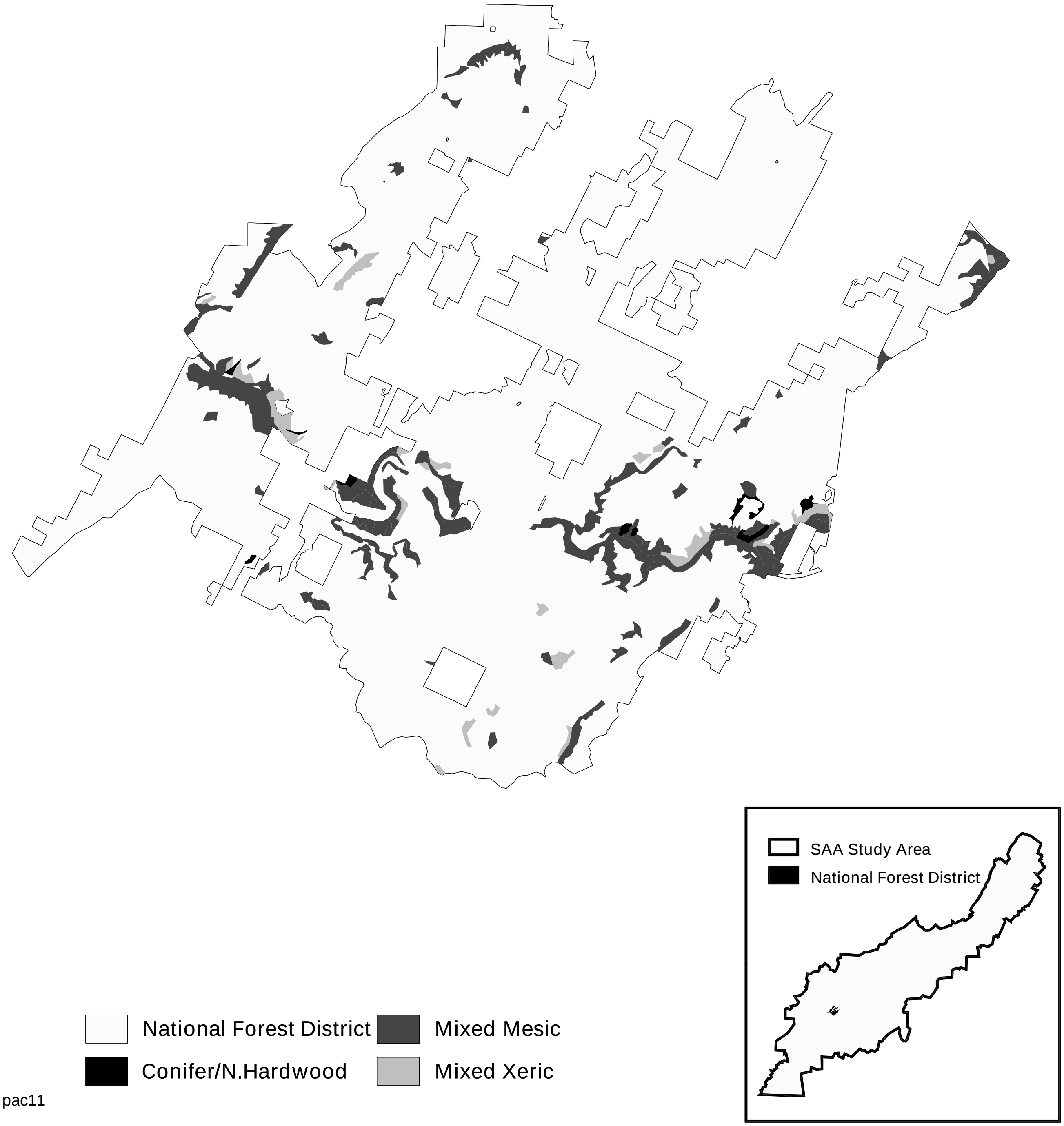

Figure C-11 The spatial distribution of the initial inventory of forest stands selected as possible old growth on national forests in the SAA area for the Hiwassee Ranger District.

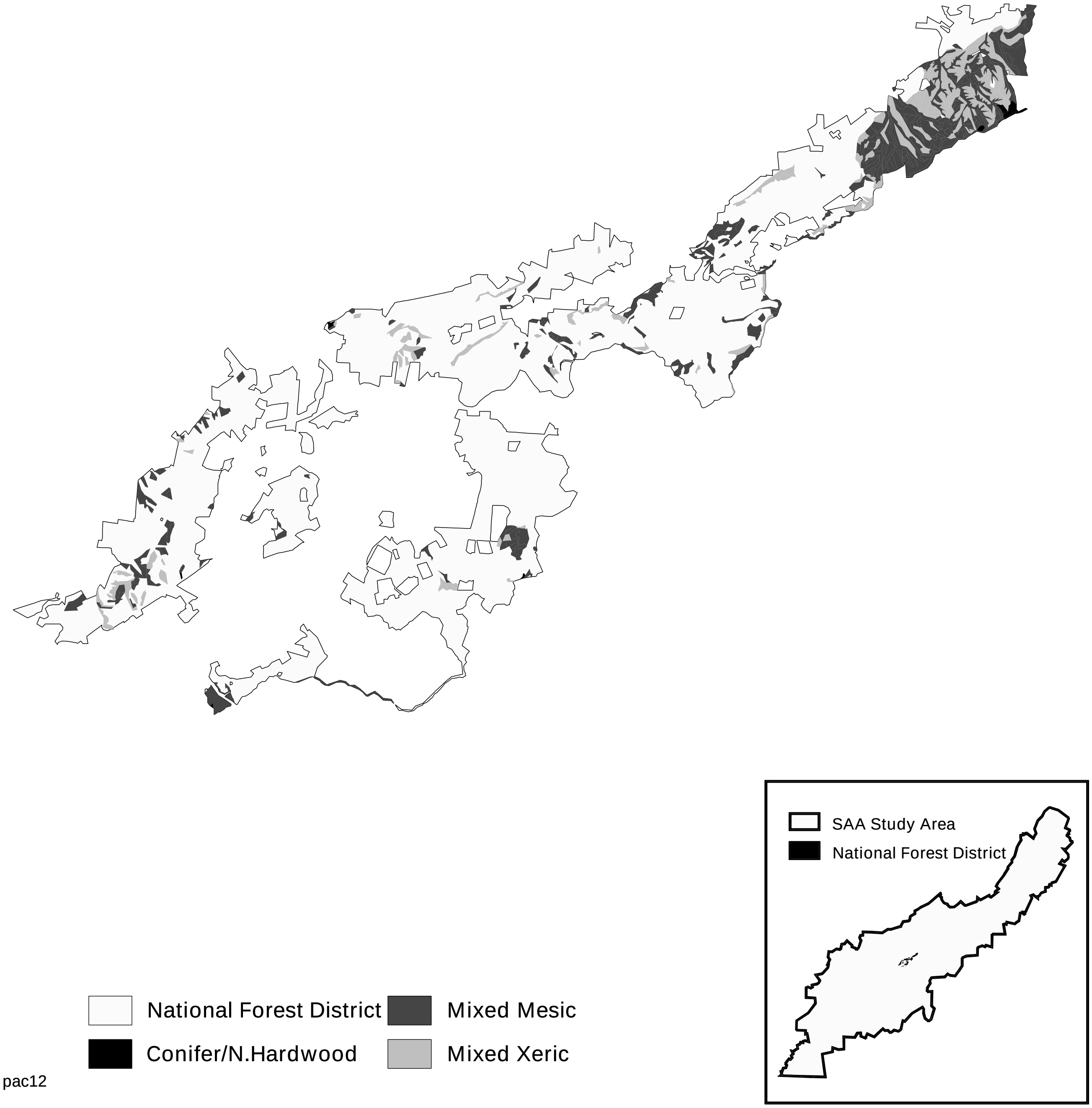

Figure C-12 The spatial distribution of the initial inventory of forest stands selected as possible old growth on national forests in the SAA area for the Nolichucky Ranger District.

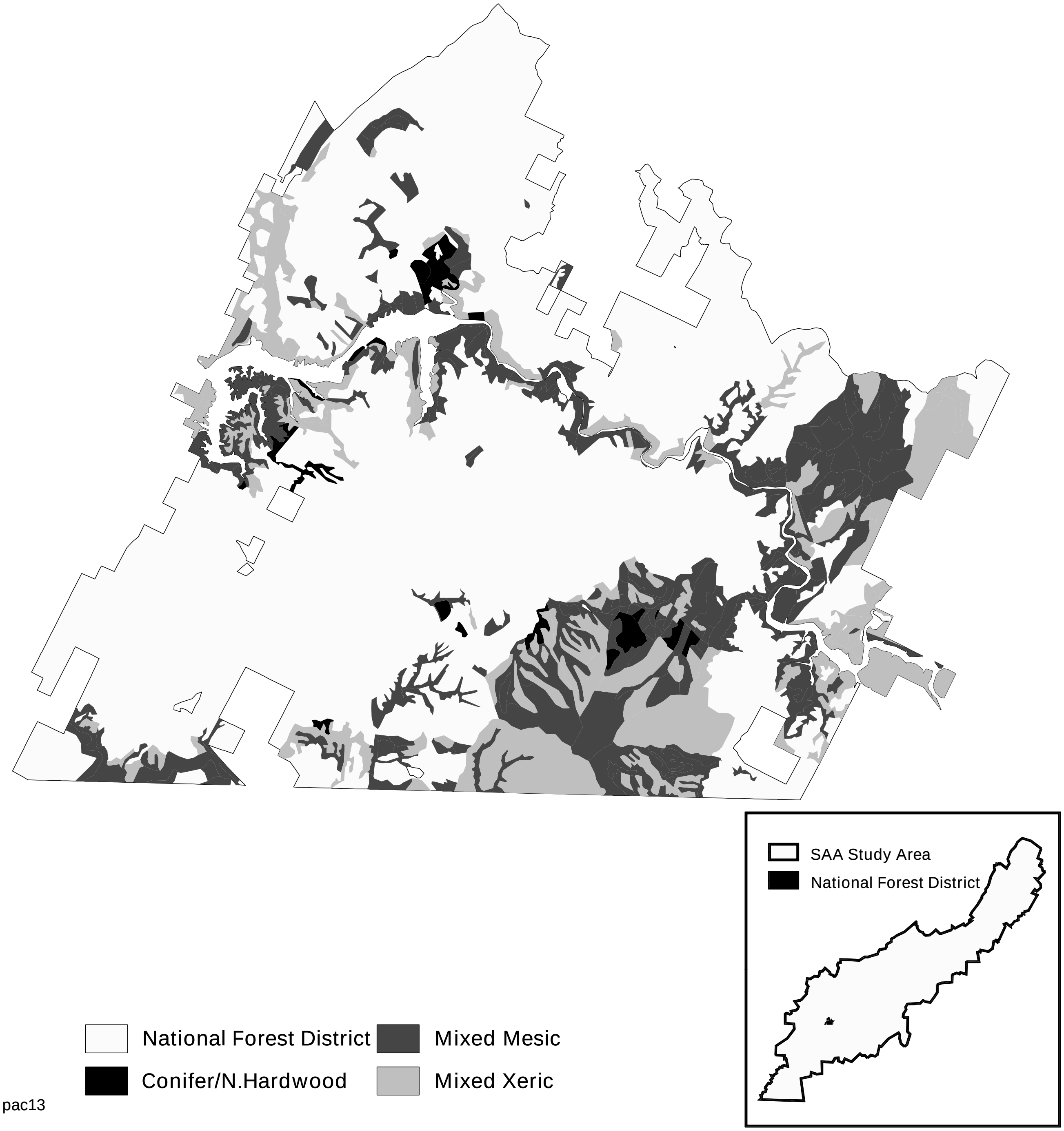

Figure C-13 The spatial distribution of the initial inventory of forest stands selected as possible old growth on national forests in the SAA area for the Ocoee Ranger District.

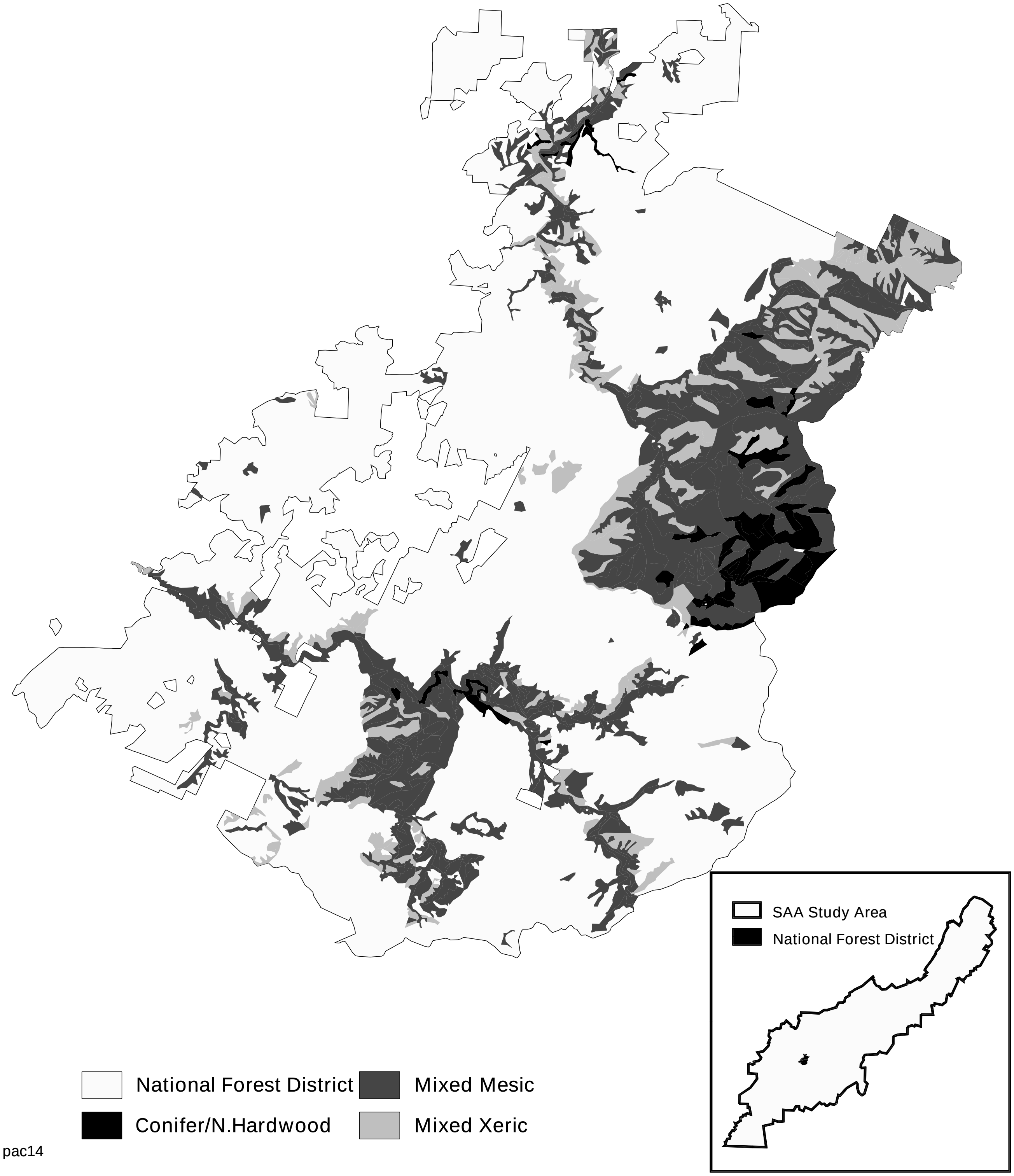

Figure C-14 The spatial distribution of the initial inventory of forest stands selected as possible old growth on national forests in the SAA area for the Tellico Ranger District.

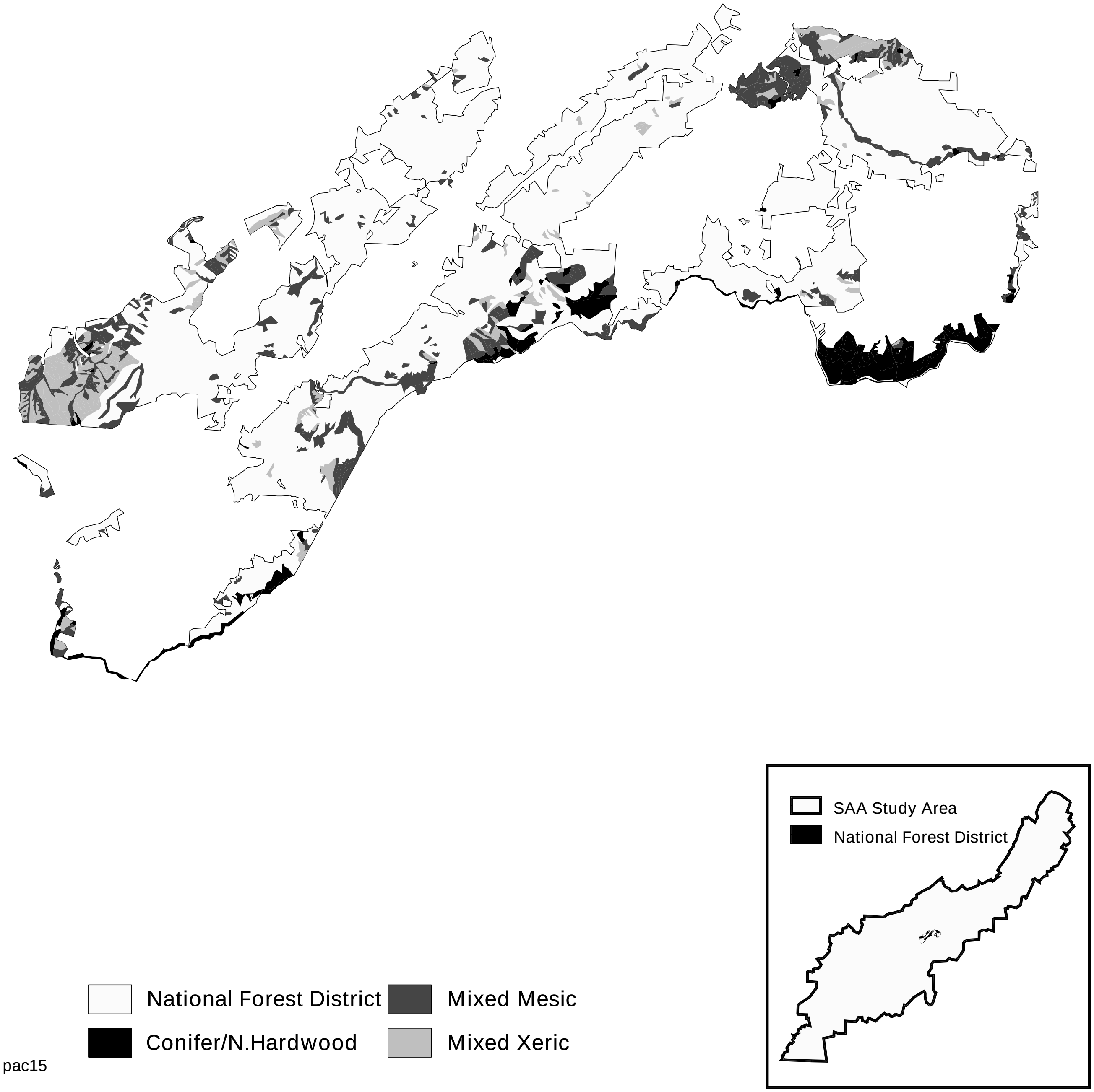

Figure C-15 The spatial distribution of the initial inventory of forest stands selected as possible old growth on national forests in the SAA area for the Unaka Ranger District.

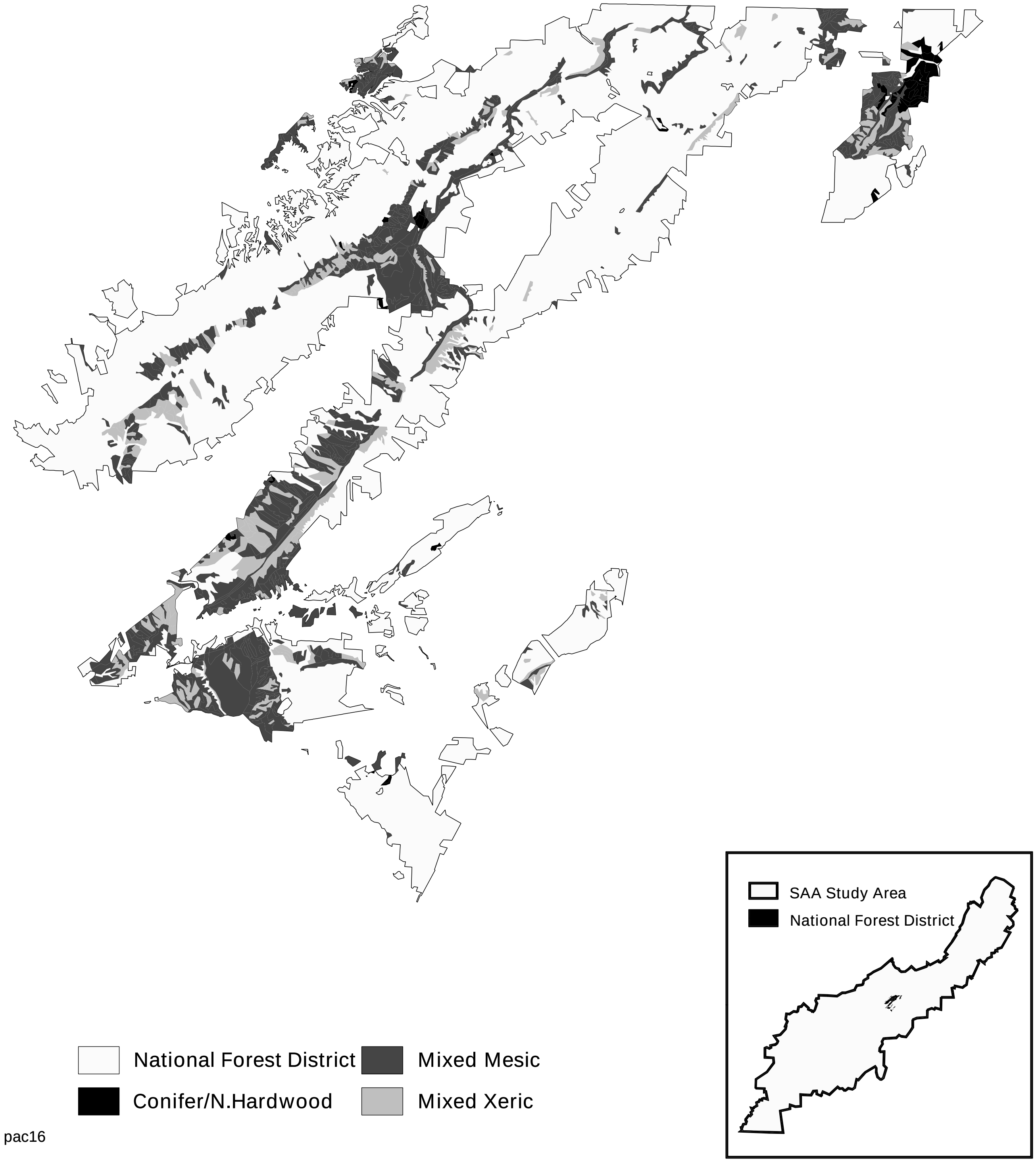

Figure C-16 The spatial distribution of the initial inventory of forest stands selected as possible old growth on national forests in the SAA area for the Watauga Ranger District.

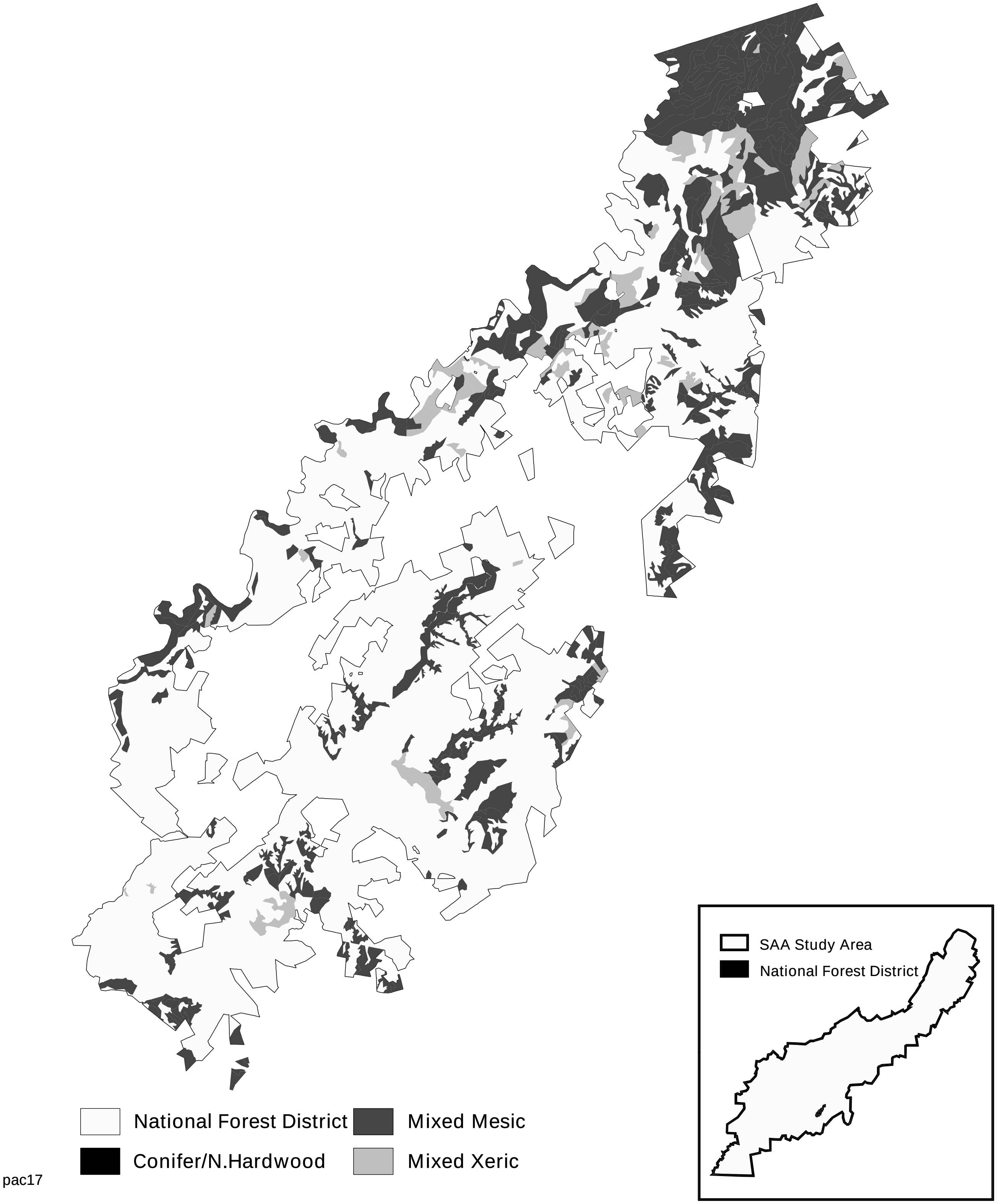

Figure C-17 The spatial distribution of the initial inventory of forest stands selected as possible old growth on national forests in the SAA area for the Andrew Pickens Ranger District.

Figure C-18 The spatial distribution of the initial inventory of forest stands selected as possible old growth on national forests in the SAA area for the Blacksburg Ranger District.

Figure C-19 The spatial distribution of the initial inventory of forest stands selected as possible old growth on national forests in the SAA area for the Clinch Ranger District.

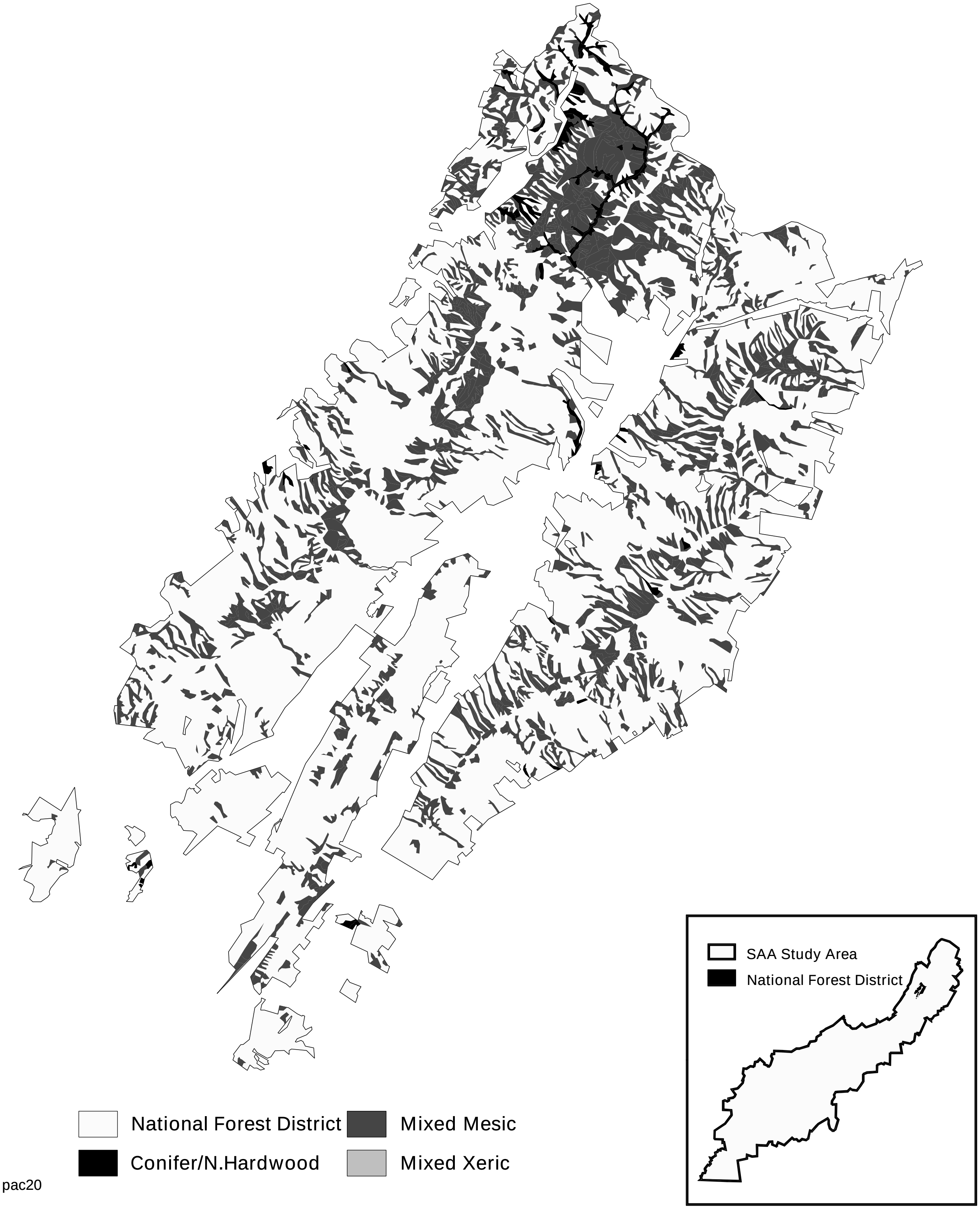

Figure C-20 The spatial distribution of the initial inventory of forest stands selected as possible old growth on national forests in the SAA area for the Deerfield Ranger District.

Figure C-21 The spatial distribution of the initial inventory of forest stands selected as possible old growth on national forests in the SAA area for the Dry River Ranger District.

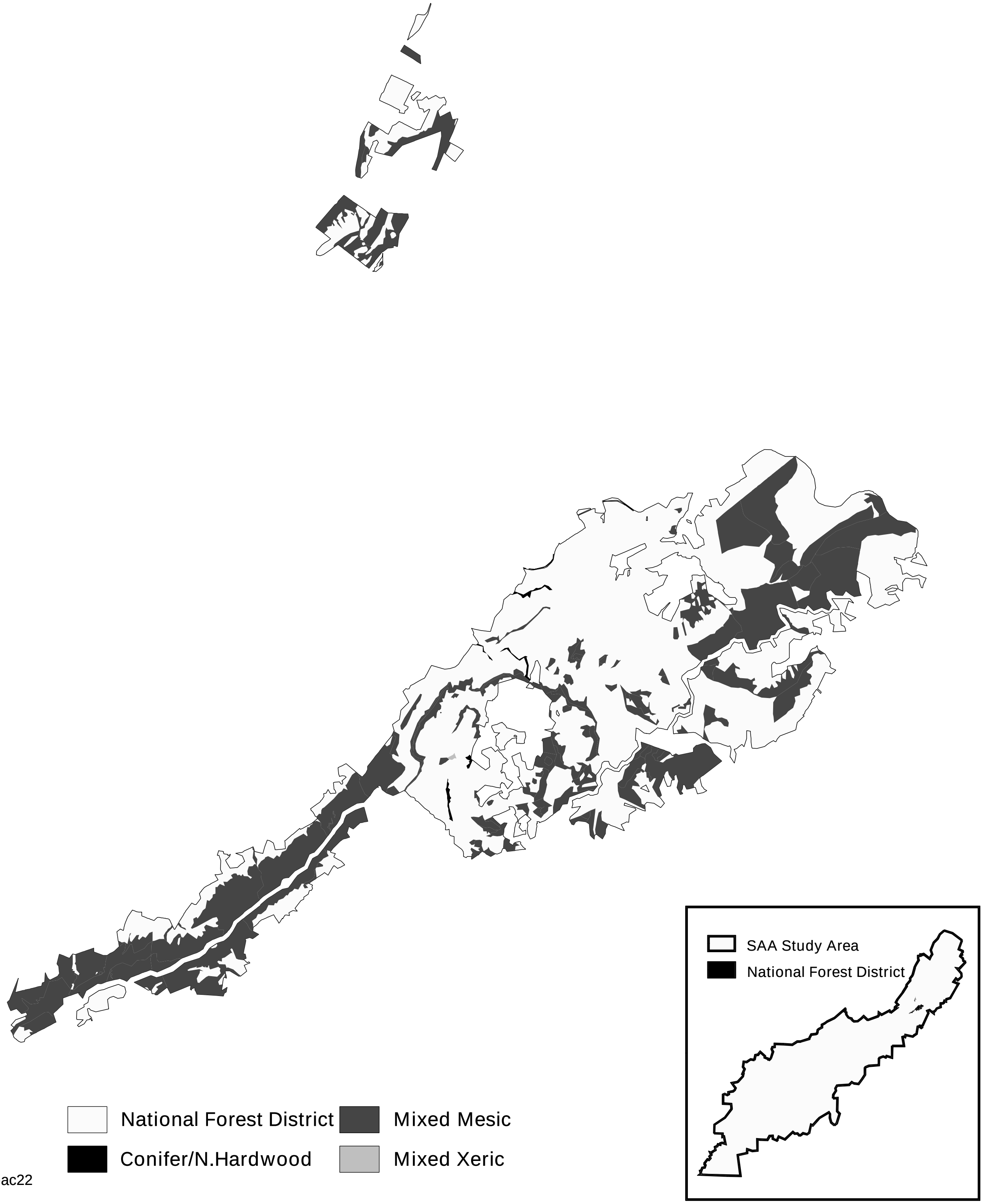

Figure C-22 The spatial distribution of the initial inventory of forest stands selected as possible old growth on national forests in the SAA area for the Glenwood Ranger District.

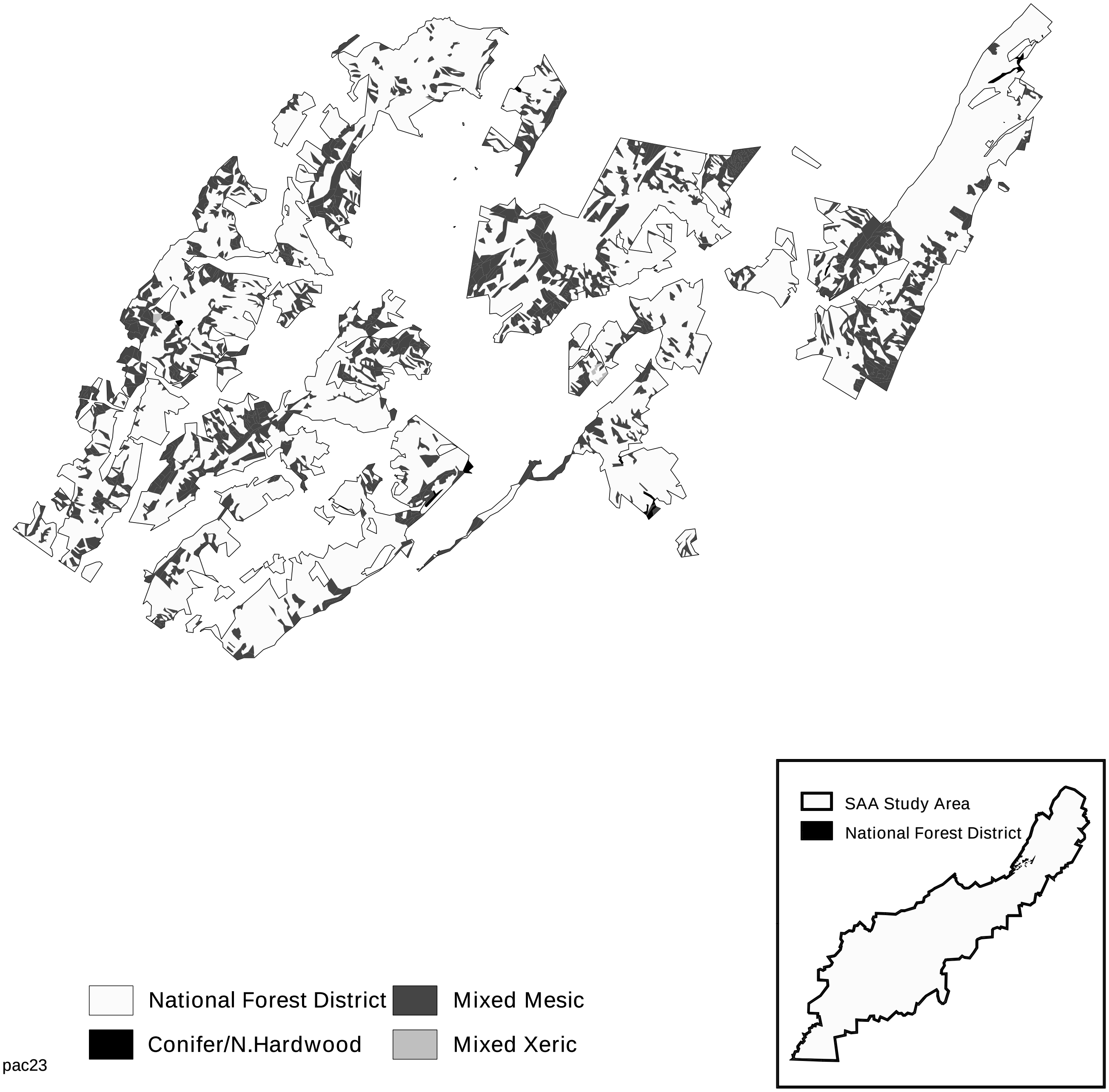

Figure C-23 The spatial distribution of the initial inventory of forest stands selected as possible old growth on national forests in the SAA area for the James River Ranger District.

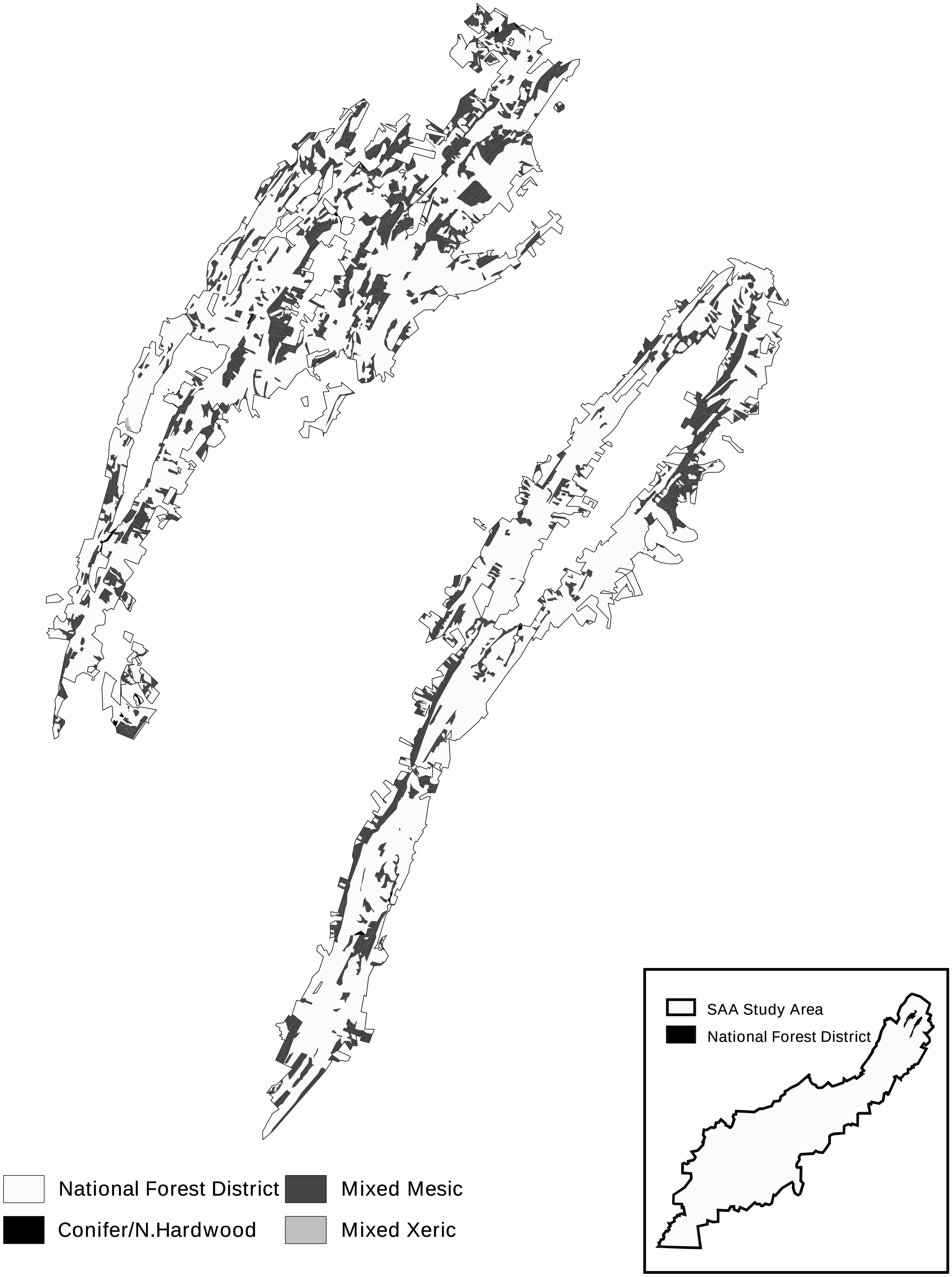

Figure C-24 The spatial distribution of the initial inventory of forest stands selected as possible old growth on national forests in the SAA area for the Lee Ranger District.

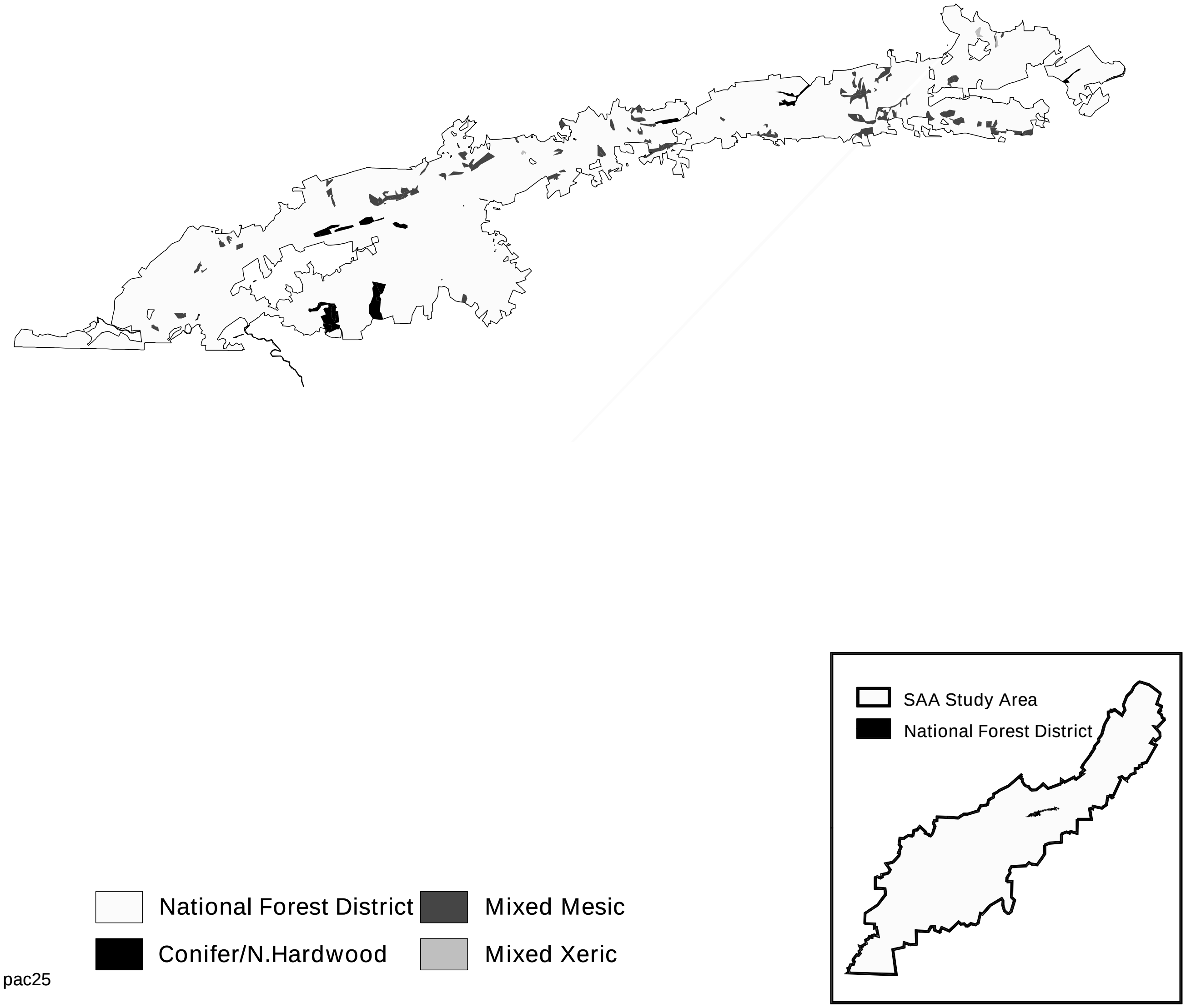

Figure C-25 The spatial distribution of the initial inventory of forest stands selected as possible old growth on national forests in the SAA area for the Mt. Rogers National Recreation Area.

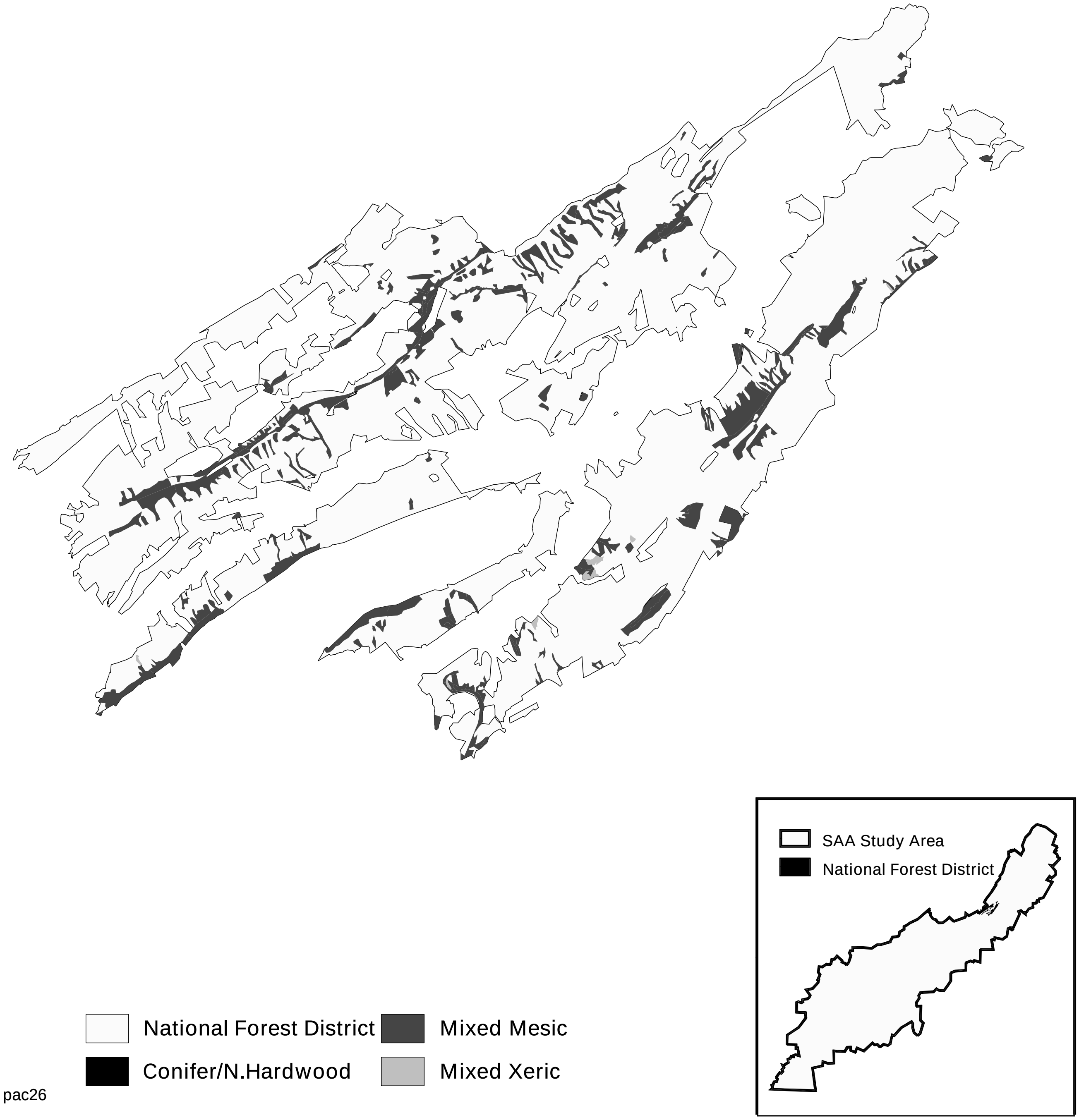

Figure C-26 The spatial distribution of the initial inventory of forest stands selected as possible old growth on national forests in the SAA area for the New Castle Ranger District.

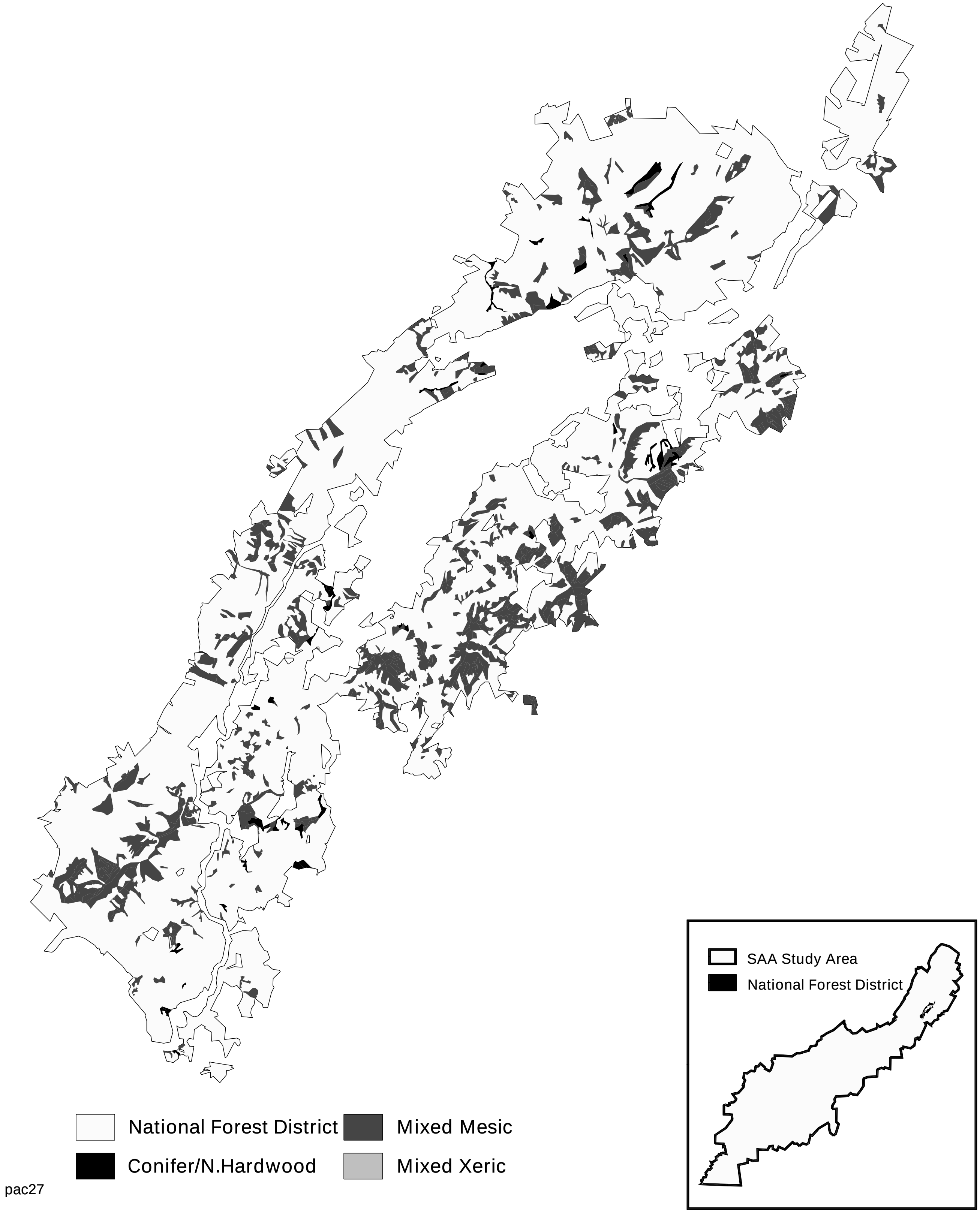

Figure C-27 The spatial distribution of the initial inventory of forest stands selected as possible old growth on national forests in the SAA area for the Pedlar Ranger District.

Figure C-28 The spatial distribution of the initial inventory of forest stands selected as possible old growth on national forests in the SAA area for the Warm Springs Ranger District.

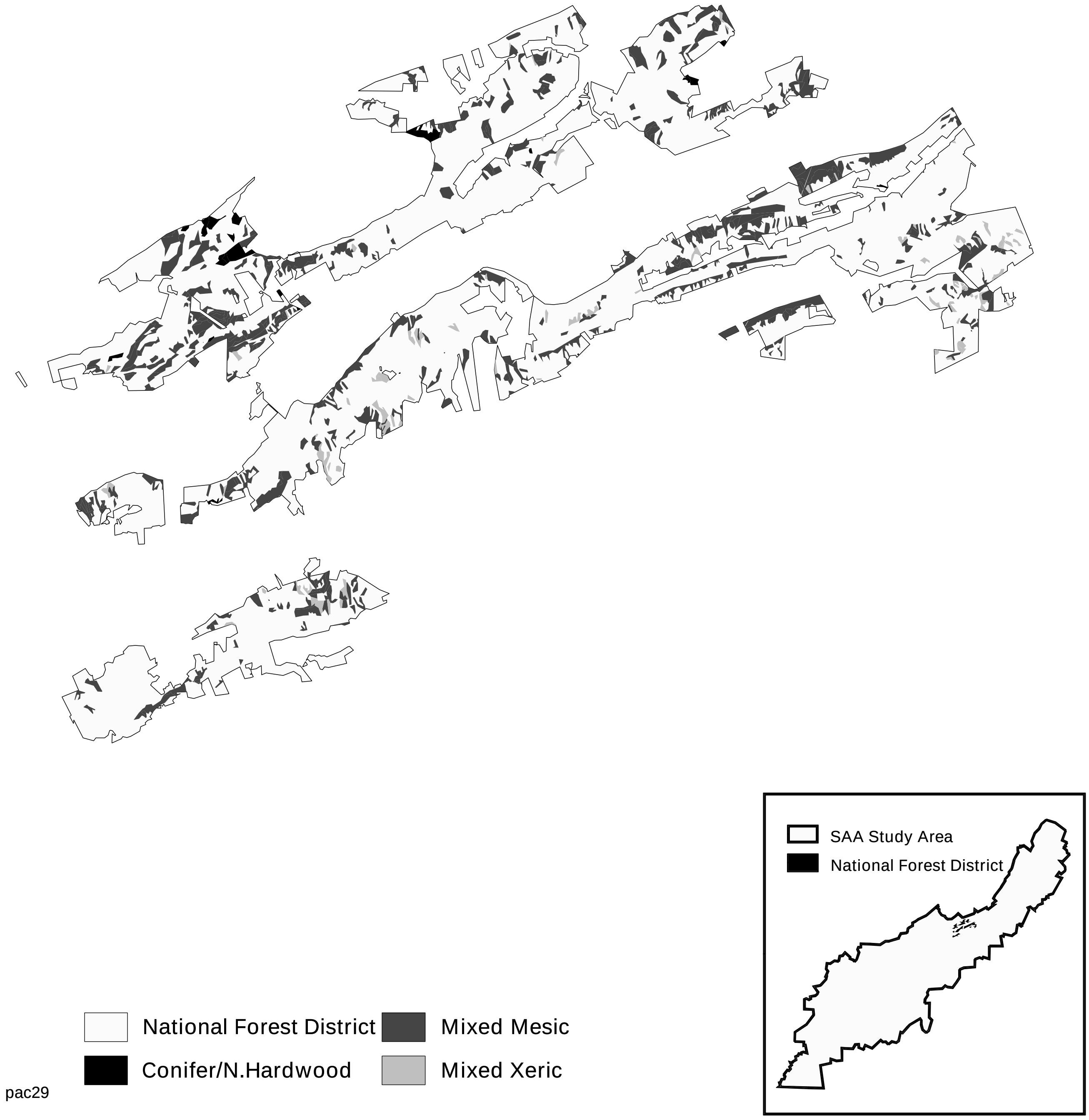

Figure C-29 The spatial distribution of the initial inventory of forest stands selected as possible old growth on national forests in the SAA area for the Wythe Ranger District.

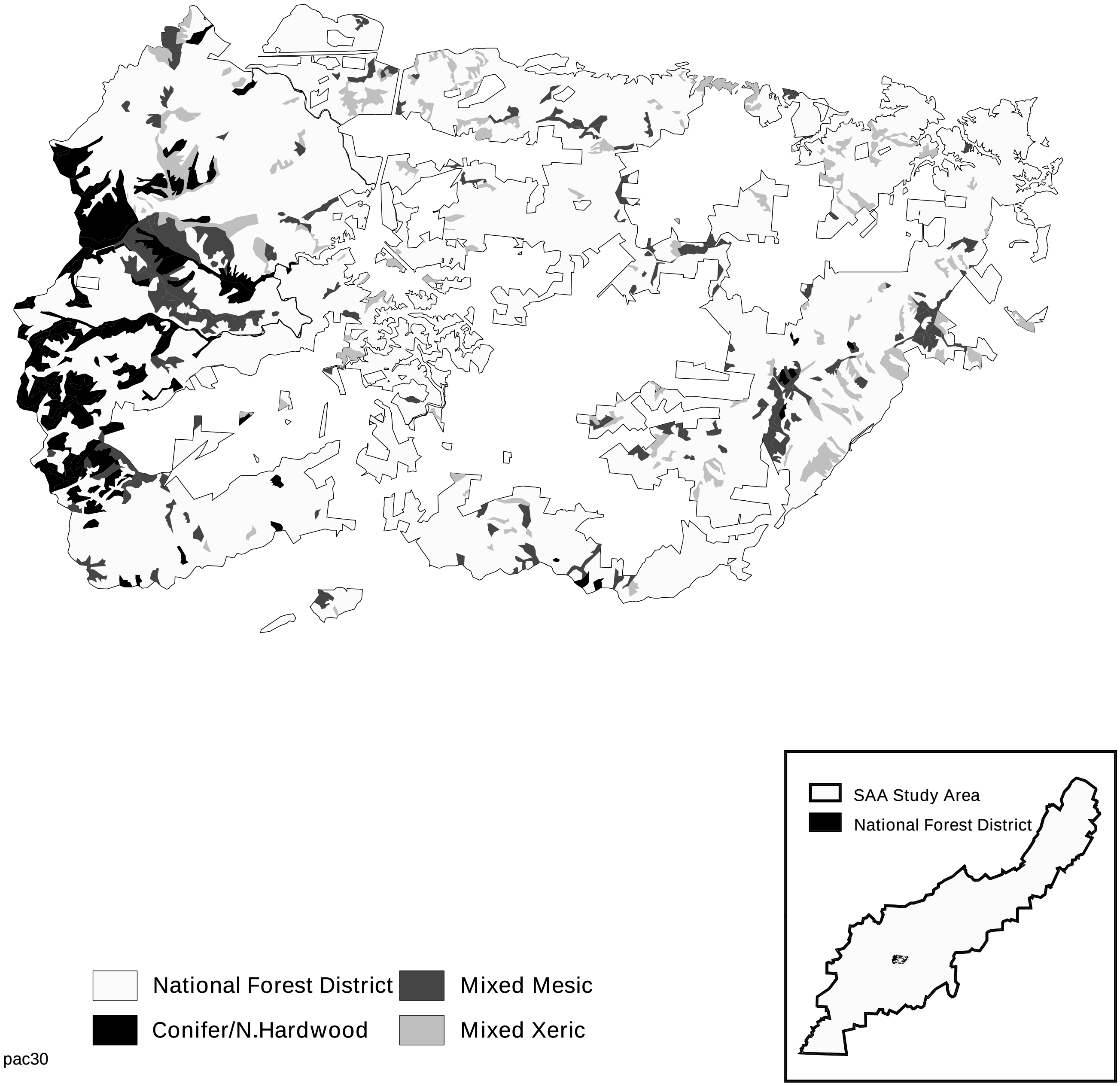

Figure C-30 The spatial distribution of the initial inventory of forest stands selected as possible old growth on national forests in the SAA area for the Cheoah Ranger District.

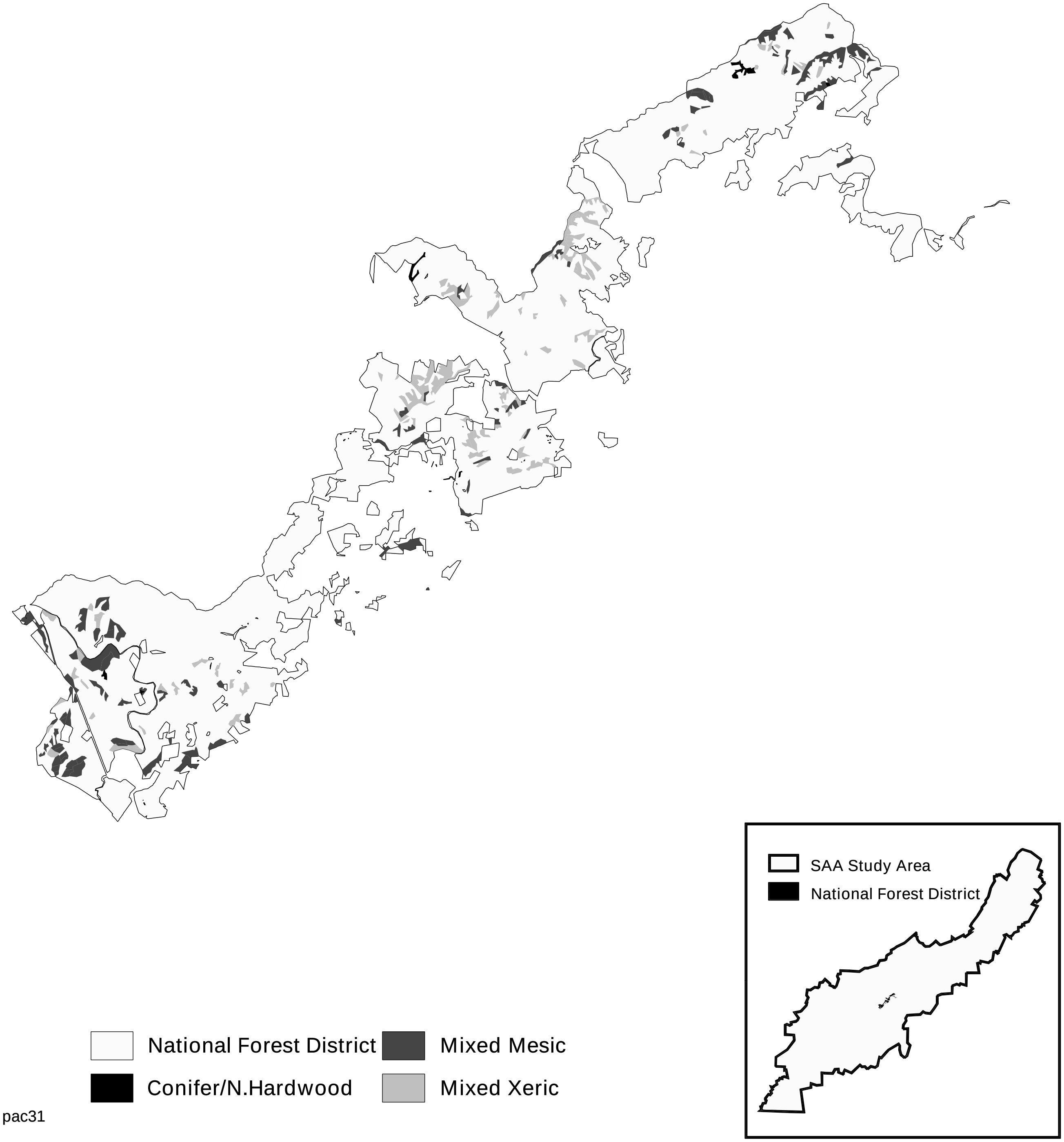

Figure C-31 The spatial distribution of the initial inventory of forest stands selected as possible old growth on national forests in the SAA area for the French Broad Ranger District.

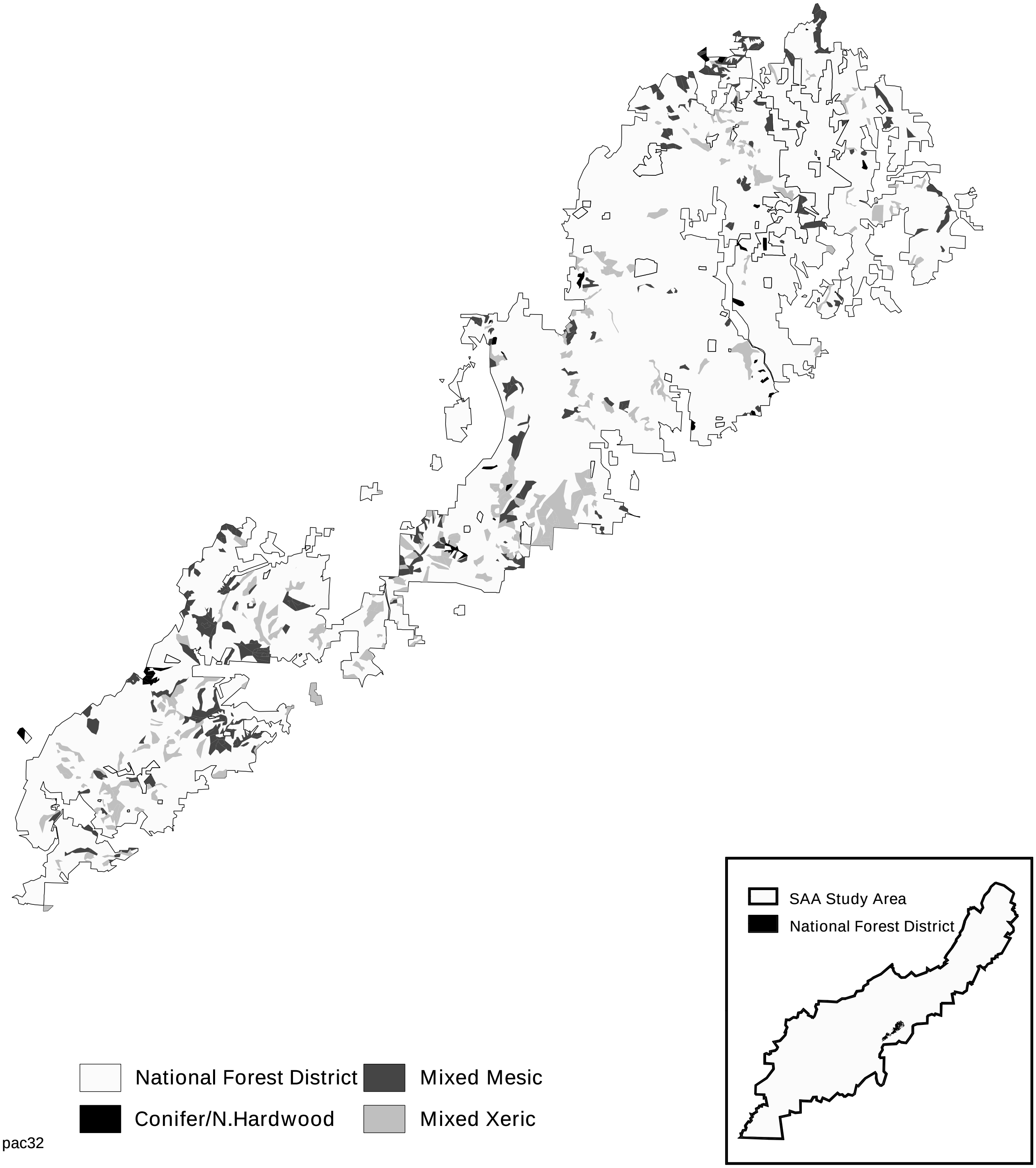

Figure C-32 The spatial distribution of the initial inventory of forest stands selected as possible old growth on national forests in the SAA area for the Grandfather Ranger District.

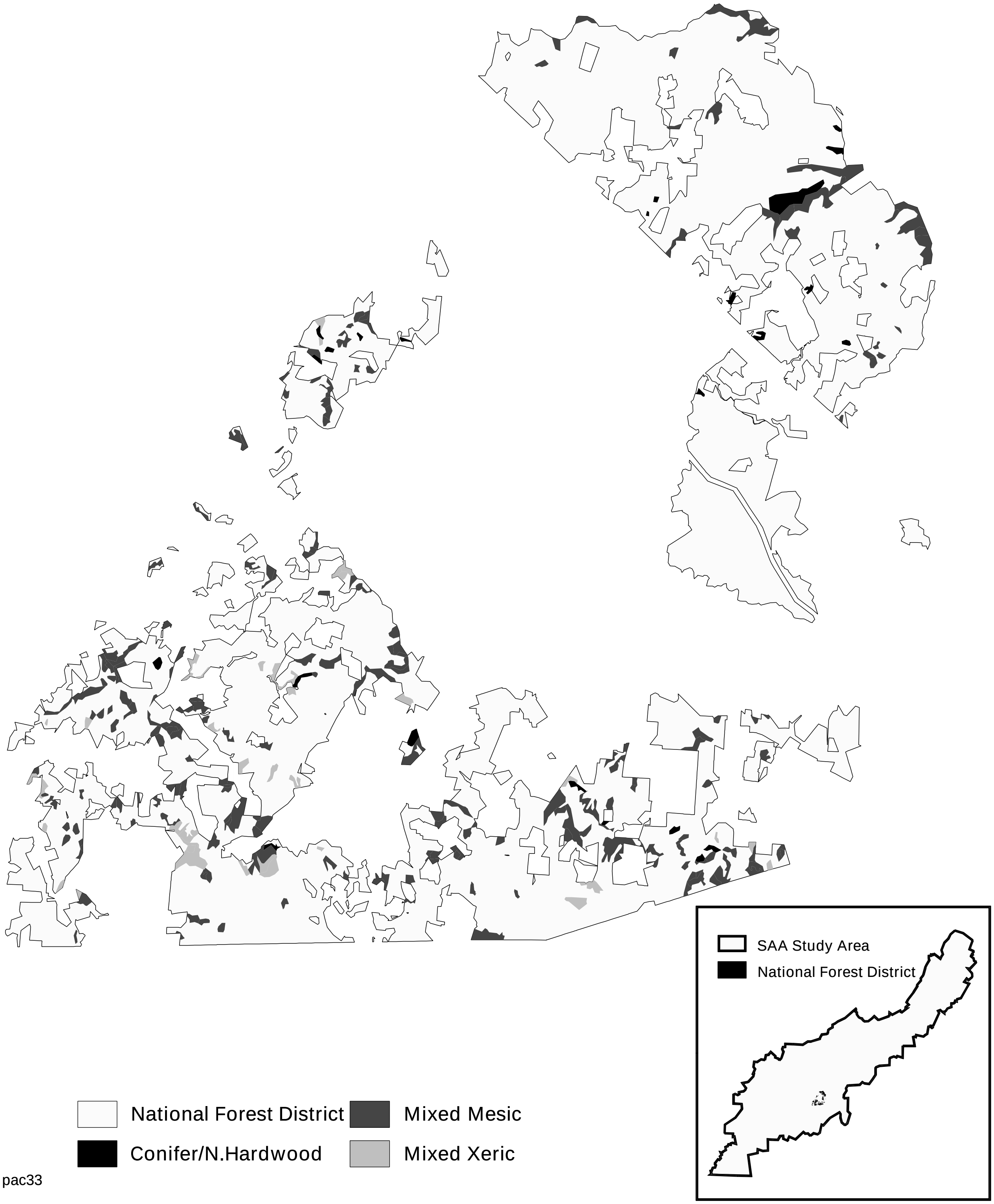

Figure C-33 The spatial distribution of the initial inventory of forest stands selected as possible old growth on national forests in the SAA area for the Highlands Ranger District.

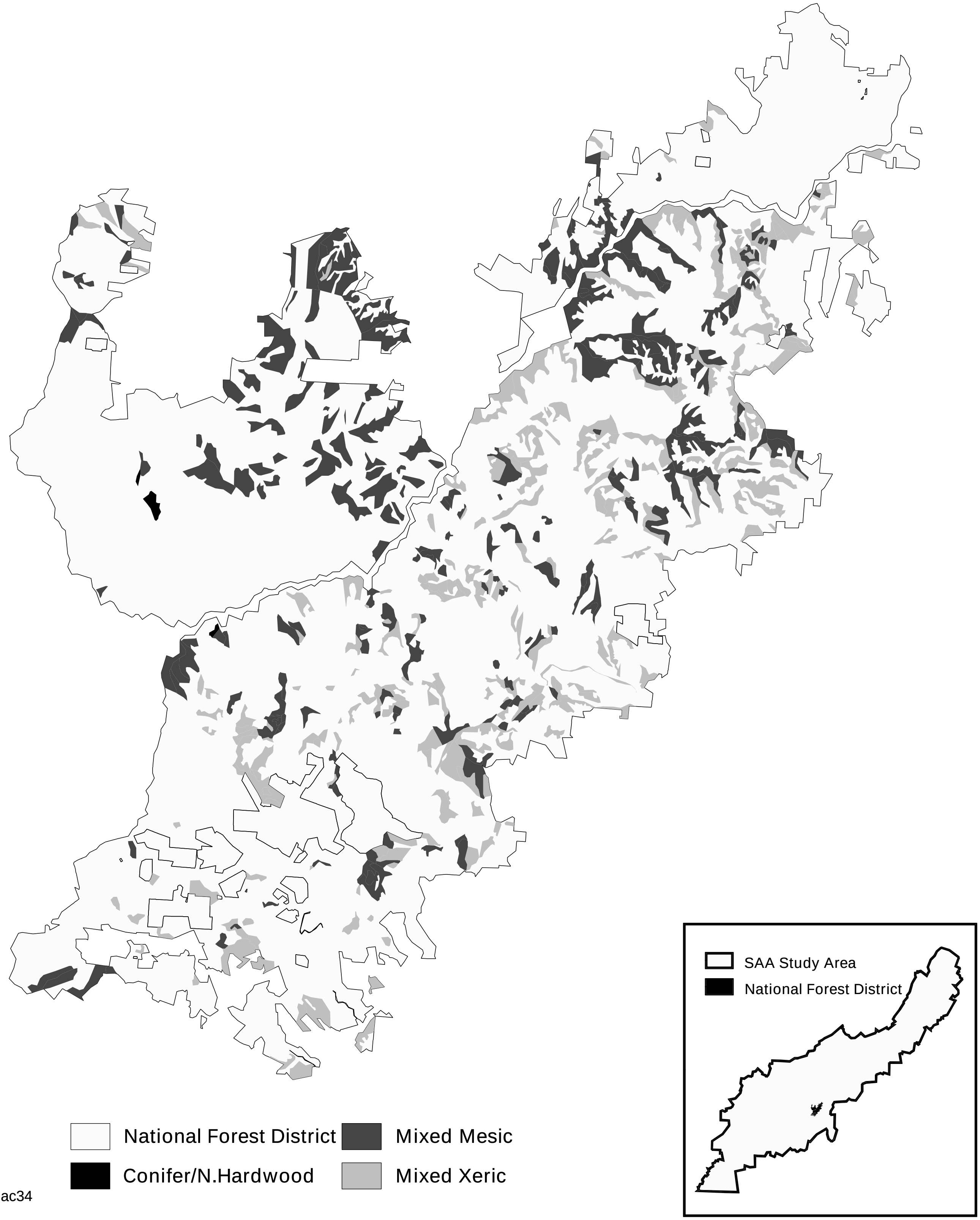

Figure C-34 The spatial distribution of the initial inventory of forest stands selected as possible old growth on national forests in the SAA area for the Pisgah Ranger District.

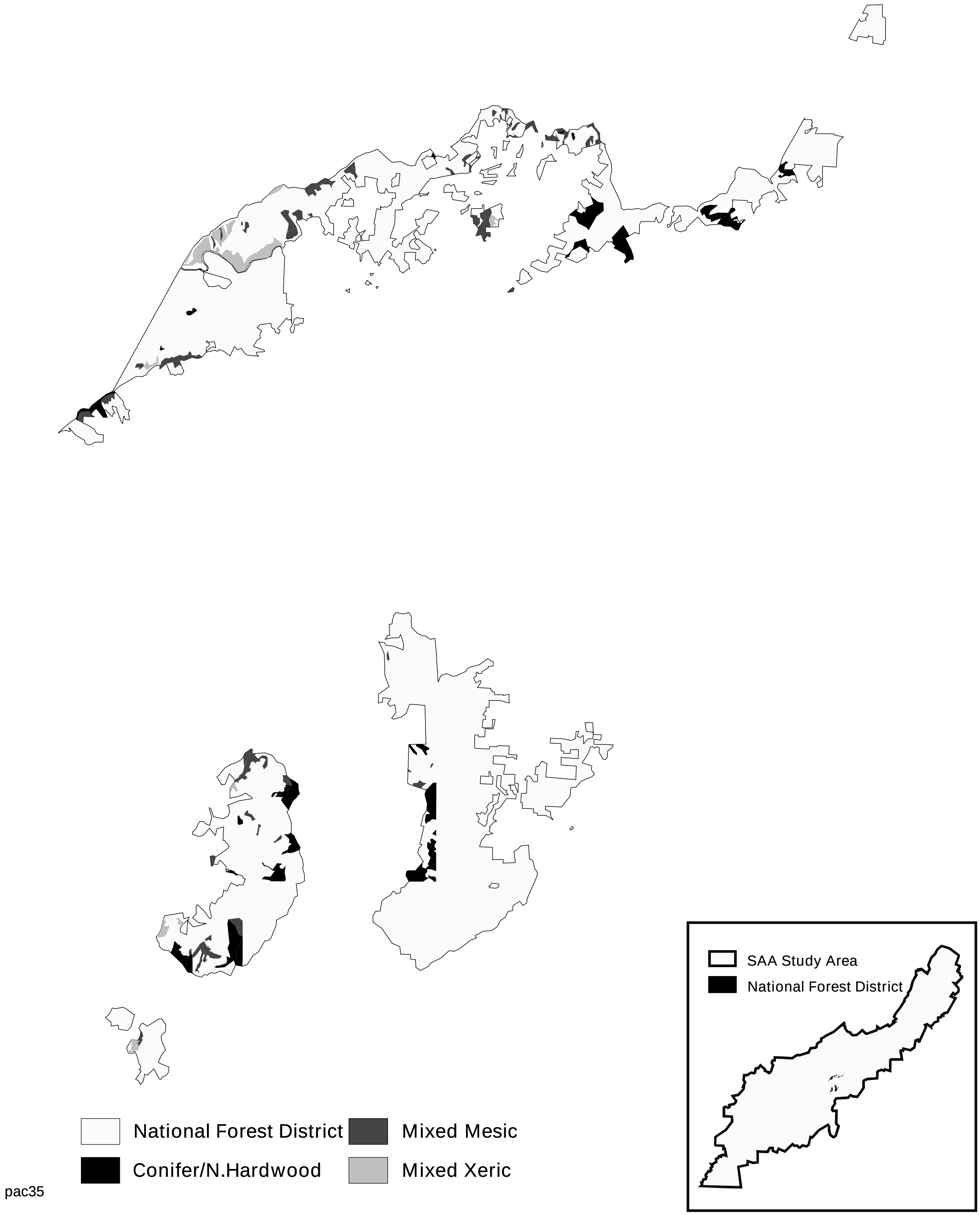

Figure C-35 The spatial distribution of the initial inventory of forest stands selected as possible old growth on national forests in the SAA area for the Toecane Ranger District.

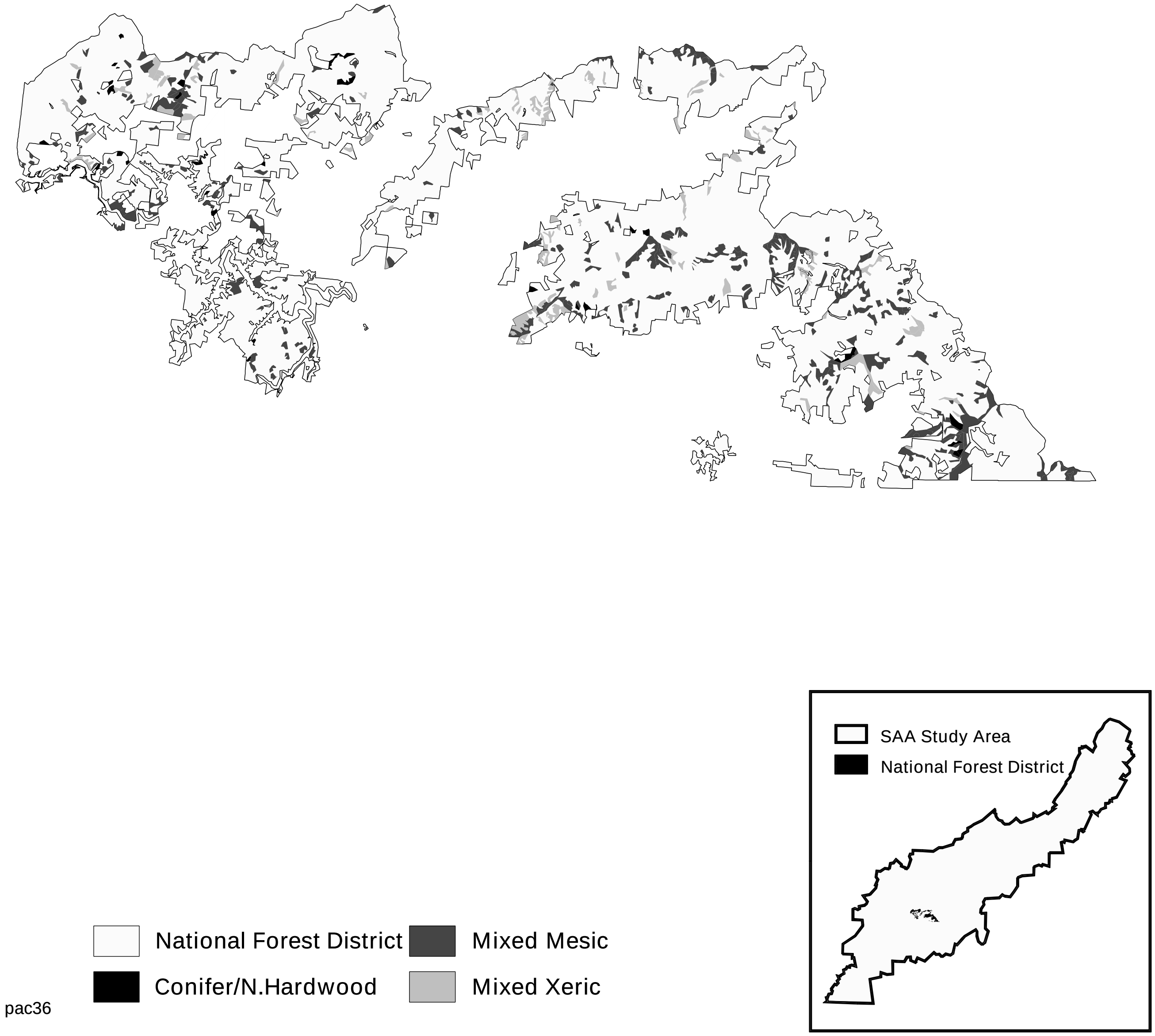

Figure C-36 The spatial distribution of the initial inventory of forest stands selected as possible old growth on national forests in the SAA area for the Tusquitee Ranger District.

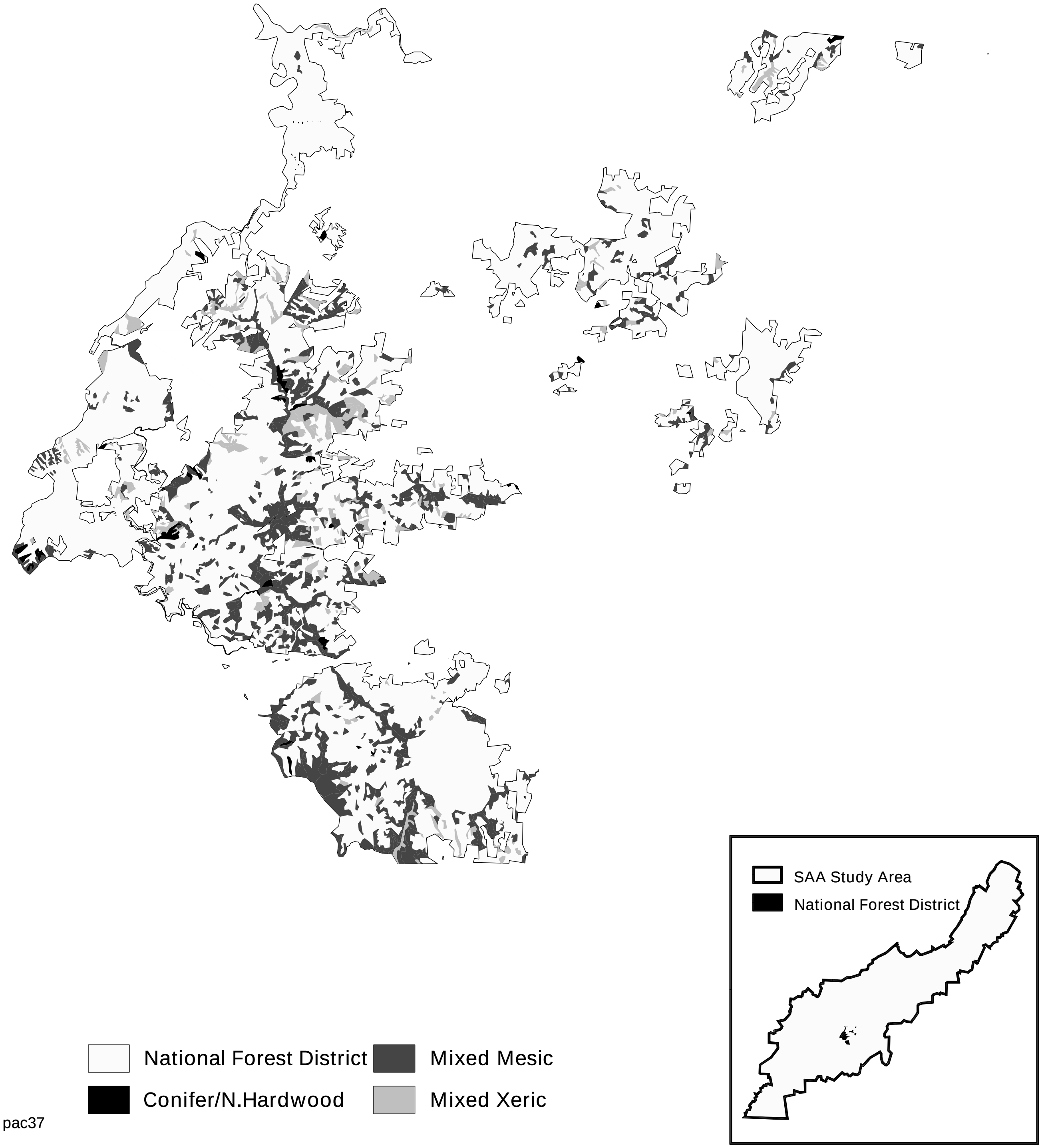

Figure C-37 The spatial distribution of the initial inventory of forest stands selected as possible old growth on national forests in the SAA area for the Wayah Ranger District.

Appendix D

Description of Ecological Units

Following is a brief description of the ecological units in the Southern Appalachian Assessment (SAA) area as they occur in the framework from domain to subsection. Information for domains, divisions, and provinces was obtained from Bailey (1995). Section descriptions are based on McNab and Avers (1994).

The Humid Temperate domain (200) encompasses the SAA area. Precipitation exceeds evapo-transpiration, dominant vegetation is forests of evergreen and deciduous species. Seasons exhibit marked differences in temperature, but precipitation is generally well distributed throughout the year. This domain is divided into two divisions based on influence of frost. They are the hot continental division and the sub-tropical division.

The Hot Continental division (220) is characterized by hot summers and cool winters with a 3- to 6-month growing season. Snow cover can be long-lasting with deep accumulations in northern areas. The prevailing climate during the growing season is dry, especially in late summer. Vegetation is mainly broadleaf deciduous trees. Soils are chiefly Inceptisols, Ultisols and Alfisols. This division consists of two provinces.

The Eastern Broadleaf Forest (Oceanic) province (221) ranges in altitude from 1,000 to 3,000 feet. Terrain is hilly with some small mountains. Winters are cold, and summers are warm. There is more precipitation in summer than in winter. Vegetation is mainly deciduous broadleaf hardwoods, with pines on drier, more exposed ridges. In the SAA area, this province has three sections.

The Northern Cumberland Plateau section (221H) ranges from about 1,200 to 2,000 feet and consists of low hills. Soils are mostly Udults that have a mesic temperature regime, a udic moisture regime, and mixed or siliceous mineralogy. Predominant vegetation is mixed mesophytic forest and Appalachian oak forests. Principal species include oaks and hickories. This section has been subdivided into two subsections.

The Southwestern Escarpment subsection (221Hc) features high hills from 500 to 1,000 feet. Soils are Hapludults or Dystrochrepts with mixed mineralogy, a mesic temperature and a udic moisture regime. Principal species include chestnut oak, northern red oak, pignut and mockernut. Mean annual precipitation is about 46 inches, and the mean annual temperature is 55 degrees F.

The Sequatchie Valley North subsection (221Hd) features open low mountains from 1,000 to 3,000 feet. The features and vegetation are similar to that of the Southern Ridge and Valley section. Soils are mostly Paleudults or Dystrochrepts with kaolinitic or mixed mineralogy and an udic moisture regime. They have a temperature regime ranging from thermic at lower elevations to mesic at higher elevations. Principal species include southern red oak, white oak (post oak), and hickories. Principal species include southern red oak, white oak, mockernut, and pignut. Mean annual precipitation is 36 to 55 inches. Mean annual temperature ranges from 55 to 61 degrees F.

The Southern Cumberland Mountains section (221I) consists of low mountains and open hills from 1,200 to 3,000 feet. Soils are mainly Udults with a mesic temperature regime, an udic moisture regime and mixed mineralogy. The oak-hickory forest type dominates vegetation, with oaks as the main species. Precipitation averages 46 inches: temperature averages about 55 degrees F. This section has two subsections.

The Pine Mountain Thrust Block subsection (221Ia) soils are Hapludults that have mixed mineralogy and mesic temperature and udic moisture regimes. Principal species include chestnut oak, red oak, and hickories. Principal species include chestnut oak, mockernut, and pignut. Mean annual precipitation is approximately 46 inches; mean annual temperature is 55 degrees F.

The Cleveland subsection (221Ib) features mountains from 1,000 to 3,000 feet. Soils are primarily Dystrochrepts with mixed mineralogy and mesic temperature and udic moisture regimes. Principal species include chestnut oak, northern red oak, white oak, southern red oak, black oaks, and mockernut. The mean annual precipitation is 46 inches and the mean annual temperature is 55 degrees F.

The Central Ridge and Valley section (221J) is a distinctive, repeating pattern of parallel ridges and valleys that have been strongly dissected by differential erosion and mass wasting. Soils are Udults with smaller amounts of Ochrepts and Paleudults. Soils depths range from shallow on sandstone ridges to deep in limestone valleys. Vegetation is Appalachian oak forest, but much of the section has been cleared for pastures, agriculture, and urban land use. This section has been subdivided into three subsections.

The Rolling Limestone Hills subsection (221Ja) has open hills from 300 to 500 feet in elevation. Soils are Paleudults, Dystrochrepts, and Hapludults that have kaolinitic or mixed mineralogy, a thermic temperature and udic moisture regimes. Principal species include chestnut and scarlet oaks, with red-cedar on soils derived from limestone. Shortleaf and pitch pines are present on disturbed sites. Mean annual precipitation ranges from 36 to 55 inches; mean annual temperature from 55 to 66 degrees F.

The Sandstone Hills subsection (221Jb) consists of open hills from 300 to 500 feet. Soils are Rhodudults, Paleudults, and Hapludults with oxidic, kaolinitic, and mixed mineralogy. They have a thermic temperature and udic moisture regime. Principal species include scarlet, chestnut, and blackjack oaks. Eastern red-cedar is commonly found on limestone soils. Mean annual precipitation ranges from 36 to 55 inches; mean annual temperature ranges from 55 to 61 degrees F.

The Holston Valley subsection (221Jc) features open hills 300 to 500 feet in elevation. Soils consist of Eutrochrepts, Hapludults, and Dystrochrepts with mixed mineralogy and mesic temperature and udic moisture regimes. Principal species include black oak, white oak, pignut and shagbark. Mean annual precipitation is 36 to 55 inches.

The Central Appalachian Broadleaf Forest-Coniferous Forest-Meadow province (M221) is in the predominately mountainous area of the Central and Southern Appalachian Mountains. Altitudes range from 700 to more than 6,000 feet. Climate is characterized by short, mild winters and long, warm summers. Precipitation is evenly distributed throughout the year, but varies widely, less than 33 inches in the Massanutten Mountains of the northern Shenandoah Valley to more than 100 inches along parts of the Blue Ridge escarpment, North Carolina. Vegetation is mainly broadleaf deciduous species with conifers on ridge crests and southern exposures. Vegetation also exhibits zonation with increasing altitudes and precipitation. Forests dominated by spruce and fir occur above 5,000 feet. This province is subdivided into four sections:

The Northern Ridge and Valley section (M221A) is characterized by a series of parallel, generally narrow valleys and mountain ranges. Elevations range from 300 to 4,000 feet. Soils are mostly Ultisols, Alfisols and Inceptisols with a mesic temperature regime and udic moisture regime. Vegetation is Appalachian oak forest, oak-hickory forest, with some northern hardwoods and mixtures of yellow pine on southern exposures. Precipitation averages 30 to 45 inches, but increases to 100 inches along the escarpment of the Allegheny Plateau on the western edge of the section. Annual temperature ranges from 4 to 14 degrees F. This section has been delineated into two subsections.

The Appalachian Ridges subsection (M221Aa) consists of plains and low mountains ranging from 1,000 to 4,500 feet. Soils are Dystrochrepts and Fragiudults with mixed mineralogy, mesic temperature and udic moisture regimes. Principal species include chestnut oak, white oak, northern red oak and black oak, mockernut, and white and pitch pines. Table Mountain pine is also present in localized areas. Mean annual precipitation ranges from 30 to 55 inches; mean annual temperature ranges from 60 to 62 degrees F.

The Great Valley of Virginia subsection (M221Ab) is dominated by a broad valley with low hills and mountains having elevations of 700 to 3,000 feet. Numerous caves and extensive karst areas are found in this subsection. Soils are Paleudults and Hapludults with mixed mineralogy, mesic temperature and udic moisture regimes. Principal species include white, chestnut, red and black oaks, mockernut and pignut. Other species include shortleaf and pitch pines along with black walnut, elm and sycamore along river courses. Historically this subsection likely had extensive acreage in grasslands and savannas interspersed with wetlands. The mean annual precipitation is 33 to 50 inches and the mean annual temperature is 46 to 55 degrees F.

The Allegheny Mountains section (M221B) is a maturely dissected plateau characterized by high, sharp ridges and narrow valleys. Elevation ranges from 500 to 800 feet. Soils are mostly Ultisols, Inceptisols, and Alfisols. The temperature regime is mostly mesic although extensive areas of frigid soils occur at the highest elevations. Vegetation is predominantly red spruce northern hardwoods, mixed mesophytic, and oak-hickory-pine. Several areas of high-elevation wetlands occur within the section. Precipitation averages 45 to 60 inches annually. Only the eastern-most edge of this section is in the SAA area.

The Northern Cumberland Mountains section (M221C) has elevations ranging from 2,000 to 2,600 feet. Landforms are mainly low mountains with a folded, faulted, and uplifted structure. Soils are mostly Ochrepts, Udults, and Aquults. The temperature regime of soils is mesic; the moisture regime is udic or aquic. Vegetation is mostly mixed mesophytic forest, Appalachian oak forest, and northern hardwoods. Precipitation averages 34 to 47 inches. A single subsection has been delineated within the SAA area.

The Central Coalfields subsection (M221Ca) consists of low mountains ranging from 1,000 to 3,000 feet. Soils are Dystrochrepts and Hapludults of mixed mineralogy, with mesic temperature and udic moisture regimes. Principal species include chestnut oak, white oak, and black oak. Sycamore, deciduous magnolias, and yellow-poplar are common along major river bottoms. The mean annual precipitation is approximately 46 inches, and mean annual temperature is around 55 degrees F.

The Blue Ridge Mountains section (M221D) contains the highest peaks in the eastern United States, with altitudes over 6,000 feet. Landforms consist of mountain peaks and ranges separated by intermountain basins. Vegetation is mainly mixed mesophytic and oak-pine mixtures at lowest elevations, oaks at moderate elevations, and spruce-fir on highest peaks. Precipitation averages 40 to 50 inches annually, but ranges to 100 inches or more on the highest peaks of the Southern Blue Ridge Escarpment. Four subsections have been delineated in this section:

The Northern Blue Ridge Mountains subsection (M221Da) consists of narrow mountains from 1,000 to 4,000 feet. Soils are Kanhapludults and Dystrochrepts with kaolinitic and mixed mineralogy, with mesic temperature and udic moisture regimes. Principal species include chestnut oak and scarlet oak on the uplands. Yellow-poplar, black cherry, red maple, and black birch are common on mesic sites. The mean annual precipitation ranges from 40 to 50 inches, and the mean annual temperature ranges from 50 to 61 degrees F.

The Central Blue Ridge Mountains subsection (M221Db) is characterized by low, plateau-like mountains ranging from 1,000 to 3,600 feet. Soils are Hapludalfs, Hapludults, and Kanhapludults with mixed and kaolinitic mineralogy and mesic temperature and udic moisture regimes. Principal species include scarlet, chestnut, white, and black oak with an abundance of white pine. There was an abundance of American chestnut before the blight. Historically, numerous small wetlands occurred. The mean annual precipitation ranges from 40 to 50 inches, and mean annual temperature ranges from 50 to 60 degrees F.

The Southern Blue Ridge Mountains subsection (M221Dc) consists of mountains from 2,000 to 6,000 feet. Soils are Dystrochrepts, Kanhapludults, and Hapludults with mixed, kaolinitic and micaeous mineralogy, respectively with mesic temperature and udic moisture regimes. Common species are white and scarlet oak, with mixed mesophytic and yellow-poplar at low elevations with pitch pine on dryer and disturbed sites, and chestnut oak, and northern red oak at moderate elevations. Red spruce and Fraser fir occur at the highest elevations, above 5,500 feet. The mean annual precipitation ranges from 40 to 50 inches, and the mean annual temperature ranges from 50 to 60 degrees F.

The Metasedimentary Mountains subsection (221Dd) features mountains ranging from 2,000 to over 6,000 feet. Soils are Dystrochrepts, Kanhapludults, and Hapludults with mixed and kaolinitic mineralogy and mesic temperature and udic moisture regimes. Common species include white, chestnut, and scarlet oaks on dry sites. Mesophytic species, such as yellow-poplar and Canadian hemlock are on moist sites. Spruce-fir vegetation occurs over 5,500 feet. The mean annual precipitation ranges from 50 to 60 inches, and the mean annual temperature ranges from 50 to 60 degrees F.

The Subtropical division (230) has a summer climate of high humidity. Winters are mild with only brief periods of prolonged freezing temperatures; snow occurs, but accumulations are uncommon. Because this division is relatively close to the Atlantic and Gulf coasts, air masses are generally oceanic in origin and summer rainfall is adequate for tree growth during most years. Vegetation is a mixture of deciduous hardwoods and evergreen conifer species. Soils are predominantly Ultisols and many are eroded due to extensive past agriculture. Within the SAA area, one province occurs in this division.

The Southeastern Mixed Forest province (231) consists of the foothills part of the Appalachian Piedmont. Precipitation averages 40 to 60 inches and is generally evenly distributed during the year. Vegetation is dominated by species of southern yellow pines and deciduous hardwoods. Understory vegetation consists of shade-tolerant trees, such as dogwood and sourwood. Soils are typically Ultisols.

The Southern Appalachian Piedmont section (231A) is a region mainly of irregular plains with smaller areas of high hills and tablelands. Elevation ranges from 330 to 1,300 feet. Udults are the predominant soils. In many areas soils are severely eroded as a result of past intensive agricultural practices, especially for cotton production. Vegetation is mostly oak-hickory-pine forest and southern mixed forest. Loblolly pine and southern red oak with an understory of dogwood and sweetgum is a common vegetative community on uplands. The annual precipitation ranges from 45 to 55 inches.

The Midland Plateau Central Uplands subsection (231Aa) is characterized by irregular plains ranging from 100 to 1,300 feet. Soils are Kanhapludults and Rhodudults of kaolinitic mineralogy with thermic temperature and udic moisture regimes. Principal species include white, chestnut, southern red and black oaks; and mockernut, pignut, and shagbark hickories. The mean annual precipitation ranges from 45 to 55 inches, and the mean annual temperature ranges from 57 to 64 degrees F.

The Piedmont Ridge subsection (231Ab) consists of table lands of moderate relief, elevations ranging from 300 to 500 feet. Soils are Kanhapludults, Rhodudults, and Udifluvents with kaolinitic and mixed mineralogy. They have a thermic temperature and a udic moisture regime. Principal species include white, chestnut, southern red, and black oaks; mockernut; and pignut. The mean annual precipitation ranges from 45 to 55 inches, and the mean annual temperature ranges from 57 to 64 degrees F.

The Schist Plains subsection (231Ac) is characterized by table lands of moderate relief with elevations ranging from 300 to 500 feet. Soils are Kanhapludults and Rhodudults with kaolinitic mineralogy and thermic temperature and udic moisture regimes. Principal species include white, chestnut, southern red, and black oaks; and mockernut and pignut hickories. The mean annual precipitation ranges from 45 to 55 inches, and the mean annual temperature ranges from 57 to 64 degrees F.

The Lower Foothills subsection (231Ad) is an area of open high hills with elevations ranging from 500 to 1,000 feet. Soils are Kanhapludults with kaolinitic mineralogy, a thermic temperature regime, and udic moisture regime. Principal species include white, red, black, and chestnut oaks; and mockernut, pignut, and shagbark hickories. The mean annual precipitation ranges from 40 to 55 inches, and the mean annual temperature ranges from 50 to 64 degrees F.

The Schist Hills subsection (231Ag) is characterized by open high hills ranging from 500 to 1,000 feet. Predominant soils are hapludults with micaeous mineralogy, a thermic temperature regime, and udic moisture regime. Principal species include white, red, black, and chestnut oaks; and mockernut, pignut, and shagbark hickories. The mean annual precipitation ranges from 39 to 55 inches, and the mean annual temperature ranges from 50 to 60 degrees F.

The Lynchburg Belt subsection (231Ak) consists of irregular plains with elevations ranging from 100 to 1,300 feet. Soils are Kanhapludults, Hapludults, and Dystrochrepts of kaolinitic and mixed mineralogy with a thermic temperature and udic moisture regimes. Principal species include white, scarlet, red, and black oaks; and mockernut, pignut, and shagbark hickories. Virginia pine is common on disturbed areas. The mean annual precipitation ranges from 50 to 55 inches, and the mean annual temperature ranges from 57 to 64 degrees F.

The Northern Piedmont subsection (231Al) is characterized by plains with high hills from 500 to 1,000 feet. Soils are Hapludults and Dystrochrepts with kaolinitic and mixed mineralogy. They have a mesic temperature regime and udic moisture regime. Principal species include white, scarlet, red, and black oaks; and mockernut, pignut, and shagbark hickories. The mean annual precipitation ranges from 35 to 45 inches, and the mean annual temperature ranges from 50 to 64 degrees F.

The Triassic Basins subsection (231Ap) consists of table lands of moderate relief ranging from 300 to 500 feet. Soils are Hapludults, Hapludalfs, and Dystrochrepts with mixed mineralogy and mesic temperature and udic moisture regimes. Principal species include white oak, red oak, shagbark, pignut, and mockernut hickories on the more mesic sites and post oak and blackjack oak on the xeric sites. The mean annual precipitation ranges from 16 to 45 inches, and the mean annual temperature ranges from 50 to 57 degrees F.

The Southern Cumberland Plateau section (231C) generally consists of open-hill landforms with some table lands and high hills. Soils are mainly Udults and Ochrepts with a udic moisture regime and a thermic temperature regime. Vegetation is mostly oak-hickory-pine forest. Principal species include loblolly pine, sweetgum, water oak, red maple, southern red oak, and white oak. The mean annual precipitation ranges from 50 to 55 inches.

The Table Plateau subsection (231Cc) is characterized by table lands of considerable relief with elevations ranging from 500 to 1,000 feet. Soils are Hapludults and Paleudults of siliceous and mixed mineralogy with thermic temperature and udic moisture regimes. Principal species include white oak, red oak, shagbark, pignut, and mockernut hickories, and eastern red-cedar on limestone soils. Mean annual precipitation ranges from 36 to 55 inches; mean annual temperature ranges from 55 to 61 degrees F.

The Southern Cumberland Valleys subsection (231Cf) is generally table land with considerable relief, elevations ranging from 500 to 1,000 feet. Soils are Paleudults and Dystrochrepts with siliceous mineralogy, a thermic temperature regime, and udic moisture regime. Principal species include white oak, red oak, shagbark, pignut, and mockernut hickories, and eastern red cedar on limestone soils. The mean annual precipitation ranges from 51 to 55 inches, and the mean annual temperature ranges from 60 to 62 degrees F.

The Southern Ridge and Valley section (231D) is an area of folded, faulted and uplifted belts of parallel valleys and ridges. Landforms are mainly plains with hills, with elevations ranging from 650 to 2,000 feet. Soils are mostly Udults with some Ochrepts. Moisture and temperature regimes are udic and thermic or mesic, respectively. Oak-hickory-pine and southern mixed forests form most of the arborescent vegetative communities. Precipitation averages 35 to 55 inches annually.

The Chert Valley subsection (231Da) consists of plains with hills from 300 to 500 feet. Soils are predominantly Paleudults with kaolinitic to siliceous mineralogy, a thermic temperature regime, and udic moisture regime. Principal species include white, chestnut, southern red, and black oaks, and mockernut and pignut hickories. Mean annual precipitation ranges from 36 to 55 inches; mean annual temperature ranges from 55 to 61 degrees F.

The Sandstone, Shale and Chert Ridge subsection (231Db) is characterized by plains and hills from 300 to 500 feet. Soils are Dystrochrepts and Paleudults with siliceous mineralogy with thermic temperature regime and udic moisture regime. Principal species include white, scarlet, red, and black oaks; and mockernut, pignut, and shagbark hickories. Mean annual precipitation ranges from 36 to 55 inches; mean annual temperature ranges from 55 to 61 degrees F.

The Sandstone Ridge subsection (231Dc) is an area of plains with hills from 300 to 500 feet. Limestone outcrops are common. Soils are Dystrochrepts with a siliceous mineralogy, a thermic temperature regime, and an udic moisture regime. Principal species include white, chestnut, southern red, and black oaks, and mockernut and pignut hickories. The dominant species of the mountain longleaf alliance includes longleaf and short leaf pines and post oak and southern red oak. The mean annual precipitation ranges from 51 to 55 inches, and the mean annual temperature ranges from 51 to

60 degrees F.

The Quartzite and Talledega Slate Ridge subsection (231Dd) is characterized by open, high hills from 500 to 1,000 feet. Soils are predominantly Hapludults with micaeous or mixed mineralogy and have a thermic temperature regime and an udic moisture regime. Principal species include chestnut, white, southern red, and black oaks; and mockernut and pignut hickories. Mean annual precipitation ranges from 36 to 55 inches; mean annual temperature ranges from 55 to 61 degrees F.

The Shaley Limestone Valley subsection (121De) consists of plains with hills ranging from 300 to 500 feet. Soils are predominantly Paleudults with siliceous or kaolinitic mineralogy with a thermic temperature regime and a udic moisture regime. The principal species include white oak, scarlet oak, red oak, black oak, and hickories. The mean annual precipitation ranges from 36 to 55 inches, and the mean annual temperature ranges from 55 to 61 degrees F.

Appendix E

Federally Listed Terrestrial Plant and Animal Species

Table E–1 The list of 51 federally listed terrestrial plant and animal species according to groupings based on habitat association.

Scientific Name	Common Name	Taxanomy	Federal Status	Global Rank	Species Group[1]
Antrolana lira	Madison Cave isopod	Invertebrate	T	1	1
Lirceus usdagalun	Lee County Cave isopod	Invertebrate	E		1
Corynorhinus townsend ii virginianus	Virginia Big-eared Bat	Mammal	E		1
Myotis grisescens	Gray Bat	Mammal	E		1
Myotis sodalis	Indiana Bat	Mammal	E		1
Helonias bullata	Swamp pink	Plant	T	3	2
Sagittaria secundifolia	Kral's water-plantain	Plant	T		2
Sarracenia jonesii	Mountain sweet pitcherplant	Plant	E	1	2
Sarracenia oreophila	Green pitcher plant	Plant	E	2	2
Platanthera leucophaea	Eastern prairie fringed orchid	Plant	T	2	4
Scirpus ancistrochaetus	Northeastern bullrush=Barbed bullrush	Plant	E	2	4
Geum radiatum	Spreading avens	Plant	E	1	5
Hedyotis purpurea var. montana	Roan mountain bluet	Plant	E	2	5
Liatris helleri	Heller's blazing star	Plant	T	1	5
Solidago spithamaea	Blue Ridge goldenrod	Plant	T	1	5
Patera clarki nantahala	Noonday globe snail	Invertebrate	T		6
Polygyriscus virginicus	Virginia Fringed Mountain Snail	Invertebrate	E		6
Asplenium scolopendrium var american	Hart's tongue fern	Plant	T	1	6
Echinacea laevigata	Smooth Coneflower	Plant	E	3	6
Sisyrinchium dichotomum	White irisette	Plant	E	1	6
Xyris tennesseensis	Tennessee yellow-eyed grass	Plant	E		6
Plethodon shenandoah	Shenandoah Salamander	Amphibian	E	1	7
Falco peregrinus anatum	American Peregrine Falcon	Bird	E		7
Amphianthus pusillus	Pool Sprite	Plant	T		7
Arabis serotina	Shale barren rock cress	Plant	E	2	7
Arenaria cumberlandensis	Cumberland sandwort	Plant	E		7
Gymnoderma lineare	Rock gnome lichen	Plant	E	2	7
Hudsonia montana	Mountain golden heather	Plant	T	1	7
Iliamna corei	Peter's mountain mallow	Plant	E	1	7
Canis rufus	Red Wolf	Mammal	E		9
Felis concolor cougar	Eastern Cougar	Mammal	E		9
Haliaeetus leucocephalus	Bald Eagle	Bird	T		11
Betula uber	Virginia round-leaf birch	Plant	T	1	11
Cardamine micranthera	Small anthered bittercress	Plant	E	1	11
Clematis socialis	Alabama leather-flower	Plant	E	1	11
Conradina verticillata	Cumberland rosemary	Plant	T		11
Marshallia morhii	Morh's Barbara's buttons	Plant	T		11
Pityopsis ruthii	Ruth's golden aster	Plant	E	1	11
Ptilimnium nodosum	Harperella	Plant	E	2	11
Sagittaria fasciculata	Bunched arrowhead	Plant	E	1	11
Spiraea virginiana	Virginia spiraea	Plant	T	1	11
Plethodon nettingi	Cheat Mountain Salamander	Amphibian	T	3	15
Microhexura montivaga	Spruce-Fir Moss Spider	Invertebrate	E		15
Glaucomys sabrinus coloratus	Carolina Northern Flying Squirrel	Mammal	E		15
Glaucomys sabrinus fuscus	Virginia Northern Flying Squirrel	Mammal	E		15
Picoides borealis	Red Cockaded Woodpecker	Bird	E		17
Apios priceana	Price's potato-bean	Plant	T		18
Hexastylis naniflora	Dwarf-flowered heartleaf	Plant	T	2	18
Isotria medeoloides	Small whorled pogonia	Plant	E	3	18
Scutellaria montana	Large- flowered skullcap	Plant	E	2	18
Trillium persistens	Persistent trillium	Plant	E		18

[1] Species Group Codes
1 = Cave Habitats
2 = Mountain Bogs
3 = Spray Cliffs
4 = Fen or Pond Wetlands
5 = High Elevation Balds
6 = High pH or Mafic Habitats
7 = Rock Outcrop and Cliffs
8 = Early Successional Habitats
9 = Wide Ranging Area Sensitive Species
10 = Mid– to Late–Successional Forest Species
11 = Seep, Spring, and Streamside Habitat
12 = Habitat Generalist
13 = Area Sensitive Deciduous Forest
14 - General High Elevation Habitats
15 = High Elevation Spruce–Fir Forest
16 = Bottomland Forests
17 = Southern Yellow Pine Habitats
18 = Mixed Mesic Habitats
19 = Mixed Xeric Habitats

Appendix F

Terrestrial Plant and Animal Species with Viability Concern

Table F–1 The list of 366 terrestrial plant and animal species with viability concern for the Southern Appalachian Assessment area sorted according to groupings based on habitat association.

Scientific Name	Common Name	Taxa	Federal Status	Global Rank	Species Group[1]
Gyrinophilus palleucus	Tennessee Cave Salamander	Amphibian	2		1
Gyrinophilus subterraneus	West Virginia Spring Salamander	Amphibian	2		1
Amerigoniscus henroti	Powell Valley Terrestrial Cave Isopod	Invertebrate		1	1
Apochthonius coecus	A cave pseudoscorpion	Invertebrate		1	1
Apochthonius holsingeri	A cave pseudoscorpion	Invertebrate		1	1
Arianops jeanneli	A cave pselaphid beetle	Invertebrate		1	1
Arrhopalites clarus	A cave springtail	Invertebrate		1	1
Caecidotea henroti	Henrot's cave isopod	Invertebrate		2	1
Caecidotea holsingeri	Greenbriar Valley cave isopod	Invertebrate		3	1
Caecidotea incurva	Incurved cave isopod	Invertebrate		2	1
Caecidotea pricei	Price's cave isopod	Invertebrate		3	1
Caecidotea sinuncus	An isopod	Invertebrate		1	1
Caecidotea vandeli	Vandel's cave isopod	Invertebrate		2	1
Chitrella superba	A cave pseudoscorpion	Invertebrate		1	1
Foveacheles paralleloseta	A cave pseudoscorpion	Invertebrate		1	1
Islandiana speophila	Cavern sheetweb spider	Invertebrate		1	1
Kleptochthonius lutzi	A cave pseudoscorpion	Invertebrate		1	1
Kleptochthonius proximosetus	A cave pseudoscorpion	Invertebrate		1	1
Kleptochthonius regulus	A cave pseudoscorpion	Invertebrate		1	1
Kleptochthonius similis	A cave pseudoscorpion	Invertebrate		1	1
Kleptochthonius species 1	A cave pseudoscorpion	Invertebrate		1	1
Lirceus culveri	Rye cove isopod	Invertebrate	2		1
Litocampa barringerorum	A cave dipluran	Invertebrate		1	1
Litocampa bifurcata	A cave dipluran	Invertebrate			1
Litocampa cookei	A cave dipluran	Invertebrate			1
Litocampa holsingeri	A cave dipluran	Invertebrate		2	1
Macrocotyla hoffmasteri	Hoffmaster's cave flatworm	Invertebrate		3	1
Microcreagris valentinei	A cave pseudoscorpion	Invertebrate		1	1
Miktoniscus racovitzae	Racovitza's Terrestrial Cave Isopod	Invertebrate		2	1
Mundochthonius holsingeri	A cave pseudoscorpion	Invertebrate		1	1
Nampabius turbator	A cave centipede	Invertebrate			1
Nesticus carolinensis	Linville Cavern spider	Invertebrate		1	1
Nesticus cooperi	Lost Nantahala Cave Spider	Invertebrate	2		1
Nesticus crosbyi	A nesticid spider	Invertebrate		1	1
Nesticus holsingeri	Holsinger's Cave spider	Invertebrate		2	1
Nesticus mimus	A cave spider	Invertebrate		2	1
Nesticus paynei	A cave spider	Invertebrate		2	1
Nesticus sheari	A nesticid spider	Invertebrate		2	1
Nesticus silvanus	A nesticid spider	Invertebrate		3	1
Nesticus tennesseensis	A cave spider	Invertebrate		2	1
Phanetta subterranea	A spider	Invertebrate		3	1
Poecilophysis extraneostella	A cave mite	Invertebrate		2	1
Poecilophysis weyerensis	A cave mite	Invertebrate		2	1
Pseudanophthalmus avernus	Avernus Cave beetle	Invertebrate	2	1	1
Pseudanophthalmus cordicollis	Little Kennedy Cave beetle	Invertebrate	2	1	1
Pseudanophthalmus deceptivus	Deceptive Cave beetle	Invertebrate	2	1	1
Pseudanophthalmus delicatus	A cave beetle	Invertebrate		2	1
Pseudanophthalmus egberti	New River Valley Cave beetle	Invertebrate	2	1	1
Pseudanophthalmus gracilis	A cave beetle	Invertebrate		1	1
Pseudanophthalmus hadenoecus	Timber ridge cave beetle	Invertebrate	2	1	1
Pseudanophthalmus hirsutus	Lee County Cave beetle	Invertebrate	2	1	1
Pseudanophthalmus hoffmani	A cave beetle	Invertebrate		1	1
Pseudanophthalmus holsingeri	Holsinger's Cave beetle	Invertebrate	1	1	1
Pseudanophthalmus hubbardi	Hubbard's Cave beetle	Invertebrate	2	1	1
Pseudanophthalmus hubrichti	Hubricht's Cave beetle	Invertebrate	2	1	1
Pseudanophthalmus intersectus	Crossroads Cave beetle	Invertebrate	2	1	1
Pseudanophthalmus limicola	Mud-dwelling cave beetle	Invertebrate	2	1	1
Pseudanophthalmus longiceps	Long-headed cave beetle	Invertebrate	2	1	1
Pseudanophthalmus nelsoni	Nelson's Cave Beetle	Invertebrate	2	1	1
Pseudanophthalmus nickajackensis	Nickajackensis cave beetle	Invertebrate			1
Pseudanophthalmus parvicollis	Thin-neck cave beetle	Invertebrate	2	1	1
Pseudanophthalmus paulus	Nobletts Cave beetle	Invertebrate	2		1
Pseudanophthalmus paynei	Paynes Cave beetle	Invertebrate	2		

Table F–1 (cont.) The list of 366 terrestrial plant and animal species with viability concern for the Southern Appalachian Assessment area sorted according to groupings based on habitat association.

Scientific Name	Common Name	Taxa	Federal Status	Global Rank	Species Group[1]
Pseudanophthalmus petrunkevitchi	Petrunkevitch's cave beetle	Invertebrate	2	1	1
Pseudanophthalmus pontis	Natural Bridge Cave beetle	Invertebrate	2	1	1
Pseudanophthalmus potomaca potomaca	South Branch Valley cave beetle	Invertebrate	2	1	1
Pseudanophthalmus potomaca senecae	Seneca cave beetle	Invertebrate	2	1	1
Pseudanophthalmus praetermissus	Overlooked Cave beetle	Invertebrate	2	1	1
Pseudanophthalmus punctatus	Spotted Cave beetle	Invertebrate	2	1	1
Pseudanophthalmus pusio	A cave beetle	Invertebrate		1	1
Pseudanophthalmus quadratus	Straley's Cave beetle	Invertebrate	2	1	1
Pseudanophthalmus rotundatus	A cave beetle	Invertebrate		1	1
Pseudanophthalmus sanctipauli	Saint Paul Cave beetle	Invertebrate	2	1	1
Pseudanophthalmus sericus	Silken cave beetle	Invertebrate	2	1	1
Pseudanophthalmus sidus	Meredith Cave beetle	Invertebrate	2		1
Pseudanophthalmus species 10	A cave beetle	Invertebrate		1	1
Pseudanophthalmus species 11	A cave beetle	Invertebrate		1	1
Pseudanophthalmus species 4	A cave beetle	Invertebrate		1	1
Pseudanophthalmus species 5	A cave beetle	Invertebrate		1	1
Pseudanophthalmus species 6	A cave beetle	Invertebrate		1	1
Pseudanophthalmus species 7	A cave beetle	Invertebrate		1	1
Pseudanophthalmus species 8	A cave beetle	Invertebrate		1	1
Pseudanophthalmus species 9	A cave beetle	Invertebrate		1	1
Pseudanophthalmus thomasi	Thomas' Cave beetle	Invertebrate	2	1	1
Pseudanophthalmus vicarius	A cave beetle	Invertebrate	2	2	1
Pseudanophthalmus virginicus	Maiden Spring Cave beetle	Invertebrate	2	1	1
Pseudosinella hirsuta	A cave springtail	Invertebrate		1	1
Pseudotremia armesi	A millipede	Invertebrate		2	1
Pseudotremia lusciosa	Germany Valley cave millipede	Invertebrate		1	1
Pseudotremia momus	A millipede	Invertebrate		2	1
Pseudotremia princeps	South Branch Valley cave millipede	Invertebrate		1	1
Pseudotremia tuberculata	A millipede	Invertebrate		2	1
Rhagidia varia	A cave mite	Invertebrate		3	1
Sphalloplana chandleri	Chandler's planarian	Invertebrate		1	1
Sphalloplana consimilis	Powell Valley planarian	Invertebrate		1	1
Sphalloplana virginiana	Rockbridge County Cave planarian	Invertebrate	2	1	1
Stygobromus abditus	James cave amphipod	Invertebrate		2	1
Stygobromus baroodyi	Rockbridge County cave amphipod	Invertebrate		2	1
Stygobromus biggersi	Bigger's Cave amphipod	Invertebrate	2	1	1
Stygobromus conradi	Burnsville cove cave amphipod	Invertebrate	2	1	1
Stygobromus cumberlandus	Cumberland cave amphipod	Invertebrate		2	1
Stygobromus ephemerus	Ephemeral cave amphipod	Invertebrate		1	1
Stygobromus estesi	Craig County cave amphipod	Invertebrate		1	1
Stygobromus fergusoni	Montgomery County cave amphipod	Invertebrate		1	1
Stygobromus gracilipes	Shenandoah Valley cave amphipod	Invertebrate		2	1
Stygobromus hoffmani	Alleghany County cave amphipod	Invertebrate		1	1
Stygobromus interitus	New Castle Murder Hole amphipod	Invertebrate		1	1
Stygobromus leensis	Lee County cave amphipod	Invertebrate		1	1
Stygobromus morrisoni	Morrison's cave amphipod	Invertebrate	2	2	1
Stygobromus mundus	Bath County cave amphipod	Invertebrate	2	1	1
Stygobromus pseudospinosus	Luray Caverns amphipod	Invertebrate		1	1
Stygobromus species 7	Sherando Spinosoid amphipod	Invertebrate		1	1
Stygobromus spinosus	Blue Ridge Mountain amphipod	Invertebrate		2	1
Stygobromus stegerorum	Madison Cave amphipod	Invertebrate		1	1
Stylodrilus beattiei	A cave lumbriculid worm	Invertebrate		1	1
Trichopetalum krekeleri	West Virginia Blind cave millipede	Invertebrate		1	1
Myotis austroriparius	Southeastern bat	Mammal	2		1
Myotis leibii	Eastern small-footed bat	Mammal	2		1
Carex barrattii	Barratt's sedge	Plant		3	2
Carex schweinitzii	Schweinitz's sedge	Plant		3	2
Chelone cuthbertii	Cuthbert's turtlehead	Plant		3	2
Helenium brevifolium	Shortleaf sneezeweed	Plant		3	2
Hypericum adpressum	Creeping St. John's-wort	Plant	2	2	2
Ilex collina	Long-stalked holly	Plant		3	2
Juncus caesariensis	New Jersey rush	Plant	2	2	2
Juncus gymnocarpus	Coville's rush	Plant		3	2

Table F–1 (cont.) The list of 366 terrestrial plant and animal species with viability concern for the Southern Appalachian Assessment area sorted according to groupings based on habitat association.

Scientific Name	Common Name	Taxa	Federal Status	Global Rank	Species Group[1]
Marshallia grandiflora	Large-flowered barbara's-buttons	Plant	2	2	2
Parnassia grandifolia	Large-leaved grass-of-parnassus	Plant		2	2
Plantanthera integrilabia	White fringeless orchid	Plant	2	2	2
Poa paludigena	Bog blue grass	Plant	2	3	2
Splachnum pennsylvanicum	Southern dungmoss	Plant		2	2
Clemmys muhlenbergii	Bog Turtle	Reptile	2		2
Bryocrumia vivicolor	Gorge moss	Plant	2	1	3
Chiloscyphus appalachianus	Liverwort	Plant	2	1	3
Eurhynchium pringlei	Pringle's eurhynchium	Plant	2	2	3
Grammitis nimbata	Dwarf polypody fern	Plant	2	3	3
Hymenophyllum tayloriae	Gorge filmy fern	Plant		1	3
Hymenophyllum tunbridgense	Tunbridge fern	Plant	2		3
Lejeunea blomquistii	Liverwort	Plant	2	1	3
Plagiochila austinii	Liverwort	Plant		3	3
Plagiochila caduciloba	Liverwort	Plant	2	2	3
Plagiochila echinata	Liverwort	Plant	2	1	3
Plagiochila sharpii	Liverwort	Plant	2	2	3
Plagiochila sullivantii var. spinigera	Liverwort	Plant	2	2	3
Plagiochila sullivantii var. sullivantii	Liverwort	Plant	2	2	3
Plagiochila virginica var. caroliniana	Liverwort	Plant	2	2	3
Plagiochila virginica var. euryphylla	Liverwort	Plant	2	1	3
Porella appalachiana	Liverwort	Plant	2	1	3
Porella wataugensis	Liverwort	Plant		1	3
Radula voluta	Liverwort	Plant		2	3
Helenium virginicum	Virginia sneezeweed	Plant	1	2	4
Isoetes virginica	Quillwort	Plant	2	1	4
Potamogeton tennesseensis	Tennessee pondweed	Plant		3	4
Cladonia psoromica	Bluff mountain reindeer lichen	Plant	2	1	4
Thryomanes bewickii altus	Appalachian Bewick's Wren	Bird	2		5
Sylvilagus obscurus	Appalachian cottontail	Mammal	2		5
Allium alleghenienses	Allegheny onion	Plant		3	5
Gentiana austromontana	Appalachian gentian	Plant		3	5
Geum geniculatum	Bent avens	Plant	2	1	5
Hypericum buckleyi	Blue Ridge St. John's-wort	Plant		3	5
Hypericum graveolens	Mountain St. John's-wort	Plant		3	5
Hypericum mitchellianum	Mitchell's St. John's-wort	Plant		3	5
Lilium grayi	Gray's lily	Plant	2	2	5
Prenanthes roanensis	Roan rattlesnakeroot	Plant		3	5
Rhododendron carolinianum	Carolina Rhododendron	Plant		3	5
Rhododendron cumberlandense	Cumberland azalea	Plant		2	5
Robinia viscosa var. viscosa	Clammy locust	Plant		3	5
Euchlaena milnei	Looper moth	Invertebrate	2		6
Arabis georgiana	Georgia rockcress	Plant	2	2	6
Arenaria godfreyi	Godfrey's stitchwort	Plant	2	1	6
Aster georgianus	Georgia aster	Plant	2		6
Astragalus neglectus	Cooper's milkvetch	Plant	2	3	6
Aureolaria patula	Spreading false-foxglove	Plant	2	2	6
Carex biltmoreana	Biltmore sedge	Plant		3	6
Clematis addisonii	Addison's leatherflower	Plant	2	2	6
Coreopsis latifolia	Broadleaf coreopsis	Plant		3	6
Delphinium exaltatum	Tall larkspur	Plant	2	3	6
Elymus svensonii	Svenson's wild-rye	Plant	2	2	6
Euphorbia purpurea	Darlington's spurge	Plant	2	3	6
Heuchera longiflora	Long-flowered alumroot	Plant		3	6
Hypericum dolabriforme	Straggling St. John's wort	Plant		3	6
Leavenworthia exigua var. exigua	Glade cress	Plant		3	6
Neviusia alabamensis	Alabama snow wreath	Plant	2	2	6
Orthotrichum keeverae	Keever's bristle-moss	Plant	2	1	6
Paxistima canbyi	Canby's mountain-lover	Plant	2	2	6
Phlox bifida ssp. stellaria	Cleft phlox	Plant	2		6
Plagiochila virginica var. virginica	Liverwort	Plant		2	6
Pycnanthemum curvipes	Tennessee mountain mint	Plant		3	6
Pycnanthemum torrei	Torrey mountain-mint	Plant		2	6

Table F–1 (cont.) The list of 366 terrestrial plant and animal species with viability concern for the Southern Appalachian Assessment area sorted according to groupings based on habitat association.

Scientific Name	Common Name	Taxa	Federal Status	Global Rank	Species Group[1]
Senecio millefolium	Divided-leaf ragwort	Plant	2	2	6
Silene ovata	Mountain catchfly	Plant	2	3	6
Tomanthera auriculata	Auriculate false-foxglove	Plant	2	2	6
Tortula ammonsiana	Ammons' tortula	Plant	2	1	6
Trifolium calcaricum	Running glade clover	Plant	2	1	6
Trillium rugelii	Southern nodding trillium	Plant		3	6
Trillium simile	Sweet white trillium	Plant		3	6
Aneides aeneus	Green Salamander	Amphibian	2		7
Plethodon petraeus	Pigeon Mountain Salamander	Amphibian		1	7
Microtus chrotorrhinus carolinensis	Southern rock vole	Mammal	2	3	7
Neotoma floridana haematoreia	Southern Appalachian eastern woodrat	Mammal	2		7
Neotoma magister	Allegheny woodrat	Mammal	2		7
Ageratina luciae-brauniae	Lucy Braun's white snakeroot	Plant	2		7
Allium cuthbertii	Striped garlic	Plant		3	7
Allium speculae	Little river canyon onion	Plant	2		7
Amorpha glabra	Appalachian indigo bush	Plant		3	7
Aster avitus	Alexander's rock aster	Plant	2	1	7
Aster surculosus	Creeping aster	Plant		3	7
Calamagrostis cainii	Cain's reedgrass	Plant	2	2	7
Carex misera	Wretched sedge	Plant		3	7
Clematis coactilis	Virginia white-haired leatherflower	Plant		3	7
Clematis viticaulis	Millboro leatherflower	Plant	2	2	7
Cyperus granitophilus	Granite-loving flatseed	Plant		3	7
Draba aprica	Whitlow grass	Plant		3	7
Heuchera alba	White alumroot	Plant		2	7
Krigia montana	False dandelion	Plant		3	7
Liatris turgida	Shale-barren blazing star	Plant		3	7
Paronychia virginica var. virginica	Yellow nailwort	Plant	2	1	7
Robinia viscosa var. hartwegii	Hartwig's locust	Plant		1	7
Rudbeckia triloba var. pinnatiloba	Pinnately-lobed brown-eyed sunflower	Plant	2	3	7
Saxifraga careyana	Golden-eye saxifrage	Plant		3	7
Saxifraga caroliniana	Carolina saxifrage	Plant	2	2	7
Scutellaria saxatilis	Rock skullcap	Plant		2	7
Sedum nevii	Nevius' stonecrop	Plant	2	2	7
Solidago simulans	Granite dome goldenrod	Plant		1	7
Talinum mengesii	Menge's flame-flower	Plant		3	7
Trichomanes petersii	Dwarf filmy fern	Plant		3	7
Xanthoparmelia monticola	A foliose lichen	Plant		2	7
Aimophila aestivalis	Bachman's Sparrow	Bird	2	3	8
Ammodramus henslowii	Henslow's Sparrow	Bird	2		8
Lanius ludovicianus	Loggerhead Shrike	Bird	2		8
Calystegia catesbiana ssp. sericata	Blue Ridge bindweed	Plant		3	8
Plethodon hubrichti	Peaks of Otter Salamander	Amphibian	2	2	10
Plethodon punctatus	Cow Knob Salamander	Amphibian	2	3	10
Juglans cinerea	Butternut	Plant	2	3	10
Desmognathus aeneus	Seepage Salamander	Amphibian	2		11
Eurycea aquatica	Dark-sided (Brownback) Salamander	Amphibian	2		11
Eurycea junaluska	Junaluska Salamander	Amphibian	2	2	11
Corynorhinus rafinesquii	Rafinesque's big-eared bat	Mammal	2	3	11
Sorex palustris punctulatus	Southern Water shrew	Mammal	2	3	11
Aspiromitus appalachianus	A hornwort	Plant		1	11
Calamovilfa arcuata	Cumberland sandgrass	Plant	2	2	11
Cardamine clematitis	Mountain bitter cress	Plant	2	2	11
Cardamine flagellifera	Bittercress	Plant		3	11
Carex austrocaroliana	South Carolina sedge	Plant		3	11
Carex ruthii	Ruth's sedge	Plant		3	11
Diphylleia cymosa	Umbrella leaf	Plant		3	11
Glyceria nubigena	Smoky Mountain manna grass	Plant	2	2	11
Hasteola suaveolens	Sweet Indian plantain	Plant		3	11
Hydrothyria venosa	An aquatic lichen	Plant		3	11
Iliamna remota	Kankakee globe-mallow	Plant	2	1	11
Marshallia trinervia	Broadleaf Barbara's buttons	Plant		3	11
Megaceros aenigmaticus	A hornwort	Plant		2	11

Table F–1 (cont.) The list of 366 terrestrial plant and animal species with viability concern for the Southern Appalachian Assessment area sorted according to groupings based on habitat association.

Scientific Name	Common Name	Taxa	Federal Status	Global Rank	Species Group[1]
Plantago cordata	Heart-leaf plantain	Plant		3	11
Sida hermaphrodita	Virginia mallow	Plant		3	11
Solidago rupestris	Rock goldenrod	Plant		2	11
Stellaria corei	Core's starwort	Plant		3	11
Viburnum bracteatum	Arrowwood	Plant	2		11
Vitis rupestris	Sand grape	Plant		3	11
Smilax biltmoreana	Biltmore carrion-flower	Plant		2	12
Dendroica cerulea	Cerulean Warbler	Bird	2		13
Glyphyalinia clingmani	Fragile supercoil	Invertebrate	2		14
Paravitrea varidens	Roan supercoil	Invertebrate	2		14
Phacelia fimbriata	Fringed scorpion-weed	Plant		3	14
Accipiter gentilis	Northern Goshawk	Bird	2		15
Cleidogona hoffmani	Hoffman's cleidogonid millipede	Invertebrate		2	15
Cleidogona lachesis	A millipede	Invertebrate		2	15
Hepialus sciophanes	A ghost moth	Invertebrate	2		15
Mesodon clingmanicus	Clingman Covert	Invertebrate	2		15
Semiothisa fraserata	Fraser Fir geometrid	Invertebrate	2		15
Abies fraseri	Fraser fir	Plant	2	2	15
Aconitum reclinatum	Trailing wolfsbane	Plant		3	15
Bazzania nudicaulis	Liverwort	Plant	2	2	15
Brachydontium trichodes	Peak moss	Plant		2	15
Cacalia rugelia	Rugel's ragwort	Plant	2	3	15
Gymnocarpium appalachianum	Appalachian oak fern	Plant	2	3	15
Leptothymenium sharpii	Mt. Leconte moss	Plant	2	1	15
Plagiochila corniculata	Liverwort	Plant		3	15
Solidago glomerata	Goldenrod	Plant		3	15
Sphenolobopsis pearsonii	Liverwort	Plant	2	2	15
Stachys clingmanii	Clingman's hedgenettle	Plant		3	15
Chelone lyonii	Purple turtlehead	Plant		3	15
Silphium connatum	Virginia cup-plant	Plant		3	16
Brachoria cedra	Cedar millipede	Invertebrate		1	18
Brachoria dentata	A millipede	Invertebrate		1	18
Brachoria ethotela	Hungry Mother millipede	Invertebrate		2	18
Brachoria falcifera	Big Cedar Creek millipede	Invertebrate		1	18
Brachoria hoffmani	Hoffman's xystodesmid millipede	Invertebrate		2	18
Brachoria separanda hamata	A millipede	Invertebrate		2	18
Buotus carolinus	A millipede	Invertebrate		1	18
Conotyla venetia	Venetia millipede	Invertebrate		2	18
Dixioria coronata	A millipede	Invertebrate		2	18
Dixioria fowleri	A millipede	Invertebrate		2	18
Nannaria ericacea	McGraw Gap Xystodesmid	Invertebrate		2	18
Nannaria shenandoah	Shenandoah Mountain Xystodesmid	Invertebrate		1	18
Pseudotremia alecto	A millipede	Invertebrate		1	18
Rudiloria trimaculata tortua	A millipede	Invertebrate		2	18
Semionellus placidus	A millipede	Invertebrate		3	18
Speyeria diana	Diana Fritillary Butterfly	Invertebrate	2	3	18
Anemone minima	Tiny anemone	Plant		3	18
Brachymenium andersonii	Anderson's brachymenium	Plant	2		18
Buckleya distichophylla	Piratebush	Plant	2	2	18
Carex manhartii	Manhart's sedge	Plant	2	2	18
Carex purpurifera	Purple sedge	Plant	2	3	18
Carex roanensis	Roan Mtn. sedge	Plant	2	1	18
Cheilolejeunea evansii	Liverwort	Plant	2	H	18
Collinsonia verticillata		Plant		2	18
Cypripedium kentuckiense	Southern lady's-slipper	Plant	2	3	18
Helianthus glaucophyllus	White-leaved sunflower	Plant		3	18
Hexastylis arifolia var. ruthii	Appalachian little brown jug	Plant		3	18
Hexastylis contracta	Mountain heartleaf	Plant	2	3	18
Hexastylis rhombiformis	French Broad heartleaf	Plant	2	2	18
Lysimachia fraseri	Fraser's loosestrife	Plant	2	2	18
Phlox amplifolia	Broadleaf phlox	Plant		3	18
Rhododendron vaseyi	Pink-shell azalea	Plant		3	18
Schlotheimia lancifolia	Highlands moss	Plant	2	2	18

Table F–1 (cont.) The list of 366 terrestrial plant and animal species with viability concern for the Southern Appalachian Assessment area sorted according to groupings based on habitat association.

Scientific Name	Common Name	Taxa	Federal Status	Global Rank	Species Group[1]
Waldsteinia lobata	Lobed barren-strawberry	Plant		2	18
Cimicifuga rubifolia	Appalachian bugbane	Plant	2	3	18
Phyciodes batesii	Tawny crescentspot butterfly	Invertebrate	2		19
Carex amplisquama	Fort mountain sedge	Plant	2	2	19
Carex polymorpha	Variable sedge	Plant	2	2	19
Shortia galacifolia var. brevistyla	Short-styled oconee bells	Plant	2	1	18
Shortia galacifolia var. galacifolia	Oconee bells	Plant	2	2	18
Solidago lancifolia	Lance leafed goldenrod	Plant		3	18
Trillium discolor	Mottled trillium	Plant		3	18
Trillium pusillum (T.p. var 1)	Least trillium	Plant	2	3	18
Trillium pusillum var. monticulum	Trillium	Plant	2	3	18
Vaccinium hirsutum	Hairy blueberry	Plant	2	3	18
Carex radfordii (=C. species 3)	Radford's sedge	Plant		1	19
Fothergilla major	Witch alder	Plant		3	19
Gaylussacia brachycera	Box huckleberry	Plant		3	19
Monotropsis odorata	Sweet pinesap	Plant	2	3	19
Nestronia umbellula	Nestronia	Plant		3	19
Phlox buckleyi	Sword leaved phlox	Plant		2	19
Prunus alleghaniensis	Alleghany plum	Plant	2	3	19
Xerophyllum asphodeloides	Eastern turkey beard	Plant		3	19
Pituophis m. melanoleucus	Northern Pine Snake	Reptile	2		19
Atheta annexa	A rove beetle	Invertebrate		2	
Atheta troglophila	A rove beetle	Invertebrate		1	
Catocala herodias gerhardi	Herodias underwing	Invertebrate		3	
Cicindela ancocisconensis	A tiger beetle	Invertebrate		3	
Cicindela patruela	Barrens Tiger beetle	Invertebrate		3	
Helicodiscus hexodon	Toothy coil	Invertebrate	2		
Paravitrea ternaria	Sculptured supercoil	Invertebrate	2		
Sigmoria whiteheadi	A millipede	Invertebrate		1	
Speyeria idalia	Regal Fritillary Butterfly	Invertebrate	2	3	
Striaria columbiana	A millipede	Invertebrate		2	
Striaria species 1	A millipede	Invertebrate		1	
Bigelowia nuttallii	Nuttall's rayless goldenrod	Plant		2	
Coreopsis pulchra	Woodland tickseed	Plant		2	
Crataegus harbisonii	Harbison's hawthorn	Plant	2		
Cuscuta harperi	Harper's dodder	Plant	2		
Diervilla rivularis	Mountain bush honeysuckle	Plant		3	
Helianthus longifolius	Longleaf sunflower	Plant		3	
Jamesianthus alabamensis	Jamesianthus	Plant	2	3	
Lysimachia graminea	Grass-leaved loosestrife	Plant		2	
Minuartia fontinalis	Water stitchwort	Plant	2		
Polymnia laevigata	Tennessee leafcup	Plant		3	
Prenanthes barbata	Bearded rattlesnake-root	Plant	2	2	
Rubus whartoniae	Wharton's dewberry	Plant	2		
Rudbeckia heliopsidis	Sun-facing coneflower	Plant	2	2	
Sabatia capitata	Rose pink	Plant		2	
Silene regia	Royal catchfly	Plant		3	
Silphium brachiatum	Cumberland rosinweed	Plant	2		
Thalictrum subrotundum	Reclined meadowrue	Plant	2		
Tomanthera pseudophyllum	Shiner's false-foxglove	Plant	2	2	
Trillium lancifolium	Narrow-leaved trillium	Plant		3	

[1] Species Group Codes

1 = Cave Habitats
2 = Mountain Bogs
3 = Spray Cliffs
4 = Fen or Pond Wetlands
5 = High Elevation Balds
6 = High pH or Mafic Habitats
7 = Rock Outcrop and Cliffs
8 = Early Successional Habitats
9 = Wide Ranging Area Sensitive Species
10 = Mid– to Late–Successional Forest Species
11 = Seep, Spring, and Streamside Habitat
12 = Habitat Generalist
13 = Area Sensitive Deciduous Forest
14 - General High Elevation Habitats
15 = High Elevation Spruce–Fir Forest
16 = Bottomland Forests
17 = Southern Yellow Pine Habitats
18 = Mixed Mesic Habitats
19 = Mixed Xeric Habitats

Appendix G

Integrated Pest Management Techniques for Gypsy Moth and Southern Pine Beetle

A brief summary of gypsy moth and southern pine beetle Integrated Pest Management (IPM) programs is presented in Appendix G.

Gypsy Moth

Prevention

(Some techniques listed under Silviculture could be classified as prevention.)

Activity:
Risk rating.

Application: Gypsy moth susceptibility (probability of defoliation if gypsy moths were present) can be predicted from the relative amount of trees that is preferred food of the gypsy moth. Vulnerability (probability of mortality after defoliation occurs) is a function of the crown condition of preferred host trees. Maps depicting relative susceptibility have been generated for the Southern Appalachian Assessment (SAA). Sufficient data for vulnerability projection are available for national forests and have been created for that ownership class.

Activity:
Training and technical assistance provided to land managers by forest pest specialists with state forestry agencies, USDA Forest Service, and Animal and Plant Health Inspection Service (APHIS).

Application: Training and technical assistance is available to land managers in all settings for evaluating gypsy moth susceptibility, vulnerability, hazard, and risk and for outlining management options.

Activity:
Quarantine (domestic and international).

Application: Quarantine reduces the probability of new introductions of the gypsy moth from areas where it does not yet exist. The domestic quarantine serves to slow the spread of gypsy moth to the south and west of the generally infested area in the eastern United States. The international quarantine prevents new introductions of the European gypsy moth into presently uninfested areas and the initial introduction and establishment of Asian gypsy moth.

Detection

Activity:
Male moth pheromone trapping.

Application: Male moth trapping with sex pheromone bait is used to delineate the boundaries of isolated infestations to guide eradication efforts, track the spread of gypsy moth, and evaluate the success of eradication efforts.

Activity:
Aerial survey.

Application: Gypsy moth outbreaks are detected through aerial survey (still photography, videography, and sketch mapping) followed by ground validation.

Activity:
Egg mass survey.

Application: The density of egg masses is an indication of the defoliation potential of the gypsy moth population in the following growing season. When used in combination with action thresholds for various social values (e.g., prevention of nuisance, noticeable defoliation, or tree mortality), egg mass surveys help to set treatment priorities. Egg mass surveys also can be used to evaluate effectiveness of treatment.

Silvicultural

Activity:
Reduction of stand susceptibility (by altering species composition, i.e. reducing species preferred by gypsy moth) and vulnerability (by removing trees that are high risk for mortality after defoliation) by thinning, harvesting, shelterwood, or changing species featured in management.

Application: Silvicultural methods are used to minimize the adverse effects of defoliation, such as mortality and aesthetic deterioration. They are most effective when applied well in advance of infestations but can be used to accomplish some objectives when defoliation is imminent or has already occurred.

Biological

Activity:
Mass trapping of male moths.

Application: Mass trapping of male moths using the sex pheromone is used on isolated populations outside the quarantine zone.

Activity:
Mating disruption with mass pheromone releases.

Application: Mating disruption with mass pheromone release to inhibit mating is used on low-level populations outside the quarantine zone.

Activity:
Sterile insect release.

Application: Release of large numbers of sterile moths to inhibit mating is used on small, isolated populations outside the quarantine zone.

Activity:
Gypchek, nucleopolyhedrosis virus (NPV).

Application: Gypchek is used for suppression and eradication of gypsy moth outbreaks where gypsy moth-specific

Activity:
Bacillus thuringensis v. *kurstaki (B.t.k.).*

Application: *B.t.k.* is widely used for suppression and eradication of gypsy moth outbreaks. It can produce effects on non-target moths and butterflies, but does not harm aquatic invertebrates.

Chemical

Activity:
Diflubenzuron (Dimilin; insect growth regulator).

Application: This compound is used for suppression and eradication of gypsy moth outbreaks. It can produce non-target effects on terrestrial and aquatic invertebrates.

Activity:
Acephate (Orthene) and carbaryl (Sevin).

Application: These compounds have a registration for gypsy moth control but are not included in the EIS.

Strategies are used in different combinations within a given area. The transition zone is where the greatest variety of strategies is employed in a pilot program called Slow the Spread (STS). This area lies between the generally infested area to the north and east, and the uninfested area. The full range of detection activities described above is used in addition to mass trapping, mating disruption, sterile insect release, *B.t.k.,* diflubenzuron, and Gypchek. Silvicultural methods are available, but not included in cost-sharing programs. No data are available concerning the area treated with silvicultural methods specifically for gypsy moth, which is a tiny fraction of the available host type.

Social values play an enormous role in the priorities used for treatment decisions, as most areas receive no treatment at all. Between 1986 and 1994, the cumulative total acres defoliated by gypsy moth in Virginia statewide exceeded 3 million acres while cooperative suppression projects were conducted on less than 1 million acres. The vast majority of this treated area was in forested residential or high-use recreation sites. General forest areas defoliated by gypsy moth are rarely treated.

Southern Pine Beetle

Prevention

(Some techniques listed under *Silviculture* could be classified as prevention.)

Activity:
Training and technical assistance provided to land managers by forest pest specialists with state forestry agencies and the USDA Forest Service.

Application: Training and technical assistance is available to land managers in all settings for evaluating southern pine beetle susceptibility, vulnerability, hazard, and risk and for outlining management options.

Activity:
Risk rating.

Application: Southern pine beetle occurrence is a function of host type (yellow pine) abundance, density, and recent radial growth. While radial growth data are not readily available on a site specific basis, an indication of relative risk can be gained by the other two factors.

Detection

Activity:
Southern pine beetle trapping.

Application: Southern pine beetles and associated insects are trapped using terpene baits. The abundance and relative frequency of southern pine beetles and clerid beetle predators indicate the intensity of outbreak and the likely course over the next growing season (increasing, stable, or decreasing).

Activity:
Aerial detection with ground truthing.

Application: Aerial detection is used when outbreaks are indicated by trapping results, ground surveillance, or local conditions. Ground truthing is used to confirm southern pine beetle activity, the actual size of the infestation, and resources threatened.

Silvicultural

Activity:
Stand susceptibility can be reduced by maintaining tree vigor with thinning and increasing diversity in structure and composition (i.e. multi-storied, multi-species stands).

Application: Reducing susceptibility is best used before outbreaks occur and throughout stand life.

Activity:
Stop existing infestations and prevent proliferation with cut-and-leave, cut-and-remove, or pile-and-burn.

Application: Cut-and-leave is used in late spring and summer to disrupt spot growth where spots are small, inaccessible, or with value too low to support removal. Emerging beetles disperse into the surrounding forest. Cut-and-remove is used year-round on spots where access and value of the attacked trees permit utilization. Beetles are removed from the site in the timber. Pile-and-burn is used in settings similar to cut-and-leave, but where destruction of beetles is desired. This method is not often used due to increased wildfire risk and high cost.

Biological

Activity:
Parasite and predator activity.

Application: Cutting methods acknowledge the importance of predators and parasites in regulating southern pine beetle populations. Dead trees without foliage support these agents after southern pine beetles have emerged. Such trees are retained where cutting methods are used to control spots so that they have an opportunity to complete their life cycle and remain available for regulating southern pine beetle populations.

Chemical

Activity:
Dursban (chlopyrifos) and lindane.

Application: These compounds are used in cut-and-spray. Infested trees are felled, cut into lengths that can be handled, and the entire bark surface of the bole sprayed with one of these chemicals registered for this purpose. Cut-and-spray is used in the same settings as pile-and-burn (i.e. where southern pine beetle brood must be destroyed).

Like gypsy moth outbreaks, not all southern pine beetle spots receive treatment. Detection efforts must identify at least one multiple tree spot per 1,000 acres of host type before suppression efforts can receive federal cost sharing. Even when this threshold is reached or exceeded, spots are treated only when justified economically or by other overriding social values (e.g. threatened or endangered species).

Post-treatment evaluations are not a routine part of IPM for southern pine beetle. However, national forests in the SAA area (and everywhere in the Southern Region) maintain the Southern Pine Beetle Information System (SPBIS). SPBIS is a continuous tracking system of spots from the first detection, through monitoring, ground checking, and salvage. Some estimate of treatment efficacy is gained by evaluating the spots that do not become active again after suppression.

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THE NORTHERN FLYING SQUIRREL (*GLAUCOMYS SABRINUS*): A CONSERVATION CHALLENGE

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The northern flying squirrel (*Glaucomys sabrinus*) has an extensive range in North America, inhabiting boreal, coniferous, and mixed forests of the northern United States and Canada and the slopes of the mountains of the east and west. Most undisturbed northern populations are apparently thriving, but those in the southern mountains are considered disjunct relicts occupying declining remnants of suitable habitat. It is clear that range contraction in the past has been associated with climate and vegetation change in the Pleistocene and the large-scale timber harvests of the early 20th century and that today a significant threat comes from forest practices and development. However, the major problem in dealing with conservation of this species is understanding its complex ecological position in its regional communities and the subtle as well as obvious influences of human activities. Thus, to preserve this species over its extensive range one will have to consider its various roles as a biological opportunist, an important prey item, a disperser of mycorrhizae, a potential victim of biological warfare, and a small, secretive glider especially vulnerable to anthropogenic and possible climatic changes in the size, arrangement, and quality of its home forests.

Key words: conservation, *Glaucomys*, heterothermy, northern flying squirrel, *Strongyloides*, truffles

The ability to develop an effective conservation strategy for a vulnerable species presupposes that one knows enough about the animal's biology and the potential threats in its environment to create a meaningful protection plan. In the case of the northern flying squirrel (*Glaucomys sabrinus*), both the acquisition of adequate data and their interpretation have been a challenge. Although concern for this species over much of its range in North America has stimulated a great number of studies over the past 20 years after a long period of limited interest, the listing of some populations as endangered fueled an intense search for that ''magic'' factor or formula that might explain its biology, guarantee its survival, and eliminate its interference with the human exploitation of its home forests. We still have much to learn. As a participant in a symposium held at the annual meeting of the American Society of Mammalogists in June 2006, I was asked to address the broad problem of flying squirrel conservation. Although this topic may be approached in a number of ways, I have chosen to attempt to provide an overview—with pertinent background and examples—of 2 interacting components of this conservation issue: the particular or salient ecological factors potentially critical to species survival; and those human activities, past and present, contributing to the species' vulnerability. I am looking for common denominators—factors important to varying degrees over the wide range and diverse habitats occupied by this species as well as special, regional threats, and I wish to raise questions about current ideas and assumptions. I maintain that in the field of northern flying squirrel conservation there may be no simple solutions but instead, within some common denominator of basic biology, an array of problems and possible management strategies dictated by regional variation in squirrel ecology and in the kinds of human influences.

With some chagrin I have recently realized that I started my studies of flying squirrels as a graduate student 43 years ago. Thus, I have decided to approach the topic partially from a personal point of view, stressing my own experiences as well as findings documented in the literature and derived from discussions with other researchers. Although my studies have included many other vertebrates over the years, I have been repeatedly drawn back to flying squirrel investigations as interesting questions and concerns have arisen. Along with a few other workers, I have become a ''marked man,'' because, over the past 25 years, inquiries have poured in from federal and state agencies, conservancies, consulting firms, and various business concerns. Everyone wants definitive information on flying squirrels in order to preserve rare or endangered squirrel populations, to find a rationale to protect threats to parks and especially significant forests, or to provide justifications for logging, road building, or development in or near the species' habitat. I would argue that the predicament of the northern

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flying squirrel is often too complicated and subtle for the pat answers these people request. Thus, I hope I can be forgiven for using my own experiences in the southern Appalachians as a starting point for a broad but not a definitive discussion of the species, linking these findings to much of the other North American research.

BACKGROUND

The northern flying squirrel is not uniformly threatened over its wide range across the boreal forests of North America and the montane and mixed forests of the south-trending mountains of the east and west (Wells-Gosling and Heaney 1984). Except in areas under heavy settlement and large-scale clear-cutting, this species is holding its own rather well in much of the northern part of its range. Its vulnerability is most pronounced in the mountain areas at the southern margins of its range—the southern Appalachians, Sierra Nevada, and Rocky Mountains.

It is quite clear from historical studies of climate and vegetation that the species has experienced a number of range contractions in the past (Arbogast 1999, 2007; Arbogast et al. 2005; Weigl 1968). During times of glacial advance in the Pleistocene, boreal forests repeatedly extended as broad southern peninsulas along the eastern and western mountains and even down the Mississippi Valley (Davis 1976; Delcourt and Delcourt 1981, 1987). One can assume, based on a few fossil records, that the northern flying squirrel then occupied a much larger southern range. The retreat of the glaciers starting 18,000 years ago would have confined squirrels to narrower strips of land and isolated massifs along the Appalachians and western mountains, but much of its remaining habitat was probably quite adequate. Then, in the late 19th and early 20th century the catastrophic clear-cut logging of Appalachian forests took place. Huge areas were denuded and burned over a short period of time—a process repeated in the west somewhat later (Loeb et al. 2000). From what we can surmise from species' habitat requirements this was a critical time of range contraction, disjunction, and probably population extinction in the mountains. However, it is unlikely that the public or even the biologists of the time were at all aware of the plight of the flying squirrels. Many of the subspecies considered endangered or rare today were unknown. Hall (see Hall and Kelson 1959) described *Glaucomys sabrinus lucifugus* of Utah in 1934, Miller (1936) described *G. s. fuscus* of West Virginia in 1936, and Handley (1953) described *G. s. coloratus* of North Carolina and Tennessee in 1953. Although some populations from the west were described in the 1890s, many subspecies remained undiscovered until well into the 20th century (Hall and Kelson 1959; Howell 1918).

Starting in the early 1980s the northern flying squirrel became the object of intensive research, but much of this work concentrated on the more abundant and widely distributed northwestern forms, whereas the rare, relict, often inaccessible populations of the mountain ridges to the south received only limited attention in spite of the listing of some subspecies as endangered in 1985 (Weigl et al. 1999). Fortunately, recent studies have provided much more background information, and the pace of research has accelerated. However, we still have much to learn about the peculiarities of the ecology of this species and both the obvious and subtle effects of human activity. And that is why conservation of this species is such a challenge.

NORTHERN FLYING SQUIRREL ECOLOGY

General

In the simplest terms one can describe the northern flying squirrel as a small, nocturnal, nonhibernating, gliding tree squirrel that occupies boreal conifer and mixed forests and uses both tree cavities and dreys for nesting (Smith 2007; Wells-Gosling and Heaney 1984). Contrary to suggestions that this squirrel is a narrow, boreal specialist, the northern flying squirrel is best described as a behaviorally plastic opportunist, capable of adjusting its biology to wide range of conditions. For example, it is quite capable of occupying deciduous and lower-elevation woodlands of the east and west, not just the spruce, fir, and other conifer forests usually cited in the literature (B. S. Arbogast, pers. comm.; Weigl et al. 2002; Weigl and Osgood 1974). Faced with cold temperatures, turbulent weather, and short periods of food limitation, the squirrel can become heterothermic, dropping its body temperature several degrees without becoming torpid (Bowen 1992). This enables it to wait out short intervals of bad weather and make the most of its body energy reserves. Unlike most squirrels, it does not depend on seeds and nuts, even when these are available (Brink 1965; Brink and Dean 1966; Hall 1991; Mitchell 2001; Thysell et al. 1997), but, although occasionally using mast, generally subsists on fungi, lichens, buds, berries, staminate cones, and animal material, none of which it appears to store. Even its reproductive biology is rather flexible. Although the squirrel commonly produces a litter in early spring, in some areas energy availability and condition of females lead either to reproductive failure or delay, with litters being observed late into the summer and even into October or December (Raphael 1984; Weigl et al. 1999; Witt 1991, 1992). Thus, compared to the smaller southern flying squirrel (*G. volans*) and most other North American tree squirrels, *G. sabrinus* possesses some unusual ecological characteristics, in keeping with the diversity of environmental conditions it must survive.

What salient features of the ecology of the northern flying squirrel need to be considered in developing conservation measures? Our knowledge of the species is still quite fragmentary, because relatively few long-term studies have been conducted (Carey et al. 1999; Cotton and Parker 2000a, 2000b; Fryxell et al. 1998; Lehmkuhl et al. 2006; Ransome and Sullivan 2002; Smith and Person 2007; Weigl et al. 1999). Most studies have been of short duration, confined to warmer months, or limited to surveys. Long-term, year-round investigations are rare. In addition, once some populations were listed as endangered in 1985 in the Appalachians and others were deemed vulnerable because of habitat modifications in the west, researchers avidly attempted to acquire and interpret new data in a quest for unitary and perhaps overly simple strategies to preserve these squirrels. Because the extensive literature on the genetics, biogeography, and ecology have been largely reviewed by Arbogast (2007) and Smith (2007) in this issue, I will concentrate on aspects of the squirrel's biology that appear essential to conservation of the species and then raise questions about the current state of our knowledge and interpretations. Some of my comments will be based on the literature, some on personal experiences.

Habitat

In reviewing the voluminous literature on the habitats utilized by northern flying squirrels, one cannot help but be impressed by certain common features as well as some regional variations that perhaps reinforce this perceived ''common denominator'' (Waters and Zabel 1995; see Smith 2007). Northern flying squirrels generally occupy boreal or north temperate conifer, mixed conifer–hardwood, and northern hardwood forests, as found in the northern United States and Canada, at various elevations of mountain regions, and in some narrow valleys subject to cold air drainage. These habitats support old-growth forest, communities with old-growth elements, or younger woodlands usually contiguous with such forest. Such areas are usually cool and moist, have cold winters, and possess a well-developed canopy, substantial ground cover, quantities of wet, dead, and downed wood, and often organic substrates. These conditions favor an abundance of snags, cavities, witches brooms, trees festooned with lichens and moss, and a diverse array of buds, berries, seeds, and fungi. In drier sites in the west, squirrels appear to select riparian areas where these cooler and wetter conditions prevail, and where there is easy access to drinking water (Meyer et al. 2005, 2007). In fact, Carey (1989, 1995) observed differences in population densities in Washington and Oregon that might be associated with moisture conditions in various forest types. Although one can point out variations in this ''typical'' habitat description, it is clear that the northern flying squirrel is versatile enough to prosper in a wide range of forest types as long as the above conditions occur in enough favorable patches and enough habitat is left undisturbed.

Although *G. sabrinus* may be a habitat opportunist and readily uses a diversity of potentially suitable forests, habitat is a major conservation problem, exacerbated by various controversial approaches to forest management. The ongoing harvest of old-growth forest, its replacement with plantations or regenerating stands, and the increasing fragmentation of much of the remaining habitat has alarmed some biologists concerned about this and other rare animal species (see Smith 2007). When rare species are declared endangered, as in the case of the northern flying squirrel, then economic forces exert tremendous pressure on researchers to develop definitive management plans that will protect the rare organisms, but also allow a return to timber harvest and development. Such is the case in Alaska (Smith and Person 2007) where the size, quality, and connectedness of planned reserves is an issue, in the Pacific Northwest where the debate over the importance of old growth versus successional forests to rare species has raged for years (Carey 1989, 1995; Lehmkuhl et al. 2006; Waters and Zabel 1995; Witt 1992; but see Ransome and Sullivan 1997, 2002, 2004; Rosenberg and Anthony 1992), and in the Sierra Nevada where thinning, fire, and harvesting may limit the size and quality of squirrel habitat (Meyer et al. 2005; Meyer and North 2005). Another example comes from the Appalachians where the currently endangered subspecies *G. s. fuscus* of West Virginia is a candidate for delisting. In the Appalachians northern flying squirrels are commonly found in older forests of spruce (*Picea rubens*), fir (*Abies fraseri*), beech (*Fagus grandifolia*), sugar maple (*Acer saccharum*), and yellow birch (*Betula alleghaniensis*), especially in the ecotones between conifers and hardwoods. However, throughout the east from Nova Scotia, Canada (Lavers 2004), to southern North Carolina (Weigl et al. 2002) the species is known to occupy hardwood habitats without spruce and fir. An array of studies have documented the squirrel's habitat diversity (Ford et al. 2004; Menzel et al. 2006; Payne et al. 1989; Stihler et al. 1987; United States Department of the Interior, Fish and Wildlife Service 2006; Urban 1988) pointing out the importance of hardwood and mixed forest habitats. *G. sabrinus* of West Virginia is more abundant and its populations more continuous than in most parts of the east. Many of the squirrels are caught in forests in which spruce is present, and this tree species supports one of the fungal genera (*Elaphomyces*) eaten by the squirrel (Loeb et al. 2000). Therefore, the United States Fish and Wildlife Service has decided that if forests containing spruce are protected in the national forests, the flying squirrel's preservation is insured, and it can be delisted, not to the ''threatened'' level but taken off the critical list entirely. The problems with this approach are many. First, it is not clear if there is any direct causality between the presence of flying squirrels and spruce. Both animal and plant may be responding independently to the same boreal conditions. Squirrels may nest in spruces occasionally and use them as one of many food sources, but there is no proof of any obligate relationship. Second, in more than 40 years of trapping and nestbox checking in various Appalachian habitats, I almost never captured animals in extensive, pure conifer stands, although telemetry revealed that they sporadically used them. Third, such a course of action fails to sufficiently protect the northern hardwood areas often used by *G. sabrinus*. Finally, the quality and connectedness of the proposed spruce-containing reserves, now and in the future, need careful study, especially in a region where timber harvest is an important part of the local economy. My main point is that economic pressures may at times influence how ecological information is interpreted resulting in overly simplistic solutions to a conservation and political issue.

Foods

One of the especially significant aspects of northern flying squirrel ecology and conservation is the direct link between the squirrel, its diet, and the perpetuation of its forest habitats. Years ago, McKeever (1960) noted high levels of fungi in the guts of California animals, and in 1965 I discovered that North Carolina squirrels were consuming large quantities of fungi and the staminate cones of fir (Weigl 1968). Subsequently, research in the Pacific Northwest documented the dependence of northern flying squirrels on the fruiting bodies of hypogeous, mycorrhizal fungi (truffles—Carey et al. 2002; Fogel and Trappe 1978; Lehmkuhl et al. 2004; Maser and Maser 1998; Maser et al. 1978, 1985, 1986; Meyer and North 2005; North et al. 1997; Pyare and Longland 2001b). The hyphae of these underground fungi form associations with tree roots, greatly increasing their surface area for the absorption of water and minerals at a small energy cost to the tree. Many tree species grow poorly or not at all without mycorrhizae. But spore dispersal to new seedlings and older trees is a problem for an underground fungus. Based on our study of the northern flying squirrel and another truffle eater, the fox squirrel (*Sciurus niger*—Weigl et al. 1989), and the work of Zabel and Waters (1997) and Pyare and Longland (2001a), the following scenario has taken shape. The truffle produces a fruiting body that gives off a chemical signal on ripening; this causes a squirrel to avidly excavate and devour the fungus (Secrest 1990). However, although the squirrel obtains energy and certain minerals (e.g., sodium and phosphorous) from these truffles, it is unable to digest the fungal spores, which are then dropped over the landscape for days or weeks afterward (Gamroth 1988). The resulting inoculation of young trees and spread of the fungus may thus have a marked impact on the perpetuation of the forest habitat on which the squirrel depends. Although *G. sabrinus* is not the only mycophagist in its home forest, it is one of the most mobile and spends much time on the ground during foraging (Bird and McCleneghan 2005; Loeb et al. 2000; Mitchell 2001; Zabel and Waters 1997). In any case, because of these food habits and their positive effect on the trees of its habitats, conservation of this species assumes a greater dimension and significance. In fact, many of the habitat models for *G. sabrinus* are now implicitly based on recognition of this squirrel, tree, and fungus symbiosis (Ford et al. 2004; Menzel et al. 2006; Odom et al. 2001; see Smith 2007).

Given the above account of the use of hypogeous fungi, it is important to link these and other foods to certain environmental factors. Truffles are the fruiting bodies of mycorrhizal fungi and appear to be most abundant in association with larger and older living trees, especially in moist, organic soils. The time course of fungal inoculation, growth, and maturation of sporocarps may vary in different forests, but old-growth conditions may be optimal. Epigeous fungi and lichens, which also are important foods, depend on abundance of dead wood and extensive tree surface areas, respectively, and, once again, cool, wet conditions. Although lichens and animal material such as insects and carrion may help support squirrels in the winter when most other foods are unavailable, some researchers also have found evidence for winter truffle use in habitats with frozen ground. Hackett and Pagels (2003) and Smith (2007) have data on the use of underground nests, but no one has reported underground foraging in winter. The other plant materials making up the squirrels diet—staminate cones, berries, beechnuts, and some seeds—are reflective of a preference for boreal habitats and old-growth conditions but also are indicative of an opportunistic species that is not limited to truffles and that might utilize additional foods.

Demographic Considerations

In spite of the spectacular increase in northern flying squirrel studies, we have surprisingly little information on the species' life history and population biology. Most studies have been dedicated to particular questions such as home range, relative density, foods, and habitat associations. Longer-term studies (e.g., Carey et al. 1999; Fryxell et al. 1998; Smith et al. 2004, 2005; Smith and Nichols 2003; Weigl et al. 1999) have begun to fill in some gaps in our knowledge, but we know very little about most population parameters and long-term temporal and spatial trends.

Smith and Person 2007 have recently reviewed much of the demography of the species and raised questions about the distribution and stability of populations. The picture of *G. sabrinus* that is developing is of a relatively long-lived (4–7 years) species with a low reproductive rate for a small mammal. In the western part of the range of *G. sabrinus*, flying squirrels appear to be more abundant than in the east and more continuous in their distribution within the old-growth forests that they commonly occupy. However, most workers report lower densities in managed or successional stands. In the east, populations often occur in distinct patches, often kilometers away from other groups in spite of what seems to be suitable intervening habitat (Weigl et al. 1999, 2002). Also in the east, population size appears to be highly variable. In some years, squirrels will be abundant in an area; in other years the populations are low or nonexistent. Have the animals died out or moved? No answer is available, but population fluctuations have been noted by other researchers (Fryxell et al. 1998). In spite of the meager data from recaptures, it is clear that at least some of the squirrels missing in intervening sampling sessions show up again months or years later (Weigl et al. 1999).

Examination of telemetry data from throughout North America suggests that home-range size is associated with habitat quality and food resources (Smith 2007). Home ranges from 2 to 60 ha have been reported. Our own work and that of others have revealed that squirrels have relatively small core home ranges (3–15 ha) that vary somewhat with sex and season, but that many individuals will display bouts of extensive linear travel, in some cases more than a kilometer, that involve both outward movement and return (Menzel et al. 2006; Weigl et al. 1999). There is some evidence that this long-distance travel is associated with a search for foods and possibly mates (Weigl et al. 1999). Such forays may affect home-range estimates if data are taken at wide time intervals. The important question here relates to the use of space by the species. If populations in a locality can fluctuate widely in numbers, have a distinctly patchy distribution in fairly uniform forest, and consist of individuals that can cover spectacular distances, it is possible that northern flying squirrels may use and thus require much larger expanses of suitable habitat than is commonly acknowledged if they are going to survive in many parts of their range. Both habitat size and connectedness assume great significance under these conditions.

Smith and Person 2007 have recently provided an intriguing example of space use that may partially relate to the preceding discussion. Working in Alaska in undisturbed habitat, they investigated populations in prime old-growth forest and adjacent groups in a wet, mixed muskeg and forest landscape. Examination of the demographic data suggested that there was a dynamic source–sink situation governing these populations. The muskeg areas were not maintaining viable squirrel populations in a steady state, but were the beneficiary of constant migration of animals from the better forest habitats. To what extent high mobility, source–sink conditions, and metapopulation distributions of squirrels are a common phenomenon is unknown, but this may be worth investigating in areas with old-growth forest adjacent to human-modified habitats. The squirrel populations reported from cutover and regenerating areas may be more variable because they are not self-perpetuating. Certainly the status of populations in West Virginia, the Sierra Nevada, and parts of the Pacific Northwest should be evaluated with this possibility in mind.

Other Species of Animals

The fate of northern flying squirrels may be closely linked to the presence of other animal species—predators, competitors, and parasites—that are in turn often of particular concern to wildlife biologists and conservationists.

Predators.—Smith (2007), Carey et al. (1992), and Weigl et al. (1999) have described some of the potential predators of the flying squirrel, but 2 in particular may be of interest in different parts of the range. Over the past 20 years it has become clear that the northern spotted owl (*Strix occidentalis*), an endangered and much celebrated species of western forests, is especially dependent on the northern flying squirrel as a prey item (Carey et al. 1992). The owl seems to thrive in extensive old-growth forests or in habitats with old-growth elements where the squirrels are most abundant (Carey 1995; Carey et al. 1999). The size and condition of the habitat ideal for supporting both the flying squirrel and the owl have been the focus of ferocious debate (Carey et al. 1992; Ransome and Sullivan 2002; Rosenberg and Anthony 1992). Old-growth forests in the west are becoming smaller in size and increasingly fragmented, but often are viewed as the economic salvation for a timber industry that is worried about an endangered species restricting the exploitation of remaining tracts. For the squirrel the issue of habitat quality, size, and connectedness is of great importance and has been the focus of several studies. Conservation of squirrel and owl thus seems inextricably linked, but doubtless shall remain a source of intense political and economic controversy.

In the eastern United States another rare animal is periodically associated with the issue of protection of *G. sabrinus*. Every few years, wildlife biologists consider the reintroduction of the fisher (*Martes pennanti*) to the southern Appalachians; this species was known to exist in the region in the recent past. In most areas fishers can probably coexist with northern flying squirrels without problems. But in small habitat islands of the southern Appalachians with few squirrels and limited alternate prey items, a predator such as the fisher might kill off these relict populations. Although there have been no introductions of fishers in areas with isolated flying squirrel populations, this idea resurfaces frequently (R. Powell, pers. comm.) and will require the careful attention of wildlife agencies in the region.

Competitors.—Smith et al. (2004, 2007) have suggested that the biology of *G. sabrinus* in the Pacific Northwest may be different from that in Alaska and the east because of the abundance of other small mammals in western forests. This diversity of sympatric rodents might then produce a greater degree of den-site and food specialization in response to direct and diffuse completion. In reality, we have little information on resource competition between northern flying squirrels and other mammals. Although red squirrels (*Tamiasciurus hudsonicus*) and Douglas squirrels (*T. douglasii*) are often mentioned as possible competitors, there is not much evidence of any severe interaction. Flying squirrels may pilfer food from red squirrel middens and the 2 species may both use cavities for nesting sites and fungi for food, but the very different overall diets of these squirrels and their nocturnal–diurnal activity separation may minimize interactions, especially in good habitat. In many years of trapping both species, I was always surprised to find that the best years for capturing northern flying squirrels also were the best for red squirrels.

The southern flying squirrel (*G. volans*) often has been considered a major competitor (Weigl 1968, 1978). Both species are nocturnal gliders that use tree cavities for dens and both may consume fungi, insects, and plant parts. Although experimental studies suggested that *G. volans* was the more active and aggressive in interactions, especially around nests (Weigl 1978), habitat preferences, diets, and climatic tolerances of the 2 species (Bowen 1992; Bowman et al. 2005) suggest only limited competition. In fact, except in the north, the 2 species usually show limited and unstable sympatry. Thus, except for the diffuse interactions suggested by Smith et al. (2005) in the west, and a few instances of resource overlap, there is little evidence that competition per se is a significant factor in the conservation of the northern flying squirrel.

Parasites.—A particularly intricate relationship between squirrel ecology and conservation grew out of some unusual discoveries in the southern Appalachians. In the 1960s I had set out to study the interaction of *G. sabrinus* and *G. volans* in the Appalachians as a model system for evaluating aspects of competition theory (Weigl 1968). Northern flying squirrels were exceedingly rare, but after several months of trapping I eventually captured enough for the experimental parts of my study. Colonies of both species were then housed in large outdoor aviaries in North Carolina. The 1st spring saw the demise of almost all of the *G. sabrinus* except those kept in the laboratory, whereas the *G. volans* seemed to thrive in an adjacent cage. With the help of 2 veterinarians and a former zoo pathologist, I narrowed down the cause of this massive die-off to an infection by the nematode *Strongyloides robustus*. *S. robustus* has a life cycle like that of the famous hookworms (*Necator* and *Ancylostoma*): embryonated eggs released with animal feces hatch and develop into infective larvae in the substrate; these penetrate the skin of a host, are carried to the lungs where they break through to the lumen, are swallowed, and finally lodge in the intestine doing marked physical and nutritional damage (Weigl 1968; Weigl et al. 1999). The parasite is most common in warmer climates where it has been reported to cause marked pathology in wild species (Davidson 1975). Once the cause of the affliction of the captive *G. sabrinus* was determined, other wild populations of squirrels were checked. All of the captive *G. volans* in my colony were parasitized (and were probably the source of the infection in the *G. sabrinus*), but had suffered no ill effects. In fact, all populations of *G. volans* studied in subsequent years carried this parasite. On the other hand, *S. robustus* could not be found in any of the *G. sabrinus* captured on the Appalachian peaks during the remaining years of the study. In the 1980s the federal listing of the Appalachian subspecies *G. s. coloratus* prompted a new 5-year study of the northern flying squirrel over a wide area of the North Carolina and Tennessee mountains. *G. volans* now also appeared intermittently in some of the capture sites of *G. sabrinus*, although there was never any stable sympatry of the 2 species (Weigl et al. 1999). *G. sabrinus* now supported varying intensities of parasite infection, and in the summer months there appeared to be some correlation between parasite loads and the condition of the animals (Weigl et al. 1999). We eventually cultured the parasite through its life cycle in the laboratory and determined its cold sensitivity (Wetzel and Weigl 1994) and its ability to be transferred by contact with contaminated nest material or soil substrates. Based on all the data to-date and some additional studies by Pauli et al. (2004) and Sparks (2005), I would suggest the following scenario. The cold, high-elevation or northern forests occupied by *G. sabrinus* only intermittently can support *S. robustus* because of the sensitivity of the infective larvae to cold. When *G. sabrinus* moves down into the more climatically moderate forests at lower elevations or when infected *G. volans* invade the upper slopes during the summer months along paths of human-modified habitat, the 2 species come into contact, especially by using the same tree cavities or feeding areas (Hackett and Pagels 2003), and *S. robustus* is then transferred. Even if the northern flying squirrels are not killed by the parasite, its effects may be sufficiently debilitating to put the species at a disadvantage. It is interesting that only in the colder parts of the range of *G. volans*—the Great Lakes area, northern New England, Ontario, and Nova Scotia—does one get reports of some degree of sympatry of the 2 flying squirrel species (J. Bowman, pers. comm.; Lavers 2004; Pauli et al. 2004). Why then doesn't *G. volans* take over the high-elevation refuges or northern habitats of *G. sabrinus*? The answer probably lies in sensitivity to cold of *G. volans*, its dependence on stored nuts and seeds for winter survival (Bowman et al. 2005; Doby 1984), and the virtual absence of these resources in most habitats of *G. sabrinus*. In summary, *G. volans* may possess a kind of biological weapon that at least in the southern and central part of its range, can prevent the persistence and spread of *G. sabrinus* (Barbehenn 1969; Haldane 1949; Hatcher et al. 2006; Price et al. 1988; P. D. Weigl, in litt.). It has been argued recently that the loss of genetic heterogeneity in the increasingly isolated, high-elevation populations of *G. sabrinus* of the east may make the species even more susceptible to parasite and other infections (Sparks 2005). What will happen if warming climatic conditions favor invasion of higher peaks and northern habitats by *G. volans* is thus an open question in considerations of species persistence.

Genetics

In many parts of the range of the northern flying squirrel, one can reasonably argue that the species is an island inhabitant, subject to most of the constraints that afflict other such populations (Brown 1971, 1978; MacArthur and Wilson 1967). Whether occupying real islands off the coast of Alaska; widely scattered habitats of the San Jacintos, Sierra Nevada, Rocky Mountains, and perhaps the Black Hills; or the upper elevations of the southern Appalachians, the species often occurs in small, disjunct populations, relicts of broader ranges in the late Pleistocene. The genetics of these populations have received intensive study over the last 10 years (Arbogast 1999, 2007; Arbogast et al. 2005; Bidlack and Cook 2001; Browne et al. 1999; Sparks 2005; Wartell 2005; A. Wartell, in litt.). Genetic structuring, private alleles, and loss of heterozygosity have been detected in many populations, most likely as a result of reduced population size, isolation, inbreeding, bottlenecks, and other drift effects. Although inbreeding tolerance and the replacement of alleles in time by mutation (Sparks 2005) might alleviate the plight of some groups, the loss of genetic diversity is usually seen as a potential threat, especially in changing environments. The persistence of reasonably large and interconnected populations thus appears to be critical to the species survival, and that means sufficiently large habitat reserves and the maintenance of forested corridors. Such a conservation solution might work if the environmental status quo can be maintained. However, in the face of continued forest destruction, drought cycles, El Niño effects, and the still largely unknown impacts of global climate change, the reduction of available habitat and of corridors could well spell the regional demise of this species from both a loss of genetic variability and the loss of viable places to live.

The Impact of Human Activity

Habitat Size and Quality

So far I have emphasized some of the complexities of northern flying squirrel ecology and its implications for species conservation. However, it is clear that the really major threats to these squirrel's persistence come from human activities, especially in areas of small disjunct populations such as those on islands or at the southern extension of the range. Clearcutting, development, or anything that destroys extensive tracts of habitat will have obvious harmful effects. The size of the remaining forest habitat and its condition then becomes critical to survival. One has only to fly over parts of the Rocky Mountains, Sierra Nevada, and Cascades or along the Appalachians to appreciate the scope of forest destruction and roadway construction in national and privately owned forests. And landscape modification is not the only concern. Successional and regenerating communities require considerable time to develop into habitats of sufficient quality to support flying squirrels. Using demographic models, Smith and

Person (2007) have questioned the adequacy of the size of planned reserves in Alaska; Carey and others (Carey 1995; Carey et al. 1999) have provided evidence that the 2nd-growth landscapes of the Pacific Northwest do not always have the same capacity as old growth for supporting flying squirrels. In the Sierra Nevada, thinning and controlled burning may have adverse impacts on the canopy and organic material on the ground, respectively. Finally, some 2nd-growth stands may well appear to support healthy densities of squirrels, but, in reality, are population sinks for migrants from neighboring old-growth habitats and thus may not permanently maintain viable populations (Smith and Person 2007). Only long-term studies can provide the conclusive data on the suitability of these special or successional areas. The small disjunct squirrel populations of the central and southern Appalachians appear particularly vulnerable to any further modification or reduction of their habitats.

Given the above problem of loss of quality habitat, one needs to recognize 2 major forces that can aggravate this threat. One is economic and political—the demand for forest products and recreation venues, for local and regional employment, and for tax revenues and investment returns. These factors are of overwhelming significance, but are beyond the scope of this paper. The other force—climate change—is more intangible. A warming climate could cause the retreat of some tree species and communities to higher latitudes and cause the substantial reduction or elimination of boreal communities on mountains. Change in the composition and the position of communities might be especially dire in areas already modified by other human influences. Thus, the persistence of northern flying squirrels in the already-disturbed forests of West Virginia could be more tenuous than many have thought during a period of global warming. In addition to modifying community composition and distribution, climate change may have another major impact. A recent paper by Westerling et al. (2006) has documented a link between progressive climate warming and changes in the phenology, desiccation, and fire frequency in western forests. Thus, climatic warming may not only cause modifications of forest distributions, but also their complete annihilation by fire. It is likely that the desiccation observed by Westerling et al. (2006) would also have a marked impact on the moisture-requiring staple foods (fungi and lichens) of flying squirrels.

Habitat Connectedness

Along with habitat size and quality, habitat connectedness assumes an important role in species preservation. The extent of unsuitable terrain between high-quality habitat and the absence of wooded corridors could be major factors in regional survival. Frequently, the greater the reduction of contiguous forest, the wider the barriers to dispersal. Such fragmentation of flying squirrel distributions could destroy the viability of metapopulation-structured groups of squirrels, and the resulting small isolates then would be susceptible to the genetic problems mentioned earlier.

The impact of barriers on movements of flying squirrels needs further study, especially the effects of the proliferation of roadways through quality habitats. One example of barrier effects comes from the southern Appalachians. A 3-year study of an extravagant economic development scheme in the North Carolina–Tennessee mountains called the Cherohala Skyway revealed such unexpected impacts (Weigl et al. 2002). Clearly, a 2-lane scenic road removes a quantity of habitat, but, of greater significance, it also can act as a barrier to dispersal to different parts of the forest. Although *G. sabrinus* is an able glider and is known to cover distances along the ground, it is unable to cross wide, exposed roadways, especially the kind of blast-and-fill rights-of-way commonly cut into the sides of mountains. In 2 years of telemetry and trapping, no squirrel was observed to have crossed the Cherohala Skyway. The resulting range fragmentation may doom this southernmost population. In addition to barrier formation, there are 2 more-subtle impacts from a roadway. One impact was detected in the winter when snow permitted the identification of mammals moving on or along the roadway. It was obvious that various predators—bobcats, coyotes, and foxes—used the roadway as patrol routes when hunting and might easily catch any small mammals on the road. Hawks and owls also hunted over the road. Thus, one can easily see that such a right-of-way is both a physical barrier and a site of increased mortality. Another effect of roadways or similar corridors is the modification of adjacent vegetation or other habitat conditions in ways that favor the invasion of potential predators, competitors, or pathogens. In the case of *G. sabrinus*, strips of oak, cherry, and other hardwood species in disturbed areas along roadways provide foods for *G. volans* and favor its invasion of high-elevation habitats, and the transfer of *Strongyloides* to *G. sabrinus*. Thus, linear disturbances of a certain width and severity are a potential source of species fragmentation and possibly increased deleterious species interactions. The impact of roads, systems of ski trails, ridge-top wind farms, recreational vistas, and other types of habitat subdivision need careful evaluation in the future—much more then they have received to-date.

Pathogens, Pests, Pollutants, and People

Another anthropogenic factor threatening northern flying squirrels is the introduction of plant pathogens, insect pests, and industrial contaminants into squirrel habitats. In the southern Appalachians, the high-elevation conifer forests have been decimated by an adelgid insect (*Adelges piceae*) that kills Fraser fir (*Abies fraseri*), a valuable timber and Christmas tree species and a source of food and habitat for northern flying squirrels (Amman 1966; Amman and Speers 1965). The staminate cones of fir and spruce are important foods for flying squirrels in the spring when they are eaten in vast quantities. Interestingly, both field and experimental studies suggest that the essential oils from these foods suppress gut parasites such as *Strongyloides* (Weigl et al. 1999). The loss of Fraser fir then would remove a source of food (truffles, staminate cones, and possibly seed), den sites, and a possible natural medicine. In any case the adelgid killing firs, a new adelgid now destroying hemlocks, the impact of pine bark beetles in some parts of the west, and the effect of acid precipitation on vegetation and soils all represent potential threats to flying squirrels.

The last intrusion mentioned in the heading of this section of the text—people—usually goes unmentioned. One of the major effects of building of roads through prime habitat is the provision of access to lands for private and commercial development. The state or federal government builds a road, and nearby landowners demand the right to connect in order to develop their forest property. During an era of explosive interest in living in natural environments or in 2nd-home ownership, the demand for newly accessible forest land is intense and is often fueled by the economic aspirations of neighboring municipalities. A short trip on the Blue Ridge Parkway in the Appalachians reveals the result of this process. The end result is the loss and fragmentation of habitat and possibly a loss of flying squirrels. Thus, the inclusion of people as a factor along with pathogens, pests, and pollutants may indeed be appropriate.

SUMMARY

In the past 25 years the northern flying squirrel has come under increasing scrutiny as new studies have been initiated, papers published, and various agencies alerted to its status and ecological significance. Because of physical, logistical, and economic difficulties associated with long-term research in remote and often rugged areas, our knowledge of this species is still fragmentary, especially in the southern Rocky Mountains, parts of the Sierra Nevada, the Black Hills, and the northeastern United States. Enough is known now to form a picture of the species' ecology and those aspects of its biology that may affect its preservation. In 2 cases, the northern flying squirrel makes a positive contribution to the forests it occupies. Throughout its range its use and dispersal of mycorrhizal fungi—both hypogeous and epigeous—make it an integral part of a squirrel–fungus–tree mutualism that may well help maintain the very forests needed for its survival. In the northwestern United States and western Canada, the flying squirrel is a critical food item for the endangered spotted owl. Thus, if its habitat is protected and the squirrel is permitted to flourish, the owl has a greater probability of survival.

In spite of the fact that the northern flying squirrel is something of an ecological opportunist, versatile enough to occupy several forest types, consume a number of foods, and reproduce when conditions permit, certain of its characteristics potentially increase its vulnerability. Its dependence on fungi and lichens during much of the year confine it to a certain array of old-growth, boreal forests with cool, moist climates and abundant dead wood and organic soils. The phenology of fungi, particularly the locality and timing of sporocarp production, may require the exploitation of a multitude of widely spaced, ephemeral patches and thus the use at times of extensive home ranges or reliance on long-distance travel. In short, the area needed to support these animals may be larger than our short-term telemetry studies have indicated. And although its diet and tolerance of cold conditions facilitate survival in habitats with severe climates, the low caloric density of much of its diet may be a factor in its relatively low metabolic and reproductive rates (McNab 1986).

The influence of others animals in the environment of the northern flying squirrel needs further study. In no part of its undisturbed range does it seem adversely affected by predators or competitors. Perhaps only in human-modified areas do these markedly assume importance. In the southern and central parts of the eastern United States the possibility that the nematode *S. robustus*, carried by the southern flying squirrel, harms the northern species is unresolved. However, the obvious ability of northern flying squirrels to occupy lowland, deciduous habitats in the absence of the smaller species, their confinement to high elevations when *G. volans* is present, and the instability of populations in contact zones argue for some kind of interaction. In Ontario, Nova Scotia, and northern Pennsylvania, the 2 species have been found in the same nest boxes (J. Bowman, pers. comm.; A. Lauers, pers. comm.; M. Steele, pers. comm.), but these are areas that are climatically unfavorable for the parasite. Thus, in part of the range of the northern flying squirrel a parasite-mediated interaction may be operating. Clearly more research on this topic is needed.

Although there is abundant evidence of the effect of small population size and isolation on the genetic diversity of northern flying squirrel populations, there is at present no evidence of a direct link between loss of genetic diversity and survival. The isolation of populations may occur naturally because of climatic responses of forest communities, but, more likely today, it is caused—or least aggravated—by human activity. We may never know when genetic impoverishment is a major or just a contributing factor to a population's disappearance.

All of the above ecological aspects of the biology of the northern flying squirrel may have varying effects on the perpetuation of populations in different parts of the range. When one adds the human component, the probability of survival can change spectacularly. Human influences on habitat size, quality, and connectedness are most likely the main threats to the species throughout its range. These critical factors in turn are the products not only of direct habitat destruction and modification, but indirect effects such introduced pathogens, pests, and contaminants and the slow, inexorable pressure of climate change. Survival of the species *G. sabrinus* is certainly critically dependent on an understanding of the species' ecology, but, even more important, an awareness of the impact of human activity on this ecology throughout its range.

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Special Feature Editor was Barbara H. Blake.

Management guidelines for enhancing Cerulean Warbler breeding habitat in Appalachian hardwood forests

Technical Report · January 2013

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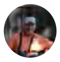

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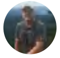

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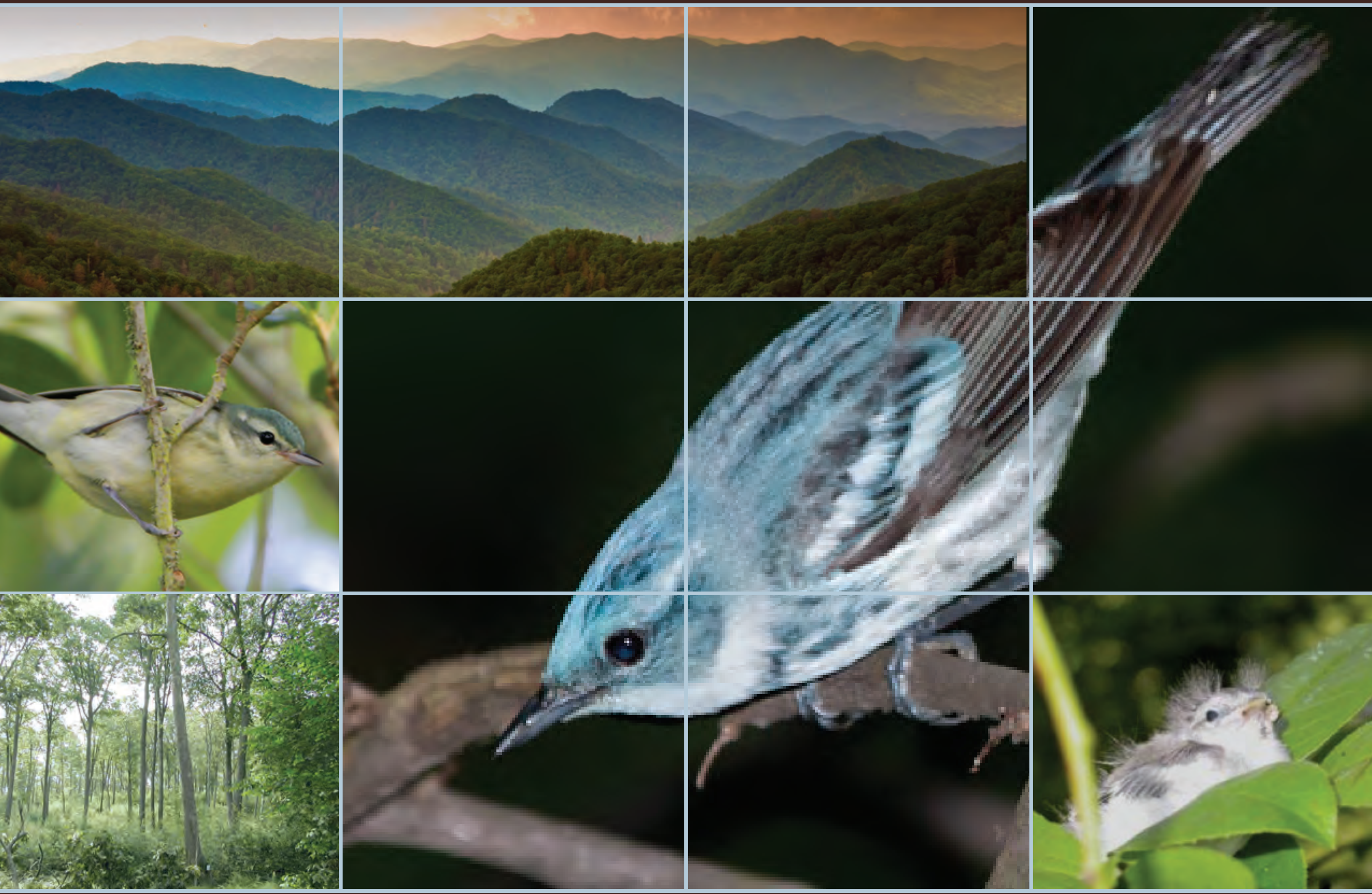
Management Guidelines for Enhancing Breeding Habitat in Appalachian Hardwood Forests



February, 2013

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Appalachian breeding habitat. Than Boves

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Photos: Cover: Clockwise: Appalacian landscape, Charlie Choc; Cerulean Warbler, Bill Hubick, http://www.billhubick.com; Fledgling, Marja Bakermans; Shelterwood, Scott Stoleson; Female, Bill Hubick

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Male Cerulean Warbler. Marja Bakermans

Introduction

The Cerulean Warbler (*Setophaga cerulea*) is a migratory songbird that breeds in mature deciduous forests of eastern North America. Cerulean Warblers (hereafter, ceruleans) require heavily forested landscapes for nesting and, within Appalachian forests, primarily occur on ridge tops and steep, upper slopes. They are generally associated with oak-dominated (*Quercus* spp.) stands that contain gaps in the forest canopy, that have large diameter trees (>16 inches diameter breast height (dbh)), and that have well-developed understory-and upper-canopy layers. Ceruleans primarily use the mid- and upper-canopy where they glean insects from the surface of leaves and conceal their open cup nests. Because they are severely declining across much of their range (Fig. 1), habitat management is a high priority. Management for this species can also improve conditions for a number of other wildlife species that depend on the same structure.

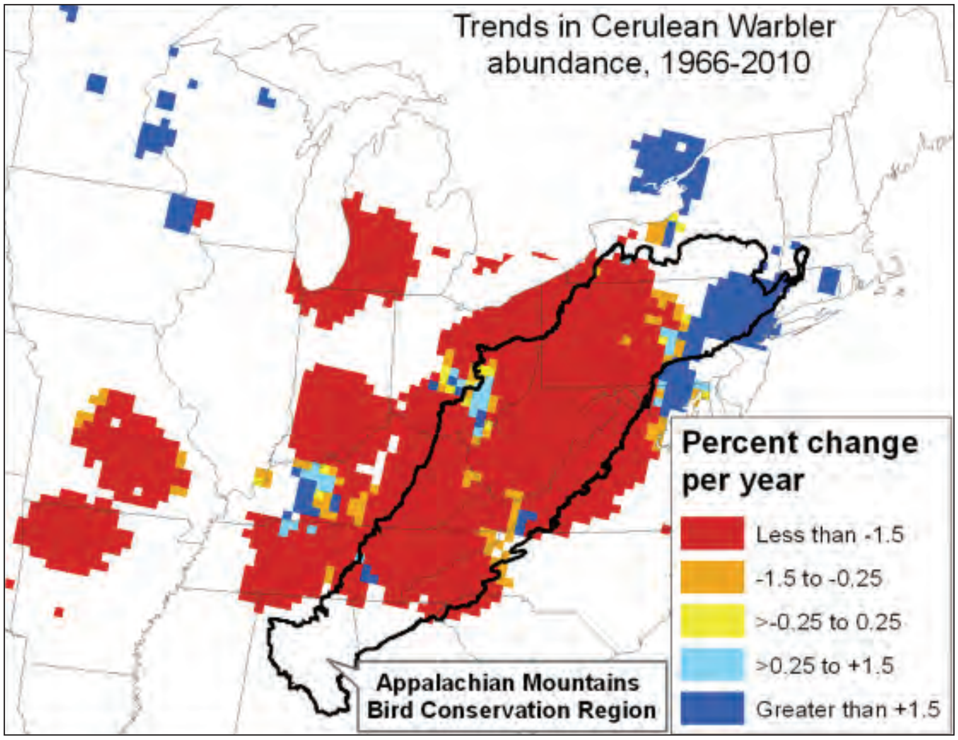


Figure 1. Cerulean Warbler distribution and trends in abundance across their breeding range from Breeding Bird Survey data (1966-2010; Sauer et al. 2011). The Appalachian Mountains Bird Conservation Region boundary is in black.

Adult Cerulean Warbler feeding chick. Wayne Miller

Goals

This document provides land managers in the Appalachian Region with guidelines for retaining and enhancing habitat for Cerulean Warblers and a diverse bird community based on the current available science. They are intended for use by federal, state and private foresters, biologists, and other land managers. These management guidelines are based to a large extent on the recently completed Cooperative Cerulean Warbler Forest Management Project (CWFMP) but also incorporate relevant findings from other research projects. All literature incorporated into this document is listed in the Reference section. The guidelines apply primarily to upland oak-dominated habitats where the majority of the research reported was completed.

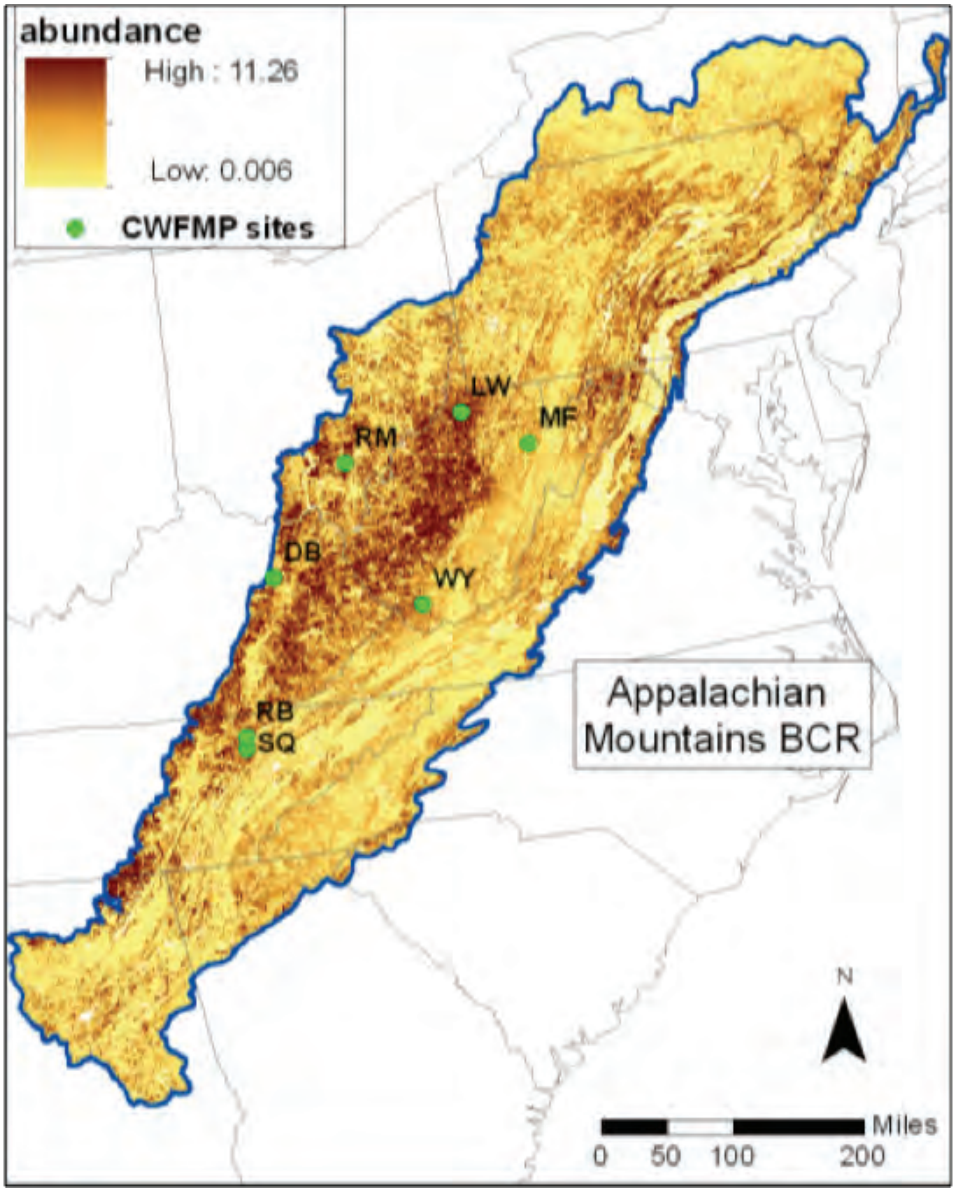


Figure 2. Cerulean Warbler abundance (number per route) estimated from Breeding Bird Survey data for the Appalachian Mountains Bird Conservation Region (BCR) (adapted from Shumar 2009). Study areas from the Cerulean Warbler Forest Management Project (CWFMP) are in the core range of the species.

Conservation


Male Cerulean Warbler. Than Boves

About 80% of the total cerulean population breeds within the Appalachian Mountains Bird Conservation Region (BCR; Fig. 1), and they are particularly abundant within the central part of the region (Fig. 2). Declines have occurred across most of their range (Fig. 1). A range-wide loss of ~70% of the population (Fig. 3) led to their designation as a species of national conservation concern by the U.S. Fish and Wildlife Service (USFWS) and as a Continental Watch List species by Partners in Flight.

Cerulean declines are primarily related to the loss and reduced suitability of habitat on breeding, migration, and wintering grounds. On breeding grounds, the second growth forests that occur throughout most forested landscapes often lack the complex forest structure favored by ceruleans. Old-growth forests naturally develop a more open and complex canopy structure, as well as multi-layered shrub and mid-story layers. Maintaining older, structurally diverse forest within cerulean breeding range may be important to sustain populations in the long-term and to support the ecosystems on which they and other organisms depend. In managed forests, however, foresters and landowners can use silviculture as a tool to develop stands with structural and compositional characteristics that are favorable for cerulean and associated species. Partial harvesting to benefit ceruleans can be consistent with forest management goals such as promoting oak regeneration and managing for a diverse wildlife community.

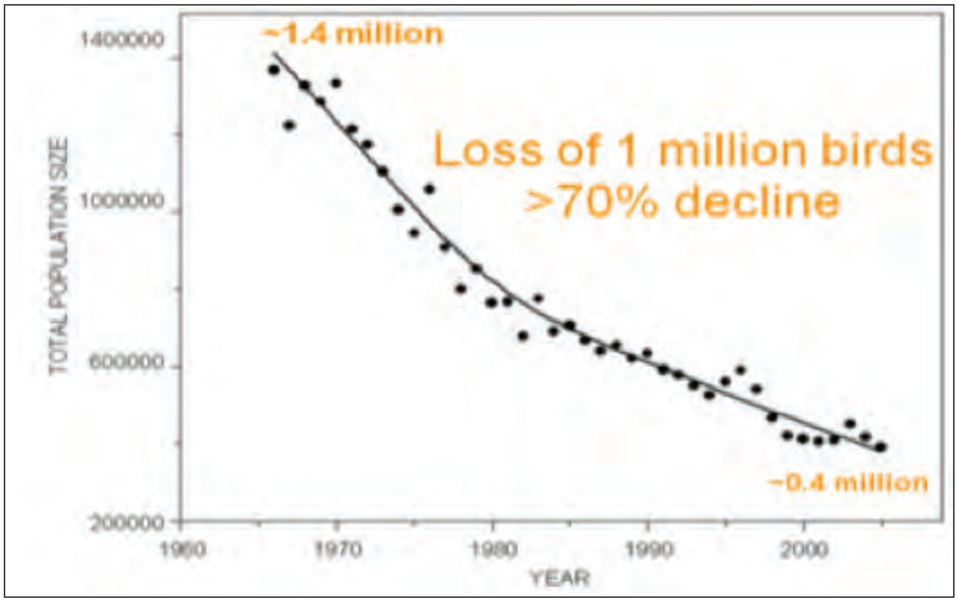
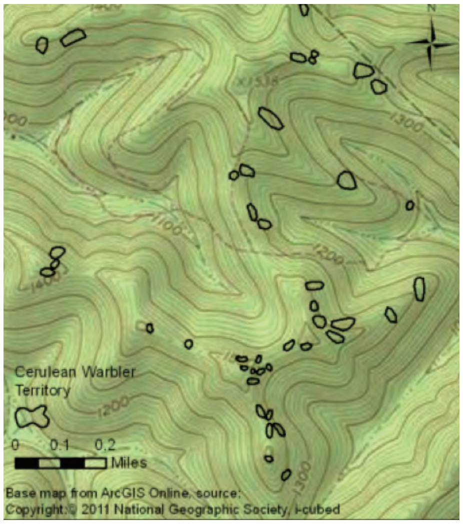


Figure 3. Cerulean Warbler population decline modeled using Breeding Bird Survey data from 1966-2006 (W. Thogmartin, unpubl. analyses).

Cerulean Warbler Habitat Association



Base map from ArcGIS Online, source: Copyright: © 2011 National Geographic Society, i-cubed

Figure 4. Cerulean Warbler territories on a topographic map of the Lewis Wetzel Wildlife Management Area, West Virginia, showing territories aligned along ridgelines and clustering near areas of local relief.

Cerulean breeding density is variable across the Appalachian region (Fig. 2). Their distribution is often patchy in part due to the patchy nature of canopy disturbance in mature forests and their strong association with ridge tops. In a southern West Virginia study, for example, they occurred at 40% of randomly placed sample points.

Landscape and Topography

Small forest tract size and the presence of large-scale edge (e.g., agricultural lands, mountaintop mines) can limit use of a site by ceruleans. Although the minimum forest tract size required by ceruleans to breed successfully is not known, smaller, more fragmented forest patches tend to have lower densities of territories and lower nest success. Ceruleans will use relatively small forest patches (~25 ac), but typically in landscapes that are primarily forested (e.g. >75% forest cover within ~6 miles of the project area). In landscapes with a relatively low proportion of forest cover (e.g. those that are dominated by agriculture), ceruleans are less likely to occur within small forest tracts. In the heavily deforested Mississippi Alluvial Valley, ceruleans require ~4000 acre tracts, in the highly fragmented Mid-Atlantic region ~1730 acres, and in the more forested Ohio Hills ~60 acres.

Ceruleans are often associated with canopy gaps and also use internal forest edges including narrow roads, narrow utility rights-of-way, narrow-cut strip mines, edges of small timber harvests, and trails. However, they are less abundant near abrupt or “hard” edges between forest cover and large expanses of open land (e.g., commercial, residential, and industrial development). In southern West Virginia, for example, cerulean abundance decreased near mountaintop mine edges and in northern West Virginia, they avoided edges of a large powerline right-of-way that was ~75 feet wide.

In the Appalachians, ceruleans primarily occur along ridges and steep, upper slopes and appear to cluster near areas of local relief such as knobs and bluffs (Fig. 4). The soil characteristics and topography of these features contribute to stratification of canopy trees so that ridge top forests often have a complex overstory structure containing large oaks with expansive crowns. Thus, ridge top forests often offer the structure and composition sought by breeding ceruleans. Within ridge top forests, ceruleans often favor mesic, north- and northeast-facing slopes, although other aspects are used. In some sections of the Appalachians (e.g. Delaware River valley), ceruleans are most dense at lower slope positions and along major waterways.

Appalachian landscape. Than Boves

Minimum patch size used by ceruleans depends on the amount of forest cover in the landscape.

Stand structure and Composition

Before extensive clearcutting in the late 19th and early 20th century, tree mortality from old age, windthrow, ice storm damage, and fire contributed to the development of structurally complex and relatively open stands in which oaks were dominant. In the even-aged stands that developed following those extensive harvests, natural canopy disturbances tended to be unevenly distributed and relatively small thereby creating a relatively homogenous canopy structure (e.g., a closed canopy forest with an undeveloped understory and/or midstory).

Important Components of Cerulean Habitat

Large Diameter Trees

Ceruleans place territories and nests in hardwood forests with well-spaced, large diameter trees (>16 inches dbh). Nests are typically in the largest trees available at a site.

Canopy Gaps and Structure

Ceruleans favor the complex canopy structure characteristic of uneven-aged stands and old growth forest. Canopy gaps allow mid- and upper-canopy trees the growing space to form long horizontal branches and develop dense foliage. Tree species composition is relatively diverse with shade-intolerant species abundant in the overstory.

Upland forest used by Cerulean Warbler. Marja Bakermans

Heterogenous stand structure including large trees, canopy gaps, and understory vegetation promote density and reproductive success of ceruleans.

A relatively open canopy structure provides ceruleans with dominant trees (i.e., taller than the surrounding canopy) where exposed perches aid the birds in broadcasting their song and whose expansive crowns offer ample foliage in which to forage and conceal nests. Nests are often placed along flat lateral branches that extend over a relatively open midstory and a relatively dense understory, conditions that occur adjacent to a regenerating canopy gap. Ceruleans preferentially use canopy gaps ~400-1000 ft^2 in size and that contain vegetative growth within them.

Oaks and Hickories

In the Appalachians, ceruleans are strongly associated with stands in which oaks and hickories (*Carya* spp.) predominate. They preferentially forage and nest in white (*Q. alba*) and chestnut oak (*Q. montana*), but they avoid red maple (*Acer rubrum*) and oaks from the red oak group (scarlet (*Q. coccinea*), black (*Q. velutina*), and northern (*Q. rubra*) and southern red oak (*Q. falcata*). On sites dominated by species other than oaks, ceruleans preferentially used black cherry (*Prunus serotina*) and black locust (*Robinia pseudoacacia*) in West Virginia and American elm (*Ulmus americana*) and sycamore (*Platanus occidentalis*) in Ohio for various activities.

Grapevines

Grapevines provide a favored source of nest material. Cerulean nest success was positively associated with density of grapevines (*Vitis spp.*) in Ohio perhaps because vines add complexity to the canopy and, consequently, reduce the search-efficiency of nest predators. In Maryland, fledglings often were observed perching within clumps of grapevines.

Understory Vegetation

Density and nest success of ceruleans have been positively associated with understory vegetation. In Ohio, vegetation surrounding nest locations had 24% greater understory vegetation density than random locations in the stand. A high density of understory vegetation is beneficial to ceruleans because 1) females frequently drop to the understory for intensive foraging bouts during incubation and brooding, and 2) fledgling birds often seek the dense vegetation for protection from predators.

Leave some grapevines
to provide nest material.

Female Cerulean Warbler incubating; note grapevine bark on the nest rim. This is a typical location for nests, i.e. on a lateral branch, next to a vertical twig, with an umbrella of leaves above the nest. Than Boves

Cerulean Warbler fledgling in thick understory vegetation. Marja Bakermans

Cerulean Warbler nest of grapevine and other materials. Marja Bakermans

Cooperative Cerulean Warbler Forest Management Project

The Cooperative Cerulean Warbler Forest Management Project (CWFMP), implemented under the auspices of the Cerulean Warbler Technical Group, was initiated to allow the scientific and management communities to test ideas about the habitat needs of ceruleans through experimental manipulations of timber harvest. The objective of the CWFMP was to study the response of ceruleans and the overall bird community to three silvicultural treatments and an unharvested control, collectively representing a canopy disturbance gradient. Seven study sites, each containing the four treatments, were established within mixed-mesophytic forest in Tennessee, Ohio, Kentucky, and West Virginia (Fig. 2). Sites were closed-canopy mature forest and located in heavily forested regions; forest cover within six miles of study areas averaged 83%. All stands were oak dominant.

Treatment plots were 50 acres in size and included an unharvested plot, a light harvest, a medium harvest, and a heavy harvest (Fig. 5). In harvested plots, treatments included a 25-acre harvest and a 25-acre section of undisturbed forest that bordered the harvest (hereafter buffers). Light harvests were single tree removals and residual basal area (RBA) averaged 93 ft^2/acre (range 84-106) resulting in stands that had~80% stocking. The goal of medium harvests was to thin the stand to

Pre-harvest, West Virginia LW study area, basal area = 121 ft^2/acre Patrick McElhone

Light harvest in 2007 (1 yr post-harvest), West Virginia LW study area, RBA=83.6 ft^2/acre. Patrick McElhone a residual stocking of 60-70% and favor the crown release of the best quality dominants and codominants. All other commercial stems (>6 inches dbh) were removed. The heavy harvests were applied with the objective of creating an understocked residual stand comprised of scattered dominants and co-dominants with all other commercial stems (>6 inches dbh) removed. After harvesting, the medium harvest had average RBA of 62 ft^2/acre (range 46-81) resulting in ~55% stocking. The heavy harvests had average RBA of 27 ft^2/acre (range 12-34). Basal area for unharvested plots averaged 117 ft^2/acre (range 95-138) with ~100% stocking.

The CWFMP is the largest forest management experiment ever conducted to evaluate cerulean warbler and associated songbird response to forest management. The results of the study demonstrate the initial response of ceruleans (first four years post-harvest) to forest management. Additional studies are needed to track cerulean response over the life of a managed stand to fully characterize the nature of the changes in habitat structure that occur in these stands and how ceruleans respond to these changes.

Medium harvest in 2010 (4 yrs post harvest), West Virginia LW study area, RBA=45.5 ft^2/acre. Jim Sheehan

Heavy harvest in 2008 (2 yrs post-harvest), Tennessee, RB study area. residual basal area (RBA)=34.5 ft^2/acre. Than Boves

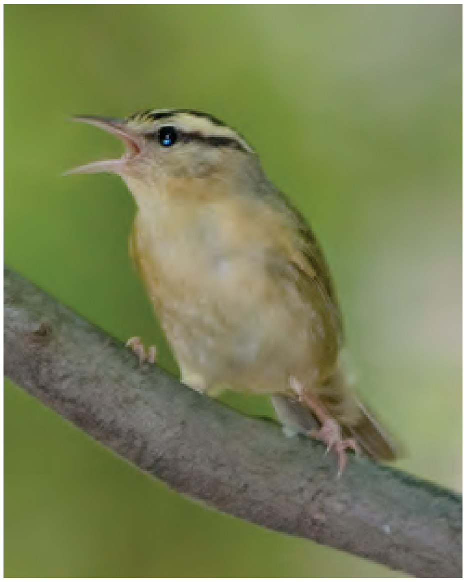
During two pre-harvest field seasons (2005-2006) and four post-harvest field seasons (2007-2010), data were collected on cerulean nest success, territory density, and habitat use. We also measured composition and relative abundance of the overall bird community to characterize response to partial harvesting and mapped territories of six other focal species in addition to Cerulean Warbler: Hooded Warbler (*Setophaga citrina*), Kentucky Warbler (*Geothlypis formosus*), Ovenbird (*Seiurus aurocapillus*), Scarlet Tanager (*Piranga olivacea*), Wood Thrush (*Hylocichla mustelina*), and Worm-eating Warbler (*Helmitheros vermivorus*).

Kentucky Warbler. Bill Hubick

Ovenbird. William Majoros

Scarlet Tanager. Bill Hubick

Wood Thrush. USFWS



Worm-eating Warbler. Bill Hubick

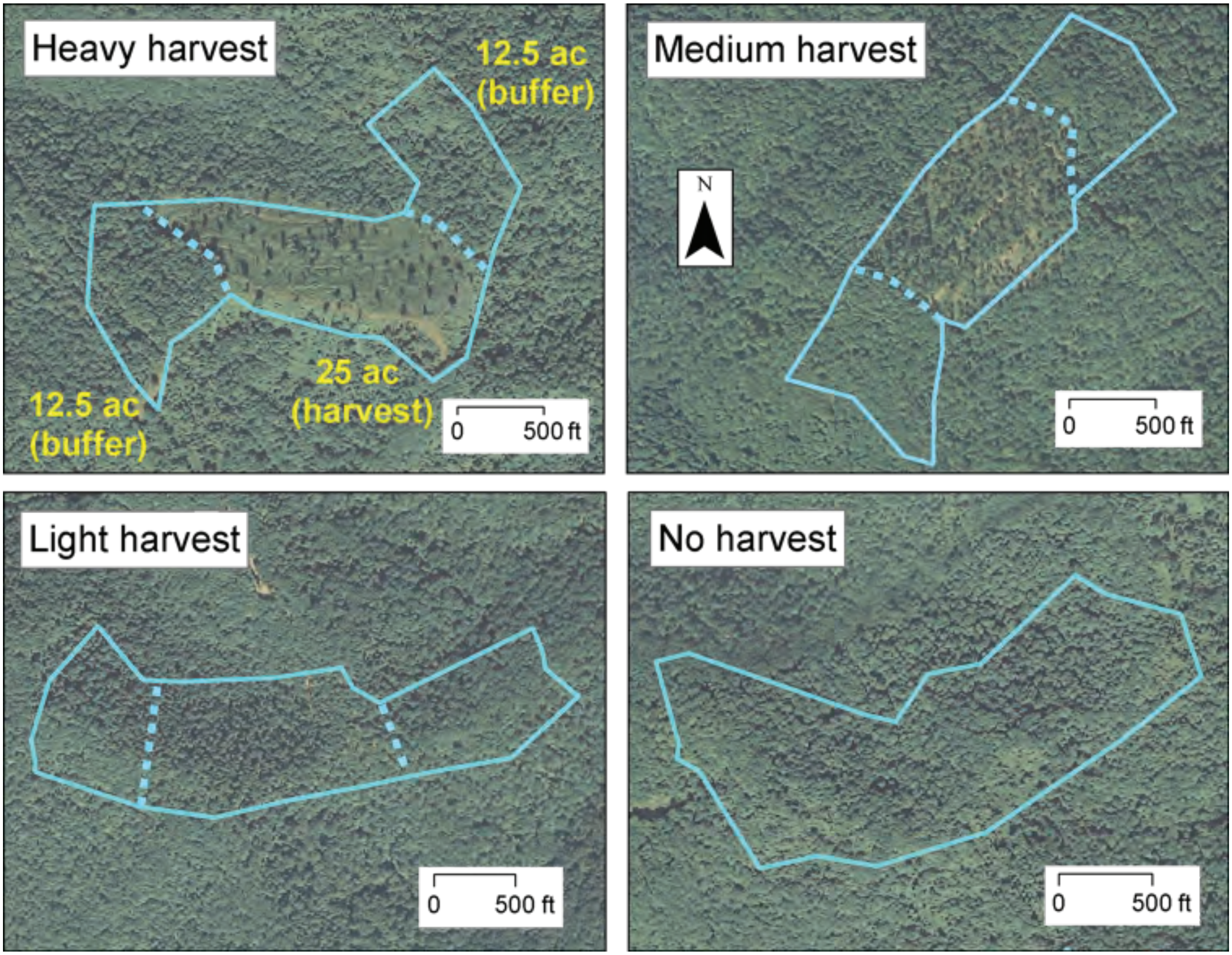


Figure 5. Plot layout in the CWFMP showing harvests and unharvested buffer areas one year after harvests were implemented on LW in WV.

Findings Relevant to Silvicultural Prescriptions

Short-term Response of Cerulean Warblers to Harvests

Territory Density

■ Across all harvests, cerulean territory density generally increased or was maintained and rarely decreased from pre-harvest densities (Fig. 6 top). The modeled response indicated that annual increases occurred (Fig. 7).

■ The largest and most consistent increases occurred when RBA was between ~40 and 90 ft²/ac (Fig 6 top, Fig 7). An extreme increase occurred in a harvest ~45 ft²/ac RBA where ceruleans were absent preharvest; post-harvest territories here were densely clustered.

■ Territory density increases that occurred at low levels of RBA (<40 ft²/ac) were typically delayed 2-3 years, likely in response to the time needed for understory foliage and structural development to occur in the residual stand. Within these heavy harvests, territories were often situated along the harvest edge (Fig. 8) and nests were rarely located within the harvest.

■ Single tree selection harvests with RBA >90 ft²/ac produced little increase in cerulean territory density (Fig 6 top).

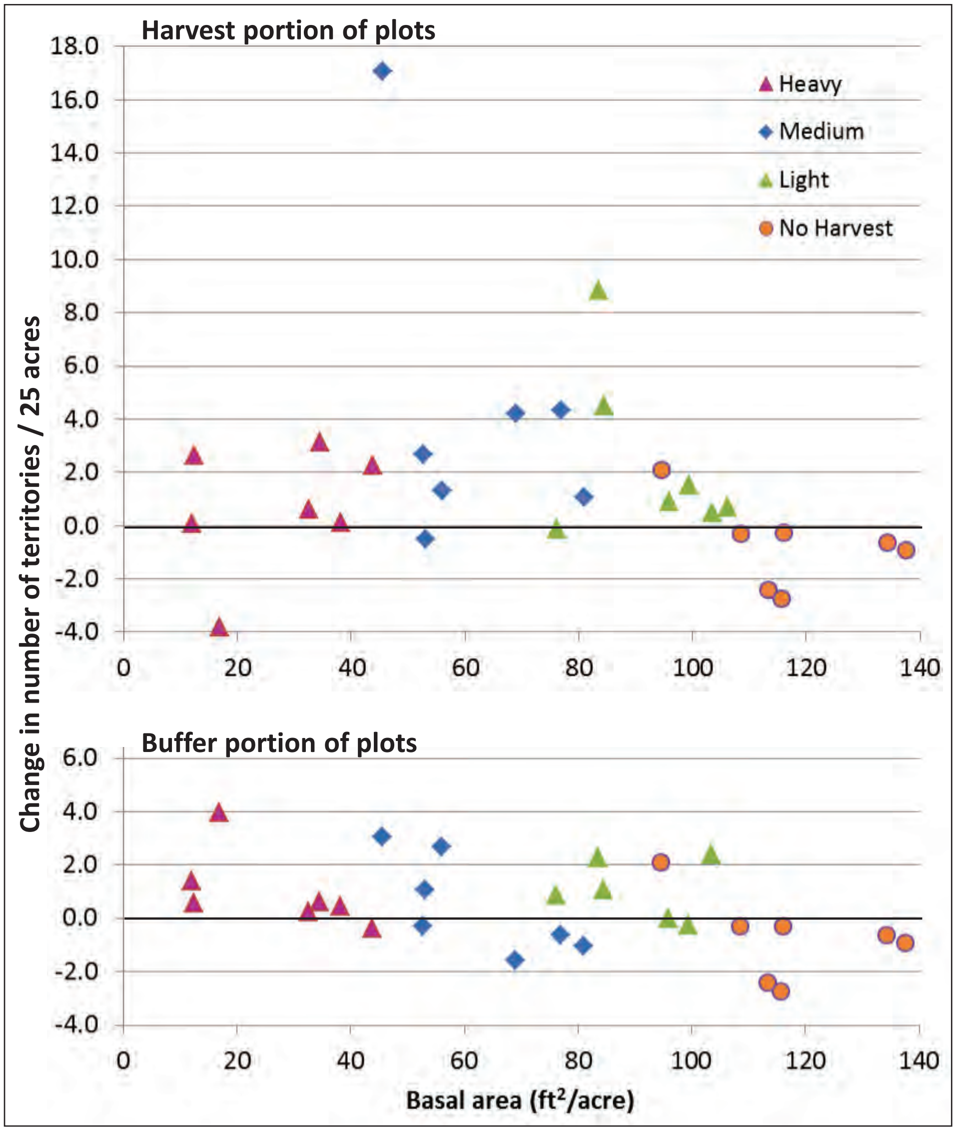


Figure 6. Mean change in number of cerulean warbler territories per 25 ac from 2006 (pre-harvest) to 2007-2010 (post-harvest) relative to post-harvest basal area and harvest intensity. Top figure is within harvests and bottom figure is within unharvested buffers. Points above the 0 line indicate plots with a mean increase in number of territories.

Ceruleans favor residual basal area of ~40 to 90 ft²/acre of canopy trees.

- Although the territory density response to harvests was generally positive (Fig. 6 top, Fig. 7) it was variable across study sites likely due to differences in pre-harvest cerulean densities, topography, and forest structure and composition.

- In the majority of unharvested buffers (Fig. 6 bottom), cerulean territory density mostly increased or was maintained regardless of intensity of the adjacent harvest.

- Some degree of thinning in the canopy of oak-dominated stands with basal area >~130 ft^2/ac would likely benefit ceruleans because territory density generally was low on these highly stocked stands (Fig 7).

Cerulean Warbler male with color bands.
Matt Shumar

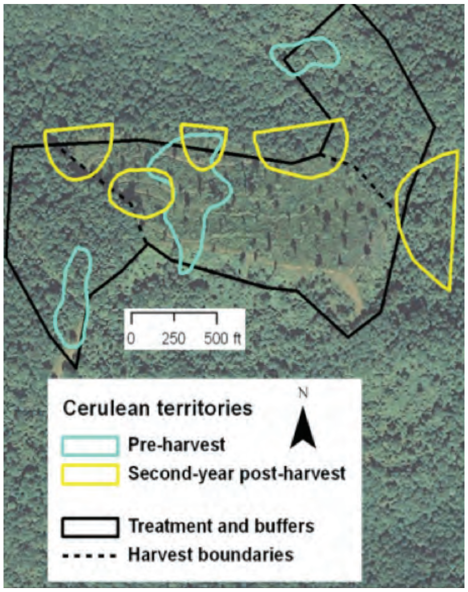


Figure 8. Cerulean Warbler territories aligned along the edge of a 20 acre heavy harvest with 12.5 ft^2/ac of residual basal area. Territories before the harvest are shown in blue and after harvest are in yellow. The birds used little of the interior of the cut.

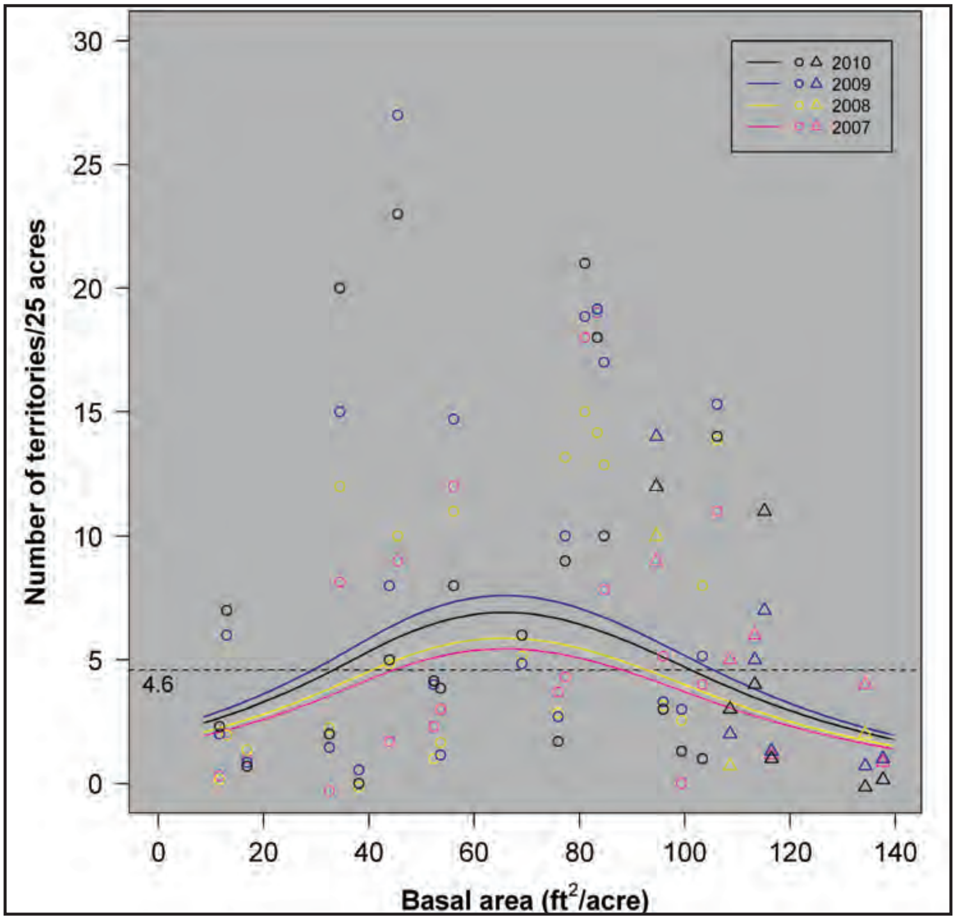


Figure 7. Annual number of post-harvest (2007-2010) cerulean warbler territories per 25 acres (circles=harvests; triangles=no-harvest control) relative to post-harvest basal area. Curved lines are the annual post-harvest predicted response for a plot with 4.6 pre-harvest territories/25 acres (the pre-harvest mean indicated by the thin dotted horizontal line).

Nest Success

- Nest success varied strongly by study site and year and was relatively low at many of the study areas. Harvest intensity had less influence on nest success than study area and year.

- Unharvested buffers adjacent to the harvests had nest success similar to that of the unharvested control stands.

- Of the three harvest treatments, medium harvests had higher nest success than light or heavy harvests (Fig. 9). However, unharvested control stands in the South region (the two Tennessee study areas) had higher nest success than any harvest.

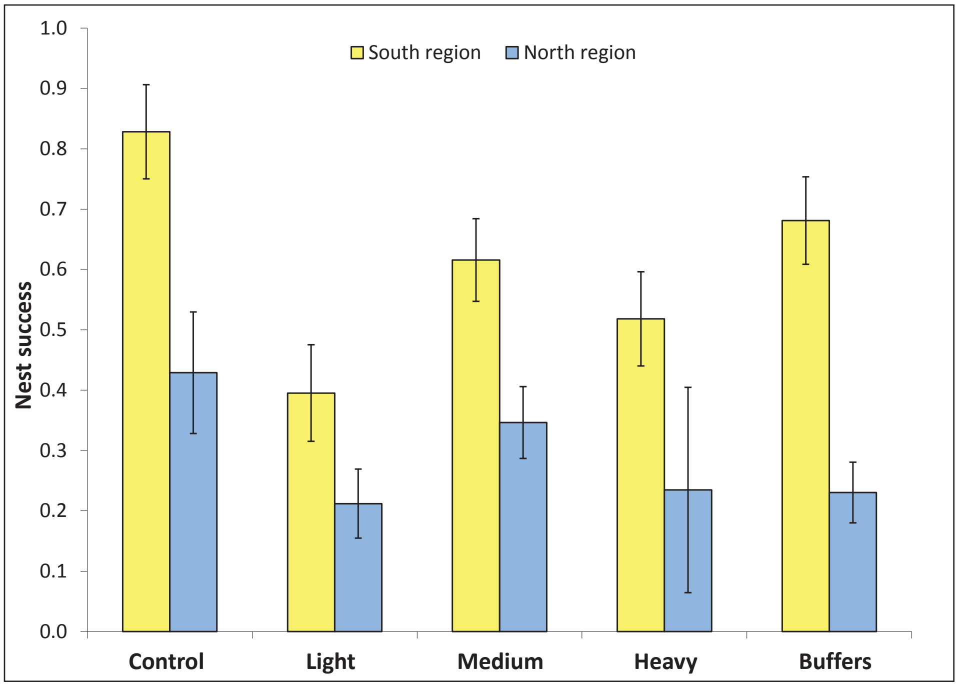


Figure 9. Cerulean Warbler nest success (with standard error bars) for the no harvest control, the three harvest treatments, and the unharvested buffers.

Male Cerulean Warbler with newly hatched chicks. Ohio DNR

Habitat Use

- For nest trees, ceruleans preferred white oak, sugar maple (*A. saccharum*), and cucumber magnolia (*Magnolia acuminata*) as nest trees and avoided red maple and oaks from the red oak group (scarlet, black, and northern and southern red oak) (Fig. 10).

- For foraging, they preferred sugar maple, chestnut oak, and hickories and again avoided oaks from the red oak group (Fig. 11).

- Ceruleans placed their nests in trees that averaged 15-19 inches dbh across the study areas. Nest trees were larger than random trees within the territory. Vegetation structure adjacent to nest trees had less mid-canopy cover and more understory cover than generally available within the surrounding territory. These conditions are characteristic of canopy gaps that have some vegetative growth within them.

White oaks, hickories, and sugar maples are favored for nesting and foraging.

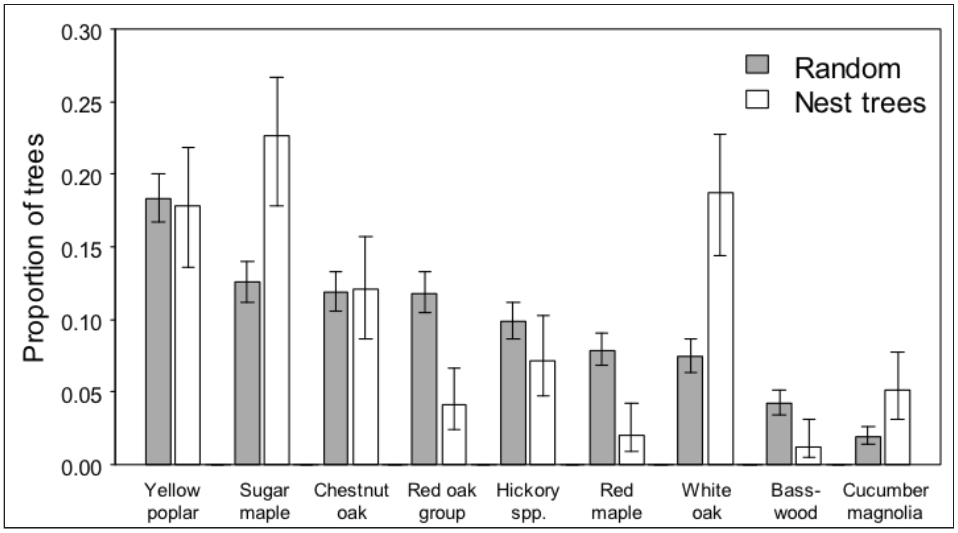


Figure 10. Nest tree selection by Cerulean Warblers at all study areas (pooled) in the Appalachian Mountains, 2008–2010. For each tree species, bars and 95% confidence intervals are the proportion of total trees within randomly sampled plots (gray) and the proportion of total nest trees (white). Red oak group includes northern red (Quercus rubra), black (Q. velutina), and scarlet (Q. coccinea) oak, and hickory species include mockernut (Carya tomentosa), bitternut (C. cordiformis), pignut (C. glabra), and shellbark (C. laciniosa) hickory. Only the most common tree species are shown.

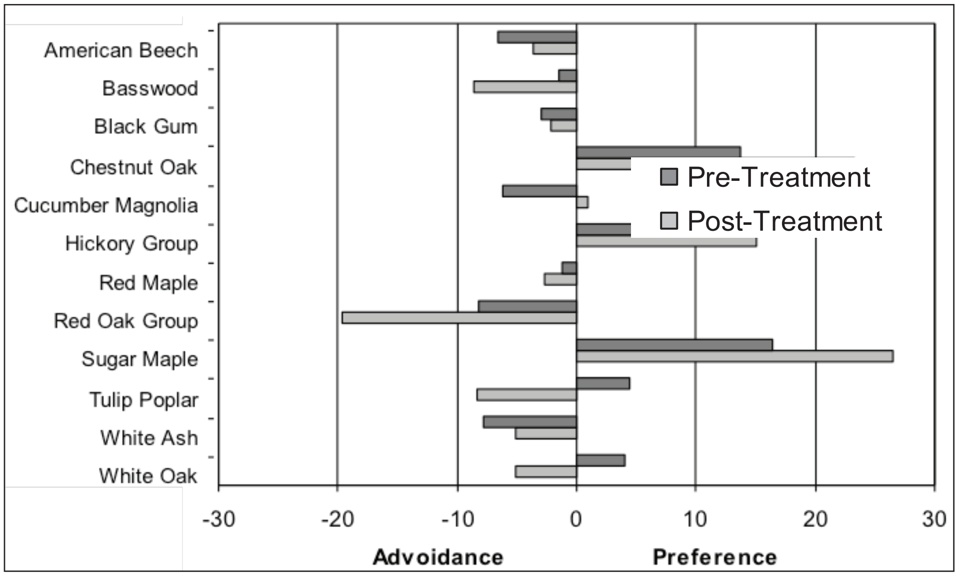


Figure 11. Pre-harvest (2006) and post-harvest (2007) indices of tree species preference and avoidance by Cerulean Warblers for the 12 most commonly available tree species.

Changes in Allied Bird Communities

Appalachian forests are considered some of the most biologically diverse temperate forests in the world. They provide breeding habitat for many avian species including those dependent on closed-canopy forest, others that require young forest habitat, and some species that require mature forest with canopy gaps. Consequently, individual species responded in various ways to different levels of RBA (Table 1).

- Ovenbird, a species that nests and forages on the ground, had its greatest abundance at high RBA (>90 ft^2/ac; Fig. 12). An immediate negative response to canopy removal persisted four years after harvests in heavy and medium harvests. Ovenbirds occurred at moderate densities in light harvests (>85 ft^2/ac).

- Species that nest in the midstory of older forests such as Wood Thrush and Acadian Flycatcher (*Empidonax virescens*), also had immediate and persistent reductions in abundance in response to canopy removal in heavy and medium harvests. This was likely in response to midstory removal and the open canopy and dense understory conditions that developed in response to these harvest levels.

- Heavy and medium harvests increased abundance and diversity of shrub-nesting species including Hooded Warbler (Fig. 12), Indigo Bunting (*Passerina cyanea*), Yellow-breasted Chat (*Icteria virens*), Kentucky Warbler, and Eastern Towhee (*Pipilo erythrophthalmus*). These species are associated with low RBA and high shrub cover. Response of some species, e.g. Hooded Warbler and Kentucky Warbler, was delayed until dense shrub cover developed.

- Certain canopy-nesting species such as Cerulean Warbler and Blue-gray Gnatcatcher (*Polioptila caerulea*) generally increased in abundance at intermediate levels of RBA across the study sites while Eastern Wood Pewee (*Contopus virens*) increased only in Ohio at intermediate RBA. Some canopy-nesters that are less sensitive to small-scale harvesting, like Scarlet Tanager, had similar abundance across the range of harvest intensities.

These short term effects are from small-scale harvesting (~25 ac) within relatively continuous mature forest. Avian species may respond differently to larger harvests, more extensive harvesting, or harvesting within landscapes with less forest cover.

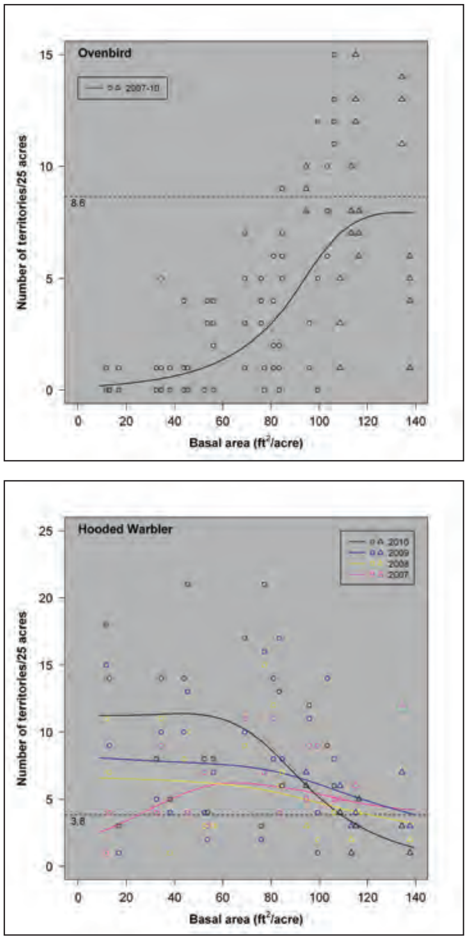


Figure 12. Number of post-harvest (2007-2010) Ovenbird and Hooded Warbler territories per 25 acres (circles=harvests; triangles=no-harvest control) relative to post-harvest basal area. Negative (Ovenbirds) and positive (Hooded Warbler) predicted responses to basal area are shown by curved lines (the pre-harvest mean indicated by the thin horizontal line). For Hooded Warbler, there was an annual increasing response during 1 to 4 years post-harvest.

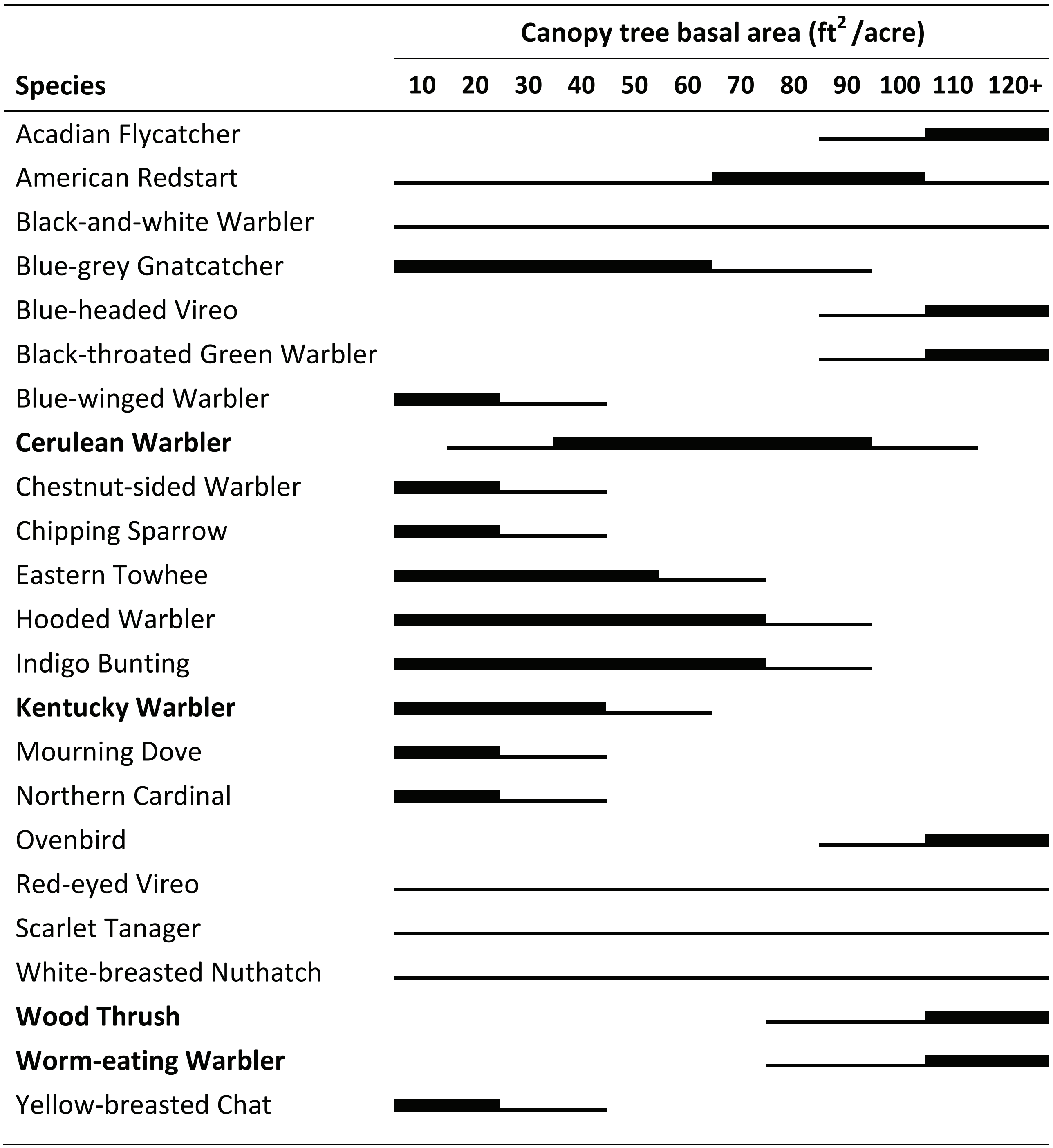


Table 1. Suitable and optimal (thickest line) basal areas for migratory songbirds that were common at CWFMP study sites. Bolded species are USFWS Birds of Management Concern. Relative abundance and/or territory density for a given species was highest under optimal basal area ranges and the species was present under suitable ranges.

Management Considerations

Female Cerulean Warbler. Ohio DNR

Cerulean Warblers occur on forested lands throughout its range. Landowners desirous of keeping their lands in forested condition can do so using the economic benefits derived from productive forest management. In mature forest stands that have high cerulean densities and high nest success, the no-harvest option is most favorable for sustaining cerulean populations. In actively managed forests, there are opportunities to use forest management practices to mimic the structure and natural disturbance regimes of old-growth forests to enhance habitat for this species. The results from the CWFMP indicate that retaining RBA levels of ~40-90 ft^2/acre after harvesting trees in 25 acre harvest units in oak-dominated stands creates a forest structure that is generally favorable for ceruleans. Small-sized harvest stands (~10-27 acres) and their edges are not avoided by ceruleans.

In addition to enhancing stand conditions for ceruleans, small-scale harvests that result in intermediate levels of RBA are consistent with promoting oak regeneration and a diverse wildlife community. These harvests create habitat for early-successional birds, many of which are experiencing long-term population declines. For example, in northeast Pennsylvania, stands of regenerating timber attract Cerulean Warblers to use both the mature forest edge and adjacent residual trees in the harvest while providing breeding habitat for Golden-winged Warblers (*Vermivora chrysoptera*). Opening the canopy also can enhance habitat for many species of forest-dwelling bats. A study of bat use of the CWFMP treatments found increased bat foraging activity within partial harvests than in unharvested plots.

Important considerations for implementing harvests for ceruleans include the following:

Landscape-scale Considerations

Forest Cover

Some studies of forest songbirds have found decreased nest success in landscapes with a low proportion of forest cover. In heavily forested regions, the abundance and productivity of ceruleans and other forest songbirds appear to be more heavily influenced by stand structure than by landscape or edge effects. Thus, habitat enhancements for ceruleans located in heavily forested regions (>70% forest cover at the six mile scale) are more likely to be effective at attracting ceruleans and landscape context may have less influence on reproductive success.

Scale of Harvesting

Even in heavily forested regions, maintaining a significant portion of the management area as mature forest cover is important for sustaining populations of forest-interior birds because many forest-interior birds are sensitive to the amount of mature forest cover at larger spatial scales. In addition, several mature forest dependent species (e.g., Wood Thrush, Worm-eating Warbler, and Acadian Flycatcher) are likely to decrease in abundance at intermediate levels of RBA. Thus, where these species are high priority, maintaining about 50% of large forest blocks in the >50 year-old age class will provide structural complexity yet retain closed-canopy forest availability.

White Oak dominated habitat. Fran Trudeau

Stand-scale Considerations

Local Cerulean Density

Where cerulean density is relatively high (>5 territories/25 acre), immediate habitat enhancements are not necessary because harvesting may reduce reproductive success which may outweigh any increases in cerulean breeding density. Ideal locations to focus management efforts are where local cerulean densities are low (<5 territories/25 acre). If no ceruleans are present near the management site (within ~5 miles), they may be less likely to colonize the managed area.

White Oak Dominance

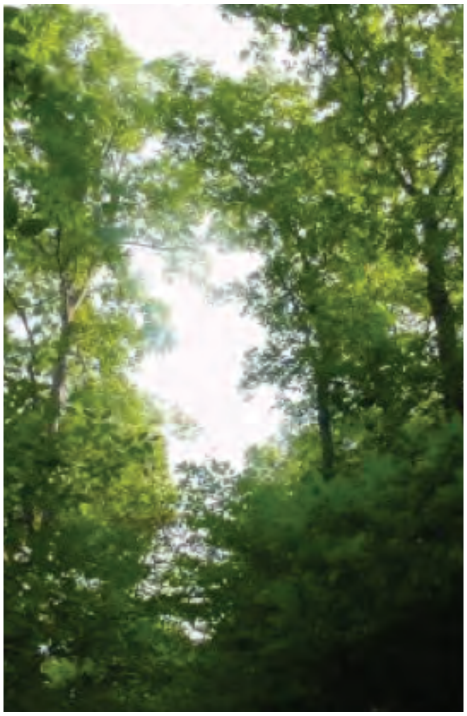
Maintaining white and chestnut oak dominance in the residual stand is a primary consideration in implementing management strategies for ceruleans. Thus, site productivity and the presence of sufficient advance regeneration of white and chestnut oaks are important considerations in management. Where feasible, favor white oak, chestnut oak, hickories, and sugar maple in the residual stand and do not retain red maple or red oaks. Retain some of the largest diameter individuals of the preferred species as residual trees. Prescribed fire at regular intervals may be necessary to promote oak regeneration, maintain small canopy gaps, and facilitate understory vegetation diversity.

Retain large diameter white and chestnut oak trees in any management scenario.

Topography

In much of the Appalachians, harvests located along ridgetops and upper slopes are likely to be more effective in attracting ceruleans. Mesic, north- and east-facing slopes are often favored by ceruleans although other aspects are used.

Size of Canopy Gaps
Ceruleans preferentially use canopy gaps that are ~400-1000 ft^2 in size, particularly those with advanced vegetative growth within them. Thus, group-selection harvests that allow already established regeneration to grow into a stratified canopy may benefit this species.


Canopy gap in West Virginia.
Scott Bosworth

Temporal and Silvicultural Considerations

A number of different silvicultural practices could achieve residual basal areas in the harvested stand that are suitable for cerulean warblers (~40-90 ft^2/acre). Some additional considerations for various silvicultural treatments are below.

■ *Single-tree selection harvests* (our light harvest treatment) were less effective in increasing cerulean numbers and rapid canopy closure may limit the duration of suitable habitat. Single-tree selection with RBA above ~90 ft^2/acre also led to lesser nest success than harvests with lesser RBA. However, if single-tree harvest is favored by a landowner for providing income, cerulean densities would still be maintained particularly if non-preferred trees are removed and preferred oaks are retained.

■ *Group selection as part of an uneven-aged system* can improve cerulean habitat and would likely be effective longer than single-tree selection. The small group openings provide for diverse canopy structure and understory development. This approach has been shown to advance stands toward late successional structure beneficial to many avian species.

■ *Shelterwood harvests* are often compatible with promoting oak regeneration and, in the CWFMP, generally resulted in increased cerulean density and intermediate levels of nest success. However, complete overstory removal during the second stage of a shelterwood harvest will substantially reduce numbers of mature forest species including Cerulean Warbler, Wood Thrush, Acadian Flycatcher, and Worm-eating Warbler. If managing for forest birds, retain the residual canopy as long as possible and until adjacent habitat has been enhanced with shelterwood or other types of harvests and colonized by ceruleans.

■ *Thinnings* as part of intermediate harvest treatments would open the canopy and provide the structure favored by ceruleans. These could take the form of a crown thinning or shelterwood seed cut.


Shelterwood harvest. Scott Stoleson

■ *Modified even-age regeneration* can be used to create future opportunities for cerulean habitat improvement. Leaving large-diameter residual stems in a harvest unit can lead to development of two-aged stands. Such stands achieve more complex canopy structure earlier in their development than similar single-aged stands and the residual stems allow for some use of the stand by forest birds. Ceruleans had increased density in RBA of >~40 ft^2/acre.

■ *Crop-tree release* is a practice that is used to accelerate development of crop-trees on higher quality sites. The practice is typically applied in 15 to 20 year-old stands. It can allow for earlier canopy differentiation by accelerating growth of dominant stems. Impact on habitat suitability for ceruleans will not be immediate, but benefits should be seen as the stand develops and where earlier entry into the stand for commercial harvest is made possible.

Complex canopy structure in a deferment cut creates future opportunites for Cerulean Warbler habitat improvements. Doug Becker

Summary

Forest management that incorporates these guidelines and that is applied to oak-dominated stands in the Appalachian region can enhance habitat for Cerulean Warblers and other avian species, as well as other wildlife. Managers can choose a range of residual basal area targets depending on their priority avian species of interest.

For ceruleans, the RBA target range of ~40-90 ft^2/acre results in the most increases for the longest time period. A variety of silvicultural approaches can achieve this range. Where cerulean densities are high (>5 territories/20 acres), habitat management is not likely to be needed.

Landscape considerations are also important. These recommendations may be most beneficial in areas with high forest cover. They have not been tested in landscapes where forest cover is low.

Sitting pretty. Bill Hubick

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Appalachian landscape. Charlie Choc

Northern Bobwhite Demographic and Population Response Following an Intensive Habitat Modification to an Agricultural Landscape

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Northern Bobwhite (*Colinus virginianus*) populations have been declining throughout most of their endemic range due to numerous factors (e.g., increased urbanization, predators); however, changing land-use practices have proved most detrimental to bobwhites. In parts of the southeastern USA, small-scale farming has been replaced by large-scale center-pivot irrigated fields and this has exacerbated habitat loss. Despite these trends, bobwhite populations in the Southeast have remained stable or increased on many areas employing intensive habitat management regimes, substantiating the importance of appropriate habitat management for long-term bobwhite persistence. In effort to reverse one such decline, we intensively modified a center-pivot, agriculture dominated landscape to benefit bobwhites by creating new habitat and improving existing habitat. Techniques utilized to modify this landscape were: establishment of linear habitats (field borders and buffer strips); planting longleaf pines; and management of existing habitat via prescribed burning and timber management. During 1998-2001, we monitored bobwhite (n = 498) demographics and population response following annual habitat restoration and management using radio-telemetry and fall abundance estimation (i.e., covey call-counts). Average survival during over-winter (0.4698, SE = 0.0721), breeding (0.3561, SE = 0 .0667) and annual (0.1673, SE = 0.0411) time-periods were higher than those reported for other agriculture studies and similar to those of intensively managed, "plantation" habitats. Bobwhite coveys and broods used newly developed longleaf pine, linear habitats (e.g. field borders/hedgerows), and managed woodlands. Further, nest site selection was commonly associated with these novel habitat types. As a result of the positive demographic response to habitat modification, bobwhite abundance also improved during the study. Consequently, we surmised that modification of agricultural landscapes may improve habitat quality and quantity for bobwhites and subsequently help to increase demographic rates and bobwhite abundance.

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Introduction

Despite being the most studied upland gamebird in North America, northern bobwhite (*Colinus virginianus*) populations have continued to decline throughout most of their endemic range. Whereas declining populations have been associated with various factors (e.g., increased urbanization, changing predator dynamics), changing land-use practices have proved most detrimental to bobwhites (Brennan 1991, 1999, Church et al. 1993, Rollins and Carroll 2001). Recent changes among agriculture landscapes have dramatically affected bobwhites by reducing habitat quantity and quality (Brennan 1999). Clean farming, larger fields, center-pivot irrigation systems and increased herbicide and pesticide use (Capel et al. 1993, Sotherton et al. 1993) have become a common rubric among these landscapes-an ecosystem which once supported high densities of bobwhites. As these habitats, which once benefited

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bobwhites, have diminished, bobwhite populations have also waned. Further, more intensive or lack of management among remaining forested habitats (e.g. silviculture) surrounding agriculture fields has also contributed to habitat loss (Burger 2002). Notably, the declining status of bobwhite populations are not unique to bobwhites, but have also affected numerous species of songbirds (Conover 2005). Contrary to these trends, bobwhite populations in the Southeast have not declined on many areas that have employed intensive habitat management regimes (Brennan et al. 2000, Palmer et al. 2002, Stribling and Sisson 2009); this substantiates the importance of appropriate habitat management to maintaining long-term bobwhite populations.

During the past decade, in an attempt to mitigate habitat loss among agricultural landscapes, federal Farm Bill programs (e.g., CRP, WHIP, EQIP) have been implemented to provide landowners monetary incentive to restore or set aside portions of their cropland to promote early-succession vegetation (Burger 2002). Numerous management practices qualify for enrollment in these programs benefiting bobwhites and other species: cool- and warm-season grass plantings; conservation tillage; exotic grass control; wildlife habitat improvement or development; pine tree management; and linear habitats (LH) including filter or buffer strips, field borders and riparian buffers. However, the utility of these linear habitats, pine plantings, and other habitats, when applied to agricultural ecosystems, to bobwhite demographics and population growth is poorly understood.

Previous research has demonstrated that bobwhites used, and in some cases preferred, LHs for normal daily activities, brood-rearing and nesting (Puckett et al. 2000, Cook 2004). Likewise, previous research has documented increases in bobwhite and songbird abundance when combined with mesomammal reduction (Bromley et al. 2000) and without mesomammal reduction (Hamrick 2002, Cook 2004, Conover 2005). Additionally, Cook (2004) found that bobwhites on areas with LHs exhibited higher survival and lower dispersal proclivities compared to areas without LHs.

However, despite the wide-spread habitat implementation gained from Farm Bill programs and the purported population increase associated with LHs and agricultural ecosystems as mentioned above, the utility of LHs relative to bobwhite demographic parameters at both the local and regional scale remains uncertain. Similarly, few studies have examined the utility of planting longleaf pines (PPs) among agriculture landscapes to improve bobwhite habitat. Whereas previous studies revealed that bobwhite abundance increased on areas with LHs compared to areas without them (Bromley et al. 2000, Hamrick 2002), their methods employed could not provide the means to adequately ascribe whether the observed population increase was a consequence of higher survival, increased reproductive success, or due to immigration. Furthermore, although Puckett et al. (2000) suggested that linear habitats were preferred among bobwhites, they reported that nest survival was low, particularly during the early nesting season. Moreover, Cook (2004) suggested that more research was needed to examine the utility of linear habitats to bobwhite broods. Thus, more research has been warranted to ascertain whether novel habitats improve demographic parameters and provide a practical utility to facilitate reversal of population declines observed among agricultural landscapes.

The primary objective of this study was to examine the utility of augmenting an agricultural dominated landscape with novel habitat types and examine bobwhite habitat-use, demographics, and population response following an intensive modification. We intensively modified the center-pivot, agriculture dominated landscape to benefit bobwhites by creating new habitat and improving existing habitat. Techniques utilized to modify this landscape were: establishment of linear habitats, field borders and buffer strips; planting longleaf pines at a conservative spacing; and management of existing habitats via prescribed burning and timber management. Finally, we compared our results, when applicable, from this study to intensively managed sites and an unmanaged agriculture site in southwest Georgia since we did not have pre-treatment demographic data.

Study Area

The study was conducted on a privately-owned property, Whitehall Plantation (3734 ha), in Laurens and Bleckley counties, Georgia, USA. This study site was located in the Upper Coastal Plain physiographic region near the fall line. Prior to intensive habitat modification during 1998-1999, the study site was comprised of dry and irrigated agriculture fields (55%), unmanaged woodlands (40%) comprised of mixed hardwoods and pines (*Pinus* spp.), and 5% other, miscellaneous-type habitats (e.g., pastures, ponds). During this time, the primary land-use objective was agriculture (i.e., row-crop farming) and the estimated bobwhite population was <1 bird/4 ha. However, during 1997, the primary land-use objective changed to management that benefited northern bobwhites, but farming remained an objective-albeit secondary.

During 1998-1999, intensive habitat management was undertaken converting the agriculture predominated landscape to a landscape more conducive to bobwhites. We employed numerous habitat techniques to improve habitat for bobwhites: dryland agriculture fields were planted in longleaf pines (*Pinus palustris*); 15 m field borders, buffer strips, and hedgerows were created in all irrigated, agriculture fields; no-tillage farming practices was implemented; annual autumn disking and fallow field management was employed to stimulate annual weed production and arthropods for bobwhite broods; and both chemical and mechanical silvicultural treatments to decrease basal area (timber density) among upland and lowland timberland areas was applied as needed. As such, the new landscape matrix was comprised of agriculture (22%), managed woodlands (21%), and planted longleaf (21%) with interspersed linear habitats (LH [12%]; hedgerows, terraces and field borders), hardwoods (10%), other (ponds, pastures, etc; 8%) and fallow fields (6%).

Methods

Trapping and Monitoring

We trapped bobwhites during October-November and March-April 1998 - 2002 using standard funnel traps (Stoddard 1931) baited with grain sorghum and cracked corn. We covered traps with brush (e.g., fresh-cut pine limbs) to minimize stress on captured birds and to conceal traps from predators. We classified bobwhites by age and gender, and we weighed, leg-banded and released them at the capture sites. We outfitted birds weighing $\geq$132 g with pendant-style (Mueller et al. 1988) transmitters (6.0 g) equipped with an activity switch (Holohil Systems Ltd., Ontario, Canada). Trapping, handling, and marking procedures were consistent with the guidelines in the American Ornithologists' Union Report of Committee on the Use of Wild Birds in Research (American Ornithologists' Union 1988) and the protocol was approved by the Auburn University Institutional Animal Care and Use Committee, IACUC (Protocol Review Numbers: 2002-0364).

Survival - We monitored bobwhites $\geq$3 times weekly using the homing method (White and Garrott 1990, pg. 42). We approached birds within 25-50 m to minimize location and classification errors; and entered the locations into a geo-database using Geographic Information Systems (GIS) and ArcView® software (Environmental Systems Research Institute, Inc.). We determined specific causes of mortality when possible, by evidence at the kill site and condition of the radio-transmitter (Curtis et al. 1988). When radio contact was lost, we systematically searched on and off the study area within approximately 5 km of the bird's last known location.

Reproduction - During nesting season, we assumed inactive birds, determined via an activity switch, observed in the same location on 2 consecutive days to be nesting. We approached inactive hens and marked their location with flagging tape at a distance of 5-10 m and recorded the location in our geo-database. We monitored nests $\geq$5 times weekly and determined exact nest location and number of eggs when the incubating hen left the nest to feed.

Table 1: Models explaining northern bobwhite survival derived via Program MARK (known-fate model; c-hat = 1.78) relative to gender- and time-dependent factors for Whitehall Plantation located in Laurens and Bleckley County, Georgia, 1999 - 2002.

Model	K	$QAIC_c$	$\Delta\, QAIC_c$	QDeviance	W_i
S(season-constant)	2	1605.8567	0.0000	1601.8554	0.4700
S(season + gender)	3	1607.4971	1.6404	1601.4942	0.2070
S(.)	1	1608.2812	2.4245	1606.2806	0.1398
S(season+gender*season interaction)	4	1608.9472	3.0905	1600.9428	0.1002
S(. + gender)	2	1609.7107	3.8540	1605.7095	0.0684
S(annual-constant)	4	1613.6336	7.7769	1605.6295	0.0096
S(season-time)	7	1614.9662	9.1095	1600.9549	0.0049
S(t)	26	1636.7524	30.8957	1584.6097	0.0000

We monitored nests daily from distances of >10 m and we determined fate of the nest as abandoned, successful, or unsuccessful. We defined a depredated nest as any nest in which ≥1 eggs was destroyed and the adult bird did not return to incubate the remaining clutch. A nest was deemed abandoned when the hen did not complete incubation and all eggs were still intact. We defined a nest successful when ≥1 egg hatched.

Statistical Analysis

Survival And Cause-specific Mortality - We used the known-fate model in program MARK (version 5.2; White and Burnham 1999) to explain variation in survival, estimate daily survival rates and estimate the probability of surviving explicit time-periods (e.g., season, year) for male and female bobwhites. The known-fate model employs a binomial likelihood (weekly in our case) and permits incorporation of individual covariates (e.g., gender) delineated by groups (e.g., years in our case) to evaluate their affect on survival. When the fate (alive, dead or censored) of every radio-marked animal is known for each survival interval, the known fates model generates Kaplan-Meier survival estimates (Kaplan and Meier 1958, Pollock et al. 1989). However, because we had missing data for some intervals (e.g., when radio-contact was lost or bobwhites were not checked during a given interval due to stochastic events [i.e., inclement weather]), the variance components of the survival estimates generated from the known-fate model in program MARK are more suitable than those calculated by traditional Kaplan-Meier methods.

We used an information-theoretic approach (Burnham and Anderson 2002, Anderson et al. 2000) to evaluate the set of candidate models. The models were developed *a priori* based on biological insight to avoid superfluous model building (i.e. data dredging). The best approximating model in the set of candidate models was determined by Akaike's Information Criteria (AIC); adjusted for small sample bias and over-dispersion ($QAIC_c$; Burnham and Anderson 2002). We used the median c-hat method as implemented in Program MARK to assess and correct for over-dispersion (c-hat = 1.78) among our data. $QAIC_c$ is a valid model selection method for both nested and non-nested sets of models (Burnham and Anderson 2002). $QAIC_c$ was used to compare each candidate model, and the model with the lowest $QAIC_c$ value was considered to be the best approximating model given the data.

Nest Survival - We estimated daily survival rate

Table 2: Predicted probability of surviving (mean survival and 95% confidence intervals) during overwinter (OW), breeding (Breed) and annual time-periods derived via Program MARK for northern bobwhites located on Whitehall Plantation in Laurens and Bleckley County, Georgia, 1999 - 2002.

Season	DSR[a]	SE[b]	LCI[c]	UCI[d]	Survival	SE	LCI	UCI
OW 1998 - 1999	0.9757	0.0054	0.9625	0.9843	0.5272	0.0748	0.3806	0.6737
Breed 1999	0.9627	0.0073	0.9454	0.9746	0.3717	0.0713	0.2319	0.5116
ANNUAL	0.9697	0.0044	0.9597	0.9773	0.2024	0.0474	0.1094	0.2953
OW 1999 - 2000	0.9693	0.0056	0.9563	0.9786	0.4448	0.0655	0.3164	0.5732
Breed 2000	0.9605	0.0067	0.9449	0.9717	0.3503	0.0625	0.2277	0.4729
ANNUAL	0.9652	0.0043	0.9556	0.9727	0.1583	0.0365	0.0867	0.2299
OW 2000 - 2001	0.9710	0.0054	0.9582	0.9800	0.4656	0.0667	0.3348	0.5963
Breed 2001	0.9600	0.0072	0.9431	0.9720	0.3463	0.0664	0.2162	0.4764
ANNUAL	0.9662	0.0044	0.9565	0.9739	0.1677	0.0392	0.0908	0.2446
OW 2001 - 2002	0.9690	0.0071	0.9517	0.9803	0.4415	0.0815	0.2817	0.6013

[a] DSR is the interval survival 7-days for this study, [b] SE = standard error , [c] LCI = lower 95% confidence interval, [d] UCI = upper 95% confidence interval

(DSR) for bobwhite nests and evaluated competing models explaining variation in nest survival using a general linear mixed model approach (Dinsmore et al. 2002, Stephens 2003, Rotella et al. 2004). We fit models using PROC NLMIXED in SAS because it provided the framework needed to model our binomially distributed data (nest fate = 0 if failed and 1 if successful) and provided a user defined link option (i.e., logit link) while concurrently considering the affects of habitat (PP [planted pines], LH [linear habitats], and other) and gender covariates and, the random effect of year on nest survival (PROC NLMIXED; Institute 1999). We considered year a random effect because we assumed that year was a random level sample and to avoid confounding fixed effects of other variables of interest (e.g., LH, PP).

We used an information-theoretic approach (Burnham and Anderson 2002, Anderson et al. 2000) to evaluate the set of candidate models. The models were developed *a priori* based on biological insight to avoid superfluous model building (i.e., data dredging). The best approximating model in the set of candidate models was determined by Akaike's Information Criteria (AIC); adjusted for small sample bias (AIC$_c$; Burnham and Anderson 2002). AIC$_c$ is a valid model selection method for both nested and non-nested sets of models (Burnham and Anderson 2002). AIC$_c$ was used to compare each candidate model, and the model with the lowest AIC$_c$ value was considered to be the best approximating model given the data. The relative plausibility of each model in the set of candidate models was assessed by Akaike weights (w_i, Burnham and Anderson 2002, Anderson et al. 2000), where the best approximating model in the candidate set has the greatest Akaike weight (Burnham and Anderson 2002, pg. 447). We used model averaging (Burnham and Anderson 2002, pg. 448) to calculate model averaged coefficients (LH, gender); and we report these coefficients, their standard errors and 95% confidence intervals, and odds ratios.

Habitat Use and Selection - We examined habitat use for bobwhite coveys (1 Oct - 31 Mar) and broods (breeding season) with 2nd and 3rd order habitat selection (Johnson 1980) for individual cov-

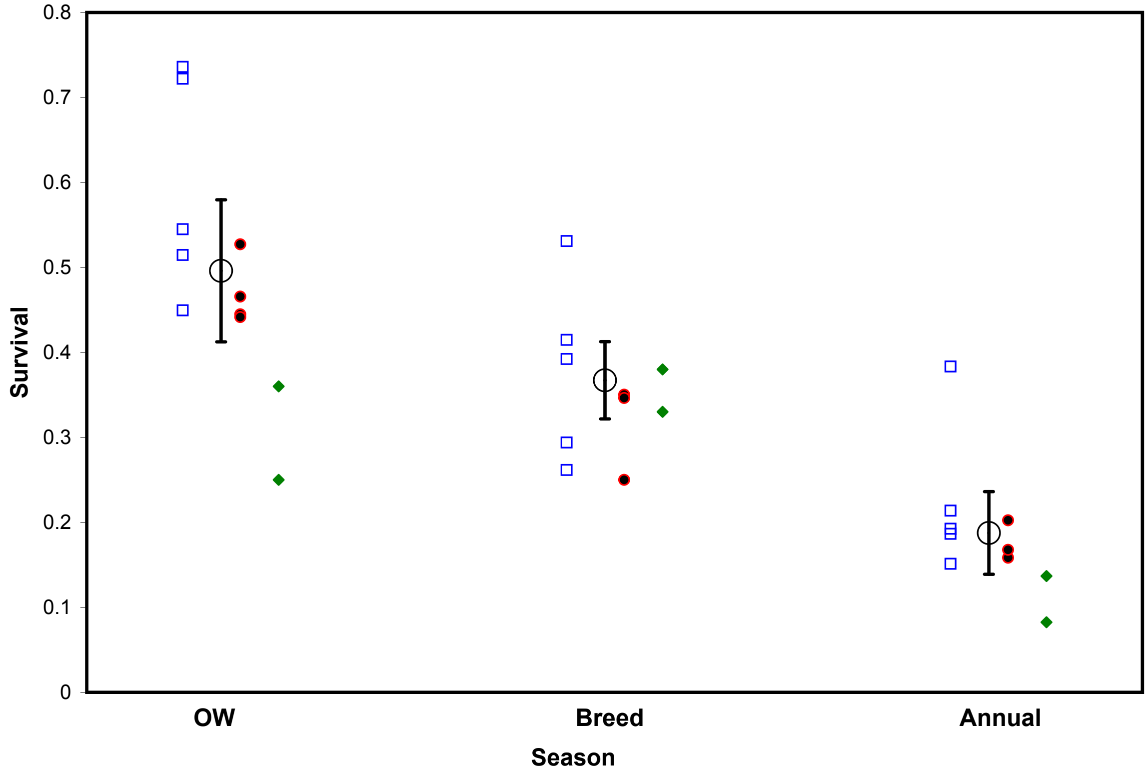

Figure 1: Predicted probability of surviving (mean survival and 95% confidence intervals) during overwinter (OW), breeding (Breed) and annual time-periods obtained via Program MARK (black circles with red outline data points), for our data compared to long-term plantation survival estimates (hollow square-shaped points) and an unmanaged agriculture site (green diamond-shaped points) in southwestern Georgia derived via Kaplan-Meier during 1998 - 2002.

eys and broods, respectively, using compositional analysis (CA; Aebischer et al. 1993, Manley et al. 2000). We defined second order availability for individual coveys and broods (only when $n > 3$ radio-tagged bobwhites/covey). The average habitat proportions within these polygons was calculated and considered to be second order availability. Second-order use was defined as the proportions of each habitat type within home ranges. We defined 3rd order availability as the proportion of each habitat type within home ranges and habitat use as the proportion of individual radio-locations within each habitat type. Prior to analysis, we replaced zero values for use with the value 0.001-an order of magnitude less than the smallest nonzero value (Aebischer et al. 1993). When a habitat was not available for use, we replaced missing values in each log-ratio with the mean of all non-missing values for the respective log-ratio (Aebischer et al. 1993). All habitat selection analyses were conducted using Compos Analysis (version 6.2; Smith 2005). We used a multivariate analysis of variance (MANOVA) test to examine habitat selection (Aebischer et al. 1993). Habitats were ranked using a matrix that indicated the difference of log ratios between habitat types, and log ratio differences were determined with paired t-tests (Aebischer et al. 1993).

We used GIS to assess metrics of habitat composition and configuration using the Animal Movements Extension (AME; Hooge and Eichenlaub 1997) to calculate fixed kernel winter home ranges (Worton 1989) using a 95% isopleth. Kenward (2001, pg. 231)

Table 3: Cause-specific mortality for known-fate radio-tagged northern bobwhites (n = 253) on Whitehall Plantation in Laurens and Bleckley counties, Georgia, 1999 - 2002.

	Causes of Mortality				
	Mammal	Avian	Snake	Harvest	Total
1998-1999	14	47	1	2	64
1999-2000	12	67	0	2	81
2000-2001	13	63	3	2	81
2001-2002	3	22	0	2	27
Pooled	42	199	4	8	253
Percent	16.6	78.66	1.58	3.16	100

indicated that ~20 locations was needed for home range size stability when using the kernel method; thus, coveys and broods with $\leq$20 locations were excluded from analysis. We also excluded mortality locations from analysis since predators may have transported birds away from the original kill site.

Results

Survival

We monitored 498 bobwhites (n_{female} = 279, n_{male} = 219) during the 3.5-year study. The most parsimonious known-fates model for our data included time-dependency as a constant-seasonal effect (Table 1) with a model weight of 0.47. The model averaged coefficient for breeding season (1 May - 30 Sep) was -0.303 (SE = 0.151). This indicated that breeding season had a negative effect (i.e., survival was lower than over-winter season) on survival for our data. Annual variation in survival was not evident (w = 0.0096; Table 1) for our data. The additive effect of gender to the best model did warrant some consideration (w = 0.2070; Table 1). Whereas the model averaged coefficient estimate for gender (i.e., female) effect was 0.124 (SE = 0.175) indicating that females survived better than males, but the confidence limits for the effect of gender included 0. Further, the model including a season and gender interaction had relatively little support (ΔQAIC$_c$ = 3.09, w = 0.1002) indicating that variation in survival relative to gender was not dependent on season (i.e., breeding or over-winter) for our data.

Generally, over-winter (OW) seasonal survival was higher than breeding (Breed) season survival (Table 2). Breeding season had a negative effect on survival (β = -0.318; 95% CI: -0.614, -0.023). The average OW and Breed season survival was 0.4698 (SE = 0.0721) and 0.3561 (SE = 0.0667), respectively. OW survival was lower than long-term bobwhite estimates from plantations but higher than unmanaged agriculture sites in southwestern Georgia (Figure 1). Breeding season and annual survival was similar among sites (Figure 1). Avian species were the most prevalent agent of mortality accounting for 78.66% of the known-fate bobwhites (Table 3). Mammals accounted for 16.60% of bobwhite mortalities, whereas snakes and harvest combined for <5%.

Reproduction

We monitored 165 nests (n_{1999} = 45, n_{2000} = 72, n_{2001} = 48) during the 3.5-year study. The constant-among years-model was the best supported model for our data (Table 4); however, models including individual-additive fixed effects of gender, PPs, LHs and a random year-effects model warranted consideration (ΔAIC$_c$ $\leq$2, w > 0.1000). The model

Table 4: Mixed models (i.e., fixed and random effects models) explaining nest survival treating year as fixed and random effects; and, linear habitats (LH) and gender as fixed effects for northern bobwhites located on Whitehall Plantation in Laurens and Bleckley County, Georgia, 1999 - 2002.

Model	K	AIC	AIC_c	$\Delta QAIC_c$	W_i
B_0	1	604.8676	604.8695	0.0000	0.3013
$B_0 + B_1{}^*(PP)$	2	606.7752	606.7808	1.9113	0.1159
$B_0 + B_1{}^*(Gender)$	2	606.8171	606.8226	1.9531	0.1135
$B_0 + B_1{}^*(LH)$	2	606.8453	606.8508	1.9813	0.1119
$B_0 + u$	2	606.8676	606.8732	2.0037	0.1106
$B_0 + B_1{}^*(Year1) + B_2{}^*(Year2)$	3	607.9712	607.9823	3.1128	0.0635
$B_0 + B_1{}^*(PP) + B_2{}^*(LH)$	3	608.7739	608.7849	3.9154	0.0425
$B_0 + u + B_1{}^*(PP)$	3	608.7752	608.7863	3.9168	0.0425
$B_0 + u + B_1{}^*(Gender)$	3	608.8171	608.8281	3.9587	0.0416
$B_0 + u + B_1{}^*(LH)$	3	608.8453	608.8563	3.9869	0.0410
$B_0 + u + B_1{}^*(PP) + B_2^*(LH)$	4	610.7739	610.7923	5.9228	0.0156

considering year as a fixed effect had relatively little support (ΔAIC$_c$ = 3.11, w =0.0731) compared to other top-ranked models. Therefore, we primarily fit models treating years as random effects so as not to confound with other fixed effect parameters.

We used model averaging to interpret coefficients for nest survival and individual covariates (Table 5). The odds ratios for gender, PP and LH were 1.07, 1.07 and 0.95 (Table 5), respectively, indicating that females and nests located in PPs were 7% more likely to be successful than nests incubated by males and found in other habitats, respectively; however, the confidence interval for these log ratios included 1. Daily nest survival (DSR) for 1999, 2000, and 2001 was 0.9727 (SE = 0.0065), 0.9645 (SE = 0.0055), and 0.9705 (SE = 0.0065), respectively. The average DSR for years pooled was 0.9687 (SE = 0.0037). Nest survival for years pooled delineated by gender and habitat type was higher for females and nests located in PPs, although these differences were not significant (Figure 2).

Habitat Use

Coveys - We combined all coveys (n = 67) during the 3.5-year study for habitat selection analysis; we determined that habitat selection did not differ between years (F$_{2,66}$ = 1.58, P = 0.214). Covey habitat selection departed from random at both the second-order (λ = 0.6467, χ^2_3 = 29.206, $P < 0.001$) and third-order (λ = 0.2644, χ^2_3 = 89.139, $P < 0.001$) levels. For our data, coveys preferred PP types over hardwoods and miscellaneous types (Table 6; 2nd order: t$_{66}$ = 3.515, $P < 0.001$; 3rd order: t$_{66}$ = 5.870, $P <$ 0.001) and AG/FAL habitat types (Table 6; 2nd order: t$_{66}$ = 3.628, $P < 0.001$; 3rd order: t$_{66}$ = 9.580, P < 0.001). LH habitat types was preferred to hardwoods and miscellaneous habitats, but the difference was not significant at the 3rd order level (Table 6; 2nd order: t$_{66}$ = 2.921, P = 0.005; 3rd order: t$_{66}$ = 0.220, P = 0.827), and LH was preferred to AG/FAL habitat (Table 6; 2nd order: t$_{66}$ = 4.247, $P < 0.001$; 3rd order: t$_{66}$ = 3.181, P = 0.002). In order of preference at the second order level bobwhite coveys preferred: planted pines, linear habitats, managed woodlands, hardwoods and thinned hardwoods, agricultural and fallow land. And at the

Table 5: Model averaged, estimated coefficients and associated precision for parameters used to model variation in nest survival for northern bobwhites located on Whitehall Plantation in Laurens and Bleckley County, Georgia, 1999 - 2002.

			Confidence Interval		
Parameter	Estimate	SE	Lower 95%	Upper 95%	Odds Ratio
Gender (female)	0.0710	0.3136	-0.5437	0.6856	1.0736
Planted Pine (PP)	0.0764	0.2518	-0.1755	0.3282	1.0793
Linear Habitat (LH)	-0.0486	0.3234	-0.3720	0.2748	0.9525

third order level bobwhite coveys preferred: planted pines, managed woodlands, linear habitats, hardwoods and thinned hardwoods, agricultural and fallow land.

Broods - We combined all broods ($n = 73$) to examine habitat selection and preference for the 3.5 year study. Brood habitat selection was not random at both the second-order ($\lambda = 0.2631$, $\chi^2_3 = 97.470$, $P < 0.001$) and third-order ($\lambda = 0.2632$, $\chi^2_3 = 97.441$, $P < 0.001$) levels. Broods preferred LHs over all other habitat types at the second-order level and all other habitat types except PPs at the third-order levels (Table 7). At the second-order level: LHs were preferred to agriculture and fallow habitats although the difference was not significant ($t_{72} = 1.034$, $P = 0.302$), PPs ($t_{72} = 3.051$, $P = 0.003$), hardwoods and other habitats ($t_{72} = 12.906$, $P < 0.001$), and managed woodlands ($t_{72} = 2.867$, $P = 0.005$); PPs was preferred to hardwoods and other habitats ($t_{72} = 7.859$, $P < 0.001$), and managed woodlands ($t_{72} = 2.867$, $P = 0.005$); agriculture and fallow lands was preferred over hardwoods ($t_{72} = 10.132$, $P < 0.001$), managed woodlands ($t_{72} = 2.034$, $P = 0.046$), and planted pines (PPs) although the difference was not significant ($t_{72} = 1.785$, $P = 0.085$); and managed woods was preferred over hardwoods ($t_{72} = 6.583$, $P < 0.001$). At the third-order level: PPs was preferred to agriculture/fallow land ($t_{44} = 4.672$, $P < 0.001$), hardwoods and other habitats ($t_{10} = 7.709$, $P < 0.001$), managed woodlands ($t_{34} = 2.876$, $P = 0.007$), and LHs, although the difference was not significant ($t_{46} = 0.371$, $P = 0.713$); and LHs was preferred to agriculture habitats ($t_{54} = 4.688$, $P < 0.001$), hardwoods and other late-succession habitats ($t_{12} = 2.154$, $P < 0.050$), and managed woodlands although the difference was not significant ($t_{72} = 1.303$, $P = 0.200$).

Population Response

We used covey call counts via the point-count method (Wellendorf et al. 2004) to determine bobwhite abundance. Using a replicated design and 14 individual, fixed points we estimated the initial bobwhite abundance at 0.86 birds/ha ($\sim$45 coveys). During fall 2001, we estimated a final bobwhite abundance of 1.48 birds/ha. Thus, we observed an estimated 75% increase in bobwhite abundance during the 3.5-year study.

Discussion

Survival

Bobwhite survival has been documented to vary both temporally and spatially (Burger et al. 1995*a*, 1998, Curtis et al. 1988, Sisson et al. 2009, Taylor et al. 2000, Terhune et al. 2007) and relative to gender (Pollock et al. 1989). During this study, variation in survival was best explained by models including seasonal effects. There was a negative effect of breeding season on bobwhite survival; bobwhites were 1.37 times (37%) less likely to survive during breeding

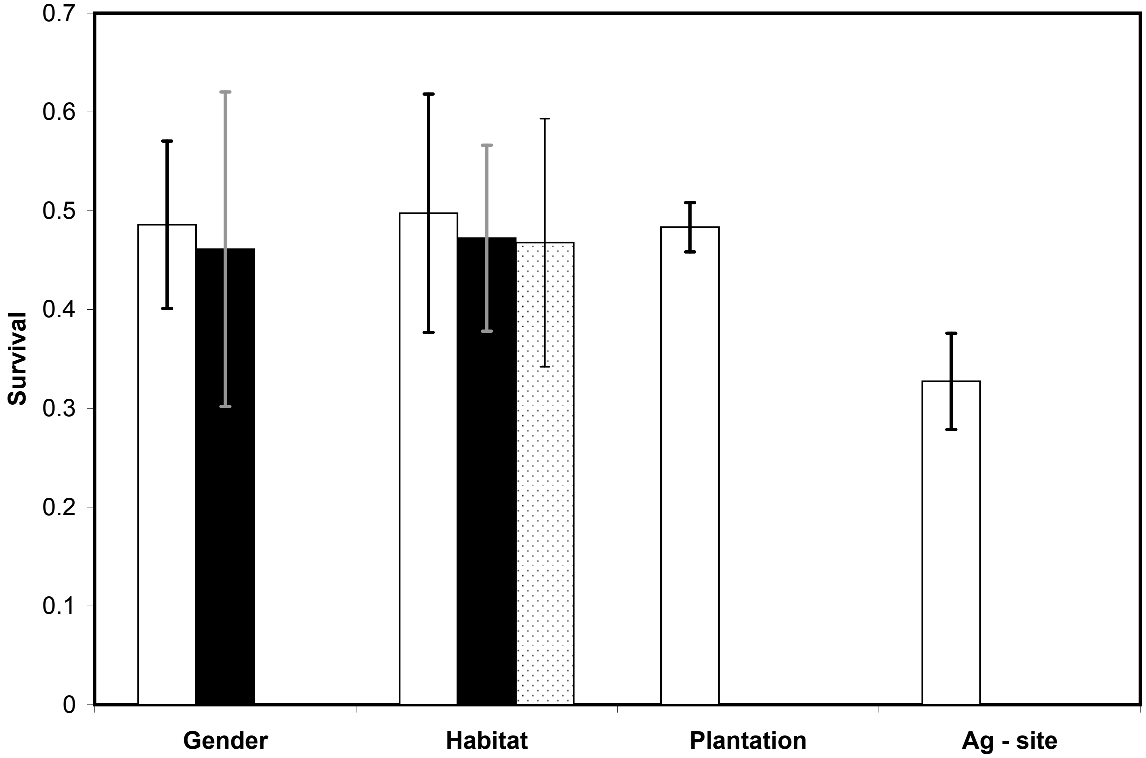

Figure 2: Predicted probability of survival (mean survival and 95% confidence intervals) of northern bobwhite nests as estimated via model averaging for gender (male [solid black bars], female [solid white bars]) and habitat (LH [black-speckled bars], other habitats [solid black bars], and PPs [solid white bars]) on Whitehall Plantation in Laurens and Bleckley County, Georgia, compared to nest survival estimates for an intensively managed plantation and an unmanaged agriculture site during 1999 - 2002.

season than OW season. Whereas the most parsimonious model including gender did warrant some consideration, the model including the interaction of gender and season was not adequately supported (Δ QAIC$_c$ = 3.09). This suggested that survival relative to gender was not dependent on season. Our survival estimates were similar to those of other reported studies (Curtis et al. 1988, Burger et al. 1998, Sisson et al. 2009, Terhune et al. 2007). Breeding season and annual survival during our study was, in general, similar to intensively managed plantation sites and an unmanaged agriculture site (Figure 1); however, OW survival was markedly disparate between sites. Interestingly, survival during OW declined relative to management strategy whereby intensively managed plantation sites, managed agriculture sites (i.e. our study site), and unmanaged agriculture sites incurred the highest, near average, and lowest survival, respectively. Similarly, Cook (2004) demonstrated that bobwhites exhibited higher survival on areas with linear habitats compared to those void of them. As such, the utility for augmenting habitat among agricultural landscapes via novel habitats (i.e. PPs and LHs) to improve survival, at least compared to unmanaged agricultural landscapes, was substantiated by our critique.

Fies et al. (2002) suggested bobwhite dispersal and movement proclivities are potentially greater among fragmented landscapes and Kabat and Thompson (1963) reported increased movements of bobwhites when landscapes typically consisted of marginal habitat. Incidentally, Cook (2004) reported that bobwhites on sites with linear habitats exhibited lower dispersal rates than sites without these

Table 6: Simplified ranking matrices for northern bobwhite coveys based on second- and third-order compositional analysis on Whitehall Plantation located in Laurens and Bleckley counties, Georgia 1999-2002.

	Ag / Fall[a]	PP[b]	LH[c]	HW / Other[d]	MW[e]	Rank
2^{nd} Order habitat selection (n = 67)[f]						
Ag / Fall		—	—	-	—	0
PP	+++		+	+++	+	4
LH	+++	-	+	+++		3
HW / Other	+	—	—		—	1
MW	+++	-		+++	-	2
3^{rd} Order habitat selection (n = 67)[g]						
Ag / Fall		—	—	-	—	0
PP	+++		+++	+++	+	4
LH	+	—	+		—	2
HW / Other	+++	—		-	—	1
MW	+++	-	+++	+++		3

[a] Ag / Fall denotes habitat types including agriculture and fallow fields, [b] PP represents planted pines (typically longleaf), [c] LH represents linear habitats: field buffers and borders, hedgerows, linear longleaf pines, and terraces, [d] HW denotes habitat types including hardwoods and other habitats not typically associated with early succession vegetation (e.g., drains), [e] MW represents managed woods: upland pines, early succession vegetation areas other than fallow fields, burned and unburned habitats, and thinned and managed mixed hardwood pine stand, [f] 2nd order analysis was based on comparing the proportional habitat use within home ranges with the proportion of total available habitat types (i.e., study area vs. home range); a triple sign indicates a significant deviation from random at the alpha level of 0.05 and positive and negative signs indicates habitat preference and avoidance, respectively, [g] 3rd order analysis was based on comparing the proportional habitat use within home ranges with the proportion of telemetry locations located within each habitat type within each bird's home range (i.e., home range vs. locations).

habitats. Additionally, Sisson et al. (2000, 2002) demonstrated that survival and home range size was dependent on resource quality and availability. Hughes et al. (2005) reported evidence to support these notions: they suggested that due to lack of resources (e.g. habitat and food availability) bobwhites were forced to utilize lower quality habitats (e.g. creek swamps, hardwoods) and traverse unsuitable habitat(s) to get to suitable habitat and/or food; however during years of abundant food resources they determined that home range size and survival improved dramatically. Although not reported herein, bobwhites on our study site generally retained high site fidelity; home range size was only marginally larger than those on intensively managed plantation sites and smaller than those on unmanaged agriculture sites (S. Mitchell, Alabama Quail Project, unpublished report), indicating that resource availability was likely not a limiting factor during our study - although supplemental feeding did occur on our study site and thus home range size may have been low from this highly available resource (Sisson et al. 2000). Collectively, novel habitats (e.g. PPs and LHs) may decrease home range size, improve survival, and reduce dispersal rates among fragmented and/or agricultural landscapes.

Covey Habitat Use - Bobwhite coveys preferred PP habitats, managed woodlands, and LHs to all other available habitats (Table 6). These findings were not contrary to what we expected because during OW months (1 Oct - 31 Mar) a significant portion of the agriculture area was disked under and was thus bare soil. The preference of PP and LH (3rd Order selection) habitats over managed woodlands

Table 7: Simplified ranking matrices for northern bobwhite broods based on second- and third-order compositional analysis on Whitehall Plantation located in Laurens and Bleckley counties, Georgia 1999-2002.

	Ag / Fall[a]	PP[b]	LH[c]	HW / Other[d]	MW[e]	Rank
2^{nd} Order habitat selection (n = 73)[f]						
Ag / Fall		+	-	+++	+++	3
PP	-		—	+++	+	2
LH	+	+++		+++	+++	4
HW / Other	—	—	—		—	0
MW	—	-	—	+++		1
3^{rd} Order habitat selection (n = 73)[g]						
Ag / Fall		—	—	+	-	2
PP	+++		+	+++	+++	4
LH	+++	-		+	+	3
HW / Other	-	—	-		+	1
MW	+	—	-	-		1

[a] Ag / Fall denotes habitat types including agriculture and fallow fields, [b] PP represents planted pines (typically longleaf), [c] LH represents linear habitats: field buffers and borders, hedgerows, linear longleaf pines, and terraces, [d] HW denotes habitat types including hardwoods and other habitats not typically associated with early succession vegetation (e.g., drains), [e] MW represents managed woods: upland pines, early succession vegetation areas other than fallow fields, burned and unburned habitats, and thinned and managed mixed hardwood pine stand., [f] 2nd order analysis was based on comparing the proportional habitat use within home ranges with the proportion of total available habitat types (i.e., study area vs. home range); a triple sign indicates a significant deviation from random at the alpha level of 0.05 and positive and negative signs indicates habitat preference and avoidance, respectively, [g] 3rd order analysis was based on comparing the proportional habitat use within home ranges with the proportion of telemetry locations located within each habitat type within each bird's home range (i.e., home range vs. locations).

was likely a result of the timing of the study. The managed woodlands were heavily disturbed when they were logged and cleaned up therefore producing mostly weeds during the first couple of years. This made good summer habitat but had not yet developed into good winter cover. We speculated that bobwhites utilized PP habitats at a higher than expected rate because of the woody vegetation component provided via the longleaf pines and the fact that groundcover was more fully developed, thereby improving the quality of "escape" cover for coveys.

Among agriculture landscapes a paucity of suitable bobwhite habitat exists throughout the year. This was evident by the extremely low OW survival and large home range sizes observed on agricultural sites without PPs and LHs or newly created early-succession habitats (Hughes et al. 2005) when compared to OW survival for our site where these habitats were available during the entire study. Furthermore, breeding season survival did not vary among sites, irrespective of management strategy, indicating that PP habitat was more critical during OW seasons, a time when habitat is likely a limiting factor among agricultural landscapes. Because bobwhites are considered an r-selected species (demonstrated by high annual mortality and high reproductive output), OW survival has been recognized as a vital demographic parameter for increasing bobwhite populations (Burger et al. 1998, Sisson et al. 2009). Under this tenet, by increasing OW survival, whether via habitat management or other means, the number of bobwhites available to reproduce is potentially augmented; and thus, improving reproductive output and subsequently increasing bobwhite abundance.

Therefore, the utility of creating novel habitats (PPs and LHs) and improving existing habitat as demonstrated in this study, under this tenet alone, may immensely improve bobwhite abundance among agricultural landscapes and other OW-habitat deficient sites.

Cause-specific Mortality - Despite numerous studies reporting agents responsible for mortalities of bobwhites, cause-specific mortality remains an enigma and is one of high observer subjectivity. Therefore, in this study, we only report assessed causes of mortality and compare our results to those studies conducted by the AQP (where protocols for ascribing causes of mortality were similar) to limit observer variability.

During this study, avian depredation was the leading cause of mortality for all years accounting for nearly 80% of all known-fate mortalities. Mammals accounted for approximately 17% and snake and harvest combined for <5%. Our results were generally similar to those reported by Sisson et al. (2009); however, avian mortality was elevated for our study site compared to their long-term results. Surprisingly, differences in causes of mortality did not vary relative to season for our data; avian species remained the leading mortality agent during both breeding and OW season. Notably, other sites exhibited variation in causes of mortality relative to season, whereas during breeding season mammals typically became a more salient cause of mortality than during OW seasons (Sisson et al. 2009). For example, Sisson et al. (2009) reported that one site in east-central Georgia experienced high OW-avian mortality (>71%) and high breeding-season mortality caused by mammals (>61%). Particularly noteworthy was the unmanaged agriculture site: avian mortality during breeding season (>61%) was much higher than mammalian mortality (<34%). Perhaps avian mortality, while ostensibly dependent on the timing and duration of raptor migration, is higher on agriculture sites whether or not PP and LH habitats are present. In such cases, habitat composition and juxtaposition may play a key role in the efficiency of avian predators to locate (and depredate) bobwhites among these types of landscapes. Thus, more research is warranted to determine whether wider linear habitats may mitigate avian mortalities and/or whether other proximate habitats (e.g. hardwoods) decrease the utility of novel habitats on specific sites.

Reproduction

Daily survival rates for northern bobwhite nests in our study did not vary among years for our data (Table 4). Since we were interested in nest survival among PPs and LHs compared to other habitat types, we treated year as a random effect to evaluate habitat type and gender effects on nest survival. The most parsimonious model was a constant survival model with no covariate effects. Our nest survival estimates were higher than those reported for other nest studies (Burger et al. 1995*b*, Puckett et al. 1995, Hughes et al. 2005), and similar to long-term nest survival estimates for intensively managed plantations (Figure 2). However, mammalian nest predator management did occur on these study sites.

Hughes et al. (2005) surmised that lack of resource availability, particularly habitat availability, was a limiting factor during their study, and suggested that habitat development - such as field borders, field buffers, and hedgerows as well as other habitat practices (e.g., no-till farming) - would benefit bobwhite nest survival and production. Previous researchers have indicated that nest predators may more efficiently forage in landscapes comprised of small and/or narrow habitats (Puckett et al. 1995, 2000). Additionally, Puckett et al. (1995) reported low nest success for nests located in filter strips, particularly during the early nesting season. We did not, however, find evidence suggesting that linear habitats negatively impacted nest survival (Figure 2). Notably, when compared to our study, the effective land area and width of filter strips was different for the study conducted by Puckett et al. (1995, 2000), mean filter strip width was 9.2 m and the effective land area comprised of filter strips was <10 percent. Further, filter strips were designed to reduce soil erosion and thus oftentimes were located along ditches (Puckett et al. 1995) - a habitat conducive to certain predators (e.g., snakes). In contrast, during our study, the effective land area created from PP (21%) and LH (12%) habitat development was >30% and the mean LH width was 15 m. Therefore, implementation of wider LHs and increased effective land area may improve nest survival, increasing the amount of habitat for nest predators to rummage. As such, recent implementation has demonstrated that when the effective land area was increased and wider LHs were constructed, bobwhite demographics and population levels increased among agricultural landscapes (D. C. Sisson, Albany Quail Project, unpublished report).

Among PP habitats, longleaf pines provided woody substrate and pine needles for nest building; nearly 42% of all nests during this 3.5-year study were located in PP habitats (S. Mitchell, Alabama Quail Project, personal communication) and nest survival among these habitat types was highest during our study. When combined with the nests located in LHs nearly 64% of all nests were constructed and incubated in these newly created habitat types. Thus, the development of these habitat types minimally improved the quantity of habitat available during nest season and, seemingly, did not render bobwhite nests more susceptible to predation.

Brood Habitat Use - Bobwhite hens preferred to raise broods in LHs, PPs and fallow areas compared to other habitat types, and they used PPs and LHs more than agriculture sites (Table 7). The higher use of LHs and PPs compared to agriculture cropland may have been attributed to later cover availability via crops in those areas combined with pesticide use (and low arthropod availability). Our results, for brood habitat use, were similar to those reported for other studies (Puckett et al. 1995, 2000, Cook 2004).

Cook (2004) and Puckett et al. (2000) reported that hens raising broods used LHs more than agriculture fields and other habitat types. Puckett et al. (2000) also reported that bobwhite chick survival was high among LHs and brood home range sizes were small. Although not reported herein, we observed that, in general, bobwhite brood home-range size was similar to those of intensively managed plantation broods (S. Mitchell, Alabama Quail Project, personal communication). Therefore, we surmised that given the preference for LHs and PPs, and similar home range size of broods during our study when compared to other intensively managed sites that these habitat types may facilitate reduced home range size for broods in agriculture landscapes. Thus, provided herbicides and pesticides are not exploited in these habitats, LHs and PPs may render the much needed niche for bobwhite broods in agriculture ecosystems whereby weedy vegetation and arthropods are prevalent and year-round habitat is made available.

Summary

In this study, we reported data that advocated novel habitat (e.g. PPs and LHs) establishment as a practical utility to promote improved demographics when compared to intensively managed plantation sites and unmanaged agriculture sites: survival was generally similar to managed sites and higher than unmanaged sites; reproduction was similar to managed sites and higher than unmanaged sites; habitat use by broods and coveys was high among novel habitats and, while anecdotal, broods and coveys benefited from PPs and LHs by reducing their home range size and providing the much needed resource availability during germane times (i.e. OW months). And, these novel habitats effectively rendered nesting habitat during breeding season and nest survival was similar among these habitat types compared to other habitat types. The combined effect of novel habitat establishment and improvement of existing habitat provided a substantial overhaul to the landscape on this study site. We effectively managed >60% of the landscape to benefit bobwhites, while maintaining farm practices on much of the remainder. As a result, bobwhites utilized novel and improved habitats heavily for covey home ranges, nest sites, and brood habitat which resulted in increased bobwhite abundance and a renewed optimism for managing bobwhites outside the traditional "plantation belt" located in southwest Georgia and north Florida. Bobwhite abundance increased each year following habitat modification whereby point counts conducted during the fall (Oct-Nov; Wellendorf et al. 2004) indicated an increase by >75 percent. Bobwhite abundance increased from <0.86 birds per hectare to >1.48 birds per hectare during the 3.5-year study.

Management Implications

Researchers and biologists have demonstrated that the most effective mode to restore bobwhite populations, both at local and regional scales, is to increase habitat availability (Klimstra 1972, Brennan 1991). Likewise, long-term research corroborates this notion where, despite the declining status of bobwhites throughout most of their range during the past decade, intensively managed bobwhite plantations have experienced stable-to-increasing bobwhite abundance (Brennan 1991, Stribling and Sisson 2009). The results from this study also illustrated the importance of habitat management to benefit bobwhites. Thus, for bobwhites to persist among agricultural landscapes, restoration (i.e., Farm Bill - CRP) programs should continue to focus on habitat management. Whereas implementation of novel habitats is by no means a panacea for reversing population declines, they may serve as pragmatic utility for at least improving bobwhite habitat among agricultural landscapes and perhaps extenuate bobwhite population declines among these ecosystems. Further, several other techniques (i.e. conservative bobwhite harvest, nest predator management, supplemental feeding) when used in conjunction with establishing novel habitats among agriculture ecosystems, may also increase restoration success.

When establishing linear habitats among agricultural landscapes, we recommend setting the target of land area affected at a minimum of 10-15% and linear habitat widths $\geq$15 m (and when applicable wider). We also recommend employing other habitat management techniques in conjunction with linear habitat establishment when appropriate: managing dry corners for early-succession vegetation, timber density reduction on adjacent sites, mid- and over-story hardwood reduction, prescribed burning, supplemental feeding and nest predator management. When planting pines, we recommend planting longleaf pines at a conservative (8X8 or greater; 600 trees/acre or less) spacing, and utilizing prescribed fire and limb pruning as needed to benefit early-succession vegetation over time. Proper management of woodlands surrounding agricultural areas should also greatly improve habitat conditions among agricultural ecosystems - this type of management was a large part of the success observed during our study. Additionally, we encourage federal and state programs implementing early-succession habitat establishment to concentrate efforts to specific-focal areas, particularly areas located near existing bobwhite populations, and expand outward in order to maximize restoration efforts at both the local and regional scale. Lastly, we recommend continued research and monitoring of bobwhite populations among these types of ecosystems to continue gleaning insight about the utility of novel habitats and learn how these ecosystems function, as a whole, both locally and regionally.

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U.S. Fish & Wildlife Service

Noonday globe snail

Petera clarkia Nantahala

Noonday globe snail, John Firdell

Status: Threatened

Description: The noonday globe snail is a moderately sized (3/4 inch wide and 1/2 inch high) land snail. Its shell is shiny and reddish in color. Their surface of the shell is sculptured with rather course lines. The area around the shell opening (aperture) is white, and a long curved "tooth" is located on the inside portion of the aperture. The animal's body is black.

Because this snail is so rare and restricted in distribution, very little is known of its biology. The species' reproductive behavior is unknown, and its food habits are also a mystery. However, other related species in the genus *Petera* feed on the subsurface hair-like structure (mycelia) of fungi. The species appears to be most active during wet weather, when it's frequently found out on the surface of vegetation rather than under the leaf litter on the forest floor.

Habitat: The snail is found in the Nantahala Gorge, on wet cliffs that are intersected by many small streams and waterfalls. The forest is mature, with many large trees and a diverse plant community. The forest floor has a thick, rich humus layer, and the area has many exposed calcareous (rich in calcium) rocks. Calcium, which is generally scarce in other cliffs in the area, is vital to snails because it is a major component of their shells.

Range: The noonday globe snail is known from only about two miles of high cliffs within the Nantahala Gorge in Western North Carolina.

Listing: Threatened, July 3, 1978. 43 FR 28932 28935

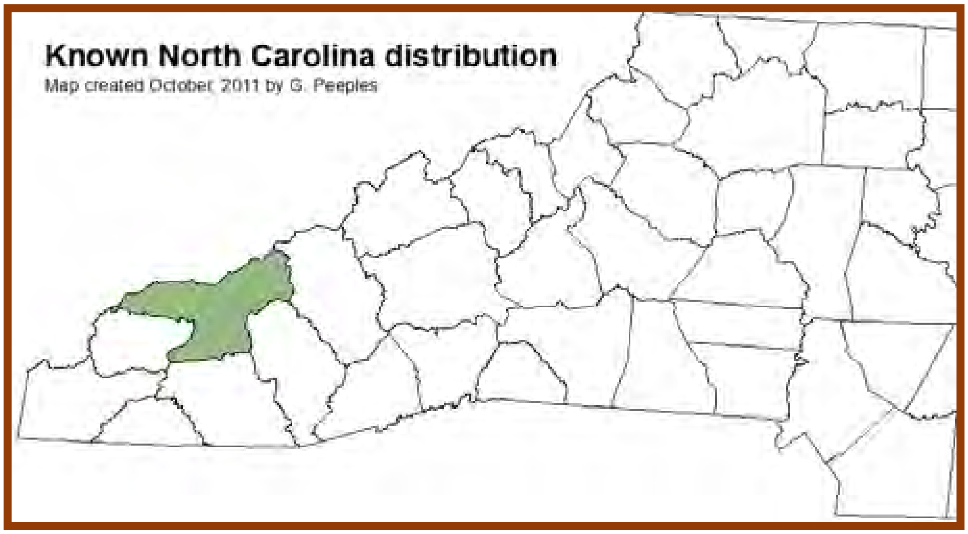
Critical habitat: None designated

Threats: The noonday globe was likely never widely distributed. Steep wet slopes with calcareous rocks are rare in Western North Carolina. However the species was likely somewhat more widely distributed within the gorge before the gorge was altered for a railroad and highway. The associated loss of the forest canopy allowed more sunlight to penetrate the gorge and likely dried the lower slope of the gorge. This habitat alteration also allowed such non-native plants as kudzu and Japanese honeysuckle to invade some roadside areas, changing the area's natural plant and animal community.

Why should we be concerned about the loss of species? Extinction is a natural process that has been occurring since long before the appearance of humans. Normally, new species develop through a process known as speciation, at about the same rate other species become extinct. However, because of air and water pollution, forest clearing, loss of wetlands, and other man-induced environmental changes, extinctions are now occurring at a rate that far exceeds the speciation rate.

All living things are part of a complex and interconnected network. We depend on the diversity of plant and animal life for our recreation, nourishment, many of our lifesaving medicines, and the ecological functions they provide. One-quarter of all the prescriptions written in the United States today contain chemicals that were originally discovered in plants and animals. Industry and agriculture are increasingly making use of wild plants, seeking out the remaining wild strain of many common crops, such as wheat and corn, to produce new hybrids that are more resistant to disease, pests, and marginal climatic conditions. Our food crops depend on insects and other animals for pollination.

Healthy forests clean the air and provide oxygen for us to breathe. Wetlands clean water and help minimize the impacts of floods. These services are the foundation of life and depend on a diversity of plants and animals working in concert. Each time



Known North Carolina distribution

Map created October 2011 by G. Peeples

a species disappears, we lose not only those benefits we know it provided but other benefits that we have yet to realize.

What you can do to help

Tread lightly and stay on designated trails.

Visit arboretums, botanical gardens, and parks and learn all you can about endangered species and the causes of their declines.
Participate in the protection of our remaining wild lands and the restoration of damaged ecosystems.

Prepared by:
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160 Zillicoa Street
Asheville, North Carolina 28801
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December, 2011

Criterion 5:

MAINTENANCE OF FOREST CONTRIBUTIONS TO GLOBAL CARBON CYCLES

Montréal Process Criterion 5 (Montréal Process Working Group 2010); Northern Area Forest Sustainability Indicators 11.1, 11.2, 11.3, 11.4 (USDA FS 2010d)

The importance of forest contributions to global carbon cycles

Northern forests cover more than 42 percent of the region and are enormous reservoirs of carbon. Through photosynthesis, live trees emit oxygen in exchange for carbon dioxide they pull from the atmosphere. As a tree grows it stores carbon in wood above and below ground, and sequestered carbon comprises about half of its dry weight. Dead trees and down logs are also reservoirs of carbon. Forest soils sequester additional carbon in the form of incorporated organic matter. In temperate northern forest ecosystems, roughly as much carbon is sequestered in forest soils as is sequestered as live biomass. Forests that are converted to other land uses release the carbon stored in the trees. Trees growing in newly established forests (afforestation) can sequester additional carbon.

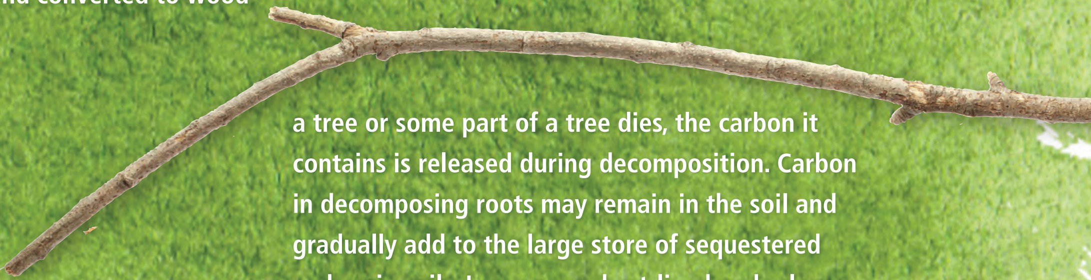
People and forests are closely linked through the carbon cycle. Human activities emit huge amounts of carbon dioxide during energy production, transportation, and other activities. Increases in atmospheric carbon dioxide have been linked to global warming. Because of their great extent and their capacity to sequester additional carbon or release carbon that is already sequestered, forests have an important role as sinks or sources of carbon in regional and global carbon cycles.

Global climate change associated with changes in atmospheric carbon dioxide levels could significantly impact the future conditions of forests, which would in turn affect the plants, wildlife, and people that depend on them. Maintenance of forest biodiversity and health are associated concerns.

Some forest management activities can increase carbon sequestration or offset human activities that emit carbon. Silvicultural practices that increase forest growth can increase the quantity of carbon sequestered in woody biomass. Wood product utilization can increase the quantity of carbon sequestered in durable wood products. Wood-based energy production can offset carbon that would otherwise be released by burning fossil fuels provided the carbon released during woody bioenergy production is reincorporated into new trees that replace those harvested for bioenergy. In contrast, energy generated from fossil fuels, such as coal and oil, emits carbon that has been sequestered underground for eons.

Key Findings for Criterion 5

- Through photosynthesis, trees pull carbon dioxide, a greenhouse gas, from the atmosphere and sequester it in wood and other tree parts.
- Forests sequester large amounts of carbon in soil organic matter and in the wood of living trees. As forests grow over time the amount of sequestered carbon increases.
- The total amount of sequestered carbon in U.S. forests is equal to approximately 27 years of carbon dioxide emissions for the U.S.
- The annual net increase in carbon sequestered in U.S. forests due to tree growth is equivalent to about 10 percent of the annual emissions of carbon dioxide and associated greenhouse gasses.
- When trees are harvested and converted to wood products, the carbon in those products remains sequestered until they eventually decompose or are burned.
- Using woody biomass to replace fossil fuels for energy production can reduce the release of carbon from the fossil fuels that would be used instead.
- In 2007, the equivalent of 2 percent of the energy consumed in the United States came from wood combustion by industrial (1.3 percent), residential (0.4 percent), utility (0.2 percent), and other (0.1 percent) users.
- Less than 1 percent of U.S. electric power is generated from wood.



Carbon and Wood

A cubic foot of wood in a living oak tree weighs about 60 pounds (green weight)—roughly half composed of water and the other half composed of dry woody biomass, about 15 pounds of which is carbon (half of the dry weight or a quarter of the green weight). Carbon is found in cellulose, hemicellulose, lignin, and other compounds that form the wood and other parts of the tree. Woody biomass may be reported in dry tons or in green tons, and carbon is more often reported as equivalent tons of carbon dioxide than as elemental carbon—distinctions that are important when interpreting and comparing biomass and carbon statistics.

When trees grow they absorb carbon from the atmosphere in the form of carbon dioxide. Through photosynthesis trees sequester the carbon in wood, bark, leaves, flowers, roots, and seeds. When a tree or some part of a tree dies, the carbon it contains is released during decomposition. Carbon in decomposing roots may remain in the soil and gradually add to the large store of sequestered carbon in soils. Leaves are short-lived and release carbon back to the atmosphere quickly as they decompose. Carbon may be sequestered for centuries in the wood of living trees. Large dead and down trees may sequester carbon for decades as they decompose slowly and gradually release carbon dioxide back to the atmosphere.

Each year, per capita emissions in the United States—largely due to combustion of fossil fuels—produce 6 tons of carbon or the equivalent of 22 tons of carbon dioxide (USDOE 2009, USDA FS 2011e). That is the amount of carbon in about 800 cubic feet of wood (roughly 10 cords). Stacked as firewood it would equal a wood pile 4 feet high, 4 feet deep, and 80 feet long.

The amount of carbon that U.S. forests sequester each year is about 10 percent of total annual U.S. emissions of carbon dioxide and related greenhouse gasses.

The passages below report on the total quantity of carbon stored in forests, how forest carbon changes over time, the role of forest products in carbon sequestration, and the capacity to avoid carbon emissions from fossil fuels by using woody biomass for energy production. For consistency with other sections of this assessment, we report carbon in U.S. tons (2000 pounds) and acres or provide metric equivalencies to help link reported values to other sources, which—by convention—report carbon in metric units (2204 pounds or 1000 kg) and hectares (2.5 acres).

Indicators of forest contributions to global carbon cycles for northern forests

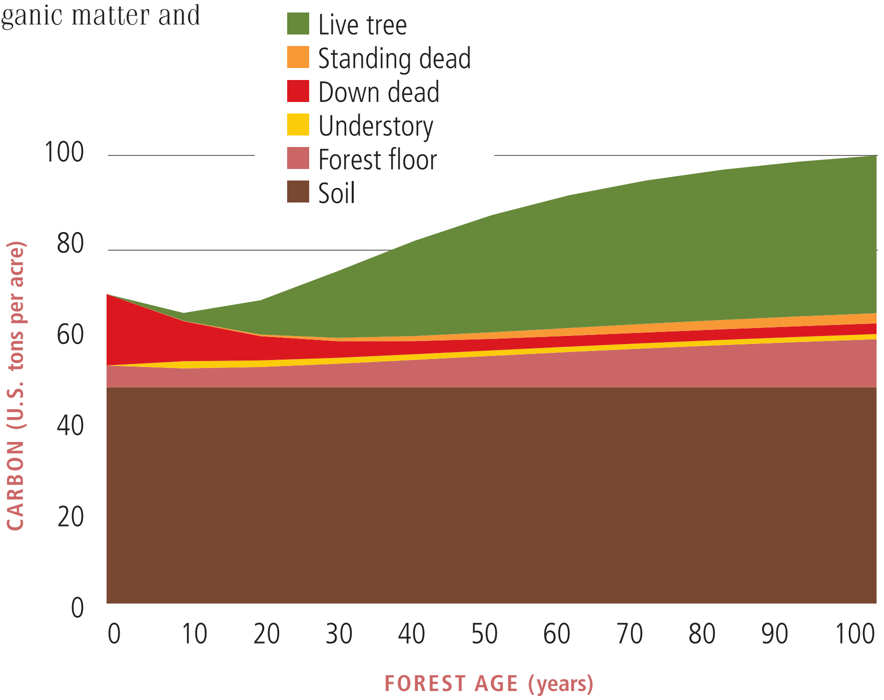
Carbon sequestered in northern forests

The two largest pools of sequestered carbon in a typical forest are in soil organic matter and in aboveground biomass (Fig. 37). Soil carbon changes slowly compared to aboveground biomass, which increases with forest growth and decreases with mortality or harvesting. Dead wood, litter on the forest floor, and tree roots are other large reservoirs of forest carbon.

The amount of carbon sequestered above ground in a forest is closely associated with wood volume or biomass. In general, more sequestered carbon occurs where more wood volume occurs (Fig. 19). However, inventorying carbon is more complicated than merely measuring aboveground forest volume because of the high proportion of carbon in soils, tree roots, and dead wood and because harvested forest products move sequestered carbon to other locations.

FIGURE 37

When and where carbon occurs in a typical forest—a composite summary for all northern forests showing average carbon by forest age and forest component; note that about 16 percent of live tree carbon is coarse roots (VanDuesen and Heath 2009).



Much of the carbon sequestered in U.S. forests is in Northern States (Fig. 38). This amount can increase over time as trees grow (above and below ground) and hold more carbon, or decrease as trees die or are harvested. As dead trees and down wood slowly decay, they release carbon gradually back into the atmosphere as carbon dioxide; if burned, they release carbon quickly. The total amount of sequestered carbon in U.S. forests is equivalent to about 27 years of carbon dioxide emissions for the United States (USDA FS 2011e). The annual increase in sequestered U.S. carbon from net annual forest growth is about 10 percent of U.S. annual greenhouse-gas emissions. Appendix Table A3 provides additional state-level detail on forest biomass and carbon.

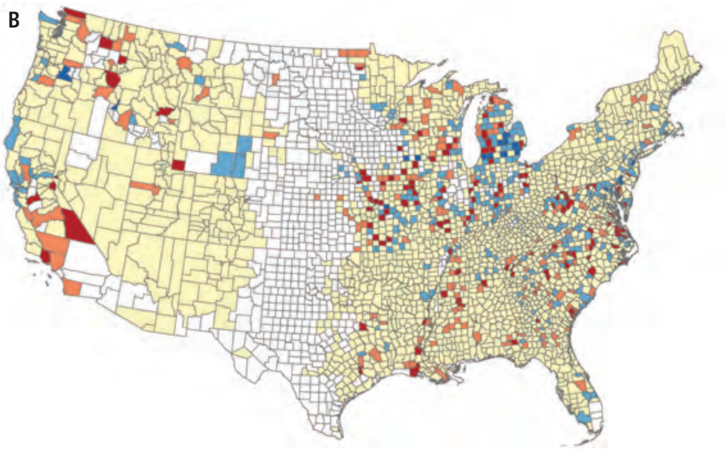
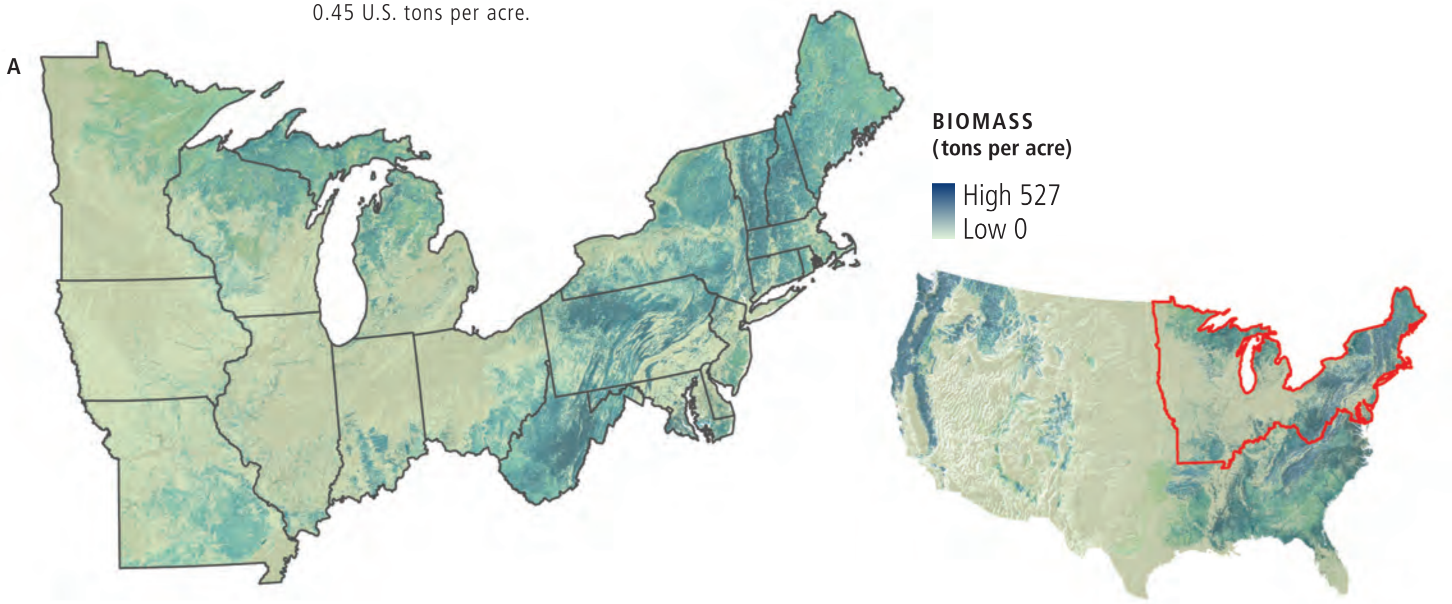


FIGURE 38

In 2006, (A) aboveground live tree biomass for Northern States (Blackard et al. 2008), and (B) estimated change in live tree carbon stock by U.S. county, accounting for harvest, land-use change and changes in live tree biomass of coarse roots, stems, branches, and foliage (Smith et al. 2009). In this case carbon change is reported as the equivalent mass in tons of carbon dioxide (CO_2) rather than carbon per se. One megagram (or metric ton) per hectare is equivalent 0.45 U.S. tons per acre.



Carbon Sequestered in Forest Products

When trees are harvested and converted to wood products, the carbon in those products remains sequestered until they decompose or are burned. Consequently, paper products typically sequester carbon for shorter periods than wood products such as building materials, flooring, or furniture. Even landfills sequester carbon in the form of discarded wood and paper products that decompose slowly because of compaction and lack of oxygen in the layers of landfill waste material. With the current mix of harvested materials and associate forest products, carbon in wood products from northern forests persists for a relatively long time.

Using Woody Biomass for Energy

The use of fossil fuels to produce energy releases carbon dioxide that was previously sequestered underground as coal, oil, or gas. By using woody biomass instead, society can reduce carbon from fossil fuels. Carbon that is already sequestered in the ground stays there (Malmsheimer et al. 2008) while carbon in woody biomass that is consumed for energy is released to the atmosphere instead. When forests harvested for biomass regenerate and grow, carbon is again sequestered in the wood growing on the regenerated forest. Thus, some carbon from using biomass for energy is cycled from the forest to the atmosphere and gradually back to the forest.

In 2007, about 2 percent of all U.S. energy consumption came from wood combustion by industrial (1.3 percent), residential (0.4 percent), utility (0.2 percent), and other (0.1 percent) users. Electric utilities throughout the North use wood for part of their energy production (Fig. 39), but less than 1 percent of U.S. electric power is generated with wood (USDOE EIA 2010).

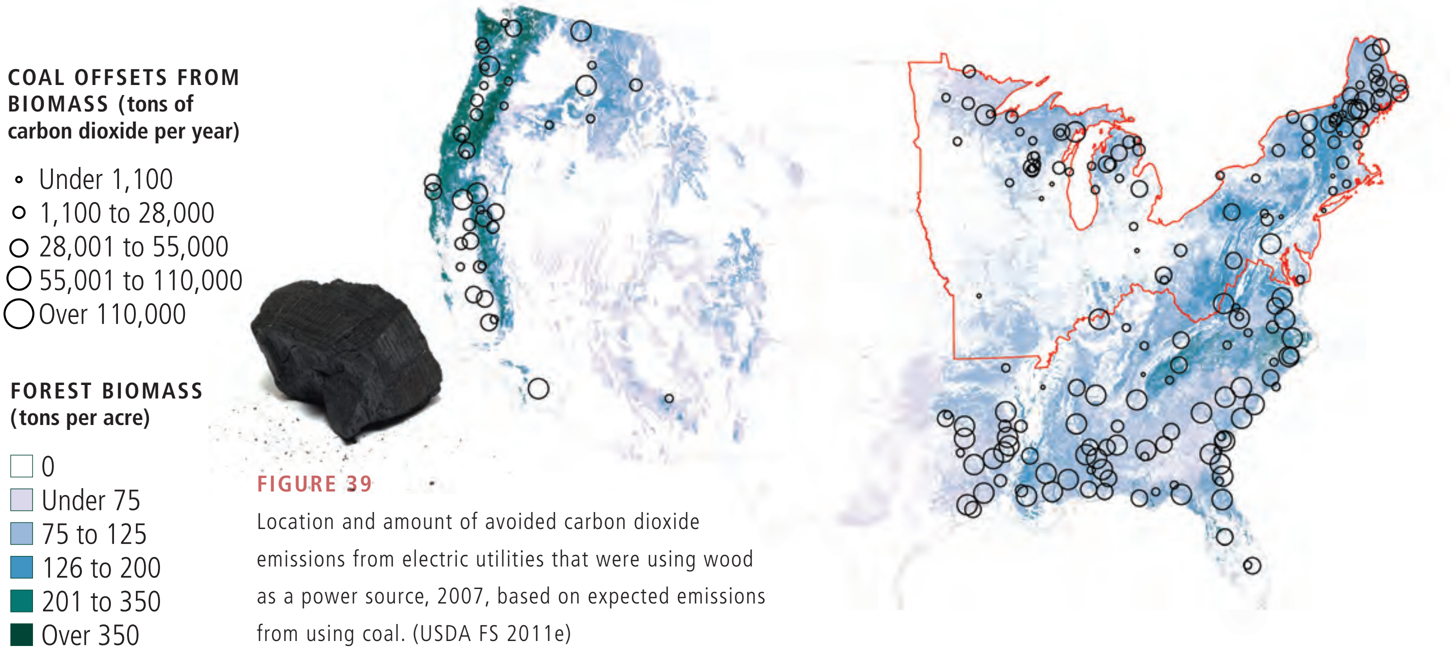


FIGURE 39

Location and amount of avoided carbon dioxide emissions from electric utilities that were using wood as a power source, 2007, based on expected emissions from using coal. (USDA FS 2011e)

The contribution of insects to global forest deadwood decomposition

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The amount of carbon stored in deadwood is equivalent to about 8 per cent of the global forest carbon stocks[1]. The decomposition of deadwood is largely governed by climate[2–5] with decomposer groups—such as microorganisms and insects—contributing to variations in the decomposition rates[2,6,7]. At the global scale, the contribution of insects to the decomposition of deadwood and carbon release remains poorly understood[7]. Here we present a field experiment of wood decomposition across 55 forest sites and 6 continents. We find that the deadwood decomposition rates increase with temperature, and the strongest temperature effect is found at high precipitation levels. Precipitation affects the decomposition rates negatively at low temperatures and positively at high temperatures. As a net effect—including the direct consumption by insects and indirect effects through interactions with microorganisms—insects accelerate the decomposition in tropical forests (3.9% median mass loss per year). In temperate and boreal forests, we find weak positive and negative effects with a median mass loss of 0.9 per cent and −0.1 per cent per year, respectively. Furthermore, we apply the experimentally derived decomposition function to a global map of deadwood carbon synthesized from empirical and remote-sensing data, obtaining an estimate of 10.9 ± 3.2 petagram of carbon per year released from deadwood globally, with 93 per cent originating from tropical forests. Globally, the net effect of insects may account for 29 per cent of the carbon flux from deadwood, which suggests a functional importance of insects in the decomposition of deadwood and the carbon cycle.

The world's forests are an important carbon sink[1], but global climate change is affecting carbon sequestration and release by altering tree growth[8,9], mortality[10,11] and decomposition[12,13]. Therefore, a comprehensive understanding of the forest carbon cycle and its climate sensitivity is critical for improving global climate change projections. Whereas previous research has focused strongly on carbon sequestration[14,15], the release of carbon—including through the decomposition of deadwood—remains poorly understood[7,16]. Deadwood currently stores 73 ± 6 petagram (Pg; 10^{15} g) of carbon (C) globally, which is about 8% of the global forest carbon stock[1] and 8.5% of atmospheric carbon[17]. The decomposition of deadwood is largely governed by climate[2–5], with the activity of different decomposer groups contributing to the considerable variation in decomposition rates[2,6,7]. Recently, the role of fungi in forest carbon cycling has received much attention[2,6] and they are believed to be the principal decomposers of deadwood[5–7]. Although local- and regional-scale studies indicate that insects can also make a considerable contribution to wood decomposition[7], global assessments that quantify the role of microorganisms and insects are lacking. Given the sensitivity of insects to climate change[18,19] and the observed declines in insect biodiversity[20–22], a better understanding of the interactions between insect decomposers and climate is needed to more robustly project carbon flux from deadwood and the role of deadwood in the global forest carbon sink[11,16,23].

Here we quantified the role of deadwood-decomposing insects relative to climate by conducting standardized field experiments of wood decomposition across 55 sites on six continents (Fig. 1a). Our sites were selected to capture the gradient of temperature and precipitation conditions under which forests occur globally. Insects and other animals (hereafter collectively termed insects for brevity) had unrestricted access to wood placed on the forest floor in the uncaged treatment in our experiment, whereas they were excluded from the wood in the closed-cage treatment using mesh cages (Extended Data Fig. 1). Our estimate of the effect of insects on wood decomposition was quantified as the difference between the decomposition rates in the uncaged and closed-cage treatments. This measure can be considered the 'net effect of insects', consisting of the direct consumption of wood by insects and indirect effects through interactions with microorganisms. The latter effects include—for example—competition for resources, grazing on fungal mycelia, creation of entry ports or vectoring, and these can therefore either increase[24] or decrease wood decomposition[25,26]. As a consequence, the direct consumption by insects could be higher than our net estimate at sites where the interactions between insects and microorganisms decrease the decomposition rates. To explore the effects of caging on microclimatic conditions and decomposition rates, we implemented a third treatment (open cage) using cages with holes, which allow insects access to the wood samples under similar microclimatic conditions to those logs in the closed-cage treatment (Supplementary Information section 1). We assessed deadwood decomposition as the loss of dry mass over a period of up to 3 years for wood samples with bark (around 3 cm in diameter, 50 cm in length) of locally dominant native tree species (142 tree species in total) as well as for standardized wooden dowels without bark. In total, we recorded wood mass loss for 4,437 individual samples. We used a Gaussian generalized linear mixed log-link model with site-specific random effects to quantify the influence of insects (uncaged versus closed cages), site-level temperature and precipitation as well as the type of wood (angiosperm versus gymnosperm) on the annual rates of wood mass loss. Although some influence of caging on microclimate cannot be ruled out, we focused on the comparison between uncaged and closed-cage treatments, because analyses across treatments indicated that this comparison provides the most robust estimate for the net effect of insects on wood decomposition (Supplementary Information section 1, Extended Data Table 1 and Extended Data Fig. 2).

To provide an estimate of the global carbon flux from deadwood decomposition (hereafter referred to as deadwood carbon release) and to quantify the functional importance of insects for global deadwood carbon, we applied the model derived from our decomposition experiment to a new global deadwood carbon map (Fig. 1a), which we synthesized from empirical and remote-sensing data. As the global modelling of deadwood remains challenging, we conducted in-depth analyses of uncertainty, evaluating the decomposition function derived from our experiment against independent empirical data[27] and quantifying the relative contribution of different sources of uncertainty in a sensitivity analysis (Supplementary Information section 2 and Extended Data Table 2). The sensitivity analysis also highlights how further research can improve the modelling of global carbon fluxes from deadwood.

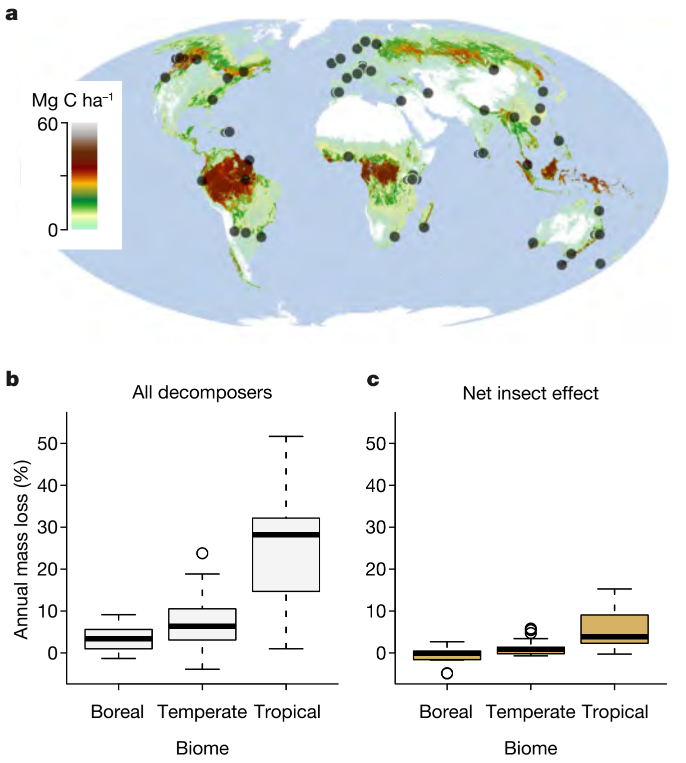

Fig. 1 | Decomposition rates and insect effects per biome. a, Estimated carbon pools in deadwood with a diameter of >2 cm (Mg C ha^{-1}) with 5 arcmin spatial resolution and the location of the 55 experimental sites (grey dots). **b**, **c**, Annual mass loss of deadwood of native tree species when all decomposer groups have access (uncaged treatment) (**b**) and the difference in annual mass loss between uncaged and closed-cage treatments that are attributed to the net effect of insects (**c**). Data are predicted values for both angiosperm and gymnosperm species at 55 and 21 sites, respectively, based on a Gaussian generalized linear mixed log-link model for 2,533 logs with site-specific random effects and temperature, precipitation, treatment and host type, as well as their interactions, as fixed effects (Table 1). Boxes represent data within the 25th and 75th percentile, black lines show the medians and whiskers extend to 1.5× the interquartile range. Note that the classification into biomes is shown for illustrative purposes, whereas the statistical model is based on continuous climate variables.

Climate and insect effects

In our global experiment, the wood decomposition rate was the highest in the tropics/subtropics (hereafter called tropics; median = 28.2% mass loss per year), and was considerably lower in the temperate (median = 6.3%) and boreal/hemiboreal (hereafter called boreal; median = 3.3%) biomes (Fig. 1b). Wood decomposition rates were highly climate-sensitive, driven by the complex interplay between temperature and precipitation (Table 1). Decomposition rates increased with increasing temperature across the full gradient of precipitation, but the effects of temperature were strongest at high levels of precipitation (Fig. 2a and Extended Data Fig. 3a). Precipitation affected decomposition rates negatively at low temperatures but positively at high temperatures. The observed positive global relationship between wood decomposition and temperature was similar to patterns observed at local-to-continental scales[2,4], as well as for the decomposition of non-woody litter[12,28], and is consistent with general theory, which predicts an increase in metabolic rates and enzymatic activity with temperature[29]. Moreover, the length of the vegetation period usually increases with temperature, which may further increase annual decomposition rates. Weaker positive effects of temperature on wood decomposition under low levels of precipitation may be the result of low levels of moisture in the wood, limiting microbial activity[30,31] and selecting for drought-tolerant fungal species that have a reduced ability to decompose wood[6]. Given that temperature is predicted to increase globally[32], our results indicate that wood decomposition rates are likely to increase in the future. The strength of this increase will be modulated by current and future levels of precipitation and the emerging water balance of a site[33]. Decomposition rates were higher for angiosperms than for gymnosperms (Table 1), which is consistent with results from a global meta-analysis and can be explained by differences in wood traits[34]. Results for standardized wooden dowels were similar to those of wood from native tree species (Extended Data Table 1).

Insect access to deadwood affected decomposition, but this effect was contingent on climatic conditions (Table 1). The net effect of insects on decomposition was particularly high in the tropics (median = 3.9% mass loss per year) (Fig. 1c). By contrast, effects were low in the temperate biome and even negative in the boreal biome (median of 0.9% and −0.1%, respectively) (Fig. 1c). The net effect of insects generally increased with temperature, with effect size strongly mediated by precipitation (Table 1). At low levels of precipitation, temperature had only a minor influence on the net effect of insects. By contrast, at high levels of precipitation, temperature was a strong driver of the net effect of insects on decomposition (Fig. 2b and Extended Data Fig. 3b). At high temperatures, increasing precipitation increased the net effect of insects, whereas at low temperatures, increasing precipitation resulted in a negative net effect of insects. Thus, decomposition rates were higher when insects were excluded at low temperatures and high precipitation. The complex relationships between insects and climate are driving several mechanisms that determine the net effect of insects on wood decomposition. First, wood-feeding termites are a key group of decomposers[7,35], but are largely restricted to regions with high temperatures (Fig. 2b). Nevertheless, considerable variation in the net effect of insects also exists among sites at which termites are present (Fig. 2b), underlining the importance of factors in addition to termite occurrence. Second, temperature affects the metabolic rate of insects, increasing consumption and accelerating larval development directly[18] as well as indirectly through enhanced food quality[36]. Third, insects can be negatively affected by high wood moisture when precipitation is high and evaporation low, as is the case in humid boreal forests, for example (Extended Data Fig. 3b), due to low aeration or high pathogen pressure[37]. Conversely, moisture is a limiting factor at high temperatures, restricting the period of high insect activity to the rainy season[38]. Fourth, interactions between insects and microorganisms can

Table 1 | Drivers of wood decomposition

Predictor	Estimate (×10^3)	s.e. (×10^3)	*z* value	*P* value	Relative effect and 95% CI
Temperature (°C)	−11.009	3.021	−3.644	<0.001	0.989 (0.983–0.995)
Precipitation (dm yr^{-1})	−3.135	3.322	−0.944	0.345	0.997 (0.990–1.003)
Host: angiosperm	−150.477	22.506	−6.686	<0.001	0.860 (0.823–0.899)
Host: gymnosperm	−82.825	24.862	−3.331	0.001	0.921 (0.877–0.966)
Treatment: uncaged versus closed	−29.228	5.694	−5.133	<0.001	0.971 (0.960–0.982)
Temperature × precipitation	−0.565	0.401	−1.408	0.159	0.999 (0.999–1.000)
Temperature × host	5.016	1.250	4.014	<0.001	1.005 (1.003–1.007)
Precipitation × host	−0.434	3.587	−0.121	0.904	1.000 (0.993–1.007)
Temperature × treatment	−4.161	0.742	−5.608	<0.001	0.996 (0.994–0.997)
Precipitation × treatment	−5.236	0.923	−5.675	<0.001	0.995 (0.993–0.997)
Temperature × precipitation × host	0.104	0.327	0.317	0.751	1.000 (0.999–1.001)
Temperature × precipitation × treatment	−0.728	0.113	−6.451	<0.001	0.999 (0.999–0.999)

Results from a Gaussian generalized linear mixed log-link model of the relative annual mass loss of wood of native tree species derived from a global deadwood decomposition experiment. The model is based on data from closed-cage and uncaged treatments, comprising 2,533 logs of native tree species from 55 sites. Fixed effects were the mean annual temperature and the mean annual precipitation, which were both centred and scaled, host tree type (angiosperm versus gymnosperm) and treatment, as well as their two- and three-way interactions, with site as a random effect. Estimates and standard errors of the maximum likelihood estimates (s.e.) for temperature and precipitation are transformed back to °C and dm yr^{-1}, respectively. The main effects for each variable are interpretable when the remaining variables are fixed to their reference value (15 °C and 13 dm yr^{-1}). A relative effect (that is, the exp(estimate)) of, for instance, 0.989 indicates that for a temperature increase of 1 °C with all other variables fixed (precipitation at 13 dm yr^{-1}, host and treatment), the deadwood dry mass after 1 year would be 98.9% of the mass without this change in temperature. This represents an additional mass loss of 1.1% induced by a 1 °C increase in temperature. The marginal R^2 of the model was 0.84.

decrease wood decomposition: insects, for example, can introduce fungal species that do not contribute strongly to wood decomposition themselves, while suppressing other principal wood-decomposing fungi, thus lowering the overall decomposition rate[25]. In cold and humid regions, such biotic interactions could outweigh the effects of direct consumption and lead to an overall negative net effect of insects on wood decomposition.

Our findings indicate that wood decomposition is driven by a complex interplay of temperature and precipitation with the decomposer community. Climate warming could accelerate wood decomposition by increasing microbial activity and insect-mediated wood decomposition, particularly in regions in which moisture is not limiting. However, increased drying as a result of global climate change could also decrease the decomposition of deadwood. Our results support that biodiversity loss of insects has the potential to affect deadwood decomposition, but that effects may vary regionally. To improve predictions of the functional effects of biodiversity loss, more research is needed on how specific components of decomposer communities (that is, biomass, species number, functional composition and species interactions) influence deadwood decomposition[7]. Our work suggests that the strongest functional effects of changes in the decomposer community will occur in regions with a warm and humid climate, which should be a particular focus of further research.

Global carbon flux estimate

To assess the role of deadwood decomposition in the global carbon cycle, we applied the relationship between decomposition rates and local climate derived from our global experiment (Table 1) to a map of the global carbon currently stored in deadwood (Fig. 1a). As our experiment focused on small-diameter deadwood over 3 years, we adjusted the decomposition rates to account for slower mass loss of large-diameter deadwood (details are provided in the Methods and Supplementary Information section 2). We evaluated our relationship between decomposition rate and local climate against 157 independent empirical observations from previous deadwood surveys[27], spanning the full range of deadwood diameters of >7 cm, time since tree death and climatic conditions. We obtained a good match between the results from our model and these independent data (Extended Data Fig. 4), suggesting that our approach is robust.

We estimate that 10.9 ± 3.2 Pg C could be released from deadwood per year globally. This suggests that the decomposition of deadwood could be an important flux in the global carbon cycle. Our estimate

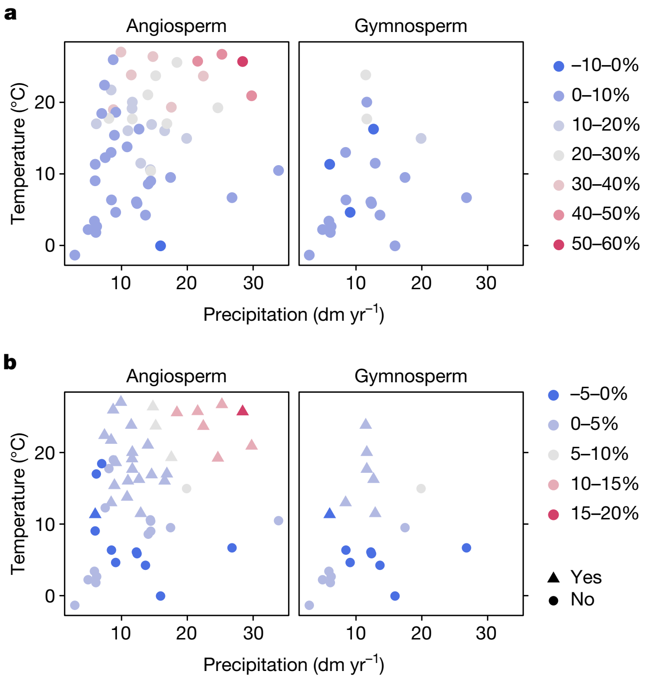

Fig. 2 | Decomposition rates and net insect effects in climate space. **a**, **b**, Annual mass loss of deadwood of native tree species, considering all possible groups of decomposers (uncaged treatment) (**a**) and annual mass loss attributed to insects (difference in mass loss between uncaged and closed-cage treatments), **b**, relative to the mean annual temperature and mean annual precipitation. Symbols indicate whether termites occur in the study areas. Points represent predicted values for angiosperm species at 55 sites and gymnosperm species at 21 sites based on a Gaussian generalized linear mixed log-link model for 2,533 logs with site-specific random effects and temperature, precipitation, treatment, host division, as well as their interactions, as fixed effects. Note that the lower sample size for gymnosperm species represents their global distribution.

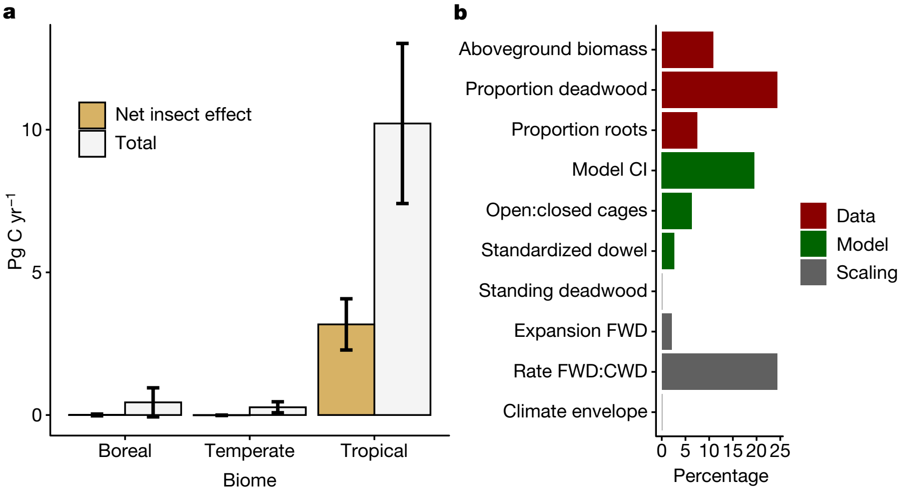

Fig. 3 | Global annual carbon release from deadwood and sensitivity analysis. a, Annual carbon released (Pg C yr^{-1}) from deadwood per biome. Error bars indicate the uncertainty of the biome-specific estimate as determined by the sensitivity analysis. **b**, Relative contributions to the overall uncertainty of the global estimate of total carbon release from the decomposition of deadwood. The colour of the bars indicates the uncertainty category. CI, confidence interval; CWD, coarse wood debris; FWD, fine woody debris. See Extended Data Table 2 for a detailed description of each factor and an uncertainty assessment of the net insect effect.

corresponds to 15–25% of the annual release of carbon from soils globally (estimated to be 50–75 Pg C yr^{-1} (ref. [28])) and is 115% of the current anthropogenic carbon emissions from fossil fuels (9.5 Pg C yr^{-1} (ref. [17])). We note, however, that not all carbon that is released from deadwood through decomposition is emitted to the atmosphere, as parts are immobilized in the biosphere or in soils[39,40]. Carbon release from deadwood is highest in tropical biomes (10.2 Pg C yr^{-1}) (Fig. 3a and Extended Data Table 3), where large deadwood carbon pools and high decomposition rates coincide (Extended Data Fig. 5). Although deadwood carbon stocks are also considerable in temperate and boreal biomes (amounting to 35% of all carbon stored in deadwood globally), the climatic limitations for wood decomposition as well as differences in decomposer communities (for example, the absence of termites) render annual carbon fluxes from deadwood much smaller in these biomes (that is, 0.44 Pg C yr^{-1} and 0.28 Pg C yr^{-1} in boreal and temperate forests, respectively), accounting for less than 7% of the global carbon release from deadwood. Globally, the net effect of insects on wood decomposition may result in a carbon flux of 3.2 ± 0.9 Pg C yr^{-1}, which represents 29% of the total carbon released from deadwood (Fig. 3a and Extended Data Fig. 5).

Our global estimates are only a first step to a better quantification of the role of deadwood decomposition in the global carbon cycle. Uncertainties related to the underlying data, the statistical models and other assumptions necessary for upscaling our experimental results were assessed in a global sensitivity analysis. This analysis bounded the uncertainty of global annual carbon release from deadwood and the net effect of insects at approximately ±25% around the mean. Of the various sources of uncertainty that were considered, the underlying data on deadwood carbon stocks contributed most strongly to the overall uncertainty (Fig. 3, Extended Data Table 2 and Supplementary Information section 2). Our results suggest that assessments of the global deadwood carbon cycle could be improved by more accurately quantifying deadwood stocks in tropical forests. Although the effects of wildfire were included in our deadwood carbon map through the underlying inventory data, we did not explicitly consider deadwood carbon release from fire. We note, however, that a large portion of the carbon stored in deadwood is not combusted in wildfires[41,42]. Further uncertainty results from our experimental design included the following. It cannot be ruled out that altered microclimatic conditions in cages affected the estimates of the net effect of insects derived from the comparison between closed-cage and uncaged treatments. Such a bias would lead to an underestimation of the net insect effect in the tropics and an overestimation in the temperate zone (Supplementary Information section 1). When the global annual net effect of insects on deadwood decomposition was derived from the comparison of closed-cage and open-cage treatments, it still amounted to 1.76 Pg C. However, this value underestimates the true effect of insects due to a reduction in insect colonization in the open-cage treatment (Extended Data Fig. 2 and Supplementary Information section 1).

Our experiment highlights that deadwood and wood-decomposing insects have an important role in the global carbon cycle. In contrast to the prevailing paradigm that insects generally accelerate wood decomposition[7], our results indicate that their functional role is more variable, and is contingent on the prevailing climatic conditions. We conclude that ongoing climate warming[32] will likely accelerate decomposition by enhancing the activity of microorganisms and insects—an effect that will be particularly strong in regions in which moisture is not limiting. To robustly project the future of the forest carbon sink[23,43], dynamic global vegetation models need to account for the intricacies of both deadwood creation (for example, through natural disturbances) and deadwood decomposition.

Online content

Any methods, additional references, Nature Research reporting summaries, source data, extended data, supplementary information, acknowledgements, peer review information; details of author contributions and competing interests; and statements of data and code availability are available at https://doi.org/10.1038/s41586-021-03740-8.

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Methods

Experimental set-up

We established 55 experimental sites in currently forested areas on six continents and in three major biomes, spanning gradients in mean annual temperature from −1.4 °C to 27.0 °C and mean annual precipitation from 2.90 dm yr^{-1} to 33.86 dm yr^{-1} (Fig. 1a). Sites were located in mature, closed-canopy stands of the dominant zonal forest type and were selected so that structural and compositional characteristics were similar to those of natural forests. To quantify the net effect of insects on wood decomposition, we compared decomposition between uncaged wood accessible to all decomposers (uncaged treatment) and wood in closed cages that excluded insects and other invertebrates (closed-cage treatment) (Extended Data Fig. 1). Cages excluded vertebrate and invertebrate decomposers, but for simplicity, and as insects comprise the functionally most important taxa, we refer to insects throughout the manuscript. To explore the microclimatic effects of caging[44], we added a third treatment of wood in cages with large openings (open-cage treatment) that not only allowed colonization by insects, but also provided similar microclimatic conditions to the closed-cage treatment (Supplementary Information section 1). Analyses across treatments showed that the most robust assessment of the net effect of insects on wood decomposition originated from the uncaged versus closed-cage treatment, as cages had a significant effect on insect colonization, but not on microclimatic conditions, and thus decomposition rates were reduced in the open-cage compared to the uncaged treatment (Supplementary Information section 1 and Extended Data Fig. 2).

Cages measured 40 cm × 40 cm × 60 cm and were made of white polyester mesh with approximately 6,450 mesh per cm^2. The honeycomb-shaped mesh holes had a width of approximately 0.5 mm. Open cages had four rectangular openings measuring 3 cm × 12 cm at both front sides and four rectangular openings measuring 10 cm × 15 cm at the bottom, representing, in total, 6% of the surface area of the cage. Furthermore, open cages had a total of ten 12-cm slits at the top and long sides. Cages were placed on a stainless-steel mesh (0.5 mm mesh width), which had the same openings as the bottom side of the cages in the open-cage treatment. The top layer of fresh leaf litter was removed before the installation of treatments. The cages and layers of steel mesh were both tightly fixed to the ground using tent pegs, to ensure that all deployed logs had close contact with the soil and to allow water uptake and fungal colonization from the soil. At each site, the three treatments were performed three times—that is, three installations per treatment per site—resulting in a total of nine installations per site (Extended Data Fig. 1). The nine installations were arranged in a matrix of 3 × 3 with a spacing of 2 m between installations, resulting in a total size of approximately 15 m × 15 m. Treatments were assigned randomly to each of the nine locations within a site. The mean spore size and hyphae width of saprotrophic fungal species (mean spore length and width, 8.9 μm and 5.5 μm, respectively[45]; hyphae width, 5–20 μm (refs. [46,47])) are smaller than the mesh width of our cages by an order of magnitude. Rhizomorphs—that is, linear aggregations of several hyphae—can be wider, but during mycelial growth each hypha extends apically rather than the whole rhizomorph[48–50]. Therefore, it is unlikely that the cages hampered fungal colonization. Data loggers recorded air temperature and humidity for the three treatments at nine sites (see Supplementary Information section 1 for details).

Decomposition measurements

Decomposition was measured as the dry mass loss of unprocessed wood of three of the locally most abundant autochthonous tree species at each study site (Supplementary Table 3-1), as well as for standardized machined wooden dowels. Unprocessed wood of local tree species with the bark retained is more likely to be colonized by local insects and fungi than machined wood without bark[44]. The latter was used to compare the decomposition based on a standardized substrate replicated across all sites. We cut wood of local tree species (around 3 cm in diameter and about 60 cm in length) from either branches or stems of young healthy trees without visible signs of insect or fungal activity. One 5-cm long section was cut from each end of all fresh logs, and the fresh mass of both the cut sections and the resulting 50-cm logs was weighed. The dry mass of all 5-cm sections was measured after drying them at 40 °C until no further mass loss was observed. We calculated the dry mass of the respective 50-cm logs as dry $\mathrm{mass}_{50\,\mathrm{cm}}$ = (fresh $\mathrm{mass}_{50\,\mathrm{cm}}$/fresh $\mathrm{mass}_{5\,\mathrm{cm}}$) × dry $\mathrm{mass}_{5\,\mathrm{cm}}$. Each installation received three 50-cm long logs of each of the three local tree species and one (closed cage) or two (open cage and uncaged) standardized wooden dowels, giving a total of 96 logs at each site. Standardized dowels (3 cm in diameter, 50 cm in length) were dried machined dowels of *Fagus sylvatica* L. without bark. They were obtained from a single producer in Germany and were then distributed to all sites. Initial dry mass of the dowels was measured directly after drying. All logs and dowels were labelled using numbered plastic tags and assigned randomly to one of the nine installations.

The experiment was established between March 2015 and August 2016 depending on the seasonality of each site. After approximately 1, 2 and 3 years, one of the three installations of each treatment per site was randomly selected and collected to measure wood decomposition. That is, all logs from one uncaged, one closed-cage and one open-cage treatment were collected per site at the same time. We chose this approach because the maximum distance between installations was 6 m and thus within-site variation was expected to be rather low. Moreover, we wanted to ensure that the same number of logs could be sampled per treatment and year and failure of cages over time would have resulted in an unbalanced number of logs per treatment. Owing to the loss of some cages, high decomposition rates at some sites and logistical restrictions, we were not able to maintain the experiment for 3 years at all sites (Supplementary Table 3-1). Litter and soil attached to the wood was removed carefully upon collection, whereas fungal fruit bodies were retained. We assessed insect colonization (presence or absence) for each log based on visible feeding marks, larval tunnels or exit holes for 3,430 (91%) of the analysed logs. The collected logs were dried at 40 °C until the mass remained constant and the dry mass was measured. At sites at which termites were present, logs were burned to account for soil that might have been carried into the wood by these insects[44]. This involved placing one sample at a time onto a steel pan atop a propane burner, and an electrical fan was used to provide aeration and to blow away ash. The residual soil was weighed and its mass subtracted from the dry mass of the wood.

Statistical analyses of the decomposition experiment

All statistical analyses were performed in R v.4.0.4[51]. For each site, we derived information on average climate conditions from WorldClim (v.2)[52], specifically BIOMOD variables 1 (mean annual temperature) and 12 (mean annual precipitation sum). We modelled relative wood mass loss of local tree species over time using a Gaussian generalized linear mixed model (function glmer in package lme4[53], v.1.1.26) with log link. The dry mass of each individual log at time t served as the response variable and the log-transformed initial dry mass ($t = 0$) was used as an offset term. For each increase of one time unit (1 year), the relative reduction is given by $\exp(\beta)$. Note that the model contained no intercept due to the constraint $\exp(\beta)^0 = 1$. The rate $\exp(\beta)$ was modelled depending on treatment (closed cage versus uncaged) and host type (angiosperm versus gymnosperm), as well as mean annual temperature (°C) and the mean annual precipitation sum (dm yr^{-1}). Temperature and precipitation were centred and scaled before modelling, but model coefficients were then back-transformed for ease of interpretation. Reference values for temperature and precipitation were 15 °C and 13 dm yr^{-1}, respectively. The model included site-specific random time slopes to deal with clustered observations. On the basis of this model, we computed the fitted annual relative mass loss (as a percentage) for each site considering temperature and precipitation. This was done separately for angiosperm and gymnosperm wood for all sites where the respective tree species were present. Note that differences in decomposition between tree species could not be tested but were subsumed in the random slope of the site, as most tree species occurred at only a few sites (Supplementary Table 3-1).

To evaluate the potential differences in decomposition rates between the wood of native tree species and standardized wood samples, we estimated the same model for the standardized wooden dowels. Further models were fitted to evaluate the potential microclimatic effects of the cages on decomposition rates and insect colonization. This included one model for the wood decomposition of native tree species for the treatments closed cage versus open cage, and one model comparing the wood decomposition between all three treatment levels (uncaged, closed cage and open cage) using a post hoc test. A binomial generalized linear mixed model was fitted for insect colonization and linear mixed models were fitted for mean daily temperature and mean daily relative humidity. Post hoc tests were applied to these models for comparisons among the three treatments.

Estimation of global carbon fluxes from deadwood decomposition

To estimate the global carbon flux from deadwood decomposition, we fitted an additive beta regression model (function gam with family betar in package mgcv[54], v.1.8) to the site-specific predicted relative annual mass loss using temperature and precipitation as predictors, separately for angiosperms and gymnosperms. On the basis of the predicted relative annual mass loss for the uncaged treatment, this model was used to predict the total deadwood carbon release globally (that is, attributable to all types of decomposers). To quantify the amount of carbon released from deadwood due to the net effect of insects, we applied the beta regression model to the predicted relative annual mass loss for the closed-cage treatment and calculated it as carbon $\text{release}_{\text{uncaged}}$ − carbon $\text{release}_{\text{closed cage}}$.

We applied this model to a spatially explicit global map of carbon stored in deadwood of angiosperms and gymnosperms, which we synthesized from empirical and remote-sensing datasets. We used mean annual temperature and the sum of the mean annual precipitation from WorldClim (v.2)[52] as predictor data. The GlobBiom (http://globbiomass.org) dataset provides high-resolution estimates of forest biomass based on Earth Observation data within the framework of ESA's GlobBiomass project. We used the GlobBiom aboveground biomass layer (that is, the stem, bark and branch compartments) for the reference year 2010, and aggregated information to the base resolution of WorldClim, that is, 5 arcmin (Extended Data Fig. 6a). We extended the aboveground biomass information provided by GlobBiom to total live carbon (including roots) by applying biome-specific root-expansion factors[55] and biome-specific biomass-to-carbon conversion factors between 0.47 and 0.49 (ref. [16]) (Extended Data Fig. 6b). The delineation of forest biomes was taken from FAO[56].

We calculated deadwood carbon stocks at a spatial grain of 5′ by relating deadwood carbon stocks to total live carbon stocks (that is, deadwood carbon fraction). To quantify the regional deadwood carbon fractions, we used previously compiled data[1], which are based on forest inventory data and represent the most comprehensive analysis of global forest carbon stocks available to date. We reanalysed their dataset and amended it with data from the FAO Forest Assessment Report[57] for cases in which values were missing (Extended Data Table 3). Our estimate of global deadwood carbon stocks therefore reflects local differences in forest productivity, mortality and land management. The previously reported values[1] defined deadwood as "all non-living woody biomass not contained in the litter, either standing, lying on the ground, or in the soil" with a diameter of >10 cm. We extended our deadwood carbon pool estimate to include all deadwood with a diameter of >2 cm by applying an expansion factor based on empirical allometric relationships[58]. Our global map of deadwood (Fig. 1a) thus represents the total amount of carbon stored in standing and downed deadwood with a diameter of >2 cm for the reference year 2010.

To differentiate between deadwood of angiosperms and gymnosperms, we used the proportion of broad- and needle-leaved biomass derived from the global land cover product GLCNMO2013[59]. The resolution of GLCNMO2013 is 1/240 degree (that is, each of our 5′ cells contains 400 land cover pixels), and it provides information on 20 land cover classes. We reclassified these to 'broadleaved', 'needle-leaved' and 'mixed forest', and aggregated to 5′ cells for each of the three forest types. The final proportion of each group was calculated assuming that carbon in mixed forests was equally distributed between angiosperms and gymnosperms (Extended Data Fig. 6c).

The experimental sites were chosen to span the global bioclimatic space inhabited by forests. Nonetheless, gaps remained in very cold and dry climatic conditions for both angiosperm and gymnosperm species as well as in very warm and wet climatic conditions for gymnosperm tree species. We constrained the application of our decomposition models to the climate space covered by the experiment to avoid extrapolation beyond our data. Specifically, we defined the bioclimatic space for robust predictions using a convex hull around experimental sites in the temperature–precipitation space (using a buffer of 3° and 3 dm, respectively). Subsequently, climatic conditions outside that convex hull were mapped to the nearest point within the hull in our modelling (Extended Data Fig. 7).

Our statistical model was derived from deadwood samples with a diameter of around 3 cm and thus overestimates annual decomposition rates when applied to the full diameter range of deadwood (Supplementary Information section 2). To address this potential bias, we used a conversion factor relating wood mass loss of fine woody debris (FWD, <10 cm in diameter) to coarse woody debris (CWD, >10 cm). We based our conversion factor on data from 11 peer-reviewed studies reporting data on both CWD and FWD decomposition, covering all major global biomes (Supplementary Table S2-1). As the relationship of the mass loss rate of CWD over the mass loss rate of FWD was robust across different climates, we used its median value (0.53) in our upscaling. An evaluation of the final deadwood decomposition rates used for deriving a global estimate of the carbon flux from deadwood was performed against independent data from 157 previously compiled observations[27]. This evaluation against independent data indicated a good agreement across all major biomes and diameter classes (Extended Data Fig. 4).

Finally, we accounted for the slower carbon release from standing deadwood relative to downed woody debris, particularly in dry regions of the boreal and temperate biome. On the basis of a wood decomposition dataset for standing and downed deadwood across several decay classes for the temperate and boreal biome[60], we estimated the decomposition of standing deadwood to be 33–80% slower compared to lying logs. This is consistent with a detailed analysis for temperate forests in Switzerland[61] that found a slowdown of 42%. In the tropics, however, the decomposition rates of standing trees have the same or sometimes even higher decomposition rates as downed trees[3,62,63]. We assumed a reduction of decomposition rates by 50% for standing deadwood in temperate and boreal forests, and no reduction in the tropical biome in our upscaling. On the basis of large-scale inventories[64–68], we estimated the proportion of standing deadwood of the total deadwood as 25% and 30% for the boreal and temperate biome, respectively.

Our global estimate of the carbon fluxes of deadwood decomposition required a number of analytical steps and assumptions, each of which is associated with uncertainties. These can be classified into uncertainties related to deadwood carbon stocks (data uncertainties), uncertainties related to the statistical modelling of deadwood decomposition (model uncertainties) and uncertainties in the upscaling of the model results to the global scale (scaling uncertainties). To assess the robustness of our estimate, we performed a global sensitivity analysis[69] in which we selected 3–4 indicators for each of these three categories of uncertainty, and estimated their influence on the overall result.

For each of the ten indicators analysed in total, we selected either a single alternative (for example, the use of the standardized dowels instead of the native species) or an upper and lower bound around the default value based on available data or indicator-specific assumptions (Extended Data Table 2). With regard to data uncertainty, we investigated uncertainties associated with the GlobBiom dataset used as the important data basis here, the deadwood carbon pool estimates[1] and the expansion factors used to derive total biomass from aboveground biomass[55]. Model uncertainties were considered by using alternative models using the 97.5th and 2.5th percentile of parameter values for fixed effects of the original model, an additional model accounting for potential microclimatic effects of cages (that is, using the open-cage instead of the uncaged treatment) and a model based on the results of the standardized dowels (instead of the native tree species). Scaling uncertainties were addressed by analysing alternative expansion factors to include deadwood <10 cm, varying the relationships between the FWD and CWD decay rates, alternative assumptions regarding the proportion and decay rate of standing deadwood, and the treatment of regions outside of the climate envelope covered by our experiment (see Extended Data Table 2 for details). All factor levels of all indicators were allowed to vary simultaneously, resulting in a total of 4,860 estimates for annual deadwood carbon release and the net effects of insects. The relative influence of each indicator on the total uncertainty was derived using an ANOVA, determining the percentage of variance explained by each factor. The contribution at the level of uncertainty categories was derived as the sum of the factors per category. The uncertainty range for the global annual deadwood carbon release estimated from this global sensitivity analysis was ±3.14 Pg C and the net effect of insects varied by ±0.88 Pg C. Data uncertainty was identified as the most important factor (around 40%), but both model and scaling uncertainty were also highly influential, each contributing 25–30% to the overall variation in the results (Extended Data Table 2).

Data availability

Raw data from the global deadwood experiment, our global map of deadwood carbon and our map of predicted decomposition rates are publicly available from Figshare https://figshare.com/s/ffc39ee0724b11bf450c (https://doi.org/10.6084/m9.figshare.14545992).

Code availability

An annotated R code including the data needed to reproduce the statistical analyses, global estimates and sensitivity analysis is publicly available from Figshare https://figshare.com/s/ffc39ee0724b11bf450c (https://doi.org/10.6084/m9.figshare.14545992).

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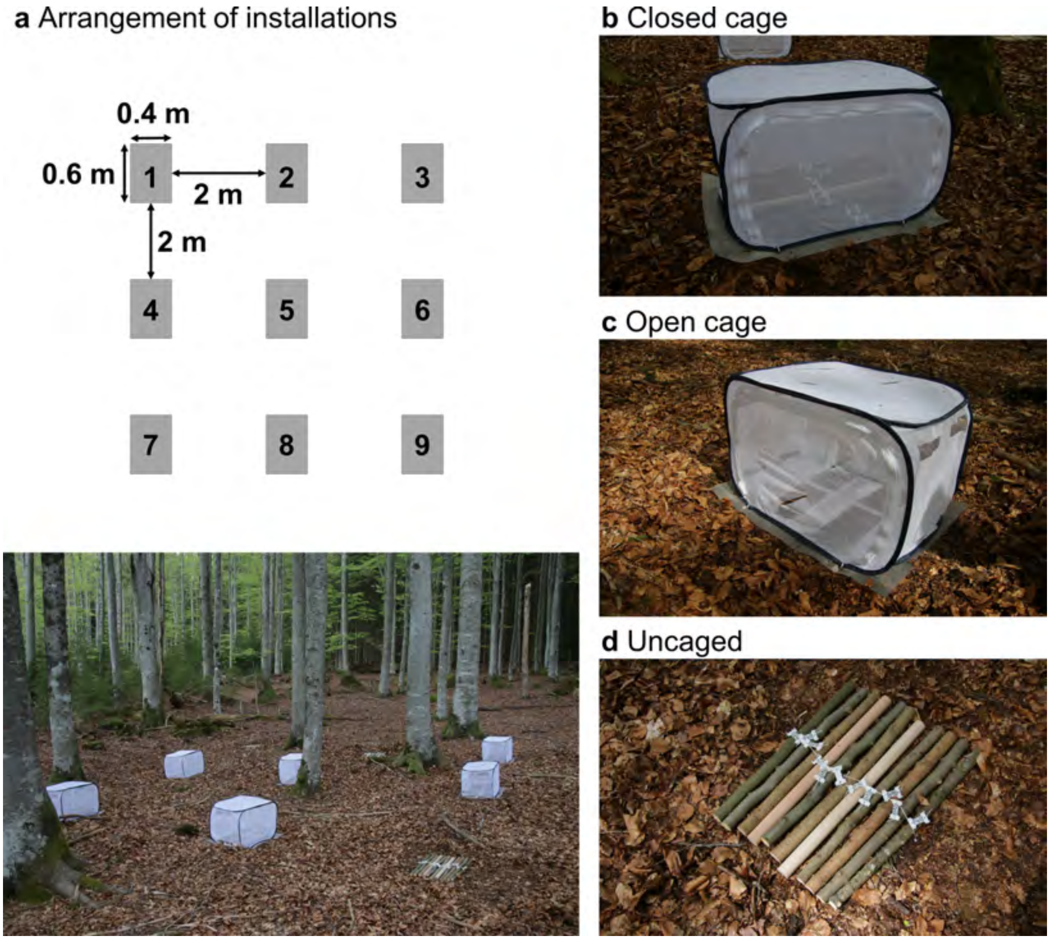
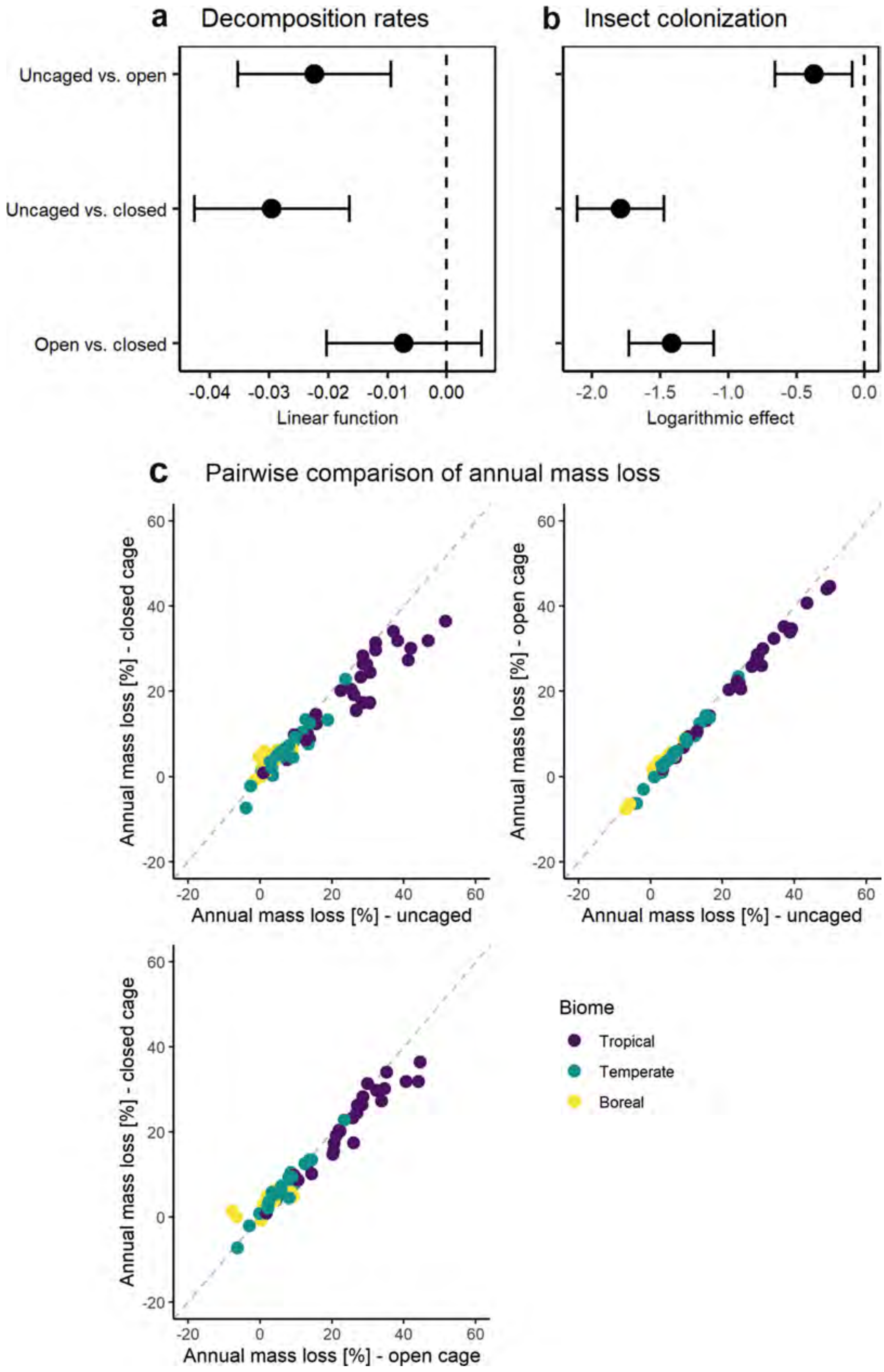


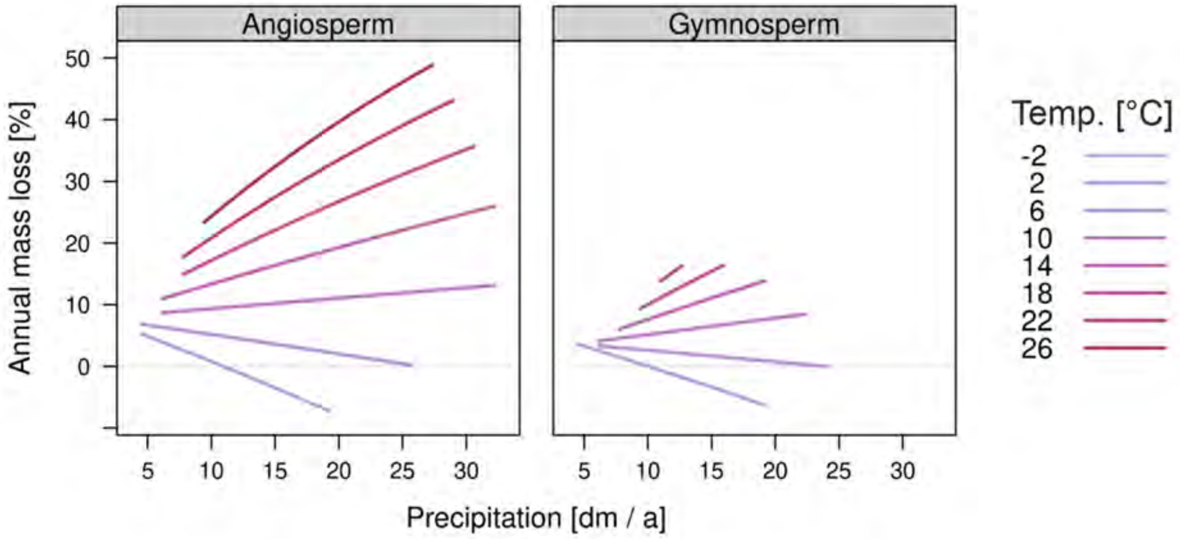
Extended Data Fig. 1 | Arrangement of installations per site and per treatment. **a**, Each site received three installations of three treatments randomly assigned to a 3 × 3 grid. **b**–**d**, Treatments included closed cages to exclude insects (**b**), open cages providing similar microclimatic conditions as closed cages but giving access to insects (**c**) and uncaged bundles of logs (**d**). Cages measured 40 cm × 40 cm × 60 cm and were made of white polyester with honeycomb-shaped meshes with a side length of approximately 0.5 mm. Open cages had four rectangular openings measuring 3 cm × 12 cm at both front sides and four rectangular openings measuring 10 cm × 15 cm at the bottom representing in total 6% of the surface area of the cage as well as a total of ten 12-cm slits at the top and long sides. All cages were placed on a stainless-steel mesh (0.5 mm mesh width), which had the same openings as the bottom side of the cages in the open-cage treatment. Photographs show the site in the Bavarian Forest National Park, Germany.



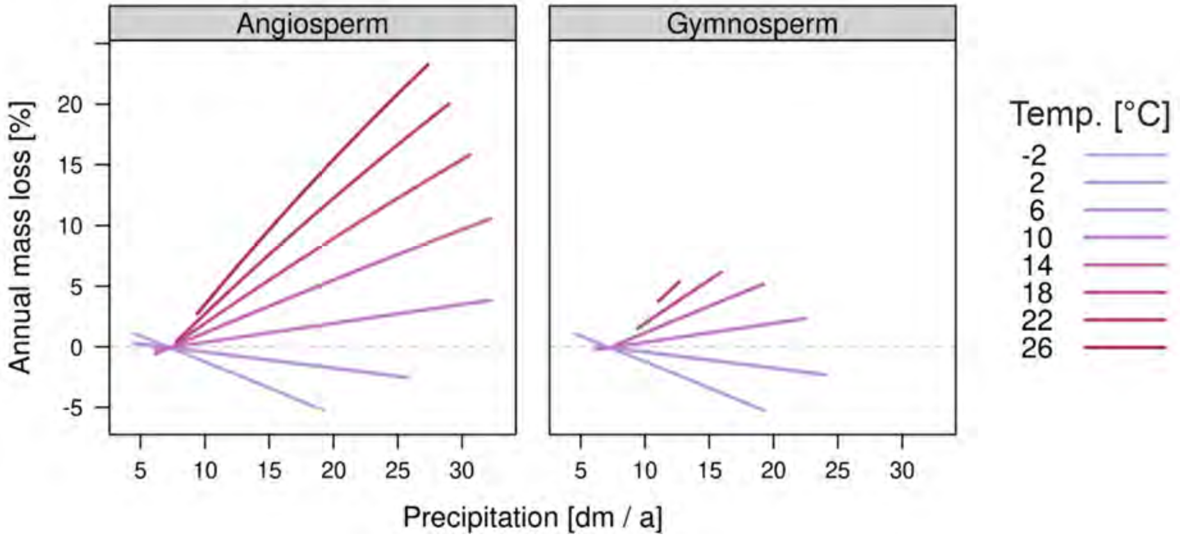
Extended Data Fig. 2 | See next page for caption.

Extended Data Fig. 2 | Effects of treatments on wood decomposition and insect colonization. a, **b**, Coefficients and confidence intervals from post hoc tests assessing all three pairwise comparisons between the uncaged, closed-cage and open-cage treatments for annual mass loss (**a**; same structure as the model shown in Table 1 based on 3,578 logs) and insect colonization (**b**; binomial model for insect presence and absence based on 3,430 logs) of wood of native tree species. The 95% confidence intervals that do not intersect the zero line (dashed) indicate significant differences. **c**, Pairwise comparison of fitted annual mass loss (%) between each of the three treatments in the global deadwood decomposition experiment. Points represent the predicted values for angiosperm species at 55 sites and gymnosperm species at 21 sites based on three Gaussian generalized linear mixed log-link models for 3,758 logs with site-specific random effects and temperature, precipitation, treatment (closed cage versus uncaged, open cage versus uncaged and closed cage versus open cage), host division, as well as their interactions, as fixed effects. In **a** and **b**, the largest differences in both response variables were observed between uncaged and closed-cage treatments. Annual mass loss was higher in the uncaged than open-cage treatment and higher in the open-cage than in closed-cage treatment, although the latter was not significant. This indicates that the open cage, despite its openings for insects, has a clearly reduced decomposition rate compared with the uncaged treatment. Insect colonization for the open cage differed significantly from both uncaged and closed-cage treatment, but was more similar to the uncaged than closed-cage treatment. This indicates that open cages were colonized by insects, but not as frequently as the uncaged treatment. Open cages thus excluded parts of the wood-decomposing insect community, which may explain the rather small difference in annual mass loss between closed cages and open cages. These results suggest that the comparison of uncaged wood versus closed cages provides a more reliable estimate of the net effect of insects on wood decomposition than the comparison of closed-cage versus open-cage treatments, which is likely to underestimate the net effect of insects. In **c**, the difference between annual mass loss in closed-cage and both treatments with insect access (uncaged and open cage) increased from boreal to tropical biomes, whereas the difference between uncaged wood and open cages hardly deviated from the 1:1 line. This indicates that the reported mass loss differences between closed-cage and uncaged treatments, as well as the accelerating effect of temperature and precipitation (Table 1), can be attributed to insects and are not an artefact of potential microclimatic effects of the cages (Supplementary Information section 1).

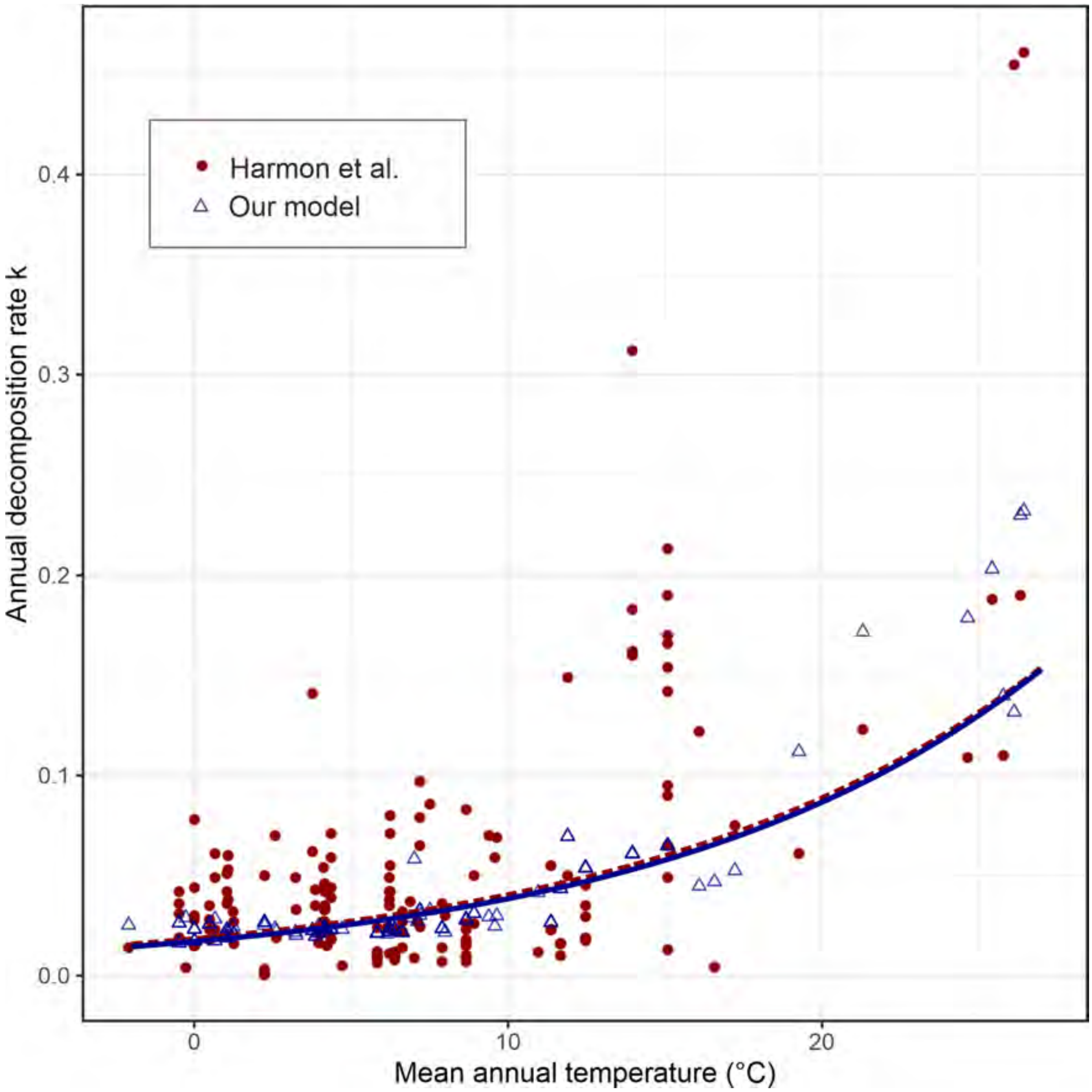
a All decomposers



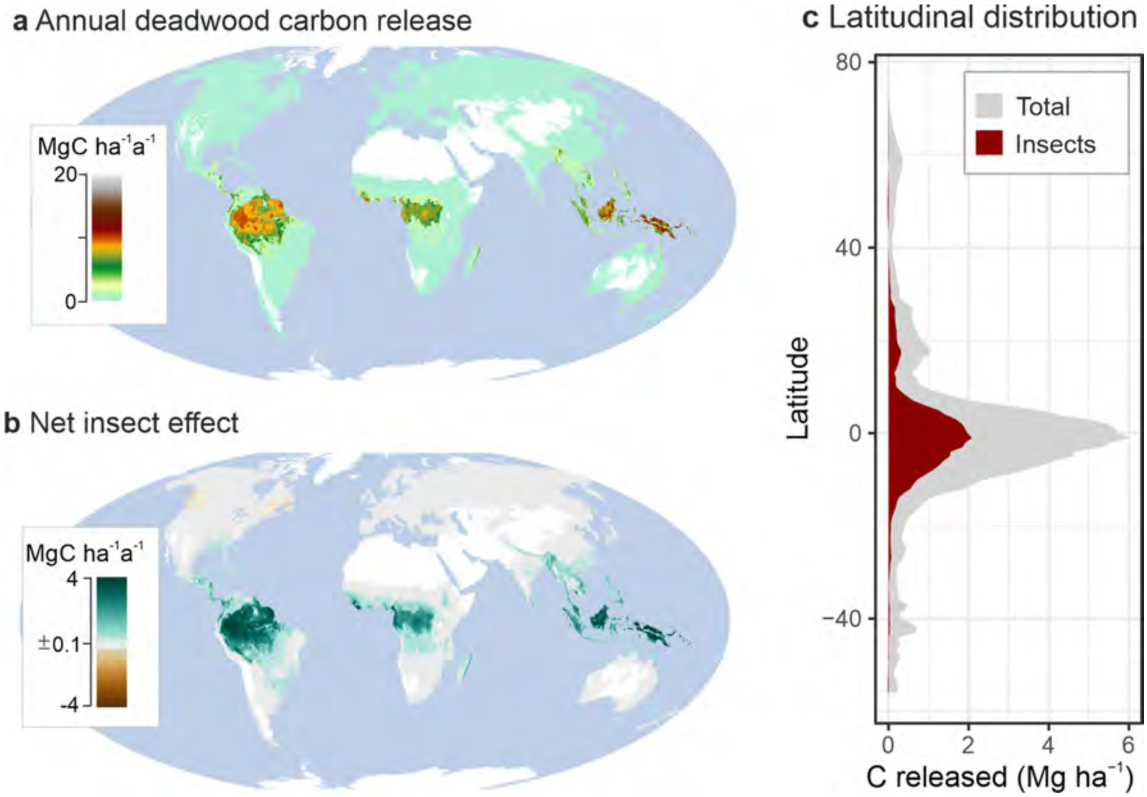
b Net insect effect



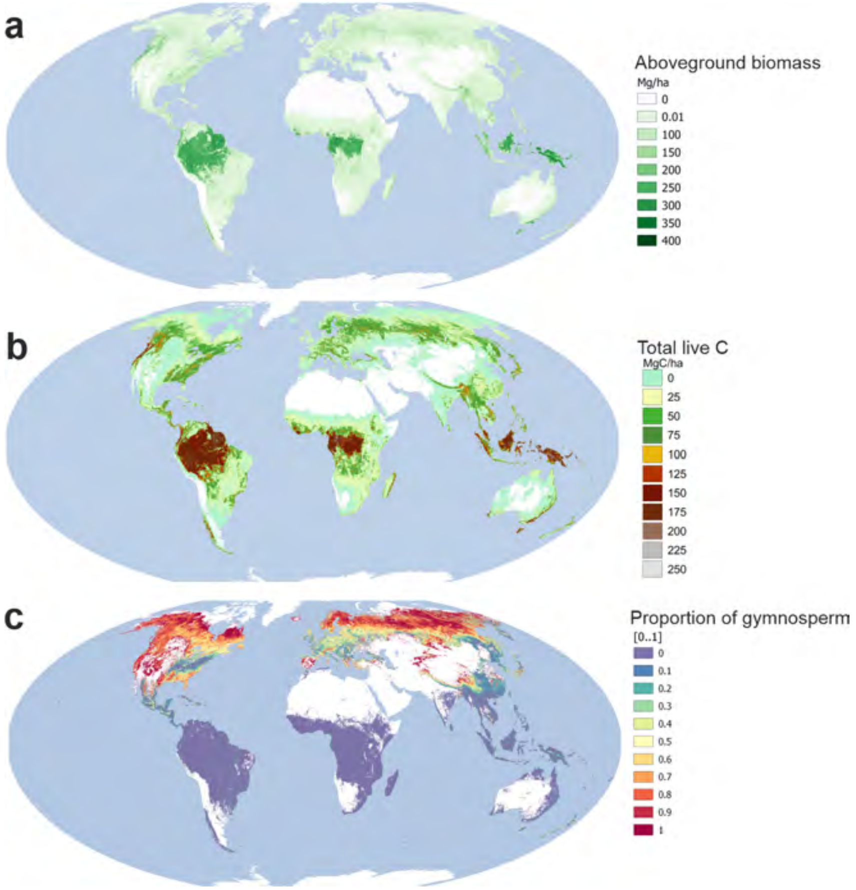
Extended Data Fig. 3 | Interaction effects of temperature and precipitation on wood decomposition. a, **b**, Predictions based on the model presented in Table 1 for annual mass loss of deadwood of native tree species (2,533 logs at 55 sites), considering all possible groups of decomposers (uncaged treatment) (**a**), and annual mass loss attributed to insects (difference in mass loss between uncaged and closed-cage treatments) (**b**), relative to temperature and precipitation. The length of the lines is limited to the gradients in precipitation covered by the sites.



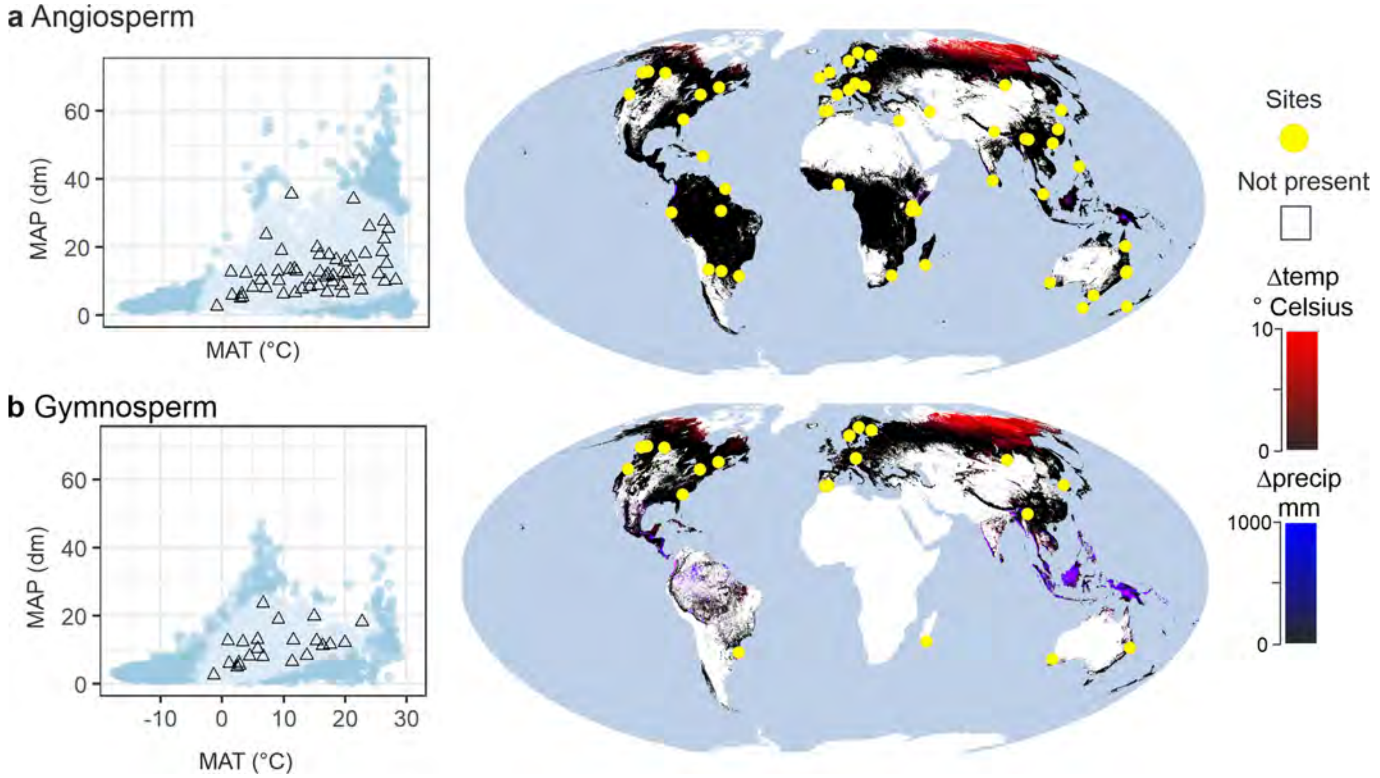
Extended Data Fig. 4 | Model evaluation against independent data. Comparison of 157 independent observations of annual deadwood decomposition rates measured for larger diameter wood in previous deadwood surveys[27] (red dots) with the predictions from our model for the same locations (blue triangles). Lines indicate the relationship between the decomposition rate and mean annual temperature from Harmon et al.[27] (red dashed line; $k = 0.0184e^{0.0787\times\text{temperature}}$) and for our model (blue line; $k = 0.0171e^{0.0812\times\text{temperature}}$). Good correspondence of both curves indicates that our models of global carbon release from deadwood provide robust estimates despite being based on experimental deadwood with a diameter of around 3 cm (for detailed discussion, see Supplementary Information section 1).



Extended Data Fig. 5 | Global deadwood carbon fluxes. a, **b**, Total annual release of deadwood carbon from decomposition including all decomposers (**a**) and annual release of deadwood carbon due to the net effect of insects (**b**). Light grey areas indicate values of ±0.1 Mg C ha^{-1} yr^{-1} and white areas are non-forest systems. **c**, Latitudinal distribution of global deadwood carbon fluxes per hectare.



Extended Data Fig. 6 | Processing steps for the global deadwood carbon map. **a**, Aboveground forest biomass (Mg ha^{-1}) aggregated to 5′ from the GlobBiom dataset. **b**, Total live carbon (Mg ha^{-1}) by extending **a** with root biomass[55] and conversion to carbon. **c**, Proportion of gymnosperm forests derived from the GLCNMO2013[59] dataset. The proportion of angiosperm cover is 1 − gymnosperm cover. White indicates non-forested area.



Extended Data Fig. 7 | Bioclimatic space for robust predictions. **a**, **b**, Climate conditions outside of the range of prediction models for angiosperm (**a**) and gymnosperm (**b**) species in climate space (left) and mapped (right). Left, dark blue points are outside of the range defined by a convex hull around the experimental sites (black triangles). Right, the colours on the maps indicate the absolute difference between the local climate and the climate used for prediction for temperature (red colour channel) and precipitation (blue colour channel) with black indicating no difference. White areas indicate that no gymnosperm or angiosperm forest, respectively, occurs there. Experimental sites are indicated by yellow dots. Temperatures outside of the range are mainly located in northeastern Siberia and northern Canada, whereas offsets in precipitation are stronger for gymnosperms in southeastern Asia, Indonesia and in the Amazon region. The land surface area not covered by our experimental data is 23.5% for gymnosperms and 17.7% for angiosperms, representing together 13.2% of the carbon stored in deadwood. These areas were included in our upscaling by mapping them to the nearest point at the convex hull in climate space.

Extended Data Table 1 | Supporting analyses of drivers of wood decomposition

Predictor	Estimate * 10^3	Std.Error * 10^3	*z*-value	*p*-value	Relative effect and 95% confidence interval
a) Standardized dowels					
Temperature (in °C - 15)	-17.918	3.922	-4.569	<0.001	0.982 (0.975 - 0.990)
Precipitation (in dm a^{-1} -13)	-6.195	4.269	-1.451	0.147	0.994 (0.986 - 1.002)
Treatment: uncaged vs. closed	-46.570	11.450	-4.067	<0.001	0.954 (0.933 - 0.976)
Temperature*precipitation	-1.597	0.502	-3.182	0.001	0.998 (0.997 - 0.999)
Temperature*treatment	-5.024	1.598	-3.144	0.002	0.995 (0.992 - 0.998)
Precipitation*treatment	-3.595	1.945	-1.849	0.065	0.996 (0.993 – 1.000)
Temperature*precipitation*treatment	-0.637	0.275	-2.320	0.020	0.999 (0.999 – 1.000)
b) Native tree species					
Temperature (in °C - 15)	-9.906	2.936	-3.374	0.001	0.990 (0.984 - 0.996)
Precipitation (in dm a^{-1} -13)	-4.343	3.268	-1.329	0.184	0.996 (0.989 - 1.002)
Host: angiosperm	-147.400	22.040	-6.688	<0.001	0.863 (0.826 - 0.901)
Host: gymnosperm	-66.540	24.460	-2.720	0.007	0.936 (0.892 - 0.982)
Treatment: open vs. closed	-6.363	5.138	-1.238	0.216	0.994 (0.984 - 1.004)
Temperature*precipitation	-0.614	0.388	-1.581	0.114	0.999 (0.999 – 1.000)
Temperature*host	5.653	1.236	4.572	<0.001	1.006 (1.003 - 1.008)
Precipitation*host	-0.251	3.189	-0.079	0.937	1.000 (0.994 - 1.006)
Temperature*treatment	-1.590	0.674	-2.358	0.018	0.998 (0.997 – 1.000)
Precipitation*treatment	-3.027	0.809	-3.742	<0.001	0.997 (0.995 - 0.999)
Temperature*precipitation*host	0.090	0.293	0.308	0.758	1.000 (1.000 - 1.001)
Temperature*precipitation*treatment	-0.486	0.102	-4.759	<0.001	1.000 (0.999 – 1.000)

a, **b**, Results from Gaussian generalized linear mixed log-link models of the relative annual mass loss of standardized wooden dowels comparing the uncaged versus closed-cage treatments (415 logs from 55 sites) (**a**) and wood of native tree species comparing the open-cage and closed-cage treatments (2,522 logs from 55 sites) (**b**). Models include the mean annual temperature and the mean annual precipitation, which were both centred and scaled, the host tree type (angiosperm versus gymnosperm; in **b** only) and treatment, as well as their two- and three-way interactions, as fixed effects and site as the random effect. Estimates and standard errors (std. error) for temperature and precipitation are transformed back to °C and decimetres per year (dm yr^{-1}), respectively. The main effects of each variable are interpretable when the remaining variables are fixed at their reference value (15°C and 13 dm yr^{-1}).

Extended Data Table 2 | Uncertainty in global carbon fluxes from the decomposition of deadwood, determined in a global sensitivity analysis

Source of uncertainty	Value range	Description	Annual deadwood carbon release		Net effect of insects	
			Uncertainty Pg	*Contribution to uncertainty (%)*	*Uncertainty Pg*	*Contribution to uncertainty (%)*
Total uncertainty			±3.22		±0.88	
Data uncertainty			±2.03	42.64	±0.59	38.22
Above-ground biomass	± 10%	Based on regional assessment against plot data for the tropical biome (GlobBiomass D17 Validation report)	±0.99	10.82	±0.29	9.73
Proportion deadwood / total C stock	± 15%	Twice the uncertainty of global stock estimates reported by Pan et al. [1]	±1.48	24.35	±0.43	21.89
Proportion of root C to total C	+10%	Based on an alternative estimate of Robinson et al.[55] (assumption iii, less conservative compared to the value given by Saugier et al.[70])	±0.90	7.47	±0.26	6.60
Model uncertainty			±1.88	28.50	±0.56	33.07
Model CI	±CI	Lower and upper bounds of model predictions with CI of fixed effects (lower and upper bounds)	±1.86	19.52	±0.16	2.06
Open:closed caged (microclimate effect)	Yes/No	Model based on data from the "*open*" instead of "*uncaged*" treatment (microclimate effect)	±0.70	2.27	±0.72	30.98
Standardized dowel	Yes/No	Model based on standardized dowels instead of native trees (tree species effect)	±1.03	6.71	±0.17	0.03
Scaling uncertainty			±1.49	26.44	±0.43	26.25
Expansion FWD	± 25%	Expansion factor to estimate deadwood pools >2cm from pools >10cm. We assumed high uncertainty as the relationship used[58] is based on only one forest type	±0.43	2.05	±0.12	1.85
Standing deadwood	slower/ faster	Fraction of standing deadwood, and assumptions for decay rate of SWD. Boreal: slower: SWD=30% of total deadwood, no decay; faster: SWD=20% of total deadwood, 50% slower. Temperate: slower: SWD=40%, 50% slower, faster: SWD=20%, decay as fast as DWD	±0.07	0.08	±0.07	0.00
Rate FWD:CWD	± 15%	The lower value assumes that the empirically determined FWD:CWD decay ratio from tropical forests[3] is applicable across the entire globe, while the upper value assumes an FWD:CWD decay ratio at the 90th percentile of the available data.	±1.48	24.35	±0.43	21.89
Climate envelope	Reduced / expanded	Reduced/expanded definition of the convex hull that limits the extrapolation to climatic conditions not covered by our experiment. Reduced: limit the convex hull to 0.5° and 0.5 dm around observed conditions, expanded: no limit to extrapolation	±0.11	0.04	±0.02	2.51

Important factors per uncertainty category were selected and allowed to vary simultaneously, resulting in a total of 4,860 analysed combinations. The uncertainty of total annual deadwood carbon released and of the net effect of insects was calculated as the s.d. over all combinations for each factor, with all other factors fixed to their default value. Similarly, the uncertainty per category was calculated over all combinations within a category, with all factors from other categories fixed to the default value. The relative contribution of each factor to overall uncertainty was derived using an ANOVA, estimating the percentage of variance explained for each factor. The contribution at the level of uncertainty categories is the sum of the respective factors in each category. Descriptions are based on refs. [1,3,55,58,70] as indicated. DWD, downed woody debris; SWD, standing woody debris.

Extended Data Table 3 | Comparison of global carbon stock estimates and results for each biome

a

	Total live carbon Pan et al.[1]	Total live carbon this study	Deadwood carbon Pan et al. [1]	Deadwood carbon this study
Boreal	53.9	60.2 (+11.7%)	16.1	17.8 (+10.5%)
Temperate	46.6	53.5 (+14.8%)	3.3	9.3 (+182.2%)
Tropical	262.1	272.7 (+4.0%)	53.6	49.4 (-7.9%)
World	362.6	386.5 (+6.6%)	73.0	76.5 (+4.8%)

b

	Annual deadwood carbon release	Net effect of insects	Deadwool carbon residence time
Boreal	0.44	0.003	40.0
Temperate	0.28	-0.009	33.3
Tropical	10.22	3.18	4.8
World	10.94	3.17	7.0

a, Global estimates of total live carbon and carbon in deadwood (>10 cm) from Pan et al.[1] compared with estimates obtained in this study (>2 cm) in Pg. Numbers in parentheses indicate the difference as a percentage. Note that Pan et al.[1] defined biomes at the country level whereas we define biomes here using the FAO Global Ecological Zones. Differences between these biome definitions are especially marked for the temperate biome, as temperate parts of Russia and Canada are included in the boreal biome in Pan et al.[1], whereas we divide Russia and Canada into boreal and temperate regions in our study. Furthermore, missing and unrealistic deadwood carbon stocks for a number of areas (specifically Japan, South Korea, China, Australia and Alaska) in Pan et al.[1] were complemented with data from the FAO Forest Assessment Report[57] in this study, which contributes to higher deadwood carbon estimates relative to Pan et al.[1]. **b**, Annual deadwood carbon release and net insect effect per biome (in Pg) and calculated residence time of deadwood carbon (years).

The Potential of U.S. FOREST SOILS to Sequester Carbon and Mitigate the Greenhouse Effect

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CHAPTER 23

The Potential of U.S. Forest Soils to Sequester Carbon

Linda S. Heath, John M. Kimble, Richard A. Birdsey, and Rattan Lal

CONTENTS

INTRODUCTION

Previous work (Lal et al., 1998; Follett et al., 2001) described the potential of U.S. cropland and grazing land soils to sequester carbon (C) and be managed to help mitigate greenhouse-gas emissions. Activities to sequester C in croplands included land conversion, land restoration, improved cropping systems, and intensified management using conservation tillage and improved water and fertility management. Lal et al. (1998) estimated that cropland soils could sequester 75 to 208 million metric tons C per year (Mt C/year) (mean = 142), while 43 Mt C/year was estimated to be emitted from production inputs. Thus, the net potential sequestration was estimated at 100 Mt C/year on a cropland area of 136.6 million hectares (Mha) (Lal et al., 1998). Activities to sequester C in grazing lands included controlling soil erosion losses, restoring eroded and degraded soils, land conversion, and improved pasture and rangeland management, which involved fertility management, planting improved species, and grazing management. The overall potential of U.S. grazing lands to sequester C ranged from 29.5 to 110 Mt C/year (mean = 69.8) during a 25-year period, with emission losses of 12.0 to 19.5 Mt C/year (mean = 15.8 Mt C/year). The net potential sequestration of grasslands was about 53.5 Mt C/year on a land base of approximately 336 Mha.

The potential to sequester C in forest soils has received little attention in the Guidelines for National Greenhouse Gas Inventories (IPCC, 1997), the Kyoto Protocol, or other studies where

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the focus has been on aboveground biomass through forest-related land-use change, such as afforestation and deforestation, or forest management (Kimble et al., Chapter 1). Yet the soil in U.S. forests contains about 60% of total forest ecosystem C (Birdsey and Heath, 1995). In terms of the dynamics of C on a global basis, attempts to balance the input and output of C have revealed a "missing sink." Houghton et al. (1998) estimated this sink at 1.8 ± 1.5 Pg C/year, some of which was likely to be contained in terrestrial ecosystems in the Northern Hemisphere, namely soils and vegetation (Pacala et al., 2001). Thus, there is a need to better understand the capacity and dynamics of vegetation or forest C sinks both above and below ground.

The purpose of this chapter is to synthesize key information from the present volume for easy reference. The main topics are the characteristics of forests and forest soils and how to measure and monitor them; C dynamics and soils processes, including the activity of soil organisms; forest management activities and their impacts on soils; and discussions of specific forest ecosystems with unique soil C dynamics or management needs. The typical managed forest in the conterminous United States is a productive, closed-canopy, temperate deciduous or coniferous forest. The soil C in boreal regions, high elevations, the arid West, wetlands, and subtropical areas, as well as urban areas and areas of agroforestry, may have distinct features, and so forests in these areas are treated separately. Finally, quantitative estimates of the potential of forest soils to sequester C are provided.

CHARACTERISTICS OF FORESTS AND FOREST SOILS

The amount of C in forests and forest soils is determined by the area of forests and the amount of C per hectare. Birdsey and Lewis (Chapter 2) present a quantitative area analysis of land use, forest management, and natural disturbance in forests. About 33% of the land area of the United States, constituting about 302 Mha, is forest. About 17% of this area is in Alaska. The total area of forestland in the conterminous United States has been stable over the last century; however, significant regional changes have occurred. The Northeast and North Central regions gained 43% and 7% in forest area, respectively, over the century, while the Pacific Coast and South Central regions both lost about 13% in forest area (Birdsey and Lewis, Chapter 2). Since 1907 deforestation has affected a cumulative area of 70 Mha of forestland; afforestation has affected a cumulative area of 62 Mha. Another major change in the latter half of the 20th century is in the area of grazed forests, which has fallen 60% or by 61 Mha. Poorly stocked forest areas declined 90% over the same period, to 1.7 Mha, due to efforts to increase stocking and to reduce uncontrolled fires. Over 64% of the forestland area in the East is in the oak-hickory (*Quercus-Carya*), maple-beech-birch (*Acer-Fagus-Betula*), oak-pine (*Quercus-Pinus*), and oak-gum-cypress (*Quercus-Liquidambar/Nyssa-Taxodium*) forest types. In the West, over 67% of the forestland area is in the pinyon-juniper (*Pinus* species *edulis, cembroides, quadrifolia,* and *monophylla-Juniperus*), Douglas-fir (*Pseudotsuga*), Ponderosa pine (*Pinus ponderosa*), and fir-spruce (*Abies-Picea*) forest types.

Whereas forests are often categorized by vegetation such as forest type, forest soils are classified using the standard Soil Taxonomy (Soil Survey Staff, 1999; Johnson and Kern, Chapter 4). Seventy percent of forests in the contiguous United States are found on four soil Orders: Alfisols, Mollisols, Inceptisols, and Ultisols. The mass of organic C in forests in the conterminous United States to a depth of 1 m is 25,780 Mt C. Approximately 25% of the C is found in Histosols. With 250 Mha of forest in the conterminous United States, the average C per hectare in soil is about 105 metric tons (t). Forest area is 52 Mha in Alaska, 0.7 Mha in Hawaii, and 0.3 Mha in Puerto Rico. Forest soil in Alaska contains about 10,430 Mt C in the first meter, forest soil in Hawaii contains 96 Mt C, and forest soil in Puerto Rico contains 33 Mt C. The estimated area by forest type in Johnson and Kern (Chapter 4) differs from the estimates of Birdsey and Lewis (Chapter 2) because Chapter 2 area estimates are based on field measurements from forest inventories, whereas Chapter 4 area estimates are based on a land classification from satellite-based imagery. The Chapter 2 area estimates are consistent with the most recent national forest inventory compilation reported by the

USDA Forest Service (Smith et al., 2001). The oak-hickory forest type contains more soil C than any other forest type in the contiguous United States, only slightly less than the soil C in the softwood forest type in Alaska (Johnson and Kern, Chapter 4). The large amount of soil C in oak-hickory is due mainly to the large area that oak-hickory covers. Forest types containing a high percentage of Histosols (and therefore high soil C per hectare) are aspen-birch (*Populus-Betula*) in the East and fir-spruce in the West, as well as white-red-jack pine (*Pinus strobus-Pinus resinosa-Pinus banksiana*), longleaf-slash pine (*Pinus palustris-Pinus elliottii*), maple-beech-birch, and oak-gum-cypress (Johnson and Kern, Chapter 4).

For context, the total stock of C in forests of the conterminous United States is 52,245 Mt C (Heath et al., Chapter 3). About 50% of the C is in the soil, 34% is in live vegetation, 8% is in the forest floor, and the rest is in standing dead trees or downed dead wood. Over the period 1953 to 1997, total forest ecosystem C increased about 155 Mt C/year on average, while the land area dropped by 3 Mha. However, Heath et al. (Chapter 3) did not include effects of land-use history on soil C, which is expected to cause C increases on the order of 20 to 40 Mt C/year. In addition, a net of 31 Mt C/year is being sequestered in harvested wood products in use and in landfills. Summing the 155 Mt C/year in forest ecosystems, plus the 31 Mt C/year in products, leads to the result that forests and forest products sequestered at least 185 Mt C/year over the period 1953 to 1997, with an additional 20 to 40 Mt C/year in soils from land-use change. Most of the 185 Mt C/year increase is due to increases in live tree biomass or the associated products produced from wood. An additional 45 Mt C/year of C, which is not counted as part of the forest sequestration because these are net estimates, is being burned for energy production (Heath et al., Chapter 3). This is a comparatively large amount that may be of interest as an offset to the burning of fossil fuels.

How do we know what we think we know about soil C? Techniques are available to measure C sequestration in forest soils (Palmer, Chapter 5). Analytical techniques for the measurement of soil C are commercially available, with sample preparation costing about $4 per sample and the median analytical technique costing $5.60 to $11 (Palmer, Chapter 5). To estimate soil C on an areal basis, bulk density and coarse rock fragment contents must be measured. To monitor total soil C change, these characteristics must also be measured over time. Careful design of the survey must include all characteristics of interest, such as forest-floor C, and plan for spatial and temporal variability. As in all surveys, a designated level of precision is met by selecting an appropriate sample size.

FOREST C CYCLE AND SOILS PROCESSES, INCLUDING THE ACTIVITY OF SOIL ORGANISMS

The factors that regulate soil properties and therefore C accumulation are climate, biota (organisms and vegetation), topography, parent material, and time (Brady and Weil, 1999; Morris and Paul, Chapter 7). Biota is the main factor that can be affected by management. Soil organisms represent only about 5% of total organic matter in forest soils (Grigal and Vance, 2000), but they control the process of transforming and decomposing soil organic C (Pregitzer, Chapter 6). Management for C sequestration in forest soils must include an understanding of soil organisms and the factors that control their growth and maintenance (Morris and Paul, Chapter 7). The potential of soil C storage in forests is great because the inputs into the soil contain compounds such as lignins, which are difficult to decompose, and forests also contain organisms that can optimize forest net primary productivity while maintaining belowground C stocks. Chapter 7 describes the major groups of soil organism and the roles that they play in forest-soil C transformations.

To understand how management can impact C cycling and storage in forests, one must also have a fundamental understanding of how C is sequestered in various pools and the mechanisms that control the flow from one pool to another (Pregitzer, Chapter 6). Most C in forest ecosystems is fixed during photosynthesis. Influencing the genetic composition of the forest and increasing the availability of resources that limit stand-level photosynthesis are the most-direct ways to increase C sequestration at the ecosystem level. However, much remains to be learned about the rest of the C cycle, including belowground allocation to coarse roots, fine roots, and mycorrhizae (Pregitzer, Chapter 6), as well as decomposition and transformation processes. A major factor controlling ecosystem C balance over decades is heterotrophic soil respiration (Pregitzer, Chapter 6). Many fundamental questions on the C cycle remain for future research.

Two C pools important in forest soils — pools that are not important in cropland and grazing land — are forest floor and downed woody debris (Currie et al., Chapter 8). The forest floor is defined as the surface organic horizon (O horizon), whose characteristics depend on climate, litter production rates, litter quality, and soil organism activities. Although the forest-floor pool was estimated to contain only 8% of C in an average U.S. forest ecosystem (Heath et al., Chapter 3), it is important as a responsive reservoir that provides much of the C ultimately stored in soil (Currie et al., Chapter 9). Downed woody debris is similar to the forest floor in that inputs to the pool occur when part of the tree dies and falls to the ground, and losses occur through decomposition and fragmentation. Results of limited studies in the United States suggest that about 10% of the mass may be lost to dissolved organic C leaching, and 25–50% may be lost from the pool due to fragmentation (Mattson et al., 1987; Currie et al., Chapter 9). Both processes have the potential to add significant quantities of C to organic C pools in mineral soil.

At the broader landscape level, disturbances are a significant component of the forest C cycle. Early in the 20th century, ecological theory adopted Clements's (1916, 1928) model of succession toward a stable climax state (Overby et al., Chapter 10). Disturbance was thought only to interrupt the development toward equilibrium. By the late 20th century, Hollings (1995) introduced the concept of nonequilibrium succession (Overby et al., Chapter 10). C accumulates in ecosystems until released by disturbance. The system then begins to reaccumulate C. Major natural disturbances are fire, insects, disease, drought, and wind. Interactions between disturbance agents complicate the study of their individual effects on forests and soil C. Overby et al. (Chapter 10) concluded that much research is needed to understand and quantify the effects that natural disturbances have on forests and forest-soil C.

DISTURBANCES AND MANAGEMENT IMPACTS ON FOREST SOILS

Disturbances and management impacts can cause mineral-soil C to increase, decrease, or remain unchanged. Land-use change has the potential for the greatest effect, due to the large changes in soil C per hectare from land conversion and because of the large area affected (Post, Chapter 12; Murty et al., 2002). Soil C is most affected when land is converted from forest to cultivation or from cultivation to forest. Studies to date on shifts from grassland to forest or forest to grassland have shown small changes. Fertilization and planting nitrogen-fixing species (Hoover, Chapter 14) have a large effect on a per-area basis, increasing the soil C density by an average of about 25%. However, the area of forest currently fertilized or planted with N-fixing species is a small percentage of the total forest. Table 23.1 is a summary of soil C information from the other chapters in this volume relating to disturbances and management impacts on U.S. forests.

Reducing soil erosion in U.S. forests would lead to only a small increase in soil C sequestration, about 0.2 to 0.5 Mt C/year (Elliot, Chapter 11). Little is known about how compaction from forest operations affects soil C (Lal, Chapter 15). Studies on wildfire or prescribed fire have produced mixed results. Generally, wildfires can affect the forest-floor C pool greatly, but there is little effect on mineral-soil C (Page-Dumroese et al., Chapter 13; Hoover, Chapter 14). Formation of charcoal in the surface soil enhances long-term C storage; however, no studies featuring estimates of C in charcoal following fire were cited (Page-Dumroese et al., Chapter 13; Hoover, Chapter 14). Individual studies have shown harvesting effects and site-preparation effects, but collectively, the results are highly variable (Johnson and Curtis, 2001). On average, there is no significant change in soil

Table 23.1 Current Forest-Soil Carbon Changes from Disturbances and Management Impacts on U.S. Forests, as Summarized from the Current Volume

Management/Event	Area (Mha)	Soil C Change	Soil C (Mt C/year)	Major Activities/Factors Potentially Affecting Soil C
Soil erosion reduction (Elliot, Chapter 11)	21[a]	Avg. annual C loss (kg/ha/year): • Forest roads: 0.12 • Forest operations: <5 • Wildfires: 5 • Landslides: 0.25	0.2–0.5	Erosion rates may be high immediately following disturbance, usually decreases rapidly; careful management will minimize soil erosion; reducing soil erosion can lead to only a small increase in C sequestration
Land-use change: forest establishment (Post, Chapter 12)	61.9 with time of establishment distributed since 1907[b]	Avg. annual C change (kg/ha/year)[c]: • Forest establishment on cropland: 338 • Grassland establishment on cropland : 332 • Species change to cool coniferous forest: 30	20, assuming afforested lands continue to be forested; over period since 1907, 805 Mt C has been sequestered.	Greatest change with forest establishment is change in vertical distribution of soil organic matter; large amount of variation in rates of actual soil organic carbon change
Wildfire effects (Page-Dumroese et al., Chapter 13)	Affects 1.6 Mha/year[b]	—	A small loss of 1 g C /m^{-2} during year of fire is equivalent to 0.016 Mt C/year	Most of C in forest floor would likely be destroyed, while long-term mineral soil losses are small; experimental studies should monitor forest floor and soil C separately
Fire suppression (in long term can lead to species composition changes) (Page-Dumroese et al., Chapter 13)	—	—	—	Example: pine stands have a greater C proportion in surface mineral soil while mixed fir/pine have more C in forest floor and down wood
Harvesting and site prep Prescribed fire and wildfire Fertilization and N-fixing species (Hoover, Chapter 14)	4.0/year, 1980–1990[b] 1.6/year, 1988–1997 No estimate	Generally not significant Mixed results Positive effect on soil C, avg. about 25% increase	—	Experiments need to be designed to measure total carbon changes, not just percent C; for harvesting and fire, results are highly site-specific, and time since activity greatly affect results; most increases to fertilization or N-fixing range between –10 to 60%, perhaps averaging about 25% (based on estimates in Johnson and Curtis, 2001)
Compaction (Lal, Chapter 15)	No estimate	Studies are needed	—	In long-term, soil compaction may have negative impacts on biomass, soil carbon, and productivity; in short-term, compaction may increase soil organic carbon density because of an increase in mass of soil per unit volume
Land-use change: Conversion from forest to agriculture (Murty et al., 2002)	70.4, with time of establishment distributed since 1907[b]	Conversion to cultivated land, loss of 20% for soils sampled to more than 45 cm Conversion to uncultivated grassland, no change on average	6.7 emitted, assuming half of area loss is converted to cultivated land[d]	Rapid initial loss, reaching a new equilibrium within 5–10 years

[a] Based on 7% of U.S. forests in a disturbed condition and total U.S. forestland of 302 Mha.
[b] Birdsey and Lewis, Chapter 2.
[c] Post and Kwon, 2000.
[d] Based on average 105 t/ha soil carbon density from Heath et al. (Chapter 3), a 20% change over 10 years, and area loss of 6.4 Mha/year in 1988–1997 (Birdsey and Lewis, Chapter 2).

C from these disturbances. However, specific studies have shown large increases or large losses for similar treatments. This suggests that results are highly site-specific and may also indicate that the differences in methodology make study comparisons difficult.

U.S. FOREST ECOSYSTEMS THAT HAVE UNIQUE SOIL C DYNAMICS OR MANAGEMENT NEEDS

Most of the forestland in the United States is in the temperate zone and is occupied by coniferous or deciduous trees that are relatively productive. Most of the chapters in the book were written to address these typical forests. However, there are some forest ecosystems that have unique soil C dynamics or that need special management considerations or both, and specific chapters were written for these areas. Some forest ecosystems of special interest with respect to C sequestration are boreal forests (Hom, Chapter 16), high-elevation forests (Bockheim, Chapter 17), arid and semiarid forests of the Interior West (Neary et al., Chapter 18), wetland forests (Trettin and Jurgensen, Chapter 19), forests managed under agroforestry (Nair and Nair, Chapter 20), urban forests (Pouyat et al., Chapter 21), and tropical forests (Silver et al., Chapter 22). The topic of grazed forests and soil C was also considered to be of interest, but this is an area requiring further research (and perhaps more researchers).

Three forest ecosystems — boreal, high-elevation, and arid and semiarid Interior West — tend to be of lower-than-average productivity (Table 23.2). Fire is a major disturbance in forests of the boreal region and the Interior West. Climate has a major influence in all these forests, because the climate tends to be marginal for tree growth, and small climatic changes can have noticeable effects. The soils in boreal forests are high in organic matter. The focus in these soils is not to increase C as much as to retain existing C. Climate change is thought to be the main factor that may affect these forest soils in the future. The dry forests of the Interior West represent a special case due to the cumulative effects of fire suppression and overgrazing. Downed wood and forest-floor C pools have significantly increased, increasing total ecosystem C reserves (Neary et al., Chapter 18). Fires in areas with such high fuel loads may be so severe as to affect soil C. Mechanical incorporation of organic material into soils is limited due to soil, physical, and institutional constraints (Neary et al., Chapter 18). Neary et al. (Chapter 18) conclude that there are virtually no management opportunities for increasing C sequestration in these forests. The most fruitful management activities are probably those that can prevent the forests from losing C if the climate changes. Almost any management activities chosen to return the forests to health will probably involve reducing the high wood-fuel loads in the forests, resulting in C emissions.

Tropical and subtropical forests store very little of the soil C in the United States because of their small area (Table 23.2). However, they offer unique opportunities for scientific study because of their variety and the fact that changes occur more rapidly in tropical forests (Silver et al., Chapter 22). Wetland forests (Trettin and Jurgensen, Chapter 19) comprise less than 10% of forestlands in the United States, but their soils are rich in C. Protecting current C stores or restoring C that has been lost due to land-use change could be substantial. Managing these areas should be explored further. Harvesting and drainage studies have had mixed results.

Two other managed forest systems that are not typically considered for management but that may provide sequestration opportunities are urban forests and forests managed for agroforestry. Agroforestry is the deliberate growing of trees on the same unit of land used for agricultural crops or animals. Nair and Nair (Chapter 20) estimated that all agroforestry practices in the United States have the potential to sequester 90 Mt C/year for a limited number of years (Table 23.2). Forests of urban areas, which are increasing in size, also may provide opportunities to increase soil C, particularly on lawns that are highly maintained for recreational use (Pouyat et al., Chapter 21). Table 23.2 is a summary of the types of forests, their areas, soil C statistics, and possible activities or characteristics related to the unique forest ecosystems discussed in this section.

Table 23.2 Summary of Forest-Soil Carbon Information Currently Available for U.S. Forest Ecosystems Having Unique Soil C Dynamics or Management Needs, as Summarized from the Current Volume

Specific Forest Ecosystem/Chapter	Area (Mha)	Soil C density (Mt/ha, 1-m depth)	Soil C (Mt C)	Major Activities/Factors Potentially Affecting Soil C
Boreal forest (Hom, Chapter 16)	47.1[a]	385[b]	18,133	Forests are of low productivity and soils contain large amounts of C; thick forest floors serve as insulation, maintaining cold soil temp.; climate is a key driver in this system Fire is a major disturbance; no experimental studies are cited; implication is fire causes increases in soil temperature, resulting in increased decomposition
High-elevation forest (Bockheim, Chapter 17)	19.0	220 ± 20.2 (range 20.8–860)	38–46	Soil C affected most by small burrowing animals Large amounts of nutrients discharged through snowmelt Climate likely to be important to C dynamics
Arid-semiarid Interior West forest (Neary et al., Chapter 18)	73.4	25–136, depending on vegetation 78 for woodland	—	Erosion can be major factor in soil loss Grazing at low to moderate levels may increase soil C; high intensity grazing may decrease Prescribed fire — no studies Climate key driver to soil C inputs Wildfire suppression and overgrazing has significantly increased FF and down wood C pools; must lose carbon for forest health Virtually no opportunities for increasing C sequestration in these ecosystems
Wetland forest (Trettin and Jurgensen, Chapter 19)	26.4	457	12,100	Land-use conversion results in a substantial loss DOC is principal pathway for hydrologic C losses Restoration may be major activity to increase C Water management or drainage biggest factor contributing to losses in short term; mixed results in long term Harvesting contributes to losses in short term; mixed results in long term
Agroforestry (Nair and Nair, Chapter 20)	—	—	—	Alleycropping could result in sequestering 73.8 Mt C/year (all C. not just soil) on 80 Mha Silvopasture may result in 9.0 Mt C/year (all C. not just soil) on 70 Mha Windbreaks on 85 Mha may sequester 4.0 Mt C (all C. not just soil) Riparian buffers, short-rotation woody crops may sequester 2.0 Mt C/year
Urban forest (Pouyat et al., Chapter 21)	28.1, with tree cover of 27%	Residential — 155 Undisturbed — 93 Other — 52 U.S. — urban avg. 82	—	Direction of change in urban soils depend on initial SOC status of native soil Highly maintained recreation use lawns have 10.3 t/ha more soil C than minimally maintained soils Atmospheric pollution effect unknown Invasive species such as Asian earthworm altering decomposition rates
Tropical/Subtropical forests (Silver et al., Chapter 22)	3.2	Hawaii — 91 (depth 0–25 cm) Puerto Rico — 75 (depth 0–25 cm) Land-use differences in PR only (0–50 cm): Forestland — 143.5 Cropland — 85.1 Grassland — 86.8	—	Peat subsidence from soil drainage; takes 100 years to form 30 cm of peat, but 10 years to oxidize due to excess drainage SOC pools may increase with stand age after reforestation; however, limited number of studies indicate soils previously under pasture may lose carbon Land use greatly affects soil C, with forests featuring higher soil C density

[a] Estimated from forest cover of aspen-birch and softwoods in Alaska in Johnson and Kern (Chapter 4).

[b] Estimated from soil carbon densities and areas of aspen-birch and softwoods in Alaska in Johnson and Kern (Chapter 4).

Table 23.3 Estimated Potential Forest-Soil Carbon Changes Resulting from Management of U.S. Forestlands for Increased Forest-Soil Carbon

Activity	Area (Mha)	Rate of C Sequestration in Soil (kg/ha/year)			Quantity Sequestered (Mt C/year)		
		Low	Medium	High	Low	Medium	High
All forest management					**24.5**	**56.4**	**103.2**
Regeneration	59.7	70	223	419	4.2	13.3	25.0
Fertilization	20.0	875	1749	3061	17.5	35.0	61.2
Restoration of degraded lands (mine reclamation)	1.0	89	487	1295	0.1	0.5	1.3
More partial cutting/less clearcuts	1.5	0	448	1195	0	0.7	1.8
Lengthen rotations	0.7	0	448	1195	0	0.3	0.9
Soil erosion reduction	21.0	9.5	16.7	23.8	0.2	0.35	0.5
Fire management	?[a]	?[a]	?[a]	?[a]	?[a]	?[a]	?[a]
Manage to increase soil C	125	20	50	100	2.5	6.3	12.5
Land-use change					**7.5**	**26.2**	**51.4**
Increase afforestation	10.0	0	338	676	0	3.4	6.8
Reduce deforestation	0.8	1740	2367	3461	1.3	1.8	2.6
Past afforestation continuing to accrue C[b]	62.0	100	338	676	6.2	21.0	42.0
Agroforestry					**16.9**	**22.3**	**28.2**
Alleycropping	80	173	230	288	14.0	18.5	23
Riparian buffers	—	—	—	—	0.4	0.5	0.6
Silvopasture	70	25	33	41	1.8	2.3	3.3
Windbreaks	85	8	12	15	0.7	1.0	1.3
Urban forest					0	1.0	3.0
Urban management	1	0	1000	3000	0	1.0	3.0
Total net C sequestration					**48.9**	**105.9**	**185.8**

Note: The estimates in Table 23.3 include only the C sequestered in soil. The amount sequestered in tree biomass and forest-floor pools can be four to six times the amount in soil C.

[a] Impact of management activities on wildfire are not clear.

[b] Although past afforestation is not an activity that can now be influenced, it is included here as a contribution to total potential soil carbon changes.

THE POTENTIAL FOR C SEQUESTRATION IN FOREST SOILS

In Chapter 1, Kimble et al. cited a study that offered a potential rate of C gain in temperate forest soils of 0.53 t C/ha/year (IPCC, 2000). Based on a productive forestland base of 204 Mha, the potential for C sequestration in forest soils of the United States is 108.1 Mt C/year. We constructed a table of activities and their potential forest C sequestration based on the information presented in this book, along with information from other forest-soil studies and two books on sequestering C in cropland and grazing land soils. The results are displayed in Table 23.3.

The activities are summarized under four main headings: forest management, land-use change, agroforestry, and urban forest management. A low, medium, and high average rate of soil C sequestration is presented to help convey the uncertainty of the estimates. There is also uncertainty in the area of land that will be affected by the activity; however, that type of uncertainty is not included here. The potential net C sequestration in soils ranges from 48.9 to 185.8 Mt C/year, with an average of 105.9 Mt C/year. Generally, the medium rates of C sequestration are lower than the 0.53 t C/ha/year cited from previous studies. This is due to the lower rates of C sequestration cited from chapters in this volume. Past studies, however, have not been designed specifically to increase soil C. Activities designed with a goal of increasing soil C should result in higher soil C sequestration rates.

Forest management activities have the potential to sequester the most soil C, mostly from fertilization, regeneration, and managing specifically to increase soil C. Note that we have not accounted for emissions that will be associated with fertilization. Question marks are shown for fire management because, based on current information about fire, it is unclear how much change, if any, will occur due to fire management. A modest afforestation program sequesters about half the total amount of the forest management activities listed, and agroforestry activities sequester about the same amount.

CONCLUSIONS

Forests of the United States sequestered a net annual average of 155 Mt C/year over the period 1953 to 1997 in biomass and aboveground mass. Sequestration into wood in products and landfills contributed an additional 31 Mt C/year (Heath et al., Chapter 3). Average soil C increases from land-use change were at least 20 Mt C/year. An average of 45 Mt/year of C in harvested wood was burned for energy or converted to an energy source, with the potential of substituting for the burning of fossil fuel.

Forest soils have not been intentionally managed for C sequestration previously. A number of processes, such as fire, deforestation, and climate change, can cause soils to emit CO_2 or methane. Some management activities, such as erosion control, managing forest ecosystems to minimize loss of C from fires, and designing silvicultural operations to minimize emissions, increase net C sequestration by reducing potential emissions. Other management activities, such as fertilization and planting nitrogen-fixing species, intensively managing urban forest soils for recreational uses, and adopting a management system like agroforestry, increase C sequestration in both soils and biomass. With many management activities, it is unclear whether C will be sequestered or emitted. Literature reviews are often mixed, indicating that results may be site-specific and dynamic. Thus, future experiments need to be carefully designed to produce useful information about C sequestration.

The potential net C sequestration in forest soils ranges from 48.9 to 185.8 Mt C/year, with an average of 105.9 Mt C/year. Forest management activities have the potential to sequester the most soil C, mostly from fertilization, regeneration, and managing specifically to increase soil C. A modest afforestation program sequesters about half the total amount of the forest management activities listed, and agroforestry activities sequester about the same amount. These potential estimates do not include the amount sequestered in tree biomass and forest-floor pools. Considering that the nonsoil C sequestration may be 4 to 6 times the amount in soil C, the potential average total C sequestration in U.S. forests may be 420 to 630 Mt C/year.

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Title: Fixing a snag in carbon emissions estimates from wildfires

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Abstract

Wildfire is an essential earth-system process, impacting ecosystem processes and the carbon cycle. Forest fires are becoming more frequent and severe, yet gaps exist in the modeling of fire on vegetation and carbon dynamics. Strategies for reducing carbon dioxide (CO_2) emissions from wildfires include increasing tree harvest, largely based on the public assumption that fires burn live forests to the ground, despite observations indicating that less than 5% of mature tree biomass is actually consumed. This misconception is also reflected though excessive combustion of live trees in models. Here, we show that regional emissions estimates using widely-implemented combustion coefficients are 59-83% higher than emissions based on field observations. Using unique field datasets from before and after wildfires and an improved ecosystem model, we provide strong evidence that these large overestimates can be reduced by using realistic biomass combustion factors and by accurately quantifying biomass in standing dead trees that decompose over decades to centuries after fire ('snags'). Most model development focuses on area burned; our results reveal that accurately representing combustion is also essential for quantifying fire impacts on ecosystems. Using our improvements, we find that western U.S. forest fires have emitted 232 ± 62 Tg CO_2 (~half of alternative estimates) over the last 15 years, which is minor compared to 4,364 Tg CO_2 from fossil fuels across the region.

Introduction

Temperate forests of the western US are significant carbon stocks (Buotte et al., 2019; Pan et al., 2011) and include some of the highest carbon density forests on Earth (Hudiburg et al., 2009). Increasing forest fire activity threatens these carbon stores in parts of the region because larger burn areas can lead to more tree mortality (Abatzoglou & Williams, 2016;

Hicke, Meddens, & Kolden, 2016; Westerling, Hidalgo, Cayan, & Swetnam, 2006). However, contemporary CO_2 emissions to the atmosphere from fire are often significantly exaggerated because of public and policy-maker misconceptions that forests commonly “burn to the ground” during fire and that mortality equals emissions (Figure 1) (California, 2018; DNR, 2018; Mater, 2017; USA, 2019; Zinke, 2018). The reality is instead negligible stem combustion of live, mature trees (i.e. <5%; Figure 2) followed by gradual decomposition over years to centuries (Campbell, Donato, Azuma, & Law, 2007; Law & Waring, 2015). Modeled estimates of fire emissions reinforce public misconceptions, as tree mortality is often mistranslated into 30-80% of tree carbon emitted immediately (Wiedinmyer & Neff, 2007), and is in conflict with observations (Lutz et al., 2017). It is important to rectify overestimates because governments are currently using mortality and emissions estimates from fire to inform land management decisions intended to mitigate climate change (California, 2018; DNR, 2018; Fears & Eilperin, 2019; Nunez, 2006; Oregon, 2005; UNFCCC, 2015; USA, 2019), emphasizing the need for model improvement using field observations.

While modeling research focuses primarily on improving representation of area burned due to the availability of validating satellite products (Hantson et al., 2016; Thonicke et al., 2010), it is critical to recognize that simulations can generate inaccurate estimates of combustion dynamics through a combination of 1) unrealistic combustion coefficients (i.e. the biomass fraction that burns), and 2) misrepresentation of forest biomass (i.e. carbon) pools. Models use assumed fractions of biomass combusted (combustion coefficients) in fire and apply that to the biomass in the area burned. These default combustion coefficients overestimate pool combustion when they exceed ranges of observed combustion across live and dead pools, effectively simulating events where forests “burn to the ground”.

The largest discrepancies between modeled and observed combustion of aboveground biomass exist for live, mature trees, which are the dominant pool of aboveground carbon across western U.S. forests (Ghimire, Williams, Collatz, & Vanderhoof, 2012; Hudiburg et al., 2009; Wilson, Woodall, & Griffith, 2013). Default values for live tree bole (stem) combustion can range from 30-80% (S1 and S2) in high severity events, but post-fire observations in the western US indicate actual combustion is nearly nonexistent for mature trees in fire-prone ecosystems (Campbell, Alberti, Martin, & Law, 2009; Campbell, Fontaine, & Donato, 2016; Lutz et al., 2017). Field experiments show that there is inadequate prolonged heat to facilitate combustion of live tree stems, even at the highest fire intensities (Smith et al., 2016; Sparks et al., 2017).

Most models also lack standing dead tree carbon pools (snags; Table S2), essential for representation of forests in the context of disturbance and mortality (Edburg et al., 2012). High-severity fires can kill live trees, which become snags and the dominant stock of aboveground carbon in burned areas (Campbell et al., 2007) (Figure 1d and Figure 2). When trees die in a ‘no snag’ model, the wood instead transfers to the forest floor, becoming downed-woody debris (Figure 1c). In drier climates, snags decompose at slower rates than downed-woody debris (Wirth, Gleixner, & Heimann, 2009), producing relatively slow emissions over decades rather than acute, large pulses through combustion. Further, biomass location matters for reburn combustion (Campbell et al., 2007; Ghimire et al., 2012); simulating snags as downed-woody debris facilitates higher rates of combustion in subsequent fires.

Generally, model fire severity is defined by the amount of biomass killed and consumed. Representation of combustion in models varies from a single severity (“static severity”, e.g. CLM 5.0; (Lawrence et al., 2018)) to a range from low-to-high (“variable severity”; e.g. LANDIS-II (Sturtevant, Scheller, Miranda, Shinneman, & Syphard, 2009); Tables 1, S1 and S2). These dynamic coefficients are either ‘categorical’ or calculated through fire sub-models that largely depend on fuel moisture and tree or woody debris size class (Table S2). Default mortality and combustion coefficients can be ‘parameterized’ to be more in line with observations, however this is often not done, especially at large scales (Liang, Hurteau, & Westerling, 2018; Wiedinmyer & Hurteau, 2010; Buotte et al, 2018) (Table S6, S7, and S8); modeling experiments instead often rely on restricting predicted burn area or fire occurrence to achieve realistic combustion (Hudiburg, Law, & Thornton, 2013; Hudiburg, Luyssaert, Thornton, & Law, 2013). There is also large variation in the biomass pools represented, with a persistent absence of snags. Even models that include dynamic combustion coefficients (e.g. LPJ-GUESS-SPITFIRE) or variable severity (e.g. LANDIS-II) can overestimate emissions because the rate at which standing wood becomes downed wood is too high without a snag pool (Figure 1c).

In this study, we compare a range of default combustion coefficients and forest structure representations of regional-to-global-scale models with observation-based combustion coefficients and a newly implemented model snag pool. Our observation-based refinements utilize carbon stock datasets that span fire events, including new, detailed field observations from the 2013 Rim Fire in California (Lutz et al., 2017). We also simulate post-fire carbon cycle dynamics using an improved version of the globally-recognized biogeochemical model DayCent (Hudiburg, Higuera, & Hicke, 2017; Parton, Hartman, Ojima, & Schimel, 1998) through addition of snag pools with varying combustion, decomposition, and fall rates

(Figure S1). We then estimate 2000 - 2016 fire emissions across the western US with our improved methods.

Materials and Methods

We calculated emissions from forest combustion in the western US states using site observations, the MTBS burn perimeter database, and ecosystem modeling. Mortality and combustion coefficients were generated from plot data collected before and after fire in the region and from commonly-used models. We developed a modified version of DayCent (Straube et al., 2018) that introduces a snag pool to improve representation of post-disturbance ecosystem structure and fluxes. DayCent was also used to simulate commonly used model combustion coefficients and mortality transfers in both snag-free and snag-enabled versions. Finally, we estimated recent western U.S. forest emissions (2000-2016) for the same range of combustion and pool structures using forest inventory derived plot biomass carbon estimates combined with the MTBS burn perimeter and severity database (Eidenshink et al., 2007).

Fire combustion coefficients from the 2013 Rim Fire were calculated using the Yosemite Forest Dynamics Plot (YFDP; CA; (Lutz, Larson, Swanson, & Freund, 2012)) dataset. The YFDP (37.77° N, 119.82° W) is part of the Smithsonian ForestGEO network of spatially-explicit monitoring plots (Anderson-Teixeira et al., 2015). YFDP is a carbon-dense, mixed-conifer forest, where live trees contained ~70% of aboveground biomass pre-fire (Table 1, Table S4). The YFDP (800 m × 320 m) was divided into ten, 160 m × 160 m quadrats, and pre-fire and post-fire aboveground carbon pools were calculated for each quadrat (Table 1, Table S3 and S4). The plot was burned in an unattended backfire set by Yosemite National

Park to check the advance of the Rim Fire (Lutz, Larson, & Swanson, 2018; Lutz et al., 2017).

At plot inception (2009 – 2010), all trees were identified, mapped, and tagged. Snags were measured as to height, diameter, top diameter, and decay class. Shrub patches were delineated as polygons and shrub biomass was calculated by plot-specific allometric equations (Lutz et al., 2014). Due to the 113-year period of fire exclusion (Barth, Larson, & Lutz, 2015), herbaceous cover was *de minimus*. Each year pre-fire (2011-2013), trees were visited to ascertain their status in May-June, and therefore the 2013 survey provided a comprehensive inventory of standing stems. In May 2014, we performed the post-fire survey, noting tree death, whether tree canopies were scorched or combusted, and measuring dimensions of partially combusted snags.

In 2011 and 2014, surface fuels were measured with 1,600 m transects following the methods of (Brown, 1974) with additional data taken on large woody debris (1000-hour fuels, ≥10 cm diameter). Live biomass was calculated using the methods of (Chojnacky, Heath, & Jenkins, 2013). Snag biomass was calculated using the same equations as when trees were killed by fire when needles were only scorched. Pre-fire biomass of snags was calculated as the mass of the bole only, calculated as a conic frustum.

Combustion estimates were also used from published studies in mature Oregon forests. (Campbell et al., 2007; Campbell et al., 2016; Meigs, Donato, Campbell, Martin, & Law, 2009) (Figure 2, Table 1). Observations from the 2002 Biscuit Fire showed that live tree combustion was limited primarily to canopy combustion and bark scorching, resulting in a maximum 7% mature tree combustion at high (stand-replacing) severity. These datasets also contained reburned plots that burned 15 years earlier in the 1987 Silver Fire. The authors did not find any significant differences between the combustion coefficients of the aboveground pools in the reburn versus the initial burn; however because significantly more of the carbon was in snag, downed wood, or small diameter tree pools, more aboveground carbon did combust.

Simulations were performed using a modified version (developed by the authors) of the biogeochemical model DayCent (Chen et al., 2016; Straube et al., 2018) that introduces standing dead pools and fluxes. DayCent is the daily time step of CENTURY, simulating fluxes of carbon and nitrogen between the atmosphere, ecosystem, and soil (for further model description see Fig S1). Our modified DayCent now incorporates standing dead pools of leaves, fine branches, and large wood into the forest sub-model, as well as accompanying fluxes of carbon and nitrogen involved in both background senescence and prescribed fire and harvest events (Figure S1). Fluxes in and out of standing dead pools are governed by inputs from death of live pools, fall rates of standing dead material, decomposition, photodegradation, and removal by harvest or fire. Attached dead leaves that fall to the ground are partitioned into surface structural and metabolic litter. When standing dead wood falls it becomes coarse and fine woody debris. Live and dead material involved in fire events may now be returned to the system as charcoal.

Simulations were performed for each of the combustion and mortality parameter sets (Table S5) extracted from the YFDP 2013 Rim Fire, 2002 Oregon Biscuit Fire, and additional regional datasets of partial aboveground combustion (e.g. Fahnestock & Agee, 1983; Kauffman & Martin, 1989; Knapp, Keeley, Ballenger, & Brennan, 2005; Meigs et al., 2009). DayCent pre-fire carbon pools and fluxes were parameterized to the 2011 and 2013 carbon stocks of the YFDP (Table 1, S3 and S4; (Lutz et al., 2012)). Model spinup (2000 years) was based on a pre-modern fire return interval of 29 years followed by 120 years of no fire, consistent with historical park records. Site soil characteristics were extracted from SSURGO (NRCS, 2010). Site climate (temperature and precipitation) was based on location data from PRISM (Daly, Taylor, & Gibson, 1997) for 1981-2017. Post-fire simulation periods in model experiments were driven with historical climate conditions. Mortality proportions were based on fire severity mortality classes (Campbell et al., 2016; Meigs et al., 2009) comparable to the mortality in the "variable-severity" model (below), facilitating comparison. Mortality classes include 0-10%, 10-50%, 50-90%, and 90-100 % for very low, low, moderate, and high severity fire, respectively.

DayCent was also used to simulate default parameter sets from the Community Land Model v 5.0 (CLM; (Lawrence et al., 2018; Oleson et al., 2013)) and Landis-II with the Net Ecosystem Carbon and Nitrogen Succession (NECN) and Dynamic Fuels & Fire System (Sturtevant et al., 2009) (Scheller et al., 2007) (Table S6, S7, and S8). These two models represent the range of coefficients and severities used by most other fire-enabled ecosystem, forest landscape, and dynamic vegetation models (Table S1 and S2). In our results, CLM and Landis-II default parameters respectively inform our "static" and "variable" severity scenarios (combustion and mortality). In total, we performed 18 scenario simulations of the YFDP representing the range of fire severity, pool combustion, and mortality transfer assumption scenarios.

CLM is the land model of the Community Earth System Model (CESM) and simulates the fluxes of energy, water, chemical elements, and trace gases between atmosphere, plants, and soil. As the land-model component of CESM, CLM is a globally utilized model in the effort to explore land-climate feedbacks, and has been used to research forest-climate interactions throughout the western United States (Buotte et al., 2019; Hudiburg, Law, et al., 2013; Hudiburg, Luyssaert, et al., 2013). During fire events, CLM employs single severity and mortality. Combustion is therefore governed by burn area. CLM first combusts litter, coarse woody debris, and live trees, and then transfers non-burned tree biomass to dead pools (Table S8).

Landis-II is a forest landscape model simulating growth and succession of tree species and age cohorts. Landis-II with NECN (derived from CENTURY/DayCent) is used to explore the potential effects of evolving climate, disturbance regimes, and management on ecosystem structure and composition. During a grid cell fire event, species cohort mortality is determined as a product of fire severity and species tolerance, with up to 100% of species cohorts killed and mortality occurring as death of all cohorts below a variable percentage of species longevity. Fire reduction parameters determine emissions and specify reduction of dead wood and litter after the above mortality scheme kills and deposits biomass on the forest floor in the same time step (Tables S6 and S7). We calculated Landis II equivalent biomass mortality estimates for the YFDP dominant stand species (White fir and Sugar pine).

Western U.S. carbon stocks were calculated from over 80,000 forest inventory plots (FIA) containing over 2.5 million tree records in the region following methods developed in previous studies (Hudiburg et al., 2009; Hudiburg, Law, Wirth, & Luyssaert, 2011; Law et al., 2018; Law, Hudiburg, & Luyssaert, 2013). Uncertainty estimates for total regional emissions were calculated using a propagation of error approach accounting for error in biomass allometrics and the MTBS fire perimeters (Law et al., 2018).

Western US fire emissions were calculated from 2000-2016 using MTBS (Eidenshink et al., 2007)) estimates of burn area and severity combined with FIA plot biomass data aggregated by ecoregion and forest type (30m pixel resolution; Table S9) and severity-specific combustion factors for each pool (large stems, small stems, downed dead wood, understory, standing dead, litter pools (Campbell et al., 2007; Meigs et al., 2009) (and Rim Fire values from this study). Areas of recurring severe fire based on the MTBS record (less than 2% of total burn area included reburns from 1984-2016; Table S10) were combusted with modified biomass pools reflecting simulated post-fire conditions using combustion observations from reburned plots in the Biscuit Fire study (Campbell et al., 2007; Campbell et al., 2016; Donato, Fontaine, & Campbell, 2016). Combustion factor scenarios were consistent with DayCent YFDP simulation sets by carbon pool (see Tables S5 - S8). Observation-based and the variable-severity model-based sets were applied by severity. The static-severity model combustion percentages were applied across all severities within burn perimeters. Comparisons with fossil fuel emissions was done using Environmental Protection Agency state CO_2 emissions data (EPA, 2018).

Results and Discussion

Fire Emissions in Carbon Dense Forests

The YFDP experienced a mixed-severity burn in 2013, consuming 22% of aboveground carbon, with dead biomass producing 95% of estimated emissions (Table 1). The fire induced ~71% tree mortality (stems ≥1 cm dbh) within one year and combusted <1% of live tree biomass. When YFDP carbon stocks burned under the range of model scenarios, default variable and static severity model coefficients resulted in up to 285% and 486% overestimated fire CO_2 emissions compared to observation-based coefficients, respectively (Figure 3a). Overestimation resulted primarily from high default bole combustion coefficients combined with existing high live biomass. High-severity fire consumed 31% and 81% (70 and 183 Mg C ha^{-1}) of aboveground live tree carbon in model scenarios, compared to 6% (14 Mg C ha^{-1}) in the observation-based scenario.

Observation-based combustion of aboveground carbon decreased from 22% (80 Mg C ha^{-1}) to 6% (22 Mg C ha^{-1}) from high- to very low-fire severity, reflecting transitions between canopy and ground fire. With variable-severity model coefficients, aboveground carbon combustion decreased from a maximum of 87% to a minimum of 10%. This wide range is explained by large modeled decreases in emissions with decreasing burn severity, averaging 20% of aboveground carbon per severity class (Figure 3a; dotted lines). By contrast, observation-based changes in emitted aboveground carbon averaged 5% per severity class. The static-severity model simulation overestimated observation-based emissions by 59-486% (high-low observed severity).

Thirty years post-fire, the static-severity scenario carbon losses still exceeded those from observation-based severities by 39-1010% (Figure 3a). The difference in emissions estimates between the variable-severity model and observation-based scenarios marginally decreased over time due to a lack of remaining biomass to decompose (Figure S3). Nonetheless, the variable and static severity models overestimated observation-based emissions by averages of 150% and 130%, demonstrating persistent unrealistic post-fire emissions over timescales relevant to greenhouse gas management. These results highlight that model estimates can both inflate fire emissions and the potential carbon benefits of severity-reduction strategies, such as thinning for fuels reduction. Further, static-severity overestimates increase dramatically at lower severities, undervaluing the persistent carbon storage capacity of forests experiencing low-severity fire.

Omission of a snag pool resulted in increased combustion of downed-woody debris (versus snags); net fire event carbon losses were 50-79% greater across no-snag scenarios (Figure 3b). Without snags, fire-killed biomass was deposited on the forest floor and decomposed at a faster rate than in the snag scenarios, where large quantities of killed biomass decayed in standing dead pools before reaching the ground (Figure S1). The combined effects of altered combustion and decomposition after 30 years yielded an average doubling of simulated net emissions across severities when snags were not represented.

From low-to-high severity, “mortality = emissions” scenarios (“public perception”; Figure 1b) exceeded observation-based emissions by 140-253% (Figure 3b); these results were similar to variable-severity scenario results (Figure 3a). At neither 30 years nor 100 years post-fire did the “mortality = emissions” scenario emissions decrease below the observation-based scenarios. Although up to 95% mortality was implemented in the observation-based scenarios, subsequent decomposition of dead biomass was largely compensated by regrowth. These results show that simulating mortality transfers that are distinct from combustion does not simply delay these carbon losses to the future (Figure S2 and S3); greenhouse gas emissions and impacts to the atmosphere are instead markedly decreased.

Emissions impacts across western U.S. forest fires in the 21st century

Across the western US, observation-based combustion emissions summed to 232 ± 62 Tg C from 2000-2016, emitting 23% of aboveground carbon stocks within ~11 million hectares of burned area (Figures 4 and 5), in agreement with estimates for Oregon over similar time periods (Law, 2014; Meigs et al., 2009). As at smaller scales, model-based live-tree combustion overestimated observation-based combustion by an order of magnitude (Figure 4), leading to regional emissions overestimates of 59% and 83%.

Forest fires in California, Idaho, and Montana accounted for 54% of total combustion emissions (Figure 5), resulting from higher burned area and aboveground carbon density relative to southern interior states. Coastal-state (CA, OR, WA) model-based scenarios exceeded observation-based emissions by 81% and 103%, compared to overestimates of 35% and 67% in the Northern Rockies (ID, MT, WY). This difference stemmed from greater aboveground carbon density in coastal versus Northern Rocky states. Thus, carbon loss is most overestimated in forests with high tree biomass.

Regional observation-based fire emissions totaled to 5% of fossil fuel emissions compared to twice that when using default coefficients (Figure 5b). Notably, Idaho and Montana fire emissions accounted for 55% and 24% of yearly fossil fuel emissions, respectively, highlighting the importance of correctly calculating fire emissions in the Northern Rockies due to large projected increases in fire (Westerling et al., 2006). Emissions in California and Washington were extremely low relative to fossil fuel emissions, likely because of population density (energy usage).

Implications

Our results illustrate that the use of inaccurate combustion coefficients in models can double forest fire emissions estimates across the western US. Overestimates increase to 3-4 times in carbon-dense forests such as the YFDP, mostly because models incorrectly combust live trees. Treating carbon released over years to centuries as an immediate emission by equating combustion with mortality is simply inaccurate. Omitting snag representation in models compounds this error, because of altered decay and combustion dynamics.

A warming climate and more frequently recurring fire (Westerling, Hidalgo, Cayan, & Swetnam, 2006) may alter some regional forest carbon stocks from the present. The field data used in this study includes area in the 2002 Biscuit Fire that contained the 1987 Silver Fire (fifteen years earlier), where reburned plots showed a 26% reduction in standing and downed dead wood loss due to fire compared to mature single-burn plots but similar pool combustion coefficients across fires (Donato et al., 2016). New observations from reburned lodgepole pine stands in the Greater Yellowstone Ecosystem show that young stands can lose a majority of the aboveground carbon (basal dia. <4cm; (Turner, Braziunas, Hansen, & Harvey, 2019)), consistent with Biscuit fire observations for the small conifer pool (Campbell 2007). This suggests a mechanism by which recurrent burning(“reburn”) could in principle lead to state changes to treeless vegetation over the mid-term because of frequent, repeated combustion of aboveground stocks *over time* (Coop, Parks, McClernan, & Holsinger, 2016). The percentage of the regional forest landscape that has recently experienced such severe reburn is less than 1% (see regional methods), but could increase in the future with climate change (Dale et al., 2001; Turner et al., 2019), and disproportionally in some areas (e.g. Southern California and US Southwest). It will be essential to accurately estimate these emissions impacts in a regional context by quantifying shifting biomass pools (e.g. dead and young pools) upon which realistic combustion coefficients are applied.

Resolving modeled inaccuracies is critical because CO_2 emissions-reduction strategies are being implemented based on these estimates (California, 2018; DNR, 2018; USA, 2019). Overestimating forest fire emissions exacerbates public and policy-maker misconceptions (Figure 1). Our simulations highlight the need for more studies on pre- and post-fire carbon pools over decadal durations in order to capture combustion dynamics in different forest types to provide observations for modelers to better constrain and validate their models. At present, even when models correctly estimate burned area, their ability to properly inform policy makers about the contributions of fires to greenhouse gas budgets can be inadequate, adding fuel to the fire when drafting forest management plans.

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The authors declare no conflict of interest.

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Table 1. Observed aboveground carbon stocks and combustion vs default model combustion. All combustion percentages are equal to combustion coefficients except for the Rim Fire snag pool, where the percentage combines combustion and transfer of snag biomass to downed-wood pools. Bold italicized numbers highlight discrepancies between the range of model coefficients (Table S6, S7, S8) (Lawrence et al., 2018; Sturtevant, Scheller, Miranda, Shinneman, & Syphard, 2009) and field observations for live trees. Field observations are from this study and previous studies (Lutz, Larson, Swanson, & Freund, 2012; Campbell, Donato, Azuma, & Law, 2007)

Rim Fire, YFDP Pool	***Stock (Mg C ha^{-1})***		***Combustion (%)***	***Model (%) (moderate-severity)***
Tree	281.3	(53.2)	***0.1 (0.0)***	
foliage	-	-	-	***80-92***
branch	-	-	-	***30-92***
bark	-	-	-	***30-46***
bolewood	-	-	-	***30-46***
Shrub	2.9	(*)	95.4 (*)	*na*
Snag	13.9	(3.0)	61.5 (8.8)	*na*
Coarse woody debris	39.5	(19.6)	58.3 (33.5)	28-50
Fine woody debris	3.3	(1.4)	94.4 (5.0)	50-100
Litter	11.9	(1.5)	90.4 (4.8)	50-100
Duff	43.6	(5.7)	88.5 (4.3)	50-100
Total	**396.4**	**(54.4)**	**21.9 (5.0)**	***na***
Biscuit Fire Pool	***Stock (Mg C ha^{-1})***		***Combustion (%)***	***Model (%) (high-severity)***
Tree	92.5		***8.7***	
foliage	5.6		***73.0***	***80-100***
branch	14.8		***7.9***	***30-100***
bark	11.7		***21.0***	***30-80***
bolewood	60.5		***0.6***	***30-80***
Snag	7.7		17.6	*na*
Coarse woody debris	7.6		34.1	28-80
Fine woody debris	1.1		78.0	50-100
Litter	9.2		100.0	50-100
Duff	6.0		99.0	50-100
Total	**124.0**		**22.5**	***na***

Figure 1. Conceptual diagram of realistic (observation-based) versus public perception and model implementation of live forest biomass combustion in high severity forest fires. A common "**public and policymaker perception**" (State of California Executive Department, 2018; U.S. Executive Office of the President, 2018; Zinke, 2018) (a), is that live, mature forests catastrophically "burn to the ground", with nearly all biomass emitted via combustion rather than remaining in the ecosystem as dead biomass (note: photograph from grassland). Flawed **"model"** fire implementations (b) are less extreme in their total ecosystem combustion, with the most significant misrepresentation being the over-combustion of live, mature trees. In "**reality"** (c), 80-90% of live stems are killed but not combusted; their mass remains as substantial dead ecosystem carbon pools after the fire. *Short-return interval reburned stands can release additional carbon from dead biomass pools, ranging from ~25% (post-mature burn) to 95% (post-young burn).

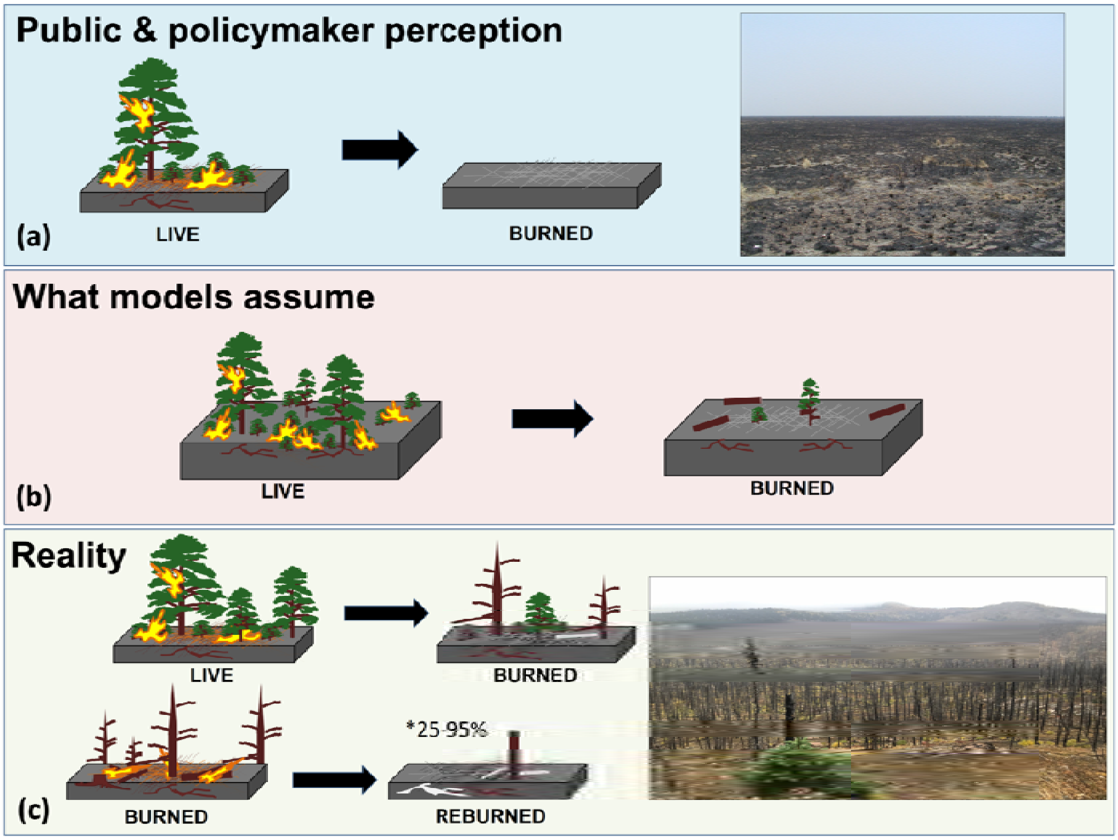
Figure 2. Post-fire forest landscapes following different, varying severity fires in Oregon. (a) Ponderosa pine – low severity patch four years after the 2003 B&B Complex mixed severity fire (28,640 ha), (b) Mixed conifer – moderate severity patch four years after the 2003 B&B complex, (c) Ponderosa pine – high severity patch two years after the 2002 Eyerly mixed severity fire (photo by T. Hudiburg) and (d) Ponderosa pine – high severity patch five years after the 2002 Eyerly fire (photo by B.E. Law).

Figure 3. Simulated ecosystem carbon losses at the time of fire (Year 0) and 30 years post-fire at the YFDP. For scenarios with variable severity, full bars indicate emissions density at high severity. Dashed lines indicate emissions at very low-to-moderate severity. Points indicate scenario means (or static emissions). **a.** Carbon losses for observation-based

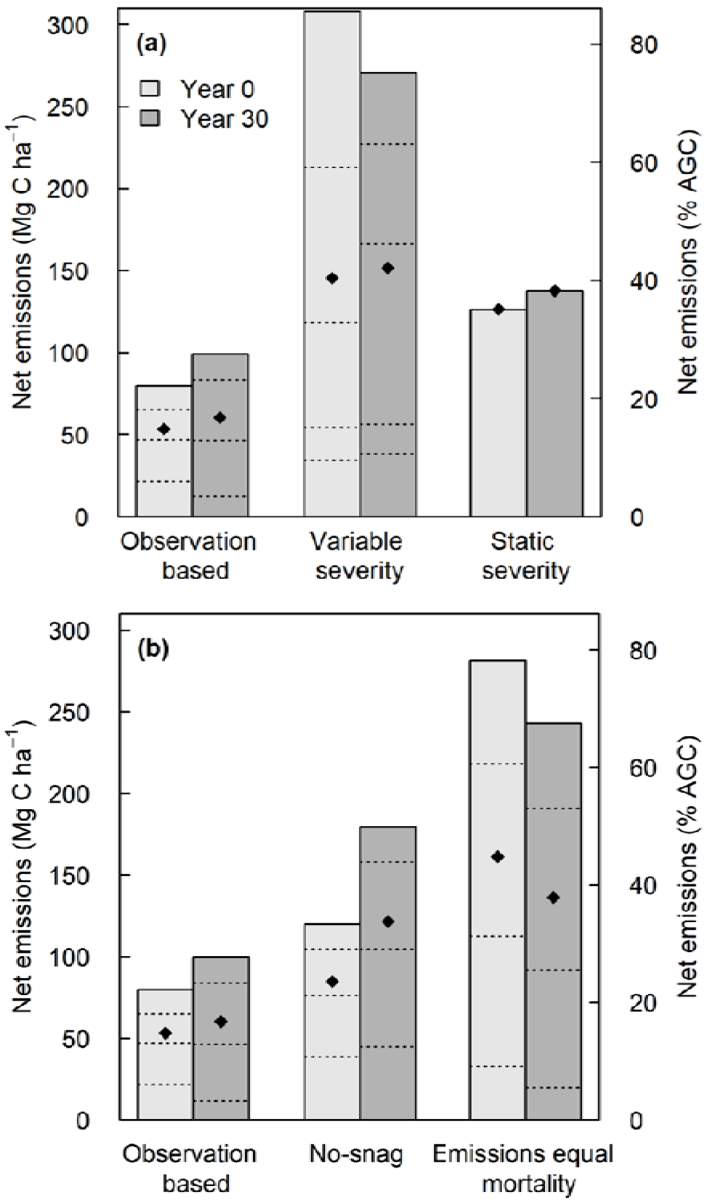
and model default parameterizations. **b.** Carbon losses for observation-based, observation-based without snags, and "mortality = emissions" scenarios.

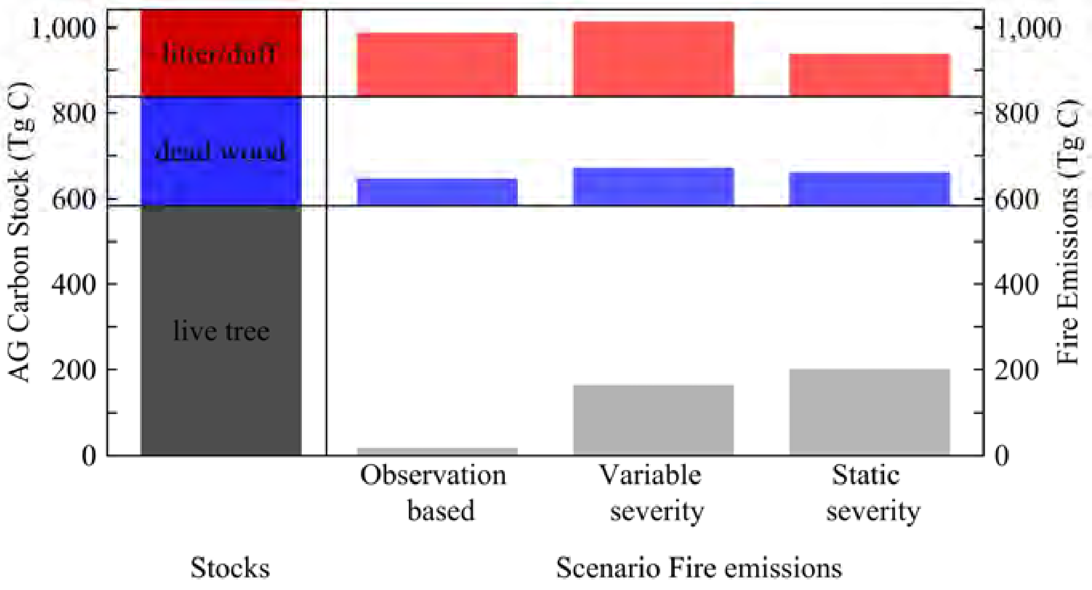
Figure 4. Western U.S. aboveground carbon pools and pool fire emissions across scenarios, 2000-2016 forest burn area. Pre-fire aboveground carbon (AG) pool totals (opaque bars) are compared to fire-event pool carbon emissions (translucent bars). Litter/duff, dead wood, and live trees account for 21%, 26%, and 53% of aboveground stocks, respectively.

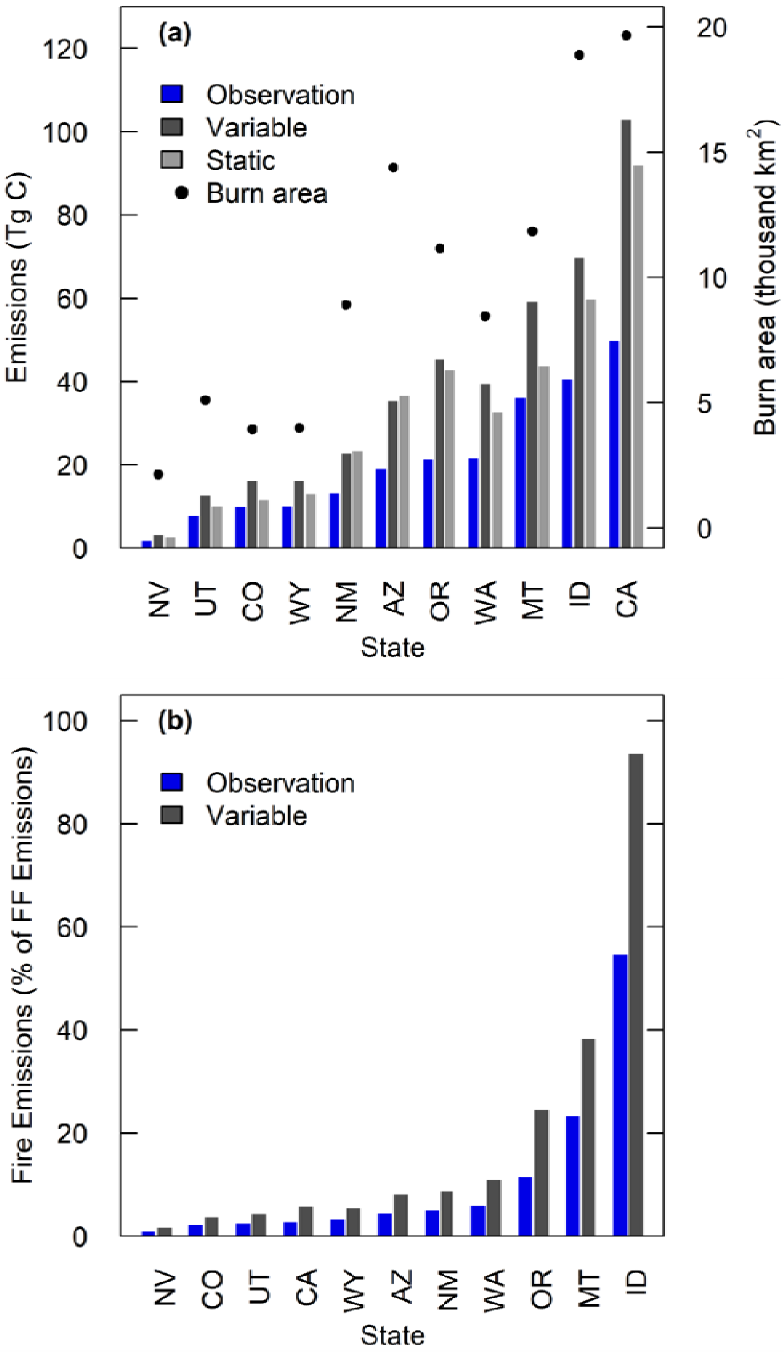
Figure 5. Total state emissions (2000 – 2016) estimated from observed combustion coefficients vs coefficients from variable and static-severity models. (a) Western state forest fire emissions and burn area. (b) Western state fire emissions as a proportion of fossil fuel (FF) emissions.


Public & policymaker perception
(a)
LIVE
BURNED
What models assume
(b)
LIVE
BURNED
Reality
LIVE
BURNED
*25-95%
(c)
BURNED
REBURNED


(a)
(b)
(c)
(d)


(a)
Year 0
Year 30
Net emissions (Mg C ha^{-1})
Net emissions (% AGC)
0
50
100
150
200
250
300
0
20
40
60
80
Observation based
Variable severity
Static severity
(b)
Net emissions (Mg C ha^{-1})
Net emissions (% AGC)
0
50
100
150
200
250
300
0
20
40
60
80
Observation based
No-snag
Emissions equal mortality


1,000
800
600
400
200
0
AG Carbon Stock (Tg C)
litter/duff
dead wood
live tree
Fire Emissions (Tg C)
Observation based
Variable severity
Static severity
Stocks
Scenario Fire emissions


(a)
Observation
Variable
Static
Burn area
Emissions (Tg C)
0
20
40
60
80
100
120
Burn area (thousand km^2)
0
5
10
15
20
NV
UT
CO
WY
NM
AZ
OR
WA
MT
ID
CA
State
(b)
Observation
Variable
Fire Emissions (% of FF Emissions)
0
20
40
60
80
100
NV
CO
UT
CA
WY
AZ
NM
WA
OR
MT
ID
State

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Carbon Balance and Management

RESEARCH

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Attribution of net carbon change by disturbance type across forest lands of the conterminous United States

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Abstract

Background: Locating terrestrial sources and sinks of carbon (C) will be critical to developing strategies that contribute to the climate change mitigation goals of the Paris Agreement. Here we present spatially resolved estimates of net C change across United States (US) forest lands between 2006 and 2010 and attribute them to natural and anthropogenic processes.

Results: Forests in the conterminous US sequestered −460 ± 48 Tg C year^{-1}, while C losses from disturbance averaged 191 ± 10 Tg C year^{-1}. Combining estimates of net C losses and gains results in net carbon change of −269 ± 49 Tg C year^{-1}. New forests gained −8 ± 1 Tg C year^{-1}, while deforestation resulted in losses of 6 ± 1 Tg C year^{-1}. Forest land remaining forest land lost 185 ± 10 Tg C year^{-1} to various disturbances; these losses were compensated by net carbon gains of −452 ± 48 Tg C year^{-1}. C loss in the southern US was highest (105 ± 6 Tg C year^{-1}) with the highest fractional contributions from harvest (92%) and wind (5%). C loss in the western US (44 ± 3 Tg C year^{-1}) was due predominantly to harvest (66%), fire (15%), and insect damage (13%). The northern US had the lowest C loss (41 ± 2 Tg C year^{-1}) with the most significant proportional contributions from harvest (86%), insect damage (9%), and conversion (3%). Taken together, these disturbances reduced the estimated potential C sink of US forests by 42%.

Conclusion: The framework presented here allows for the integration of ground and space observations to more fully inform US forest C policy and monitoring efforts.

Keywords: Forests, Disturbance, Harvest, Insects, Fire, Drought, Greenhouse gas, Land use, Climate change, FIA, UNFCCC

Background

The 2015 Paris Climate Change Agreement, with consensus from 192 signatories, calls for achieving a balance between anthropogenic emissions by sources and removals by sinks in the second half of this century [1]. Forests are currently responsible for the capture and storage of an estimated 25% of global anthropogenic emissions [2]. If Paris goals are to be achieved, further enhancement of forest-based carbon (C) removals to mitigate emissions in other sectors will be a critical component of any collective global strategy [3], especially as no alternative sink technologies have yet been proven at scale. Thus, spatially identifying terrestrial sources and sinks of carbon, and understanding them well enough to predict how they will respond to management decisions or future climate change, will pose major science and policy challenges in the years to come.

Remote sensing products can provide regular and consistent observations of Earth's surface to help identify the condition of forest ecosystems and changes within them at a range of spatial and temporal scales [4]. Over the past

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several years, the remote sensing research community has used these products to monitor tropical deforestation, forest C stocks and associated C emissions, largely in support of REDD+ initiatives in developing countries [5–12]. In many developed countries, periodic national forest inventories form the basis of annual greenhouse gas (GHG) reporting to the United Nations Framework Convention on Climate Change (UNFCCC). The sample-based design of these inventories may offer little in the way of detailed and spatially-explicit information on the distribution of forest biomass [13], timing and location of timber harvesting in managed forests, or the cause and timing of other types of forest disturbances. If the ultimate aim of the Paris Agreement is to introduce practices that lead to reduced emissions and enhanced removals of C from the world's managed forests, including in temperate and boreal biomes, then a lack of disaggregated, spatially-explicit information could pose challenges over the coming years related to knowledge of where changes are occurring and where interventions are likely to be most effective.

Several C budget models have been developed to simulate ecosystem response to climate drivers and other disturbances, and these models represent an established approach to estimating C fluxes at national to regional scales. For example, Canada's National Forest Carbon Monitoring Accounting and Reporting System (NFCMARS) uses the Carbon Budget Model of the Canadian Forest Sector (CBM-CFS3), and is used also as a decision support tool for forest managers to quantify forest C dynamics at a landscape scale. Different models emphasize different aspects of ecosystem dynamics, with some accounting for competition between plant functional types, nutrient limitation, and natural disturbances. Time series of anthropogenic land-cover changes are usually prescribed based on spatially explicit data. The models can reflect spatial and temporal variability in C density and response to environmental conditions, but their modeled C stocks may differ markedly from observations [14].

Such models are not used explicitly in the GHG inventory for the US to report forest C fluxes. Instead, the current US inventory system uses the C stock-difference accounting approach [15] enabled by the annual national forest inventory conducted by the United States Department of Agriculture (USDA) Forest Service Forest Inventory and Analysis (FIA) program. The difference in C stocks in five C pools is estimated via sequential re-measurements of permanent ground inventory plots. When forest stocks decline, it is assumed that C emissions have occurred from the land to the atmosphere if not reconciled with a transfer to another land use category. Conversely, when forest C stocks increase it is assumed that C has been sequestered from the atmosphere by terrestrial vegetation. In this way, estimated net C change in the US forest sector is the integrated result of both anthropogenic and natural processes—harvest, land use change, fire, drought, insect infestation, wind damage—all of which influence the magnitude of forest C stocks in each pool. Results are most statistically robust when compiled at large spatial scales (e.g., state or regional), such that quantification of finer-scale spatial patterns is less precise. Though changes are well constrained via sequential re-measurements on inventory plots, the US [16, 17] has only recently begun using methods to disaggregate the effects of various disturbance types on forest stocks and fluxes (although this separation is not a requirement of IPCC Good Practice Guidance, [18]).

The objective of this study was to synthesize information from remote sensing observations of forest carbon stocks and disturbance with information collected by various US agencies into a framework that (1) more explicitly attributes C losses to major disturbance types (land use change, harvesting, forest fires, insect damage, wind damage and drought); and (2) disaggregates net C change into relevant IPCC reporting categories of non-forest land converted to forest land, forest land converted to non-forest land, and forest land remaining forest land. This framework allows for the integration of ground and space observations to more fully inform US forest C policy and monitoring efforts.

Methods

We built a spatially-explicit empirical model that combines information from many data sources to infer disturbance and resulting C dynamics within each hectare of forest land in the 48 conterminous states of the US, totaling an area of more than 2.1 million km^2. For the purposes of regional comparison and analyses, we divided the US into three broad regions (North, South, West) based on similar histories of forestland use ([19], Fig. 1) and into nine smaller subregions based on those used in the US FIA program. Forest types were defined as hardwood or softwood, following the National Land Cover Data (NLCD) classification (deciduous forest class: hardwoods; evergreen forest class: softwoods). The time period of analysis is 1 January 2006 to 31 December 2010.

Data inputs

Forest area map (2005)

Forest extent in the base year 2005 was determined from the NLCD and the global tree cover and tree cover change products of Hansen et al. [8]. Specifically, an area was determined to be forested if categorized as

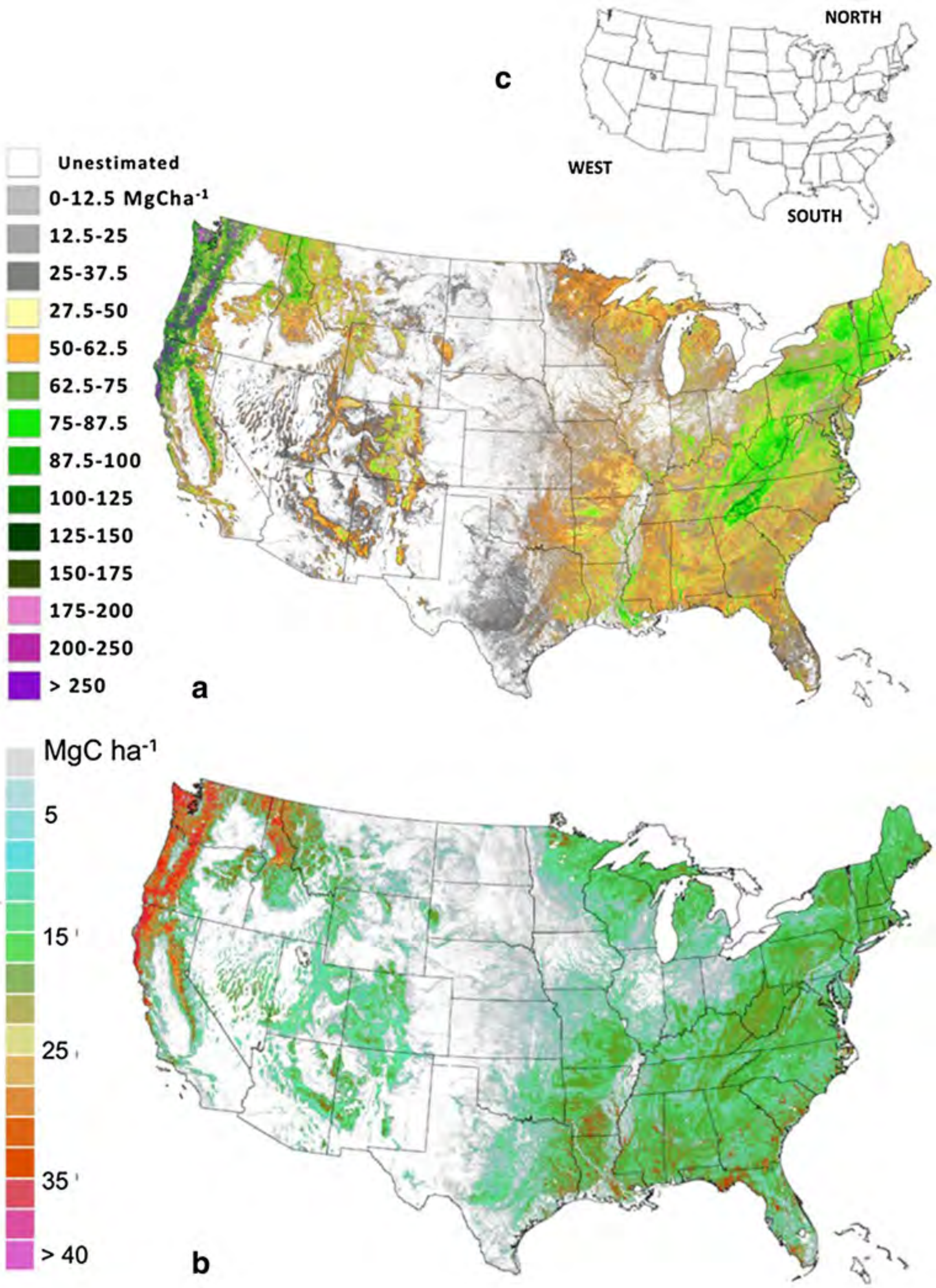

Fig. 1 **a** Map of aboveground live woody biomass carbon density (Mg C ha^{-1}) and **b** uncertainty across forest lands of the conterminous US at 1-ha resolution for circa the year 2005. **c** The regional analysis was performed by dividing the US into three sub-regions as recommended by Heath and Birdsey [19]. The above and belowground carbon density maps and the uncertainty maps can be downloaded from NASA's distributed Data Active Archive Center (http://dx.doi.org/10.3334/ORNLDAAC/1313)

hardwood or softwood in the NLCD 2006 dataset[1] and, according to the Hansen et al. [8] dataset, it (a) met the tree cover threshold of 25% in the year 2000 and was not lost between 2001 and 2005 or (b) did not meet the tree cover threshold of 25% in 2000 but was identified as having gained tree cover (i.e., afforestation/reforestation) between 2000 and 2012. The NLCD has been shown to significantly underestimate tree cover [20] and thus the forest area estimates used in this analysis—defined by both NLCD and Hansen et al. [8]—are likely to be conservative. However, these two data products currently represent the best available spatially explicit data for forest extent in the conterminous US (CONUS).

Forest biomass density maps (circa 2005)

We developed maps of C stocks (50% of biomass) in aboveground live biomass in US forest land as part of NASA's C Monitoring System (CMS) program based on a combination of remote sensing observations and FIA data (Fig. 1). The overall methodology used in mapping the aboveground live forest biomass C density is described in Saatchi et al. [5]. After filtering for cloud effects, slopes, and signal-to-noise ratio, more than 700,000 samples of lidar (light detecting and ranging) data acquired between 2003 and 2008 from the Geoscience Laser Altimeter System (GLAS), onboard the Ice, Cloud and land Elevation Satellite (ICESat) were used as samples of the vertical structure of US forest land. We used the Lorey's height [21] measured in 65,000 single-condition FIA plots (i.e., plots with a single domain mapped on each plot) to calibrate the lidar-derived height metric and used the relationship between Lorey's height and aboveground C density for 28 forest types to convert the lidar data into estimates of aboveground live C density. All FIA plots with a probability of disturbance causing reduced canopy cover (<50%) were removed from the height-biomass model development to reduce any potential discrepancy between ground and lidar height metrics. Lidar-derived biomass samples were then extrapolated over the landscape using a combination of optical and radar satellite imagery that captures the variations of forest structure and cover to create wall-to-wall maps of forest aboveground live biomass C density. We used nine remote sensing imagery layers as spatial predictor variables. Optical and thermal data from Landsat imagery (bands 3, 4, 5 and 7) were aggregated to 100 m spatial resolution from 30 m native resolution along with the leaf area index derived from Landsat imagery [22]. In addition, we used the advanced land observing satellite (ALOS) phased area L-band synthetic aperture radar (PALSAR) imagery at two polarizations (HH and HV backscatter) along with topographical data of surface elevation and slope from Shuttle Radar Topography Mission (SRTM) resampled to 100 m resolution from 20 and 30 m native resolutions, respectively. ALOS PALSAR plays an important role in quantifying variation in forest biomass. In particular, the HV polarization provides the largest contribution among the data layers to predicted biomass because it has a strong direct sensitivity to biomass up to 100–150 Mgha^{-1} (depending on forest type), is less impacted by soil moisture and other environmental variables, and may contribute significantly in extrapolating larger biomass forests through texture and spatial correlation. Similarly, SRTM data include information on topography and also forest height. We used the national elevation data (NED) to represent the ground surface elevation and used the difference between SRTM and NED as an indicator of forest height. This variable also contributed significantly to explaining the spatial variation of biomass over forests with biomass values >150 Mgha^{-1}.

The aboveground C density samples derived from GLAS data were combined with satellite imagery using the maximum entropy estimation (MaxEnt) algorithm to estimate aboveground biomass density for each 1-ha pixel. MaxEnt is a probability-based algorithm that estimates the posterior likelihood distribution of a variable by maximizing the entropy of said probability distribution while maintaining the constraints provided by the training samples [23]. We selected a random subset consisting of 70% of the samples (~500,000 samples) for model input and used the remaining 30% for model evaluation and validation. The product from the MaxEnt estimator includes both the mean aboveground carbon (AGC) density for each 1-ha pixel and the estimation of the error derived from a Bayesian probability estimator for each pixel. Spatial uncertainty analysis and uncertainty propagation were used to evaluate the overall uncertainty of AGC at the pixel level. This process included the quantification of error at each step of the process and the use of the Gaussian error propagation approach:

$$\mathrm{Error} = \sqrt{\varepsilon^{\mathbf{2}}_{\mathbf{measurement}} + \varepsilon^{\mathbf{2}}_{\mathbf{allometry}} + \varepsilon^{\mathbf{2}}_{\mathbf{sampling}} + \varepsilon^{\mathbf{2}}_{\mathbf{prediction}}}$$

where each of the terms are the relative errors at that pixel and represent the measurement errors of lidar for capturing the forest height, the error associated with the lidar aboveground C allometry model for each forest type, the error associated with sampling the 1-ha pixel

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[1] Within each 1 ha pixel, the wet woodland class was included as forest but was not used to determine whether the pixel was hard- or softwood. Hard- or softwood was determined based on the plurality of NLCD hard- or softwood 30 m pixels within the hectare, ignoring the sub-fraction of wet woodlands and selecting softwood when hard- and softwood fractions were equal.

with GLAS footprint size (~0.25 ha), and the MaxEnt prediction error. In evaluating the errors at the state and county level, we also included the spatial correlation of the prediction error from the MaxEnt approach [24].

In the FIA, belowground forest biomass is quantified using a root-shoot ratio [25]. Knowledge of root biomass dynamics is fundamental to improving our understanding of carbon allocation and storage in terrestrial ecosystems [26]. We used the relationship between belowground carbon (BGC) and AGC from the FIA data to develop a BGC spatial distribution at the same scale as AGC [5, 27]. In estimating the uncertainty in BGC, we followed the same approach as AGC with the addition of including the errors associated with the model used in relating AGC to BGC.

FIA stock change data (2006–2010)

To estimate average net changes in the stock of live AGC and BGC between 2006 and 2010 in forests disaggregated by disturbance type, we queried the FIA database (http://apps.fs.fed.us/fiadb-downloads/datamart.html) to extract more than 141,000 records associated with re-measured permanent plots, where each extracted record represents a "condition" (i.e., domain(s) mapped on each plot according to attributes such as land use, forest type, stand size, ownership, tree density, stand origin, and/or disturbance history) of a measured plot at two points in time, typically 5 years apart. Disturbed plots were stratified into a lookup table by geographic region (North, South, or West), forest type (hardwood or softwood), disturbance type (fire, insect, wind, conversion, or harvest), and disturbance intensity (Table 1). A similar lookup table was developed for undisturbed plots stratified by geographic region, forest type, and base C stock in the year 2005 (Table 2).

Disturbance maps (2006–2010)

Sources of disturbance data used in this analysis are summarized in Table 3 and include spatially-explicit data on locations of fire, insect damage, wind damage, land use change, drought, and timberlands. The timberlands map was used to attribute net carbon gains occurring within vs. outside timberland areas. Because harvested wood may come from intermediate treatments (treatments not intended to cause regeneration), partial harvest or clearcutting forests, deforestation, and non-forest land trees, the area of clearcuts as observed within timberland areas through remote sensing imagery cannot represent all these wood sources [28]. Therefore for estimating C losses from timber harvest, we used data collected in the US based on mill surveys rather than remote sensing observations.

Timber product output data (TPO 2007)

The volume of roundwood products, mill residues and logging residues reported in the TPO database (Table 3), separated by product class and detailed species group, were used to estimate C losses from wood harvest. The spatial resolution of the data was the "combined county", which represented the minimum reportable scale from the timber product output (TPO; FIA Fiscal Year 2013 Business Report, [29]) data while retaining necessary confidentiality.

Model assumptions

IPCC Tier 2 estimation

The terrestrial C cycle includes changes in C stocks due to both continuous processes (i.e., growth, decomposition) and discrete events (i.e., disturbances such as harvest, fire, insect outbreaks, land-use change). Continuous processes can affect C stocks in all areas every year, while discrete events (i.e., disturbances) cause emissions and redistribute C in specific areas in the year of the event. In accounting for net C change in this analysis, we use country-specific data (Tier 2) and apply the simplifying methodological assumption [15] that all post-disturbance emissions (after accounting for C storage in harvested wood products) occur as part of the disturbance event, i.e., in the year of disturbance, rather than modeling these emissions through time as in IPCC's Tier 3 approach. The application of lower tier methods also assumes that the average transfer rate into dead organic matter (dead wood and litter) is equal to the average transfer out of dead organic matter, so that the net stock change in these pools is zero [15]. This assumption means that dead organic matter (dead wood and litter) C stocks need not be quantified for land areas that remain forested. The rationale for this approach is that dead organic matter stocks, particularly dead wood, are highly variable and site-specific, depending on forest type and age, disturbance history and management. Because the FIA data used in this analysis do not include measurements of soil C or dead C pools and no robust relationships currently exist that relate these pools to a more easily measured pool (such as the derivation of belowground biomass from aboveground biomass using root:shoot ratios), we excluded the soil C and dead C pools from our analysis. As a result, our estimate of net C change using the stock-difference approach is equal to the net change in C stocks in the aboveground and belowground live biomass pools only, with a fraction of the aboveground live biomass assumed to be transferred to the wood products pool, where a portion is permanently sequestered in long-lived products and the remainder emitted to the atmosphere (see below).

Table 1 Look-up table of annual fractional change (average = μ; standard error = σ) in aboveground carbon (AGC) and belowground carbon (BGC) in disturbed forests based on FIA plot data

Region	Forest type	Disturbance	Initial C	N	AGC μ	AGC σ	BGC μ	BGC σ
North	Softwood	Fire	Low	2	−0.003	0.012	−0.001	0.013
North	Softwood	Fire	Medium	3	−0.052	0.031	−0.053	0.031
North	Softwood	Fire	High	5	−0.150	0.030	−0.157	0.030
North	Softwood	Weather	Low	63	−0.013	0.016	−0.014	0.016
North	Softwood	Weather	High	10	−0.163	0.013	−0.169	0.013
North	Softwood	Insect	Low	85	−0.003	0.007	−0.003	0.008
North	Softwood	Insect	Medium	82	−0.044	0.023	−0.046	0.023
North	Softwood	Insect	High	45	−0.126	0.035	−0.133	0.032
North	Softwood	Harvested	Low	521	−0.046	0.035	−0.048	0.036
North	Softwood	Harvested	High	246	−0.152	0.026	−0.158	0.025
North	Hardwood	Fire	Low	40	−0.003	0.009	−0.003	0.009
North	Hardwood	Fire	Medium	29	−0.045	0.024	−0.048	0.023
North	Hardwood	Fire	High	11	−0.131	0.034	−0.136	0.034
North	Hardwood	Weather	Low	412	−0.011	0.016	−0.011	0.016
North	Hardwood	Weather	High	34	−0.160	0.017	−0.164	0.016
North	Hardwood	Insect	Low	656	−0.002	0.008	−0.002	0.008
North	Hardwood	Insect	Medium	432	−0.045	0.020	−0.046	0.020
North	Hardwood	Insect	High	118	−0.132	0.029	−0.136	0.028
North	Hardwood	Harvested	Low	2177	−0.047	0.035	−0.047	0.035
North	Hardwood	Harvested	High	806	−0.154	0.023	−0.157	0.023
South	Softwood	Fire	Low	127	−0.002	0.007	−0.003	0.008
South	Softwood	Fire	Medium	174	−0.048	0.021	−0.052	0.022
South	Softwood	Fire	High	52	−0.124	0.027	−0.131	0.028
South	Softwood	Weather	Low	78	−0.016	0.016	−0.017	0.016
South	Softwood	Weather	High	16	−0.161	0.026	−0.168	0.023
South	Softwood	Insect	Low	46	−0.002	0.008	−0.004	0.008
South	Softwood	Insect	Medium	66	−0.054	0.022	−0.059	0.023
South	Softwood	Insect	High	60	−0.135	0.030	−0.142	0.029
South	Softwood	Harvested	Low	1787	−0.044	0.034	−0.048	0.036
South	Softwood	Harvested	High	586	−0.149	0.025	−0.157	0.024
South	Hardwood	Fire	low	112	−0.002	0.008	−0.003	0.008
South	Hardwood	Fire	Medium	86	−0.042	0.021	−0.045	0.022
South	Hardwood	Fire	High	37	−0.131	0.033	−0.139	0.030
South	Hardwood	Weather	Low	484	−0.014	0.016	−0.015	0.016
South	Hardwood	Weather	High	32	−0.162	0.019	−0.167	0.017
South	Hardwood	Insect	Low	145	0.000	0.013	−0.002	0.011
South	Hardwood	Insect	Medium	121	−0.047	0.022	−0.051	0.022
South	Hardwood	Insect	High	38	−0.133	0.031	−0.138	0.031
South	Hardwood	Harvested	Low	1235	−0.048	0.036	−0.051	0.036
South	Hardwood	Harvested	High	609	−0.146	0.029	−0.152	0.027
West	Softwood	Fire	Low	13	−0.007	0.008	−0.007	0.008
West	Softwood	Fire	Medium	8	−0.049	0.023	−0.050	0.026
West	Softwood	Fire	High	0	*−0.126*	*NA*	*−0.133*	*NA*
West	Softwood	Weather	Low	5	−0.003	0.008	−0.003	0.008
West	Softwood	Weather	High	0	*−0.162*	*NA*	*−0.168*	*NA*
West	Softwood	Insect	Low	12	0.001	0.007	0.001	0.007
West	Softwood	Insect	Medium	3	−0.041	0.016	−0.044	0.018
West	Softwood	Insect	High	0	*−0.131*	*NA*	*−0.138*	*NA*

Table 1 continued

Region	Forest type	Disturbance	Initial C	N	AGC μ	AGC σ	BGC μ	BGC σ
West	Softwood	Harvested	Low	28	−0.027	0.030	−0.028	0.031
West	Softwood	Harvested	High	0	*−0.150*	*NA*	*−0.157*	*NA*
West	Hardwood	Fire	Low	4	−0.002	0.008	−0.002	0.008
West	Hardwood	Fire	Medium	3	−0.057	0.021	−0.059	0.021
West	Hardwood	Fire	High	0	*−0.131*	*NA*	*−0.138*	*NA*
West	Hardwood	Weather	Low	0	*−0.013*	*NA*	*−0.013*	*NA*
West	Hardwood	Weather	High	0	*−0.161*	*NA*	*−0.165*	*NA*
West	Hardwood	Insect	Low	13	−0.003	0.008	−0.003	0.009
West	Hardwood	Insect	Medium	3	−0.041	0.025	−0.044	0.028
West	Hardwood	Insect	High	0	*−0.132*	*NA*	*−0.136*	*NA*
West	Hardwood	Harvested	Low	4	−0.039	0.031	−0.039	0.033
West	Hardwood	Harvested	High	0	*−0.151*	*NA*	*−0.155*	*NA*

Italics imputed from other regions

Disturbance attribution

Forest land was assumed to be disturbed if included in at least one of the disturbance maps (Table 3) during the 2006–2010 time period: (1) maximum burn severity score of at least two (low) over the 5 years of fire data; (2) insect damage of at least three trees per acre over the 5 year study period; (3) within a path of a tornado or a buffered region around the hurricane path where wind speeds typically exceeded 95 miles per hour (category 2 hurricane)[2] between 2006 and 2010; (4) converted to agriculture, barren land or settlement in the NLCD layer between 2006 and 2011 (considered as deforestation events); or (5) had an average drought intensity score of more than two in the NDMC Drought Monitor map between the years of measurement. For fire and insect disturbance, three levels of disturbance intensity were assigned based on burn severity score (from the MTBS dataset) or insect damage per acre (from the Aerial Detection Survey), respectively. Two levels of wind disturbance intensity were assigned and areas determined to have been converted to agriculture or settlement were assumed to experience one uniform intensity of disturbance. All other forest land was assumed to be undisturbed between 2006 and 2010. In areas where multiple types of disturbance were identified within a 1 ha forest land pixel, we assumed only one disturbance type was driving the C loss. Disturbance type priority was set based on the intensity of the disturbance and level of confidence in the data sets. In general, more intense disturbances and higher quality products took priority over less intense disturbances and those products assessed as having more uncertainty. The disturbance location and intensity products were assumed to be in the following quality order, from least to most inherent uncertainty: conversion, fire, wind, insect damage. For instance, a pixel identified as experiencing an intense fire disturbance and a low intensity insect disturbance was assigned the high intensity fire disturbance as the single disturbance driving loss. This assumption simplified the processing but added additional uncertainty to the estimates. The assigned disturbance type priority varied across multiple iterations of our uncertainty analysis. It was not possible to attribute harvest disturbance to specific pixels, therefore C losses from harvest were estimated at the county scale using TPO data.

Estimation of net carbon change

Net carbon change from fire, wind, insect damage, land use change, and drought

If a hectare of forest land in the US was categorized as disturbed between 2006 and 2010 based on the disturbance maps, then the intensity and type of disturbance was identified. The pixel was then linked to an annualized percent net change in C stock estimate, based on its identified category in the FIA-based lookup tables. These annualized percent change values were multiplied by the initial base C stock in 2005 in each pool (aboveground biomass, belowground biomass) and multiplied by 5 years to estimate total net change in C within the pixel between 2006 and 2010.

Net carbon change from harvest

Annual C losses associated with harvest activities were estimated using mill surveys compiled into the USDA

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[2] This wind speed threshold was selected based on the Saffir Simpson Hurricane Wind Scale, which indicates that trees start to be uprooted and fall at category 2 sustained wind speeds between 96 and 110 mph. The hurricane tracks were buffered to a symmetrical width of 100 km.

Table 2 Look-up table of annual fractional change (average = μ; standard error = σ) in aboveground carbon (AGC) and belowground carbon (BGC) in undisturbed forests, based on FIA plot data

Region	Forest type	Drought	Initial C	n	AGC μ	AGC σ	BGC μ	BGC σ
North	Softwood	No	<25	5167	0.064	0.135	0.080	0.199
North	Softwood	No	25–50	3459	0.023	0.034	0.023	0.034
North	Softwood	No	50–100	2085	0.016	0.024	0.016	0.024
North	Softwood	No	≥100	345	0.013	0.034	0.013	0.034
North	Softwood	Yes	<25	50	0.028	0.030	0.031	0.035
North	Softwood	Yes	25–50	50	0.008	0.034	0.008	0.035
North	Softwood	Yes	50–100	12	0.016	0.040	0.016	0.040
North	Softwood	Yes	≥100	2	0.013	0.017	0.013	0.016
North	Hardwood	No	<25	12,559	0.074	0.102	0.087	0.131
North	Hardwood	No	25–50	13,656	0.025	0.036	0.025	0.036
North	Hardwood	No	50–100	14,173	0.014	0.026	0.014	0.026
North	Hardwood	No	≥100	3265	0.010	0.030	0.010	0.030
North	Hardwood	Yes	<25	19	0.016	0.058	0.016	0.062
North	Hardwood	Yes	25–50	12	0.006	0.040	0.006	0.041
North	Hardwood	Yes	50–100	7	0.001	0.026	0.000	0.027
North	Hardwood	Yes	≥100	1	0.006	NA	0.005	NA
South	Softwood	No	<25	3648	0.314	0.355	0.452	0.621
South	Softwood	No	25–50	2940	0.082	0.069	0.085	0.072
South	Softwood	No	50–100	2345	0.039	0.049	0.039	0.050
South	Softwood	No	≥100	673	0.021	0.050	0.020	0.051
South	Softwood	Yes	<25	464	0.340	0.407	0.487	0.694
South	Softwood	Yes	25–50	348	0.081	0.071	0.084	0.074
South	Softwood	Yes	50–100	299	0.038	0.039	0.038	0.041
South	Softwood	Yes	≥100	110	0.020	0.038	0.020	0.039
South	Hardwood	No	<25	6585	0.133	0.191	0.176	0.291
South	Hardwood	No	25–50	6180	0.040	0.044	0.041	0.045
South	Hardwood	No	50–100	8244	0.021	0.032	0.021	0.032
South	Hardwood	No	≥100	2697	0.014	0.032	0.014	0.032
South	Hardwood	Yes	<25	630	0.140	0.184	0.185	0.272
South	Hardwood	Yes	25–50	498	0.042	0.062	0.044	0.064
South	Hardwood	Yes	50–100	756	0.021	0.029	0.021	0.030
South	Hardwood	Yes	≥100	275	0.011	0.029	0.011	0.029
West	Softwood	No	<25	56	0.061	0.102	0.079	0.123
West	Softwood	No	25–50	45	0.027	0.048	0.028	0.049
West	Softwood	No	50–100	61	0.022	0.026	0.022	0.027
West	Softwood	No	≥100	80	0.014	0.019	0.014	0.019
West	Softwood	Yes	<25	0	*0.310*	*NA*	*0.443*	*NA*
West	Softwood	Yes	25–50	0	*0.072*	*NA*	*0.075*	*NA*
West	Softwood	Yes	50–100	0	*0.037*	*NA*	*0.037*	*NA*
West	Softwood	Yes	≥100	0	*0.020*	*NA*	*0.020*	*NA*
West	Hardwood	No	<25	33	0.037	0.055	0.043	0.061
West	Hardwood	No	25–50	26	0.023	0.026	0.025	0.028
West	Hardwood	No	50–100	45	0.026	0.041	0.027	0.043
West	Hardwood	No	≥100	38	0.019	0.025	0.020	0.027
West	Hardwood	Yes	<25	0	*0.137*	*NA*	*0.180*	*NA*
West	Hardwood	Yes	25–50	0	*0.041*	*NA*	*0.043*	*NA*

Table 2 continued

Region	Forest type	Drought	Initial C	n	AGC μ	AGC σ	BGC μ	BGC σ
West	Hardwood	Yes	50–100	0	*0.021*	*NA*	*0.021*	*NA*
West	Hardwood	Yes	≥100	0	*0.011*	*NA*	*0.011*	*NA*

Italics imputed from other regions

Table 3 Fourteen independent datasets were integrated and used to produce net carbon change estimates by disturbance type

Product	Source	Spatial coverage	Temporal coverage	Url
Tree cover Tree cover change	[8]	Complete CONUS	Tree cover: single snapshot in 2000 Loss: annual 2001–2010 Gain: 2000–2012	http://earthenginepartners.appspot.com/science-2013-global-forest/download_v1.1.html
Fire	Monitoring trends in burn severity	Complete CONUS	Annual 2006–2010	http://www.mtbs.gov/products.html
Wind	NOAA's storm prediction center—tornado tracks	Complete CONUS	Annual 2006–2010	http://www.spc.noaa.gov/gis/svrgis/
Wind	NOAA's storm prediction center—hurricane paths	Complete CONUS	Annual 2006–2010	http://nhc.noaa.gov/gis/
Insect	USFS aerial detection survey	Sub-set of CONUS	Annual 2006–2010	http://www.fs.fed.us/foresthealth/technology/adsm.shtml
Forest type	National land cover database—hardwood or softwood	Complete CONUS	Single snapshot in 2000	http://www.mrlc.gov/
Conversion	National land cover database	Complete CONUS	Snapshots in 2006 and 2011	http://www.mrlc.gov/
Drought	NDMC drought monitor	Complete CONUS	Weekly between 2006 and 2011	http://droughtmonitor.unl.edu/
Timberlands	Mark Nelson USFS for 2007 resources planning act	Complete CONUS	Snapshot in 2007	N/A
Biomass density Carbon stocks	Sassan Saatchi	Complete CONUS	Snapshot in 2005	http://dx.doi.org/10.3334/ORNLDAAC/1313)
Harvest	USFS timber products output	Combined county CONUS	Survey in 2007	http://www.fia.fs.fed.us/program-features/tpo/
FIA	USFS forest inventory and analysis program	Sites in CONUS	Between 1997 and 2013	http://www.fia.fs.fed.us/

TPO database for the year 2007. Due to the periodic nature of the TPO report for 2007 data, harvest emission estimates were assumed to be representative for all 5 years included in our analysis (2006–2010). Volumes of roundwood products, mill residue and logging residues were converted to biomass using oven-dry wood densities [30]. The fraction of C in primary wood products remaining in end uses or in landfills after 100 years per product class[3] was assumed to be permanently sequestered, and was estimated from values published in Smith et al. [31]. Fuelwood, posts/poles/pilings and miscellaneous product classes were assumed to be fully emitted. Emissions from mill residues were considered equal to the summed mill residues from fuel by-products, miscellaneous by-products and unused mill residues, plus emissions from fiber by-products. All fiber by-products were assumed to form pulp and to follow the emissions assumptions of pulp products. All logging residues were assumed to be emitted. Timberlands were delineated based on the boundaries of the US timberlands map (Table 3), and annual net C gains within timberlands were estimated following the look-up tables for growth in undisturbed forests as described below.

Net carbon change from forest growth/regrowth

Forest land in the US that did not experience deforestation through land use conversion or significant damage by wind, insect, fire, or drought over the analysis period, as well as new forest land (i.e., afforestation/reforestation), were linked to values of annual net change

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[3] The TPO and Smith et al. [31] product classes were mapped to one another as follows: Sawlog = softwood/hardwood lumber (depending on species); veneer = softwood plywood; pulp = paper; composite = oriented strandboard.

in C stock, based on the area's identified category in the lookup tables derived from FIA measurement data. These annualized percent change values were multiplied by the initial C stock in 2005 in each pool (aboveground biomass, belowground biomass) and multiplied by 5 years to estimate total net change in C within each 1-ha pixel between 2006 and 2010.

Total annual net carbon change

The FIA-based estimated net change in C represents the sum of net C losses (caused by disturbances) and net C gains (caused by forest growth) that occurred between FIA measurement dates at the site. Similarly, our estimate of net C change (ΔC_{net}) during the 5-year period at the combined county scale was calculated as:

$$\begin{aligned}\Delta C_{net} = {} & \Delta C_{undist} + \Delta C_{A/R} + \Delta C_{conversion} \\ & + \Delta C_{timberlands} + \Delta C_{insect} + \Delta C_{fire} \\ & + \Delta C_{wind} + \Delta C_{drought}\end{aligned}$$

where ΔC_{undist} is the net C change in forest land outside of timberlands that did not experience land use conversion or significant damage by wind, insects, fire or drought. $\Delta C_{A/R}$ is the net C change in new forest land. $\Delta C_{conversion}$, ΔC_{wind} ΔC_{insect}, and ΔC_{fire} represent the net C change in forestland that was converted or significantly disturbed by conversion, wind, insects, and fire, respectively. $\Delta C_{drought}$ is the net C reduction in sequestration in forest land experiencing drought from what was expected during non-drought periods. $\Delta C_{timberlands}$ is the net C change on timberlands (as delineated by the timberlands map), calculated as the sum of net C gains (as estimated from FIA lookup tables) and C losses (as estimated from the TPO data, accounting for the fraction of harvested C stored permanently in the long-lived product pool). By convention, C losses are represented as positive values and C gains as negative values. Consequently, various forms of disturbance result in a weaker (i.e., less negative) overall sink than would occur otherwise in the absence of disturbance.

Uncertainty analysis

We estimated statistical bounds for the estimates of net C change by conducting a Monte Carlo uncertainty analysis [32]. The four sources of uncertainty included in the simulation were associated with the forest biomass density maps, the stock-change lookup tables derived from FIA data, each of the disturbance maps, and the TPO data. The simulation was conducted at the combined county scale. Uncertainty in the biomass density maps was derived from a secondary simulation in which the input datasets were resampled to generate 100 replicate training datasets, or realizations, that had the same qualities of the original training dataset, but different random error. A new MaxEnt model was fit to each of these 100 replicated datasets and used to create 100 full resolution biomass maps. Uncertainty in the FIA-based ΔC values were calculated using the variance in the look-up tables:

$$uncertainty\% = \frac{\frac{\sigma}{\sqrt{n}} * 1.96}{\mu} * 100$$

Uncertainty in the area affected by disturbance was estimated to be 30%, with an estimated 5% bias in under reported area. We conducted the simulation using three separate rule sets for selecting a disturbance type for pixels identified as experiencing multiple disturbances during the 5-year study period. Uncertainty in the TPO data at the combined county scale was also assumed to be 30%.

We ran 10,000 Monte Carlo simulations with stochastic elements in place for the four uncertainty components. We assumed that 80% of the randomly generated error was random and 20% of the error was systematic within the simulation. To implement this assumption, we estimated the error associated with each component twice—once at the simulation iteration level and again for each individual combined county. The iteration level uncertainty was multiplied by 0.2 before it was added to the original combined county estimate, while the combined county level stochastic element was multiplied by 0.8 before it was added. In this way, we accounted for both random error as well as systematic error in our estimates.

This uncertainty analysis was intended to provide context to the estimates and assist in the process of identifying methods and data in need of refinement or replacement. The uncertainty analysis is not exhaustive, in the sense that additional sources of uncertainty exist that are not accounted for in the analysis presented here. These additional sources include but are not limited to (a) potential temporal mismatch between the biomass data providing initial carbon stocks in 2005 and the activity data beginning in 2006 and (b) uncertainty in the equations and factors used in the FIA to convert tree measurements to estimates of wood volume and carbon stocks. Given these additional sources of uncertainty, the uncertainty bounds presented here are almost certainly an underestimate of the actual uncertainty.

Results

Forest land in the conterminous US, as defined here totaling 221 million ha in 2005, sequestered -460 ± 48 Tg C year^{-1} between 2006 and 2010, while average C losses from forest disturbances were 191 ± 10 Tg C year^{-1}. Combining estimates of net C gains and net C losses results in net C change of -269 ± 49 Tg C year^{-1} (Fig. 2). These results are broadly

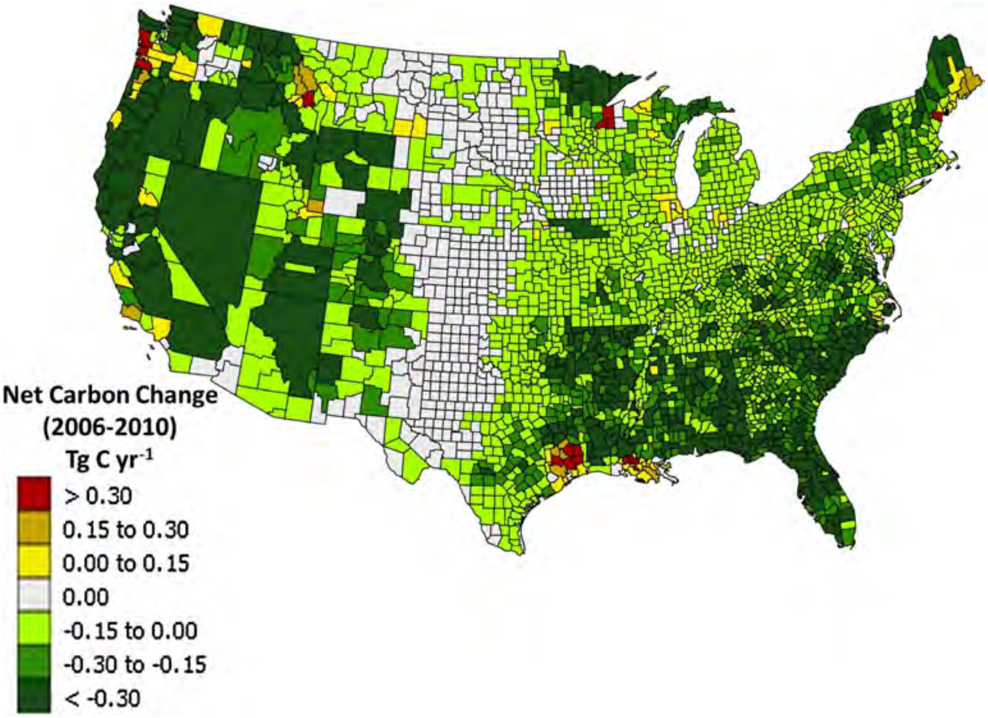

Fig. 2 Average annual net carbon change (Tg C year^{-1}) at the combined county scale across the CONUS. Most combined counties (91%) are net C sinks while areas with extensive forest disturbance can be net C sources to the atmosphere

consistent with estimates reported in the US. GHG inventory for forests in 2010 (−293 Tg C year^{-1}, [33]) but we estimate a larger net sink than reported in Zheng et al. [28] (−181 Tg C year^{-1}), although the spatial and temporal domains varied across these analyses, as did the C pools included.

New forests, averaging 0.4 million ha per year, sequestered -8 ± 1 Tg C year^{-1}, while deforestation, averaging 0.1 million ha per year, resulted in C losses of 6 ± 1 Tg C year^{-1}. Forest land remaining forest land lost 184 ± 10 Tg C year^{-1} to disturbance (13% from natural disturbance, 87% from harvest); these were compensated by net carbon gains of 452 ± 48 Tg C year^{-1}, 75% of which occurred within timberland areas (Table 4). C losses from natural and human induced disturbances reduced the potential net C sink in US forests by 42% compared to the potential sink estimated without disturbance effects included, an estimate that is similar to other studies [28, 34].

Regional variation in net C change across the nation was substantial. The South sequestered more C in growing forests (-271 ± 28 Tg C year^{-1}) than the North (-97 ± 10 Tg C year^{-1}) or the West (-92 ± 11 Tg C year^{-1}), while at the same time losing more C to the atmosphere from disturbances (105 ± 6 Tg C year^{-1}) than the other regions (41 ± 2 Tg C year^{-1} for the North and 44 ± 3 Tg C year^{-1} for the West). Forest C change in the South was substantial, in terms of both C losses and gains, because this region is home to a majority of the wood harvest occurring in the US (60% of all C loss from harvest occurred in the South), and is therefore also home to the largest area of regenerating forests that are sequestering C at high rates. At the state level, the highest C losses occurred in the forests of Georgia, Alabama, Washington, Mississippi, Louisiana, and Oregon, with each of these states losing more than 11 Tg C year^{-1} (Table 5). Georgia, Florida, Alabama, Mississippi, and North Carolina gained the most forest C in the time period, with each sequestering at least 24 Tg C year^{-1}. C gains exceeded C losses in all states. Forests in approximately 6% of combined counties were a net source of C to the atmosphere (Fig. 2).

We estimated net C losses from six separate disturbance processes: fire, insect infestation, wind, timber harvest, land use conversion, and drought (Fig. 3). C losses from harvest (162 ± 9.9 Tg C year^{-1}) were more than five times higher than losses from all other processes combined (30 ± 2.6 Tg C year^{-1}). Fire (7 ± 1.0 Tg C year^{-1}), wind (5 ± 0.7 Tg C year^{-1}), insect infestation (10 ± 1.3 Tg C year^{-1}), and deforestation (6 ± 0.7 Tg C year^{-1}) each contributed a similar magnitude of C losses across the CONUS, while drought

Table 4 Average annual net C change (Tg C year^{-1}) across US forests between 2006 and 2010, disaggregated into categories of non-forest land to forest land, forest land to non-forest land, and forest land remaining forest land

Category	Area (Mha year^{-1})	Net C gain (Tg C year^{-1})	Net C loss (Tg C year^{-1})
Non-forest land to forest land	0.4	−8 ± 1	
Forest land to non-forest land	0.1		6 ± 1
Forest land remaining forest land	221.1	−452 ± 47	185 ± 10
Insect damage	0.9		9 ± 1
Forest fire	0.6		7 ± 1
Wind damage	0.6		5 ± 1
Drought	0.8		1 ± 0
Timberlands	152.0	−342 ± 42	162 ± 10
Undisturbed forest	54.9	−109 ± 19	
Total	221.6	−460 ± 48	191 ± 10
Net C change			−269 ± 49

Results are further disaggregated by disturbance type within the forest land remaining forest land category

accounted for about 1 ± 0.2 Tg C year^{-1}. Individual disturbances had spatially distinct distributions (Fig. 4a). On average, drought affected areas had C sequestration rates 20% lower than drought-free areas.

C losses in the South were highest (105 ± 6 Tg C year^{-1}) with the highest fractional contributions from harvest (92%) and wind (5%), with a particularly high concentration of loss coming from the South Central region (including the states of Texas, Oklahoma, Mississippi, Louisiana, Kentucky, Tennessee, Alabama, and Arkansas; Fig. 4b). The West had the second highest C loss (44 ± 3 Tg C year^{-1}) with significant contributions from harvest (66%), fire (15%), and insects (13%). The North had the lowest C loss (41 ± 2 Tg C year^{-1}) with most significant proportional contributions coming from harvest (86%), insect damage (9%), and conversion (3%).

Our results can also be used to estimate net C impacts of localized disturbances at finer spatial scales. A tornado struck Lakewood, Wisconsin on 7 June 2007 and caused severe forest damage, resulting in net C loss of more than 0.3 Tg C across a 13,000 ha swath (Fig. 5a). The wild fire in southern California's Santa Barbara County, termed the "Zaca" fire, started on 4 July 2007 and caused extensive damage to more than 97,000 ha of forest in the Los Padres National Forest, resulting in net C loss of more than 4 Tg C (Fig. 4b).

The highest fractional contribution of C loss in all states was from harvest (Table 4), and 64% of these losses were from logging residues [both above- (19%) and belowground (23%)] and mill residues (22%). Across all wood product classes, the production of pulpwood resulted in the highest forest C losses (26 Tg C year^{-1}), followed by saw logs (18 Tg C year^{-1}), although a high proportion of C in saw logs is in use or in landfills, both which are considered to be long-term C storage (Fig. 6).

Discussion

Comparison with other studies

We estimate that Hurricanes Gustav and Ike in 2008, the only two hurricanes above category 2 to make landfall during the study period, damaged forests in Texas and Louisiana and led to net C change of more than 22 ± 2 Tg C (or 4 ± 0.5 Tg C year^{-1} on average over the 5 year period). Other studies report average annual C loss in US forests due to hurricane damage in the 20th century of 14 Tg C year^{-1} [35]. Zhou et al. [36] estimate total C emissions from wood harvest in 35 eastern US states as 168 Tg C year^{-1} between 2002 and 2010, while our estimate for the same geographic extent is 132 ± 8 Tg C year^{-1} between 2006 and 2010. Other national scale estimates of emissions from wood harvest are lower, such as that of Williams et al. [37] (107 Tg year^{-1} in 2005) and Powell et al. [34] (74 Tg C year^{-1} between 1986 and 2004). Hicke and Zeppel [38] estimated that bark beetles and fire together resulted in gross emissions of 32 Tg C year^{-1} in the western US between 1997 and 2010. We estimate that insects and fire resulted in net C change of 17 ± 2 Tg C year^{-1} between 2006 and 2010. We conclude that, given the different spatial extents, time periods and C pools included, results from our analysis that cover all disturbance types are broadly consistent with these and other more specialized studies (see Williams et al. [39] for a comprehensive review).

Priorities for improved forest carbon change estimates

Results generated from this analysis are dependent on the algorithm that assigns each hectare of forest land to a category that is then associated with a C stock change value. By including spatial data sets of carbon stocks and disturbance from remote sensing observations, the

Table 5 State level estimates of forest area in 2005 (millions of ha), net C gains, net C losses, and net C change (Tg C year^{-1}) together with the percent of C loss attributable to harvest, drought, fire, wind, insect infestation, and land use conversion within the state

State	Forest area	C gain	C loss	Net C change	Fire (%)	Insect (%)	Wind (%)	Conversion (%)	Drought (%)	Harvest (%)
Alabama	8.5	−27.3	12.5	−14.9	0	1	0	1	0	97
Arizona	2.0	−2.4	0.4	−1.9	22	0	1	0	0	77
Arkansas	7.4	−22.6	8.6	−14.0	1	2	0	2	0	95
California	9.3	−16.8	9.4	−7.4	32	0	7	1	0	60
Colorado	5.1	−6.7	0.3	−6.3	8	0	0	1	0	92
Connecticut	0.9	−1.2	0.2	−1.0	0	0	1	31	0	68
Delaware	0.2	−0.2	0.1	−0.1	0	0	0	4	0	95
District of Columbia	<0.1	0.0	0.0	0.0	0	0	0	100	0	0
Florida	6.4	−28.5	6.3	−22.2	3	0	0	3	0	94
Georgia	9.4	−33.2	14.4	−18.8	1	1	0	2	0	96
Idaho	7.1	−10.2	4.9	−5.3	29	0	23	0	0	48
Illinois	2.3	−2.8	1.1	−1.7	0	0	0	3	0	97
Indiana	2.3	−2.8	1.7	−1.1	0	0	3	1	0	95
Iowa	1.2	−1.5	0.4	−1.1	0	1	0	3	0	96
Kansas	0.9	−1.1	0.2	−0.9	0	1	0	3	0	95
Kentucky	5.7	−11.5	3.3	−8.2	1	0	0	6	0	93
Louisiana	5.4	−18.0	11.1	−6.9	0	19	0	1	0	79
Maine	6.8	−7.7	6.7	−0.9	0	0	15	1	0	84
Maryland	1.2	−1.5	0.8	−0.8	0	0	6	7	0	86
Massachusetts	1.5	−1.9	0.6	−1.3	0	0	4	18	0	78
Michigan	8.5	−10.3	4.3	−6.0	0	0	1	1	11	87
Minnesota	7.7	−9.5	3.2	−6.3	1	0	3	1	0	96
Mississippi	7.0	−24.3	11.6	−12.7	0	2	0	2	0	96
Missouri	7.1	−8.7	2.7	−6.0	1	2	0	4	0	93
Montana	7.3	−8.6	5.0	−3.5	14	0	49	0	0	37
Nebraska	0.3	−0.4	0.1	−0.2	2	1	0	0	0	97
Nevada	0.7	−0.8	0.1	−0.7	15	0	0	0	0	84
New Hampshire	2.1	−2.6	0.8	−1.8	0	2	4	6	0	88
New Jersey	1.0	−1.3	0.5	−0.8	2	0	40	14	0	43
New Mexico	2.6	−3.2	0.3	−2.8	33	0	16	0	0	51
New York	8.3	−10.7	3.1	−7.6	0	0	5	4	0	91
North Carolina	7.6	−23.7	9.6	−14.1	0	0	0	1	2	95
North Dakota	0.2	−0.3	0.0	−0.3	0	1	0	2	0	96

Table 5 continued

State	Forest area	C gain	C loss	Net C change	Fire (%)	Insect (%)	Wind (%)	Conversion (%)	Drought (%)	Harvest (%)
Ohio	3.6	−4.4	1.2	−3.2	0	0	7	7	0	86
Oklahoma	3.6	−9.0	1.6	−7.3	2	2	0	3	0	94
Oregon	9.2	−20.6	11.1	−9.6	4	0	2	6	0	88
Pennsylvania	7.6	−9.8	4.0	−5.8	0	0	13	3	0	84
Rhode Island	0.2	−0.2	0.1	−0.2	0	0	3	11	0	85
South Carolina	4.8	−18.4	6.5	−11.9	1	1	0	2	0	97
South Dakota	0.5	−0.6	0.2	−0.3	2	0	0	0	0	98
Tennessee	6.2	−14.2	4.0	−10.1	0	1	0	3	0	95
Texas	7.9	−23.3	9.8	−13.6	1	23	0	2	0	74
Utah	2.2	−2.2	0.3	−1.8	24	0	38	0	0	38
Vermont	2.0	−2.5	0.6	−1.9	0	0	2	1	0	96
Virginia	6.7	−16.5	6.1	−10.4	1	0	0	2	0	97
Washington	7.9	−17.3	11.7	−5.6	3	0	8	19	0	70
West Virginia	5.3	−6.9	2.5	−4.4	0	0	1	6	0	93
Wisconsin	7.2	−8.4	6.3	−2.0	0	1	23	0	5	70
Wyoming	2.7	−3.3	0.8	−2.5	21	0	25	0	0	54
Total	221.5	−459.5	191.1	−268.4	4	3	5	3	1	85

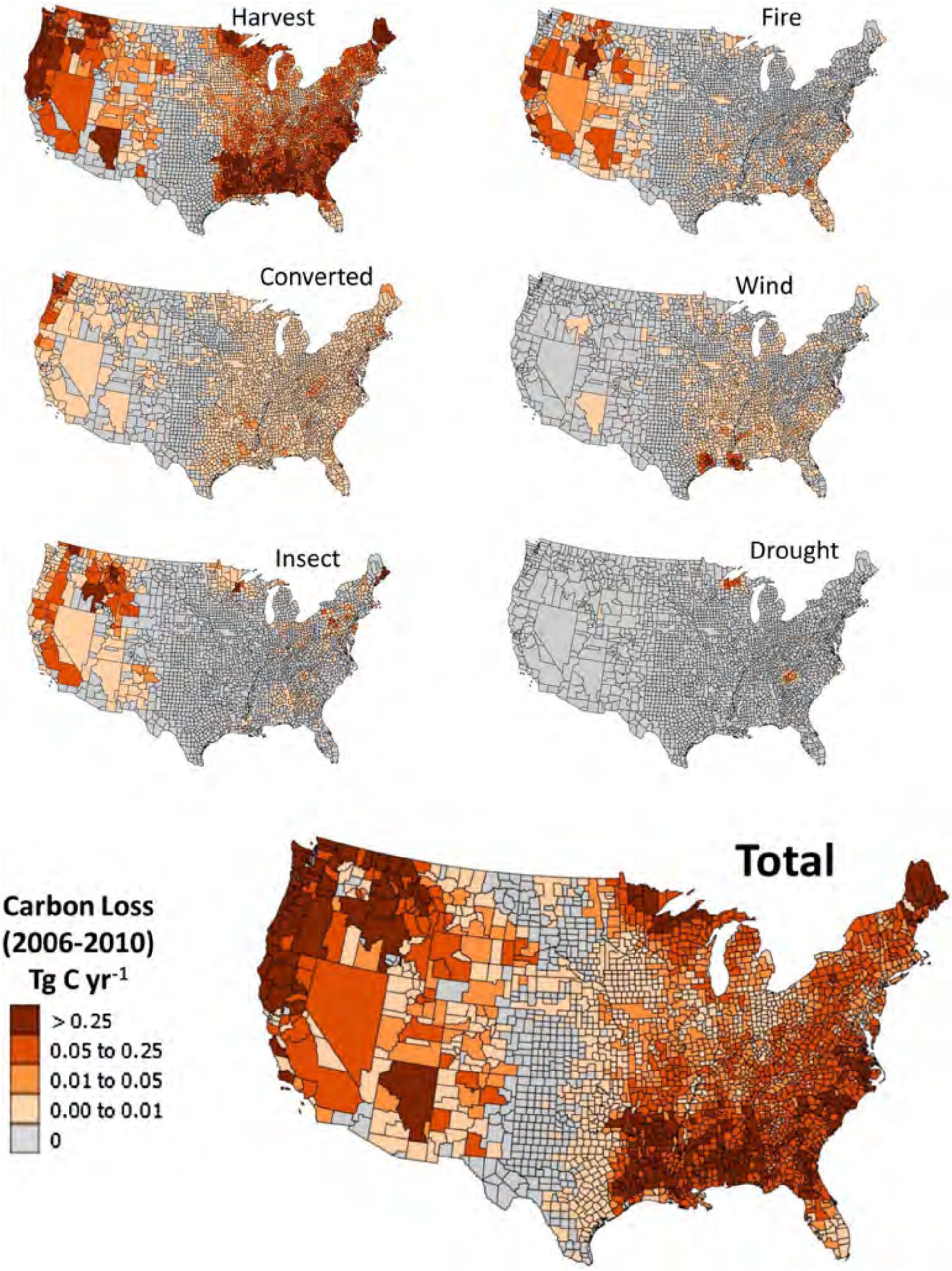

Fig. 3 Average annual net carbon loss (Tg C year^{-1}) attributed to the most likely disturbance type and estimated at the combined county scale for harvest, fire, land use conversion, wind, insect, and drought. Combining these six sources results in estimates of total annual net C loss from disturbance occurring between 2006 and 2010

methodology avoids making gross assumptions on the regional distribution of carbon stocks and disturbance, thus improving estimates of C loss. The strength of this approach is estimated in the uncertainty analysis. Our framework is therefore completely dependent on the underlying data sources and, as the data improve, so will the estimates. Although the US is among the world's leaders in technology and open data, where high quality geospatial datasets are publicly available and inventory programs are maintained by various federal and state agencies, opportunities for improvement remain.

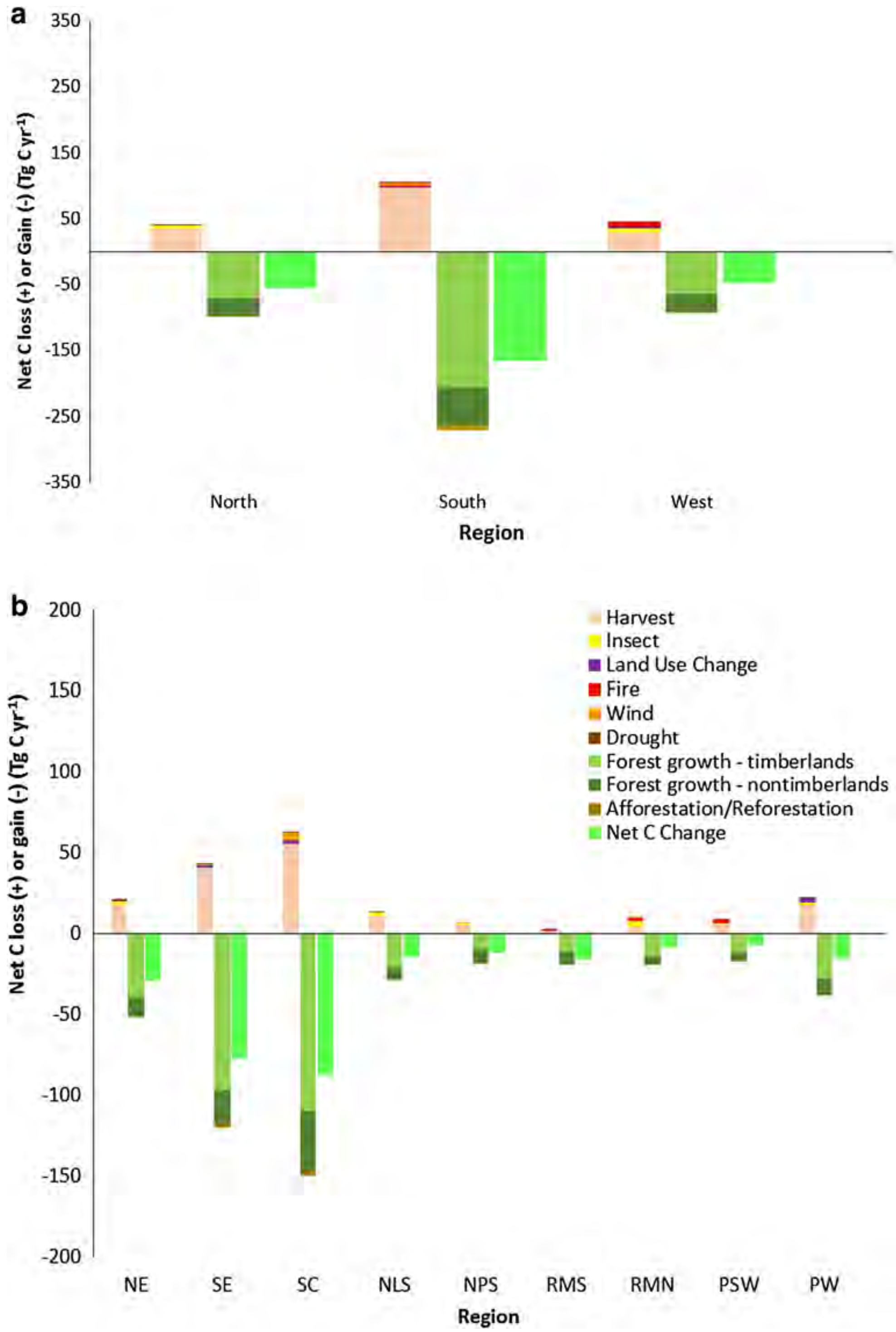

Fig. 4 Average annual net carbon change by disturbance type in **a** the North (79 million ha of forest), South (87 million ha), and West (56 million ha) regions and **b** by FIA region: northeast (NE; 41 million ha), southeast (SE; 35 million ha), southcentral (SC; 52 million ha), northern lake states (NLS; 23 million ha), northern plains states (NPS; 15 million ha), pacific west (PW; 17 million ha), rocky mountain northern (RMN; 14 million ha), rocky mountain southern (RMS; 15 million ha), and the pacific southwest (PSW; 9 million ha)

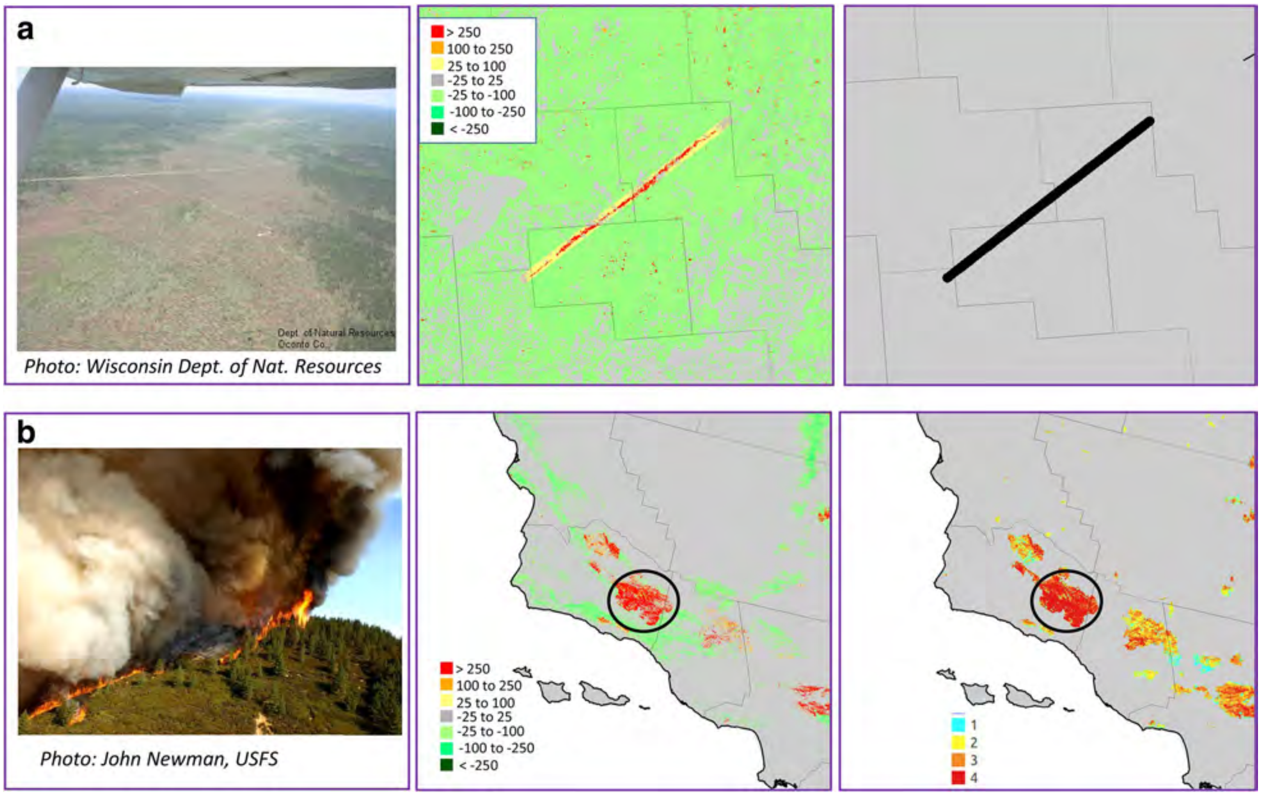

Fig. 5 The forest carbon accounting framework implemented here can be useful in assessing carbon impacts of localized disturbances. **a** 2007 tornado in Lakewood, Wisconsin. The tornado track from NOAA (*right*) resulted in extensive impacts to the forest, which is evident in an aerial photo (*left*) and in the resulting estimate of net carbon change (center, in units of Mg C ha^{-1}). **b** 2007 wild fire in southern California's Santa Barbara County, termed the "Zaca" fire. A photo of the blaze (*left*) highlights the fire intensity, which is mirrored in the burn severity map (*right*, MTBS) and the resulting net carbon change estimate (*center*, in units of Mg C ha^{-1})

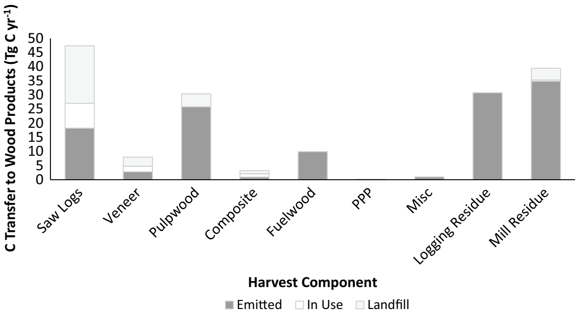

Fig. 6 Fate of C harvested from US forestlands in the year 2007, with some stored in use and landfills and the rest emitted within 100 years. *PPP* posts, poles and pilings

Priorities for FIA data collection

All forest inventory data used to estimate changes in the above- and belowground C stocks in this analysis come from FIA plots measured more than once. However, many more FIA plots have been re-measured in the North and South regions of the US than in the West. The limited number of re-measured FIA plots in the West resulted in higher uncertainties in net C stock change estimates and, in some disturbance categories, required the imputation of estimates obtained from other regions (Tables 1, 2). As the FIA program continues national implementation of an annual inventory (including re-measurement), the FIA data used in this analysis can be revised accordingly so that the sample size of plots per disturbance type increases and uncertainties decrease. Until the early 2000s, the FIA program measured only live tree attributes (e.g., tree diameter) allowing for the estimation of aboveground C and modelling of the other pools based on regions, live tree, and site characteristics (although the dead wood pool was measured in some states). Therefore, we estimated changes in the aboveground C pool using measured data while we relied on models to estimate belowground C. The FIA program is in the process of replacing model predictions of C in the dead wood, litter, and soil organic C pools with estimates obtained from measurements of these pools on a subset of FIA plots [40]. These pools, excluded from the current analysis, can be included in our framework as new data are collected.

Priorities for non-forest lands

Our analysis focused on forest areas defined in part by the NLCD data that is based on the interpretation of Landsat imagery. Comparison of our 1-ha map of carbon density of forestlands based on NLCD with high resolution lidar data over the state of Maryland has shown a significant underestimation of carbon stocks in highly fragmented and mixed urban and forest landscapes [41]. These small scale forests cover substantial areas of densely populated and fragmented landscapes of the eastern United States and appear to be highly dynamic. There is information on the disturbance and recovery of these forests over the time frame of our study, but our analysis has ignored carbon sources and sinks from these lands. By improving the carbon inventory and satellite observations to capture small scale changes, the uncertainty of carbon fluxes, particularly over the Eastern states, may be reduced. In the future (post-2020), planned satellite observations of the aboveground structure of forests by GEDI and NISAR from the National Aeronautics and Space Administration (NASA) and BIOMASS from the European Space Agency should improve the annual inventory of forest C change, as should the planned collection of FIA plot data in urban and woodland areas.

Priorities for UNFCCC reporting

Although the US has data on the magnitude of area change across land use categories, it does not have reliable and comprehensive estimates of C stocks across the entire reporting time series (e.g., 1990–2014 for the most recent UNFCCC submission) and full matrix of land use and land-use change categories to report these changes separately. For this reason, in its GHG inventory submission the US has historically deviated from IPCC guidance by reporting together C stock changes from afforestation and forest management as "forest land remaining forest land", while emissions associated with a land use conversion from forest land to a non-forest land use are reported in the non-forest land use category (per IPCC guidance). For the first time in its 2016 submission [16, 17], the US delineated net C stock changes from afforestation separately from forest land remaining forest land. An additional data need is refined C stock monitoring on non-forest lands and better coordination among land use categories to ensure complete accounting and avoidance of double counting. Our spatially resolved analysis approach allowed us to disaggregate net C change into subcategories of non-forest land to forest land (-8 ± 1 Tg C year^{-1}), forest land to non-forest land (6 ± 1 Tg C year^{-1}), and forest land remaining forest land (-267 Tg C year^{-1}). While the sole focus on net processes within the forest land use category in this study does not fully solve complete C accounting issues across all land uses, the methods used in this research are an incremental improvement toward resolving components of net C change within the forest land category, and these results can help inform and refine US reporting in the future.

Priorities for improving disturbance attribution

Insect and disease aerial detection surveys (ADS) are conducted annually using a variety of light aircraft by the USDA Forest Service in collaboration with other state and federal cooperators. Overview surveys map the current year's forest impact, and some regions have been conducting ADS for more than 60 years while others have become more active only within the last decade. Therefore, annual maps of insect damage with full coverage of all US forestlands are not available, but areas most likely to be affected by insect damage are surveyed more frequently. We accounted for the lack of continuous data coverage in our uncertainty analysis by assuming a 5% bias in underreported area. The Monitoring Trends in Burn Severity (MTBS) dataset, sponsored by the Wildland Fire Leadership Council, consistently maps the burn severity and perimeters across all lands of the US since 1984. Although 30 m resolution imagery is used for analysis, the minimum mapping unit for delineating fire perimeters is greater than 1000 acres (404 ha) in the West and 500 acres (202 ha) in the East. Therefore, burned forest areas smaller than these patch sizes were excluded from our analysis.

Priorities for wood harvest data collection

Information on the primary anthropogenic source of C loss in US forests—wood harvest—is available only at the level of combined counties. TPO data allow for the estimation of C losses from the extraction of wood products that are not readily detected by remote sensing observations, including the most recent Landsat based tree cover loss data from Hansen et al. [8]. We examined the relationship between TPO estimated C losses and a remote sensing-based estimate of C losses from forest disturbance that could not be readily linked to another disturbance type (i.e. wind, insect, fire, or conversion). For this comparative analysis, we assumed all tree cover loss pixels in Hansen et al. [8] data that could not be linked to another disturbance type were harvested, and subsequent C loss was estimated via our FIA look-up table approach. When aggregated to the state level, these two independent estimates of C loss associated with harvest were highly correlated (Fig. 7), and the remote

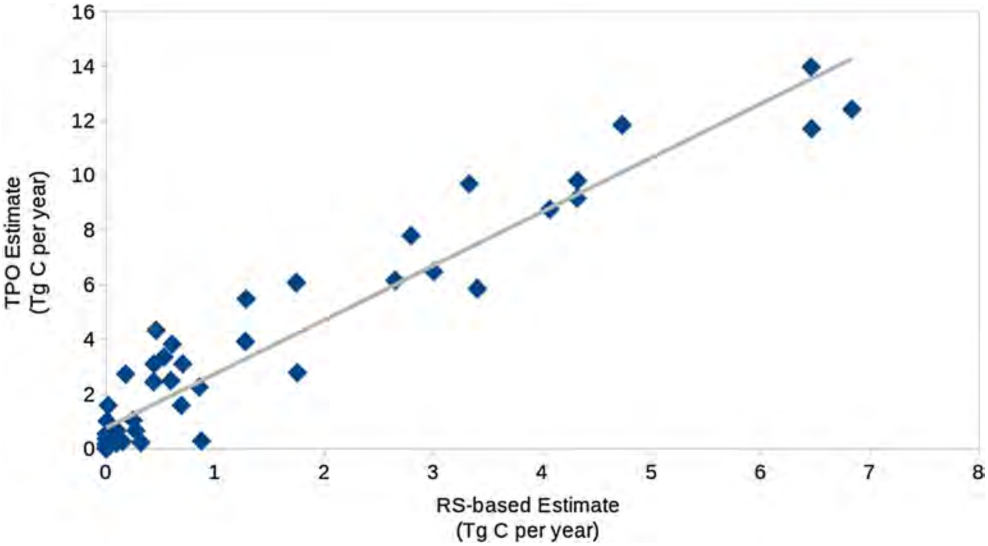

Fig. 7 Relation between C losses from harvest as estimated from timber product output (TPO) data and from an independent remote sensing-based estimate. TPO = 1.98 × RS + 767,777; $R^2 = 0.91$). Data points represent results aggregated to the state-level

sensing-based estimates of (net) C loss from harvest were approximately half of the (gross) TPO-based estimates. This provides indications that: (1) Landsat-based remote sensing observations likely miss a significant proportion of harvest activity due to partial loss, rather than full loss, of tree canopy cover; and (2) the additional C loss not identified by the remote sensing approach is spatially proximate to larger scale C losses from harvest, at least at the state scale. Increased transparency on the spatial location, timing and type of harvesting occurring across the US would allow more explicit attribution of forest C fluxes to specific forest management activities.

Managing US forests for climate change mitigation

Globally, the US ranks fourth in terms of forest area [42, 8]. Although large C losses occur from US forests as a result of an active wood products industry, particularly in the US South, 76% of the total US net carbon sink (342 Tg C year^{-1}) occurred within timberland areas, more than half of which are privately owned [43]. The income received by landowners from Intensive forest management may reduce the likelihood of forest conversion to development, but in the absence of all disturbance effects, we estimate a potential C sink between 2006 and 2010 of −460 and −436 Tg C year^{-1} if only non-harvest disturbance effects (fire, drought, wind, insect damage, land-use conversion) are considered. The US has also committed to restoring 15 Mha of forest land [44], which could further increase the C sink capacity of US forests. This implies that the US C sink could be increased substantially if existing forest land were managed to achieve this goal.

In addition to sequestering and storing atmospheric carbon, US forests also generate wood products that support the energy, industry, transport and building sectors both domestically and internationally. Given that wood harvest represents the majority of C losses from US forests, increasing the US net forest C sink would require shifts in current forest management practices as well as more refined and disaggregated information to reduce the uncertainty of these estimates and resolve these with correct estimation of net C change. For example, national debate has grown over the production of wood pellets as a renewable energy source, particularly from the southeast US, with demand driven by European policies to reduce emissions of greenhouse gases and increase the use of renewable energy. Georgia, Florida, Alabama and Virginia currently account for nearly all US wood pellet exports [45]. Although wood pellets are claimed by the industry to be made from residues at lumber mills or logging sites, the industry's growth could lead to a substantial increase in demand on Southern forests, potentially creating incentives to expand plantations. The potential of bioenergy to reduce greenhouse gas emissions inherently depends on the source of the biomass and its net land use effects; bioenergy reduces greenhouse gas emissions only if the growth and harvesting of the biomass used for energy sequesters carbon above and beyond what would be sequestered anyway [46]. This additional carbon must result from land management changes that increase tree C uptake or from the use of biomass that would otherwise decompose rapidly.

New global emphasis on climate change mitigation as one of the many benefits that forests provide gives US decision makers the opportunity to re-evaluate national and state policy agendas to consider not only the production of merchantable wood volume and biomass for bioenergy, but also enhanced C sequestration and storage for climate change mitigation. As recognized in the 2014 Farm Bill [47], there is a growing need to both reduce the uncertainty associated with estimating forest biomass and the associated monitoring of C dynamics across US forests. As it currently stands, the statistical power of detecting changes in forest C stocks exists only at large regional scales [48], disallowing the detection of C change at policy-relevant scales such as encountered in the pellet industry. Continued research to both downscale forest C inventories and correctly attribute C change to natural and anthropogenic disturbance events is needed to empower forest management policy decisions.

Conclusions

Achieving a global, economy-wide "balance between anthropogenic emissions by sources and removals by sinks" [1] will require both more emission reductions and more C sequestration from the forest sector. Results from this analysis indicate the location and estimated magnitude of C losses from different disturbances in absolute and relative terms, and can be used to track more explicitly which losses result from natural or anthropogenic disturbances. Our national net C change estimate of -269 ± 49 Tg C year^{-1} is within the range of previously reported estimates, and provides spatially explicit estimates and attribution of changes to different types of disturbances. Data are synthesized from various US agencies into a common framework, which could improve inter-agency dialogue to ensure complete accounting and to avoid double counting within and between land use categories. This work may also improve collaboration that drives a more efficient and participatory process for allocating resources towards activities that meet common goals, including an increased focus on climate change mitigation. The methodological framework and accompanying results allow US policymakers and negotiators to better understand the causes of forest C change more completely so that they can participate more effectively in domestic policy discussions about forest management and monitoring as well as in international negotiations. Integration of results from this and other studies should further enable the development of future US GHG inventories that include disturbance attribution and full land use change accounting in expectation of post-2020 commitment requirements.

Authors' contributions

NH, SH, SS, CW, SB and WS designed the study. SH, NH and TP conducted the analysis. CW, BW and GD compiled the FIA and TPO datasets. SS, YY and AF produced the biomass maps. SH implemented model runs and designed and conducted the uncertainty analysis. BB provided guidance on C modeling and on technical implementation of the methods. SH produced figures and tables. NH and SH wrote the paper. All authors reviewed the final manuscript. All authors read and approved the final manuscript.

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Competing interests

The authors declare that they have no competing interests.

Availability of data and materials

The datasets supporting the conclusions of this article are available on the ORNL DAAC website (http://daac.ornl.gov/).

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