EFFECTS OF WILDFIRE ON THE STRUCTURE AND COMPOSITION

OF MIXED OAK FORESTS IN THE BLUE RIDGE OF VIRGINIA

by

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(ABSTRACT)

Canopy mortality, species abundance, and the species composition and
origin of regeneration in mixed oak forests were studied following a
1900 ha man-caused early growing season wildfire in Shenandoah National
Park, in the Blue Ridge Physiographic Province, Virginia. Pre-fire
stands were dominated by chestnut, scarlet, and northern red oaks
growing on medium quality sites (50 year upland oak site index 16-20 m).
Permanent vegetation plots were established in four stands for each of
high and low levels of fire intensity, and unburned stands.
High-intensity fire killed 67% of the basal area and 81% of the trees,
whereas low-intensity fire resulted in mortality of 8% of the basal area
and 15% of the trees. The mean height of stem-bark char accounted for
91% and 95% of the variation in percent of the basal area and number of
trees killed by fire, respectively. Logistic regression models were
developed to predict the probability of fire-induced individual tree

mortality (top-kill) as a function of DBH and height of stem-bark char
for chestnut oak, scarlet and northern red oaks, pignut hickory,
blackgum, and red maple. Blackgum and chestnut oak were the most fire
resistant species studied. High-intensity fire resulted in the
development of a new stand, whereas low-intensity fire resulted in
thinning of the overstory and understory. Most species present in
pre-fire stand reproduced via basal sprouts, chestnut oak sprouts were
the most abundant stems taller than 1 m two growing seasons after
high-intensity fire. Tree-of-heaven was the only species to regenerate
by seed and able to match the height growth of oak basal sprouts.
This research suggests that oaks are capable of replacing themselves on
medium quality sites following wildfire-induced canopy removal in 55 to
60 year old stands.

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INTRODUCTION and JUSTIFICATION

Wildfire has long been a major ecological force affecting the structure and composition of North American forests. Many authors report the occurrence of fossil charcoal (fusain) in soil under forests as evidence of fires occurring before the evolution of man (Soper 1919, Hansen 1943.) Fires occurring before human habitation were presumably ignited by volcanic activity and lighting strikes.

The habitation of North America by the American Indians brought another source of fire to the landscape. Indians used fire to drive game, enhance berry crops, expose nuts on the forest floor, and clear land for agricultural use (Day 1953).

Forest fire intensity and frequency increased with the arrival of Euro-American man. Fire was used extensively to clear land for agricultural use and provide forage for livestock. Large, severe wildfires also became common, typically burning the slash left following logging operations.

The damaging effects of wildfire, both on human property and on the productivity of the land, were recognized during the late 1800's and early 1900's. Legislation and funding aimed at prevention and suppression of wildfires followed this realization. The efficacy of these efforts was astounding, and the acreage of land ravaged by wildfires decreased dramatically. However, the costs associated with fire suppression became increasingly high, rising to annual expenditures

in excess of $300 million in the mid-1970's for the USDA Forest Service
alone (Albini 1984).

Greater understanding and appreciation of the ecological importance
of fire in the development of existing vegetational patterns paralleled
increasing concern over continually rising costs of fire suppression. An
increasing number of ecologists and foresters began to view attempts to
exclude fire from the majority of American wildlands as not only futile,
but also unwise in some cases. The perpetuation of some economically
and aesthetically desirable forest communities is dependent on periodic
burning (Ahlgren and Ahlgren 1960). As people recognized the
potentially beneficial effects of fire in certain circumstances, the use
of prescibed fire increased. Presribed fire was used most frequently in
the management of various coniferous species, for preparation of sites
for seeding and planting, competition control, and reduction of fuel
loadings to decrease wildfire hazard. Prescribed fire has also been used
to improve the quality of wildlife forage and habitat as well as to
control certain forest pathogens.

Historically, the use of prescribed fire in forest management has
been primarily restricted to coniferous forest types. Many commercially
important hardwood species are highly susceptible to basal wounding by
fire, leading to subsequent decay and considerable loss of value (Nelson
et al. 1933). The steep slopes and poor accessibility found in much of
the eastern hardwood forest make control of prescribed fires quite
difficult. Consumption of the forest floor by intense fires can be
damaging to soil and cause severe erosion. Another important reason why

use of prescribed fire is uncommon in eastern hardwood forests is the lack of knowledge of how fire affects community processes in most hardwood forest types. In an extensive review of the fire effects literature, Simard (1983) found that fire effects have been observed in eastern hardwood forests for more than fifty years. However, less than three percent of the published fire effects literature is concerned with eastern hardwoods, indicating that scientists know less about the ecological effects of fire in eastern hardwood forests than in any other timber type where fires are common (Simard 1983). Intelligent use of prescribed fire to achieve management objectives is severely inhibited by this lack of knowledge.

On May 2, 1986, a camper ignited a wildfire in the South District of Shenandoah National Park while trying to light a campstove. The fire started near the park boundary in the watershed of Big Run, in Rockingham County, Virginia. Fire hazard was extreme due to an unusually dry Spring, and strong winds pushed the fire up into the large drainage basin of Big Run. By the time the fire had been contained three days later, approximately 1900 ha of forested land had been burned. Various forest cover types representative of those found throughout the Eastern Deciduous forest had been burned, and fire behavior had varied from low-intensity surface fires to crown fires of extreme intensity. The variable nature of the fire combined with the wide variety of forest cover types affected provided a valuable opportunity to study the ecological effects of forest fires in eastern hardwood forests.

The objectives of this study were to determine the effects of an early growing season wildfire of variable intensity on canopy mortality, and on the abundance, species composition, and origin of regeneration in a mixed oak forest in the Blue Ridge Physiographic Province of Virginia. Permanent vegetation plots were established during this study, and will allow monitoring of stand development as well as initial regeneration, which should improve understanding of the effects of fire on the regeneration and development of mixed oak forests over time. Monitoring stand development allows the study of successional processes in eastern hardwood forests, and also provides information which may facilitate the use of prescribed fire in the management of Appalachian hardwood forests in a silviculturally and environmentally sound manner.

LITERATURE REVIEW

In order to understand the effects of wildfire on canopy mortality, and on the abundance, species composition, and origin of regeneration in mixed oak forests in the Appalachians it is necessary to be familiar with the structure and composition of Appalachian oak forests, general principles of oak regeneration, characteristics of forest fires, and the effects of fire on the physical and biotic environment. The following literature review is aimed at developing this background.

Appalachian Oak Forests

The forests of the Blue Ridge of central Virginia are part of the Appalachian Hardwood Region of the Eastern Deciduous Forest (Braun 1950). One of the most common forest types in this region is the mixed oak forest. The dominant overstory species include five major species of upland oak; white oak (Quercus alba L.), northern red oak (Quercus rubra L.), chestnut oak (Quercus prinus L.), black oak (Quercus velutina Lam.), and scarlet oak (Quercus coccinea Muench.). Pignut hickory (Carya glabra (Mill.) Sweet) is commonly associated with the above oak species. The mixed oak type is most commonly found on medium quality sites, with 50 year upland oak site index ranging from 15 to 23 meters. Many higher quality sites are currently occupied by mixed oak forests, presumably because severe disturbances due to logging and fire have resulted in sub-climax communities occupying these sites (Merritt 1979).

General Principles of Oak Regeneration

The manner in which oak communities regenerate, and the
implementation of silvicultural practices to promote oak regeneration
form a complex subject, and in spite of considerable study, much remains
to be learned. Most oak species are intermediate to intolerant to shade,
therefore even-aged regeneration methods are the most appropriate means
of regenerating oak stands. Following canopy removal, the oak component
in the newly regenerating stand will originate primarily from stump
sprouts and stems of advance regeneration, including seedlings and
seedling sprouts which became established under the mature canopy. But
not all advance regeneration stems will be competitive in the emerging
stand. Small stems, or stems of low vigor, are not capable of the rapid
height growth necessary to ensure a dominant position in the developing
stand. Seedling sprouts, which have a root system older than the
above-ground portion of the stem, are often the most competitive advance
regeneration stems. Seedling sprouts arise from repeated dieback of the
above-ground portion of the stem followed by resprouting from dormant
buds at the base of the stem. Various studies of oak reproduction show
that seedling sprouts typically comprise about 75 percent of the oak
reproduction, and that the root systems may be as much as 30 years older
than the tops (Liming and Johnson 1944, Merz and Boyce 1956). Causes of
stem dieback include low light intensity under the canopy, grazing, and
fire (Liming and Johnson 1944, Ross 1982). The pattern of repeated

dieback and resprouting facilitates the buildup of a reservoir of desirable oak regenerants which will grow vigorously following release. Although oak seedlings may become established in the understory at low levels of light intensity, they require 30 percent of full sunlight to achieve maximum photosynthesis (Phares 1971). Light intensity under a closed canopy is typically below these critical levels, resulting in the development of oak seedlings with broad, flat-topped crowns, which display poor height growth upon release (Carvell 1967, Sander 1971, Ross 1982). Resprouting following stem dieback or even breakage during logging operations reestablishes a more favorable crown conformation with an active leader that is capable of vigorous height growth upon release (Carvell 1967, Ross 1982). Sander (1971) correlated stem diameter of seedling sprouts with height growth following release. Stems with a groundline diameter of at least 1.3 cm displayed increased annual height growth during the first three years following canopy removal, thus ensuring at least a codominant position in the developing stand. However, stems greater than 2.5 cm in diameter tended to produce undesirable multiple sprouts. Therefore, Sander (1971) concluded that seedling sprouts with basal diameters between 1.3 and 2.5 cm are the most desirable form of advance regeneration. Size of the rootstock, not stem diameter, is believed to be the factor controlling growth potential, but stem diameter is correlated with size of the rootstock, and was used due to relative ease of measurement. Oak seedling sprouts often have root systems with larger diameter and biomass than the above-ground portion of the plant (Sander 1971). The high root/shoot

ratio allows understory oaks on dry, nutrient deficient sites, to develop large carbohydrate reserves. Carbohydrate reserves in the root system enable oak seedling sprouts to respond to release with vigorous height growth, which makes them competitive in the developing stand. The high root/shoot ratio allows understory oaks to develop good growth potential (by developing an extensive root system), even when the nutrient and moisture capital required to sustain large above-ground biomass is limiting (Monk 1966).

Height of oak advance regeneration has also been found to be an important factor influencing competitive ability following canopy removal (Sander 1972). However, Ross and others (1986), working in mixed-oak forests in the Ridge and Valley of Virginia, found that stems smaller than 1.4 m in height were capable of sufficient growth to attain a competitive position in the developing stand. Ross and his co-authors attributed this to the fact that fast-growing competitors such as yellow-polar (Liriodendron tulipifera L.), red maple (Acer rubrum L.), and black birch (Betula lenta L.) are not abundant on these sites, and concluded that site quality and stand composition should be considered as well as stem height and density in evaluating the adequacy of oak advance regeneration. Recent work by Sander et al. (1984) in the Missouri Ozarks also indicates that size of oak advance regeneration, site quality, and species composition are important when assessing the regeneration potential of oak stands. The size of oak regeneration relative to the size of competing individuals is more important than the absolute size in determining the competitive status of oak regenerants:

if the competing individuals are small or slow-growing, then smaller oak regeneration will have greater competitive ability in the developing stand.

Stump sprouts are also capable of becoming part of the dominant stand following canopy removal. Stems cut during the dormant season tend to produce more sprouts which grow more vigorously than stems cut during the growing season, presumably the result of decreased carbohydrate reserves in the root systems of stems cut during the growing season (Kays 1985). Ross and others (1986) found that chestnut oak stumps had the highest frequency of stump sprouting and fastest growth rate of all upland oaks, followed by scarlet oak and black oak. Sprouting frequency was greatest on lower quality sites, with small, slow growing, young trees tending to sprout most frequently. However, considering only trees which sprouted, trees of larger diameter, that had displayed faster growth rates prior to harvest, produced the greatest number of sprouts per clump, and taller trees produced the tallest sprouts . Since oak sprout clumps were taller than all vegetation (closely followed by advance regeneration), frequency of sprouting exerts a more significant effect on early dominance than varying rates of sprout growth, which will likely become more important after canopy closure (Ross et al. 1986).

Numerous factors influence the development of an oak advance regeneration pool of sufficient abundance and vigor to ensure regeneration of an adequately stocked oak stand following canopy removal. Seed production, dissemination, and germination are the

initial steps in the development of the advance regeneration pool. For both the red oak group (subgenus Erythrobalanus) and white oak group (subgenus Leucobalanus) good seed crops are highly variable and unpredictable and may be produced every 2 to 10 years. In years of lower seed production the vast majority of viable seed is lost to predation by deer, bear, rodents, insects, and birds; therefore, appreciable numbers of oak seedlings will become established only in years of bumper seed crops.

Acorns of the white oak group develop in one growing season and germinate during the autumn in which they fall; whereas acorns of the red oak group take two years to develop and germinate during the spring after falling. This variation in season of acorn germination results in a greater percentage of white oak acorns successfully germinating (Carvell 1979). Carvell hypothesized that the smaller size of white oak acorns enables them to be more easily covered by litter, thereby making them less susceptible to predation. The germination of white oak acorns in the fall allows the development of a taproot, which reduces winter desiccation, a known loss of viability for red oak acorns (Carvell 1979).

In a study of the abundance of advance oak regeneration in 59 mature oak stands in West Virginia, Tryon and Carvell (1958) found that although acorn production was fairly constant across all stands, the abundance of oak advance regeneration was highly variable. They hypothesized that environmental factors controlling germination of acorns and establishment and survival of seedlings were more important

in controlling the abundance of advance regeneration than variation in acorn production. Evidence consistent with this idea was presented by Weitzman and Trimble (1957), who reported finding that the greatest amount of northern red oak advance regeneration in West Virginia and western Maryland was found on sites with fifty year upland oak site indices ranging from 15 to 18 m. Ross (1986) also found the greatest amount of oak advance regeneration in the Ridge and Valley of Virginia on sites with upland oak site indices (base age 50 years) ranging from 15 to 20 m. Results of both studies showed that the abundance of oak advance regeneration declined above and below this range of site quality.

Carvell and Tryon (1961) used multiple regression analysis to evaluate the effects of various environmental factors on abundance of oak advance regeneration. The percent of sunlight reaching the forest floor was identified as the environmental factor most closely correlated with amount of oak advance regeneration. An even closer correlation was obtained by combining percent sunlight with aspect, with open stands on relatively dry aspects (westerly through southeasterly) having the most abundant advance regeneration. This is logical, indicating that relatively shade-intolerant oak species can become established and survive underneath somewhat open canopies on relatively dry sites, where they form the climax community. The degree of stand disturbance (including grazing, fire, and thinning, resulting in some degree of canopy opening) within the past 20 years was also closely correlated with abundance of oak regeneration. Oak regeneration was more abundant

in stands disturbed more recently and more severely (Carvell and Tryon 1961). The greater amount of oak advance regeneration in more highly disturbed stands is thought to reflect the greater amount of sunlight reaching the forest floor following these disturbances, and has important silvicultural implications.

Numerous authors have documented the difficulty of regenerating oaks on high quality sites, even where oak species are dominant in the pre-harvest stand (Beck and Hooper 1986, McGee 1979, Merritt 1979). The difficulty of regenerating high quality sites to oak is caused by competition from species such as yellow-poplar, which is highly competitive on high quality sites and can out compete oaks if site requirements are met. Severe competition from shade-tolerant species such as red maple, sugar maple (Acer saccharum Marsh.), and American beech (Fagus grandifolia Ehrh.) is also a problem. The difficulty of regenerating oak forests on high quality sites suggests that oaks are seral on high quality sites. Carvell and Tryon (1961) suggest that thinnings late in the rotation may be used to stimulate development of oak advance regeneration by allowing sunlight to reach the forest floor. Sander (1979) advocates the use of a shelterwood system to achieve the same purpose. Although most research involving shelterwood cuttings to stimulate oak regeneration has yielded poor results, Sander (1979) contends that as research provides more information to improve our understanding of the principles involved, effective use of shelterwood cuttings will be useful to stimulate development of oak advance regeneration. Recent observations indicate that a bumper seed crop

immediately following the seed cut may be necessary for successful establishment of oaks using the shelterwood system.

Fire has also been identified as a potentially useful tool for promoting oak regeneration, as scientists have recognized that many of the mature oak stands of today originated under conditions of abusive logging practices followed by severe wildfires (Merritt 1979). If fire is to be beneficial in the management of oak stands, it must either increase establishment of advance regeneration prior to harvest, improve growth following cutting, or improve the competitive position of oaks, regardless of whether absolute growth rates are increased. Whether or not prescribed fire can be controlled in eastern hardwood forests, accomplish the above objectives, and be applied without unduly damaging crop trees or degrading site quality, must be addressed through research before fire will be widely used in the management of oak forests in the eastern United States.

Characteristics of Forest Fires

The ecological effects of forest fires are determined by the nature and properties of forest fires. Fire is a thermochemical reaction in which organic matter is consumed and heat, smoke, and ash are produced. Fire is a combustion reaction (one of many types of oxidation reactions), in which carbohydrates are oxidized yielding carbon dioxide and water . Combustion is a chain reaction which takes place rapidly at high temperatures (Brown and Davis 1973).

The amount of heat released during flaming and glowing combustion varies depending on the type of fuel. In general, flaming combustion rapidly releases large amounts of heat; whereas glowing combustion can produce large amounts of heat energy, with the rate of energy release typically being slower (Brown and Davis 1973).

The energy released by the burning of forest fuels sustains the combustion reaction and is called the heat of combustion. In natural conditions, however, the complete heat of combustion is never released during the burning of forest fuels. The energy loss is due to such factors as incomplete combustion of fuels and the heat energy required to release bound water from the fuel and to vaporize moisture contained in the fuel. Heat yield is the amount of heat produced by combustion of fuels after accounting for these losses (Brown and Davis 1973).

Heat energy released in forest fires is transferred by three primary mechanisms: conduction, convection, and radiation. Conduction is the means by which heat is transferred through solids. Convection is the transfer of heat by the movement of a gas or liquid. Radiation involves the transfer of heat with no movement of matter. Radiative heat travels in straight lines at the speed of light. These three mechanisms of heat transfer exert important influences on the behavior and ecological effects of forest fires.

Convection and radiation transfer heat to the surface of both unburned and burning fuel, which causes the preheating of fuels ahead of an advancing fire front as well as providing heat which may damage living plants and animals. Conduction is responsible for the transfer

of heat through individual pieces of fuel and into the internal structures of living organisms. Convection has the greatest impact in determining fire behavior. The rate of spread of fires is strongly influenced by the geometry and arrangement of fuels and by convective heat transfer as affected by topography and wind. The characteristic large vertical smoke plume associated with forest fires is formed by convective heat transfer and is called the convection column. Firebrands (pieces of burning material) transported into the atmosphere by the convection column, transferred by wind, and ultimately deposited some distance downwind are responsible for the phenomenon known as spotting.

The physical characteristics of forest fuels are very important in determining fire behavior and effects. The total quantity of fuel per unit area (fuel loading) has obvious significance with regard to fire behavior. Fuel geometry and arrangement are also important. For a given fuel loading the rate of burning increases as the surface area of fuel increases. Moisture content of fuel exerts a significant effect on rate of burning and heat yield. The moisture content of fuel, especially nonliving fuel, varies over a wide range. The environmental factors of rainfall, temperature, and relative humidity exert their influence on fire hazard and behavior largely through their influence on the moisture content of fuels (Brown and Davis 1973).

Forest fire behavior

Forest fires can be divided into three categories; 1) ground fires,

which burn within the soil organic layer; 2) surface fires, which burn in surface litter and low vegetation; and 3) crown fires, which burn in the crowns of trees forming the forest canopy. Any given forest fire may actually be a combination of more than one of these types of fire (Brown and Davis 1973).

Fires are also classified according to the direction in which they spread relative to wind direction. Headfires move in the same direction as the wind and tend to spread relatively rapidly. Backfires move into the wind and spread rather slowly. Different flanks of the same fire may behave as headfires or backfires (Brown and Davis 1973).

Fire behavior is complex and highly variable, both between fires and within a single fire. In spite of obvious difficulties, much research has been performed in an attempt to develop quantitative measures of fire behavior, with remarkable success. Quantitative descriptors of fire behavior such as fire intensity, rate of spread, and heat per unit area provide more precise means of expressing and comparing the behavior of different fires than commonly used subjective terms such as hot, cold, smoldering, running, and spotting. Quantification of fire behavior also facilitates the necessary correlation of fire behavior with the ecological effects of fire (Rothermel and Deeming 1980). Although flames in forest fires are random, pulsating events and exhibit considerable variability, several quantitative measures have been developed which are useful in describing fire behavior.

Frontal fire intensity, or fireline intensity, defined as the rate

of energy output per unit length of flame front, is one such

quantitative measure (Byram 1959). Fire intensity is directly related

to flame size and is solely a physical attribute of the fire itself

(Alexander 1982). All active fire fronts have three characteristics in

common: they spread, consume fuel, and produce heat energy in flaming

combustion (Van Wagner 1970). Fire intensity, in units of kilowatts per

meter, can be determined by the following equation (after Byram 1959,

Eq. 3.3):

$$I = Hwr \hspace{4cm} [1]$$

where:

I = fire intensity (kW/m)

H = heat yield (kj/kg)

w = weight of fuel consumed in the active flaming

zone (kg/m^2)

r = linear rate of spread of fire front (m/s)

Fires burning in areas containing fuels of similar heat yield and

fuel loading can have identical fire intensities by having different

rates of spread and fuel consumption. Van Wagner (1965) recommends

reporting rate of spread as well as intensity to give a more complete

description of fire behavior. It is also sensible to quote rate of

spread in meters per minute as opposed to the units of meters per second

used in calculations since meters per minute facilitates visualization

of the rate of spread of the fire front, especially for slow-moving

fires (Van Wagner 1978).

The heat released per unit area is also a useful quantity to be used for correlation of fire behavior and fire effects and is calculated by dividing fireline intensity by rate of spread (Rothermel and Deeming 1980).

$$H_A = I/r \qquad\qquad [2]$$

where:

H_A = heat released per unit area (kj/m^2)

I = fireline intensity (kW/m)

r = rate of spread (m/s)

Heat released per unit area essentially conveys the same information as quoting both fire intensity and rate of spread. The interpretation of both of these approaches is that if fireline intensity is constant, a slower rate of spread will direct more heat to the site (and any organisms present).

Residence time, the length of time required for the zone of active combustion to pass a given point, is another quantity useful for relating fire behavior to fire effects (Rothermel and Deeming 1980). Living tissue is damaged almost instantaneously by exceptionally high temperatures, but the same degree of damage can be inflicted by exposure to lower temperatures if the exposure is of long enough duration (Rouse 1986). For example, headfires tend to burn with greater intensity than

backfires, but with faster rates of spread. Backfires may direct as much heat to the site due to their longer residence time; this can exert a large effect on living organisms (Harmon 1980). Residence time can be used to account for this phenomenon and can be measured with a stopwatch, or with thermocouples wired to a strip chart recorder, or calculated by the following equation (Rothermel and Deeming 1980).

$$t_r = D/R \tag{3}$$

where:

t_r = residence time (min)

D = flame depth, distance at the base of flames

from the leading edge to the rear (m)

R = rate of spread (m/min)

The relationship of fire intensity to the length of flames at the fire front can be expressed mathematically and used to estimate fire intensity as a function of flame length, since field measurements of flame length are easier to obtain than are measurements of heat yield and weight of fuel consumed. The relationship between fire intensity and flame length (from Byram 1959, Rothermel and Deeming 1980) is:

$$I = 258(F_L)^{2.17} \tag{4}$$

where:

I = fireline intensity (kW/m)

F_L = flame length (m)

Time and temperature profiles of fires are also attributes of fire behavior that are strongly related to fire effects, but they have received little use for several reasons. Time-temperature profiles of forest fires are extremely difficult to measure, let alone predict (Rothermel and Deeming 1980). Since time-temperature profiles cannot be expressed in simple physical units, such profiles must be considered in relation to the biological and physical environment that produced them (Van Wagner 1970, Alexander 1982). The difficulties associated with describing fires in terms of temperature can be avoided by accepting the fact that fires which have similar behavior (as indicated by intensity, rate of spread, etc.) have similar temperature patterns (Van Wagner 1970).

Past fire-effects literature has tended to view fire as a binary phenomenon (Alexander 1982): either an area burned or it did not burn. As a result of this lack of consideration of fire behavior, or only using subjective qualitative descriptors, there has been little opportunity to compare and correlate results of different studies (Rothermel and Deeming 1980). In order to accurately relate the results of fire effects studies to situations beyond the study itself, and to effectively use fire to achieve specific land management objectives, it is necessary to relate fire effects to quantifiable characteristics of fire behavior (Rothermel and Deeming 1980, Alexander 1982).

The relation of quantitative measures of fire behavior to the categories of forest fires described previously facilitates greater

understanding of both the behavior of different types of fires and the information conveyed by the fire behavior parameters themselves. Headfires tend to burn with greater intensity and faster rates of spread than backfires, although backfires may direct as much or more heat per unit area to the site due to lower spread rates and longer residence time (Fahnestock and Hare 1964). Ground fires burn with relatively low intensity and rate of spread, but may release considerable heat per unit area, due to the long duration of smoldering combustion. Crown fires tend to have the greatest fireline intensity and spread rates, with surface fires generally being intermediate, but heat released per unit area may vary, depending on the interaction of fire intensity and spread rate.

It is useful to think of fire intensity as increasing in a discontinuous fashion, such as a step function, because fire intensity tends to increase in jumps as conditions become suitable for combustion of a different type or size of fuel. A good example of this is when the convection column of a surface fire transfers enough heat into the canopy that the tree crowns burst into flames. The transformation of a surface fire into a crown fire occurs rather rapidly, resulting in a tremendous increase in fire intensity within a very short time interval.

Quantitative measures of fire behavior are most easily employed in the study of fires which are set and controlled by researchers; however, many studies focus on the effects of wildfires, where measurement of fire behavior while the fire is burning may not be possible, as a result of either timing or danger to personnel. Wildfires present valuable

opportunities to study fire effects, and in spite of the above
limitations it is desirable to attempt to quantify fire behavior.
McCarthy and Sims (1935) used the percentage of mortality of oak stems
in the eight cm diameter class as a measure of fire intensity in
studying the relation of tree size and fire-induced mortality in
southern Appalachian hardwoods. Although they used a rather crude
measure of fire intensity, their work is significant in that it was a
very early attempt to quantify fire behavior in the study of fire
effects.

Van Wagner (1973) found that height of crown scorch in conifer
stands in Canada is closely related to fire intensity (R^2=0.98).
Unfortunately measurement of crown scorch is difficult in deciduous
forests during the dormant season when leaves are not present. Height of
stem-bark char has been substituted for flame length into Byram's (1959)
equation to calculate fire intensity (McNab 1977), but no research
exists supporting the validity of this procedure. Cain (1984) measured
flame length, height of crown scorch, and height of stem-bark char in a
controlled burn in a sapling-sized stand of loblolly (Pinus taeda L.)
and shortleaf pines (Pinus echinata Mill.). Neither height of crown
scorch nor height of stem-bark char, when used in the Van Wagner (1973)
and Byram (1959) equations, respectively, yielded fire intensities as
high as were calculated using direct measures of flame length (Cain
1984). However, the ratio of fire intensities on thinned and unthinned
plots (corresponding to different fuel loadings) were very similar for
all three methods of calculation. These comparisons indicate that height

of crown scorch and height of stem-bark char, although inadequate as
absolute measures of fire intensity, provide useful measures of relative
fire intensity (Cain 1984). It is likely that the relationship between
fire intensity and height of crown scorch varies depending on
environmental conditions such as ambient temperature, wind speed, and
relative humidity, as well as season of the year and moisture content of
foliage. Inexplicably, Cain (1984) reported no attempt to quantify the
relationship of crown scorch or bark char with flame length or fire
intensity using regression analysis.

Ecological Effects of Forest Fires

Forest fires, whether controlled fires set by man, or wildfires
burning out of control, exert considerable influence on the structure
and function of ecosystems. Fire effects are defined as the combination
of the immediately evident effect of fire on the ecosystem in terms of
biophysical alterations and population reductions, and long term
response to these alterations (Muraro 1971, Alexander 1982). Simard
(1983) expanded upon this by noting that primary effects of fire are
changes in an ecosystem brought about by fuel consumption, chemical
reactions, or heat, and secondary effects are changes resulting from
primary effects. Primary autecological effects of fire such as death,
injury, and stimulation are all heat related, and primary effects on
fuels, soil, and microclimate have secondary physical effects on each
other as well as on the vegetative community (Simard 1983). Fire exerts

considerable impact on ecosystems at both autecologic and synecologic levels.

The effects of fire on ecosystems occur as a result of fire altering the availability of sunlight, water, and nutrients, changing the microclimate of a site, and changing the source of regeneration (Harmon 1984). Severe fires also result in the transfer of large amounts of biomass to the detrital food chain (Harmon 1984), leading to a release of nutrients for plant uptake, which is likely an important aspect of ecosystem recovery (Bormann and Likens 1979).

Fire can alter successional trends: the most commonly held belief is that fire tends to set back succession, resulting in the perpetuation of seral communities. Some ecologists recognize the existence of fire climax communities on sites subject to frequent fires. The influence of fire on ecosystem succession depends to a large extent on the fire regime found in that ecosystem.

The fire regime is the relatively consistent pattern of fire occurrence resulting from the interaction of climate, vegetation, physiography, and ignition sources (Irving 1983). Fire frequency, intensity, size, and the season in which fires occur are key factors of fire regimes which influence the impact of fire on ecosystems. The fire regime can greatly impact species composition and forest structure (Irving 1983).

Given the above definition of fire regime it is readily discernible that fire is both an exogenous and endogenous factor. Research has provided evidence supportive of this claim in terms of both fire

occurrence and ecosystem effects. The perpetuation of oaks on high quality sites in the eastern United States, where they are seral, has been attributed to ignition of fires by man, an exogenous influence (Merritt 1979, Brown 1960). Mutch (1970) found evidence which led him to hypothesize that fire dependent communities tend to burn more readily because natural selection has favored development of characteristics which increase flammability. Longleaf pine (Pinus palustris Mill.) is commonly regarded as a species which requires recurrent fire to maintain dominance, and the high flammability of longleaf pine litter appears to facilitate the occurrence of fire. Harmon (1980), in a study of fire occurrence and effects in the southern Appalachians also found evidence that the relatively high flammability of pitch pine (Pinus rigida Mill.) and Table Mountain pine (Pinus pungens Lamb.) promoted high intensity crown fires which increased the ability of these species to dominate sites which would otherwise succeed to hardwoods such as scarlet oak and chestnut oak.

Effects of fire on soil

The effects of fire on soil have a significant long term impact on ecosystem structure and function. The degree of response of all soil properties to fire is dependent on fire intensity and the resulting degree of exposure of mineral soil to heat (Wells et al. 1979). Land productivity and soil stability are both adversely affected by excessive heat; however, land productivity may be enhanced by low-intensity fires.

Low-intensity fires which do not completely destroy the leaf litter layer can facilitate nutrient cycling, control pathogens, and do not increase the rate of erosion (Wells et al. 1979). Conversely, intense fire can completely consume the forest floor, volatilize nitrogen, disrupt soil structure, and induce water repellency in some soils. Long term effects of fire on soil are influenced by the frequency and intensity of burning and the mitigating effects of vegetative cover between successive fires.

Fire effects are typically confined to the upper layers of the soil profile. Mineral soil is a poor conductor of heat; the soil temperature 5 cm below the surface is only raised about $10^{o}C$ by a surface fire (Wells et al. 1979).

Fire alters both chemical and physical properties of soils, with effects on physical properties typically being detrimental and effects on chemical properties being either detrimental or beneficial. A study in Louisiana indicated that ten years of fire protection improved soil physical properties but not soil chemical properties (Hare 1961). Litter consumption by fire exposes mineral soil to weather, with damaging results. Soil aggregates are dispersed by raindrop impact, and macropores become clogged (Wells et al. 1979). Soil porosity, infiltration, and aeration are all reduced in proportion to the degree of organic matter consumption (Wells et al. 1979, Hare 1961). Soil erosion can also be accelerated; soil characteristics, slope, and fire intensity all exert an important influence on erosion rates (Wells et al. 1979).

The combustion of organic matter also alters the chemical properties of soils. Nitrogen is volatilized when leaf litter is burned, and nitrogen content in the upper inch of soil is reduced (Hare 1961, Wells et al. 1979). However, nutrients such as phosphorus, potassium, calcium, and magnesium largely remain in the ash, and their increased availability can cause a temporary fertilization effect (Wells et al. 1979). Nutrients in the ash are susceptible to loss by leaching and runoff; losses are greater following fires of greater intensity (Wells et al. 1979). The results of a Louisiana study indicated that potassium, phosphorus, and magnesium availability remained high for 2 years following prescribed burning (Hare 1961). Soil acidity is also decreased by fire, and this increases both symbiotic and nonsymbiotic fixation of nitrogen (Wells et al. 1979). In some cases no net change in available nitrogen in the forest floor and litter was found after 10 to 20 years of annual or periodic burning (Wells et al. 1979).

Heat from fire can kill soil microflora and fauna. A study in the Pacific Northwest found soil bacteria and actinomycetes to increase following severe fire, but soil fungi were found to decrease (Hare 1961). In general fungi are destroyed more easily than bacteria, and both are killed more easily as soil moisture content increases, due to the higher thermal conductivity of moist soil (Wells et al. 1979).

Effects of fire on plants

Heat from fires can cause injury, mortality, and increased seed

production in plants. However, as no studies were found which indicate that fire promotes production or release of seed in eastern hardwoods, the following discussion will focus on fire-induced injury and mortality.

Aside from consumption of living material by combustion, the injury and mortality caused by fires is the result of excessive heat. Five theories exist as to the mechanism by which heat injury to protoplasm occurs (Hare 1961). The most widely accepted theory is that heat causes coagulation of protoplasm (Hare 1961). Other theories include heat destruction of enzymes, asphyxiation, and lipoid liberation (Hare 1961). Dehydration increases both heat and cold resistance of plants, but no evidence has been found which indicates that well-hydrated, physiologically active protoplasm of one species of plants has higher thermal tolerance than that of another species (Hare 1961).

The temperature required to kill living plant tissue varies depending on the duration of exposure. The relationship between time and temperature is exponential: a relatively high temperature may cause no apparent injury during seemingly long exposures, but a slightly higher temperature may kill rather quickly (Hare 1961). Plant tissue dies almost instantaneously if exposed to temperatures of $60^{\circ}C$ or greater (Hare 1961).

The season of burning also exerts an important influence on the amount of heat required to produce injury. The physiological state of trees during the dormant season reduces their susceptibility to fire injury (Greulach 1973). The ambient air and tree tissue temperature

also tends to be lower during the dormant season, therefore a greater amount of heat must be supplied to raise the temperature of tree tissue to critical levels (Loomis 1974). However, direct sunlight can increase the temperature of plant tissues as much as $14^{o}C$ greater than ambient temperature (Hare 1961).

Fire-induced tree mortality may be caused by cambial girdling or crown scorch (Van Wagner 1973, Harmon 1980, Rouse 1986). Heat from fires is directed to bark surfaces and tree crowns by radiation and convection, and heat received at the bark surface is transferred to the cambium by conduction (Brown and Davis 1973).

The degree of crown scorch is dependent on the temperatures reached at various heights in the convection column, which is a function of fire intensity, ambient temperature, relative humidity, and wind speed (Van Wagner 1973). Increases in wind speed decrease the temperature in the convection column by mixing cooler air with fire-heated air. Species differences in resistance to crown scorch are related to growth form and rate of height growth (Hare 1961). Elevated, open crowns receive less heat energy than low, compact crowns under similar conditions in the convection column.

The amount of heat reaching the cambium of trees is related to the nature, magnitude, and duration of the heat source, as well as tree bark properties (Spalt and Reifsnyder 1962). The leeward side of a tree receives more heat than the windward side due to the chimney effect, and the difference is greater with increasing height along the bole (Fahnestock and Hare 1964). Headfires direct more heat to higher

positions along the bole on the leeward side of trees than backfires
(Fahnestock and Hare 1963). Cambial temperature increases with
increasing temperature and duration of the heat source at the bark
surface, although there is a time lag and internal temperatures do not
get as high as surface temperatures (Fahnestock and Hare 1964). Harmon
(1980) observed that backfires tend to cause more girdling than
headfires, due to the longer duration of the heat source, and trees near
large fuels such as downed logs are particularly susceptible to
girdling. However, Ferguson (1957) found that although headfires caused
more top-kill in understory oaks, there was no difference between
headfires and backfires in the amount of complete kill. The time lag
associated with conduction of heat through bark causes cambial
temperature to continue to increase after removal of the heat source,
and temperatures can remain high for long periods of time (Hare 1965).
The rate of cooling after removal of the heat source is considerably
slower than the rate of heating, due to the temperature gradient between
the bark surface and the cambium being less steep (Hare 1965).

Many authors correlate fire resistance with bark thickness (Stickel
1934, Hare 1965, Harmon 1984, among others). In general, bark thickness
increases with DBH. Some species show a linear relationship between DBH
and bark thickness, while other species show increasing or decreasing
rates of bark accumulation with increasing DBH (Harmon 1984). Fast
growing trees produce thick bark faster than slow growing trees, so
crown class can have an important effect on bark thickness (Harmon
1984). Hare (1965) found that time to cambium kill using a small flame

produced by a propane torch was exponentially related to bark thickness. If this relationship applies to flames in forest fires then a small increase in bark thickness would greatly increase fire resistance.

Bark thickness influences fire resistance, but physical properties of bark have an important effect on fire resistance as well. The numerous air cells and corky nature of bark make it an excellent insulator (Hare 1965). Hare (1965) found differences between species in the insulating ability of bark independent of thickness. The insulating ability of bark is related to thermal conductivity, specific heat, and thermal diffusivity (Harmon 1984). Thermal conductivity is linearly related to bulk density, and increased heat resistance has been associated with an increase in the ratio of corky outer bark (dead cells) to inner bark (living cells) (Spalt and Reifsnyder 1962). The ratio of outer bark to inner bark increases with age and DBH (Spalt and Reifsnyder 1962). Thermal conductivity also increases with increasing moisture content, which varies by season; maximum bark moisture content occurs in the spring and minimum bark moisture content occurs in the fall for most tree species (Hare 1961). Differences in heat resistance between species are more evident for stems of larger diameter and greater bark thickness, presumably because of the increase in the amount of the factor responsible for the difference in heat resistance (Hare 1965).

Species with thick bark such as chestnut oak commonly have deep fissures in the bark surface, and it has been hypothesized that deep fissures may partially offset the heat resistance provided by thick bark

(Spalt and Reifsnyder 1962). However, in a study of prescribed burning
in southern pines, Fahnestock and Hare (1964) found that irregularities
in the bark surface affect the degree of heating, and maximum
temperatures and total heat absorbed were lower in bark fissures than on
surface plates. No similar studies were found in the literature
pertaining to eastern hardwoods.

Numerous authors have attempted to quantify fire-induced tree
mortality in relation to fire type and intensity and tree
characteristics. Quantification of fire-induced tree mortality is often
complicated somewhat because it can often take as long as two growing
seasons after a fire for fire-injured trees to die (Loomis 1974).
McCarthy and Sims (1935) developed a series of curves for predicting
mortality of upland oaks by diameter class based on percent mortality in
the 8 cm diameter class. Height of crown scorch has been used to
predict mortality in conifer stands (Van Wagner 1973), however, this
method seems difficult to apply in mature hardwood stands, especially
during the dormant season. Useful equations for predicting percent
mortality of black oak, scarlet oak, white oak, and post oak (Quercus
stellata Wangenh.), as a function of height of stem-bark scorch (any
discoloration of bark due to fire) and DBH, have been developed for both
growing and dormant season fires (Loomis 1974). Although R^2 values for
most of these equations were in the range of 0.66 to 0.77, the standard
errors were as high as 19.7%, and the linear models were not constrained
to predict probabilities only in the range of 0-100%. It would be
desirable to improve the predictive power of these equations, as well as

constrain the models so that predicted probabilities cannot occur at values less than 0% or greater than 100%. Harmon (1984) developed an equation for prediction of the probability of chestnut oak surviving low-intensity surface fires. Bark thickness was the predictor variable and the equation had an R^2 of 0.91. Greene and Schilling (1987) developed logistic multiple regression models (Walker and Duncan 1967) for prediction of the probability of girdling for loblolly pine (Pinus taeda L.), water oak (Quercus nigra L.), and sweetgum (Liquidambar styraciflua L.) saplings treated with a propane fueled backfire simulator using diameter at groundline and mean temperature exposure as independent variables. The logistic regression procedure has several distinct advantages over the methods used by Loomis (1974) and Harmon (1984) for prediction of fire-induced tree mortality (or for prediction of event probabilities in general) which have been described extensively (Walker and Duncan 1967, Monserud 1976, Daniels et al. 1979).

Most attributes which influence species specific fire resistance increase with age, including bark thickness, ratio of outer bark to inner bark, and height of the crown above the ground. The attributes which influence fire resistance are all positively correlated with DBH, especially for shade-intolerant species in even-aged stands. The fire return interval has a significant interaction with the increasing fire resistance of a given species with increasing age in determining species composition in an area with periodically recurring fires. A shorter fire return interval would decrease the relative importance of slow-growing, thin-barked species and increase the relative importance

of fast-growing, thick-barked species (Harmon 1984).

Effects of fire on oak forests

The effects of fire on plant communities is dependent on characteristics of both the fire and the plant community. The fire regime exerts an important influence on the forest community in a given area over time. Fire characteristics which influence the effect of a single fire on a vegetational community include fire intensity and rate of spread. Characteristics of plant species which influence the nature and magnitude of the effect of a single fire on a plant community include resistance to fire induced mortality, reproductive success, and competitive ability in the post-fire environment. Environmental factors such as ambient temperature, relative humidity, wind direction and speed, topography, and soil characteristics affect fire behavior as well as plant structure and function. The amount of fuel accumulation and the flammability of fuels are influenced by the plant community and environmental factors, and greatly affect fire occurrence and behavior.

Fire is typically believed to return plant communities to earlier successional stages, but it requires a relatively severe fire to produce this effect. Barden and Woods (1976) found that surface fires in mixed pine-hardwood stands in the southern Appalachians did not shift species composition to pine unless 85% or more of the pre-fire basal area was removed by the fire. Similarly, Brown (1960) found a direct correlation between the occurrence of American beech, sugar maple, and yellow birch

(<u>Betula</u> <u>alleghansiensis</u> Britton.) forests in New England and the length of time since an area had been burned. Oaks tended to dominate the canopy in more recently burned areas (Brown 1960). Swan (1970) found similar results in southern New York.

Oaks possess a number of traits which enable them to resist fire-induced mortality better than many associated species. Relatively rapid development of thick, corky bark increases the fire resistance of chestnut oak (Harmon 1984). Oak stems of seedling, sapling, and larger sizes resist root-kill better than many competing species, and respond to top-kill with vigorous basal sprouting (Swan 1970), although sprouting vigor declines with repeated top-kill (Johnson 1983). Oak saplings (2.5 to 10 cm DBH) produced sprouts in proportion to their pre-burn importance following wildfire in New York, and produced a greater number of sprouts per clump than northern hardwoods (Swan 1970). Number of sprouts per clump was greater for larger stems, and chestnut oak produced the greatest number of stems per clump of all oak species (Swan 1970). Density of oak stems tends to increase following fires, largely due to the tendency of top-killed oaks to produce multi-stemmed sprout clumps (Thor and Nichols 1974, Teuke and Van Lear 1982, Huntley and McGee 1983).

Several researchers have investigated the possibility that use of prescribed fire may promote greater success in regenerating oaks on high quality sites. Fire may benefit oak regeneration by increasing seedling establishment due to reduction of excessive litter, or by increasing the proportion of oaks in the advance regeneration pool (McQuilkin 1983).

Litter depths of about 2.5 cm are optimal for germination of acorns, and fire reduces excessive litter accumulations, which should improve conditions for germination and establishment of oak seedlings (Thor and Nichols 1974, Swan 1970). The literature contains contradictory results as to whether fire increases the proportion of oaks in the advance regeneration pool. The proportion of red maple, yellow-poplar, and black cherry (Prunus serotina Ehrh.) stems of regeneration size were found to increase following both pre-harvest and post-harvest burns in the Southern Highlands of northeastern Alabama (McGee 1979, McGee 1980, Huntley and McGee 1983). The proportion of oak regenerants remained fairly constant following fire, but in all cases the proportion of oaks in the pre-harvest understory and in the developing stand was less than the proportion of oaks in the pre-harvest overstory.

Teuke and Van Lear (1982) found the relative importance of oak species in the understory increased and the relative importance of red maple regeneration decreased following a single prescribed fire in the southern Appalachians. Understory oaks sprouted frequently following a single prescribed fire in the transition oak forests of New York, but competitors such as red maple also had high sprouting frequency, and tended to produce undesirable multi-stemmed clumps (Nyland et al. 1983). Oak species responded to a second prescribed fire by sprouting vigorously, whereas most competitors displayed decreased sprouting frequency and vigor following the second fire (Nyland et al. 1983). Periodic and annual burns favor other shade-intolerant species which are prolific sprouters in addition to oaks. Sassafras (Sassafras albidum

Nutt.) was very common on burned plots on the Highland Rim in Tennessee, but was considerably less abundant on unburned plots. The lack of light in the understory appeared to kill rootstocks of intolerant species including oaks (Thor and Nichols 1974).

In spite of contradictory evidence that has been found by various studies concerning the effect of fire on oak regeneration, the results of most research indicates the existence of some general trends. Low-intensity fires kill the above-ground portions of small stems: larger stems and species with greater fire resistance are top-killed as fire intensity increases. Most hardwood species are capable of resprouting from dormant buds at the base of the stem following top-kill. Complete kill of stems (no resprouting) is greater for fires of high intensity and for individuals in poor physiological condition (Ferguson 1957). Sprouting frequency and vigor decreases following repeated top-kill for most species, although oaks seem more resistant to complete kill than competitors such as red maple. A single fire in a mature hardwood stand will increase understory density, decrease the size of stems, and generally result in improved form of regenerants, by replacing stems with broad, flat-topped crowns with stems possessing a definite leader. The greatest detriment to burning a young hardwood stand is the loss of several years growth.

Fire may be a useful tool to assist silviculturists in regenerating high quality oak stands on good sites. However, the knowledge and techniques required to use fire to achieve specific land management objectives has not yet been developed. Studies are needed in which

characteristics of pre-fire vegetation, fire behavior, and site
characteristics are carefully monitored and related to fire effects in
order to identify the possible role of fire in regenerating oak forests.

METHODS

Study Area

The study area is located on the western slope of the Blue Ridge Physiographic Province, in the South District of Shenandoah National Park (SNP), in Rockingham County, Virginia (Figure 1). The area is characterized by the northeast to southwest trending main ridge, with numerous spur ridges, steep slopes, rock outcrops, and deep valleys. Elevations in the study area range from 300 to 800 m above mean sea level. Slopes vary from near 0 to 60 percent, with 30 to 50 percent slopes being the most common. The area is underlain by Cambrian sandstones, quartzites, and shales of the Chilhowee Group (Gathright 1976). Soils are mostly Dystrochrepts and occasionally Hapludults, and are shallow, coarse-textured, rocky, and low in fertility. Mean July temperature at the nearby Big Meadows Weather Station averages 18° C, while mean January temperature averages -2° C. Precipitation averages 1219 mm yearly and is distributed throughout the year. The frost-free growing season is 146 days. However, the Big Meadows Weather Station is situated at 1077 m above sea level, so the annual precipitation in the study area is likely to be slightly lower, while mean temperatures are higher and the frost-free growing season is longer.
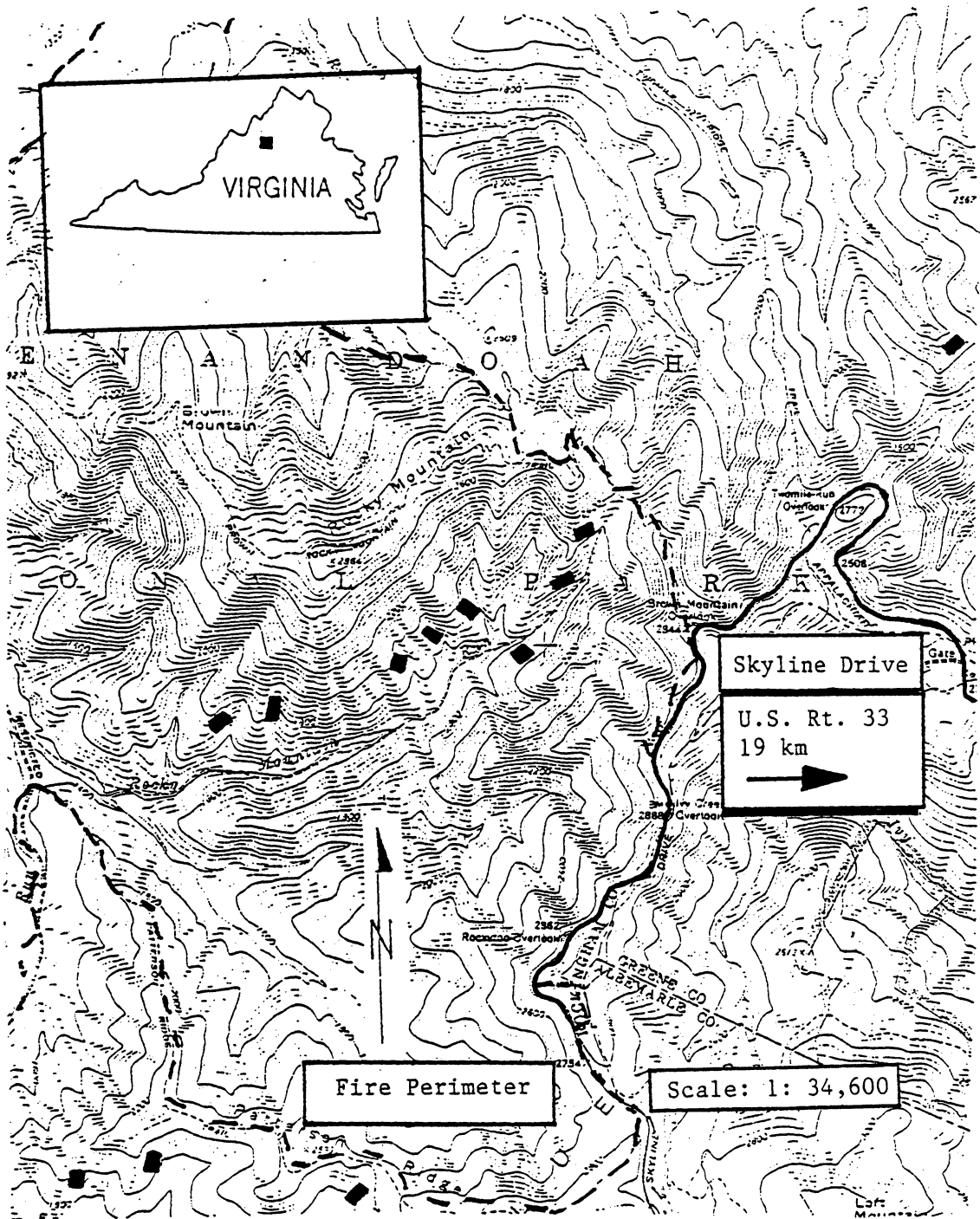
Figure 1. General Location (inset) and specific location of study
sites (■) in the South District of Shenandoah National Park, in
the Blue Ridge Physiographic Province, in Rockingham County, Virginia.

Study Site Selection

Selection criteria were established with the goal of studying
stands of sufficient quality to be economically feasible to manage for
hardwood timber production. This enables the results of this study to
not only improve understanding of the ecological effects of wildfire in
mixed oak forests, but also provides insight into potential uses of fire
in the management of mixed oak forests in the Appalachians. Selection
criteria were designed to allow the study of the highest quality sites
that had experienced a wide range of fire intensities. Sites of higher
quality than those delineated by the above criteria exist within the
study area, but fire intensity was invariably low on these sites.

Forest cover type maps prepared from aerial color-infrared slides
were obtained from the USDI National Park Service (NPS). A map showing
four classes of fire-induced canopy scorch prepared from aerial
color-infrared slides was also obtained from the NPS (Appendix I). These
maps were digitized and entered into the COMARC Geographic Information
System (GIS) at the Virginia Polytechnic Institute and State University
(VPI) in order to overlay forest cover type maps, the fire severity map,
and topographic maps, which facilitated identification of potential
study stands. Site quality of potential study sites was estimated on
the topographic maps using the Forest Site Quality Index (FSQI), an
index which was developed in the Ridge and Valley Physiographic Province
of Virginia, West Virginia, and Maryland to estimate relative site

quality based on the topographic parameters of aspect, slope position, and slope inclination (Wathen 1977) (Appendix II). Use of this index is justified because the soil parent materials are quite uniform throughout the study area and are similar to soil parent materials found in the Ridge and Valley. A number of potential study sites were located using the GIS; plots were established in four randomly selected stands for each of the three levels of fire intensity only after field verification of all selection criteria for each stand. In order to ensure that the variability within each level of fire intensity was less than the variability between levels of fire intensity, three classes of fire intensity (high, low, and unburned) were used instead of the four classes delineated on the NPS map. Although this was an observational study, rather than a controlled experiment, statistical techniques used for analysis of designed experiments were employed. Since the treatments were not applied to the experimental units at random, potential study sites had to be of similar species composition, stand density, age, and site quality to be considered for inclusion in the study, and actual study stands were chosen at random from a list of potential study stands. The data were analyzed as if the study was designed as a completely randomized experiment.

Permanent vegetation plots were established in four stands for each of three levels of fire intensity, for a total of twelve stands. Plots were established in areas burned in May of 1986 by headfires of high and low intensity, and in nearby unburned stands. Burned stands were selected based on pre-fire stand characteristics. The pre-fire

overstory of all study sites was dominated by chestnut oak, in association with scarlet oak, northern red oak, and pignut hickory, and did not differ between levels of fire intensity (Table 1). Stand ages ranged from 55 to 61 years, average height of dominants and codominants was similar for all stands, and stands were of similar site quality, with 50-year upland oak site index (Schnur 1937) of about 18 m (Table 2). All stands were between 485 and 700 m in elevation, and mean elevation did not vary greatly between levels of fire intensity. Pre-fire stand basal area did not differ between levels of fire intensity, and averaged 25.5 m^2/ha (Table 1).

Characterization of Fire Intensity

The Big Run Fire was ignited by a careless camper on the evening of May 2, 1986, and burned 1889 ha of the Big Run drainage basin before it was contained on May 5, 1986 (Figure 2). The areas in which study sites were located were burned on May 3 and May 4, 1986. It was on these days that the fire burned with the highest intensity and most rapid rates of spread. Since this fire was an uncontrolled wildfire and the study was implemented after the fact, direct quantification of fire behavior was not possible, therefore fire intensity was quantified in a relative manner using the height of stem-bark char measured from groundline to the highest point of blackening on the uphill face of the tree. National Park Service records of environmental conditions for each day of the fire were also obtained (Table 3). Data used for

Table 1. Relative basal area and number of stems per hectare by species for mixed oak stands in Shenandoah National Park, Virginia, before being affected by wildfire.

Species	Relative Basal Area	Relative Number of Stems per Hectare
	---- Basal area (%) ----	Stems/hectare (%)
Chestnut oak	65.1 [1]	59.4
Scarlet oak	15.3	11.1
N. red oak	6.4	4.7
Pignut hickory	7.1	19.9
Black oak	1.0	0.8
Blackgum	1.3	5.2
Red maple	1.4	6.2
Black locust	0.3	0.6
Sassafras	0.1	0.2
Serviceberry	0.5	2.4
Dogwood	0.2	1.1
White oak	0.3	0.2
Pitch pine	0.5	0.2
White pine	0.3	0.1
	Basal area (m^2/ha)	Number of stems/hectare
All species	25.5	1156

[1] Analysis of variance indicated no significant differences between levels of fire intensity, therefore the values above are means for twelve stands, four in each of three levels of fire intensity.

Table 2. Pre-fire stand attributes for mixed oak stands affected by wildfire of three levels of intensity in Shenandoah National Park, Virginia.

Stand Attribute	Fire Intensity		
	High	Low	Unburned
Total Age (yr)	55	57	61
Height[1] (m)	21	22	21
Upland Oak Site Index[2]	18	19	18

[1] Average height of dominant and codominant chestnut, scarlet, and northern red oaks.

[2] Base age 50 (Schnur 1937).

Figure 2. Big Run Fire, May 2 - 5, 1986, in the South District of Shenandoah National Park, Rockingham County, Virignia. Photograph courtesy of Resource Management Staff, Shenandoah National Park, USDI National Park Service.

Table 3. Environmental parameters during the Big Run Fire, Shenandoah National Park, Virginia.[1]

Date	Temperature		Relative Humidity	Wind	
	High	Low		Direction	Speed
	--- °F ----		------- % -------		mph
May 2, 1986[2]	86	42	22	N-NW	10-20
May 3, 1986	65	30	24	N	10-20
May 4, 1986	55	28	24	W	2-5

Precipitation		
January through April		April
Yearly Average	1986	1986
------------- mm -----------------		
307	142	21

[1] Source: Unpublished record "Big Run Fire Review", Shenandoah National Park, United States Department of the Interior National Park Service.

[2] Fire was ignited by a campstove on the evening of May 2, 1986.

characterization of fire intensity were recorded in June of 1987, 1-1/2

growing seasons following the fire. Mean height of stem-bark char was

1.67 m and 0.33 m for stands burned by fire of the high and low levels

of fire intensity, respectively (Table 4), (Figures 3 and 4) . Average

canopy mortality (mortality as used in this thesis refers to top-kill,

or mortality of the above-ground stem) for the stands burned by

high-intensity fire was 67% of pre-fire basal area and 81% of the

pre-fire number of stems/ha (Table 4). Stands affected by low-intensity

fire experienced 8% reduction in basal area and 15% reduction in the

number of stems/ha (Table 4).

Plot Design and Data Collection

Three permanent vegetation plots (subsamples) were randomly located

in each of 12 stands for a total of 36 plots. Plots were established in

March, 1987, and June, 1987. Data for the overstory were recorded in

June, 1987, in order to accurately assess the status of trees which were

too tall to allow examination of terminal buds. Data for the understory

were recorded between September 15, 1987, and October 15, 1987, in order

to characterize the understory at the end of the second growing season

following the fire.

Plot configuration followed a nested design (Figure 5). Main plots

are 24 x 24 m (0.0576 ha) in size and were oriented with the uphill side

parallel to the contour. The dimensions of the plot axes which were

oriented down the slope were corrected so that all plots represented a

Table 4. Mean height of stem-bark char and wildfire-induced canopy mortality for three levels of fire intensity in mixed oak stands in Shenandoah National Park, Virginia. Values are means of four stands for each level of fire intensity.

	Fire Intensity	
	High	Low
Height stem-bark char[1] (m)		
Mean	1.67a[2]	0.33b
Range	1.23 - 2.08	0.18 - 0.48
Basal area mortality[3] (m^2/ha)		
Mean	17.8a	2.1b
Range	8.5 - 27.3	0.6 - 4.2
Percent basal area mortality		
Mean	67.1a[4]	7.9b
Range	31 - 93	2 - 16
Number of trees/ha mortality		
Mean	881.1a	201.1b
Range	682 - 1105	69 - 324
Percent mortality (%trees/ha)		
Mean	81.4a	14.8b
Range	68 - 97	5 - 25

[1] Height of stem-bark char was measured from groundline to the highest point of bark blackening on the uphill face of the tree.

[2] Means within rows followed by different letters do not differ at alpha= 0.05 (One-Way ANOVA for completely randomized design).

[3] Mortality refers to death of the above-ground portion of the stem (top-kill).

[4] Statistical analysis was performed on arc-sine transformed percentages.

Figure 3. Mixed oak stand in March, 1987, which was burned by a
high-intensity surface headfire in May, 1986, in Shenandoah National
Park, Virginia. Note height of stem-bark char and presence of basal
sprouts. Photograph by the author.

Figure 4. Mixed oak stand in March 1987, burned by low-intensity
surface headfire in May, 1986, in Shenandoah National Park, Virginia.
Note moderate degree of stem-bark char. Photograph by the author.
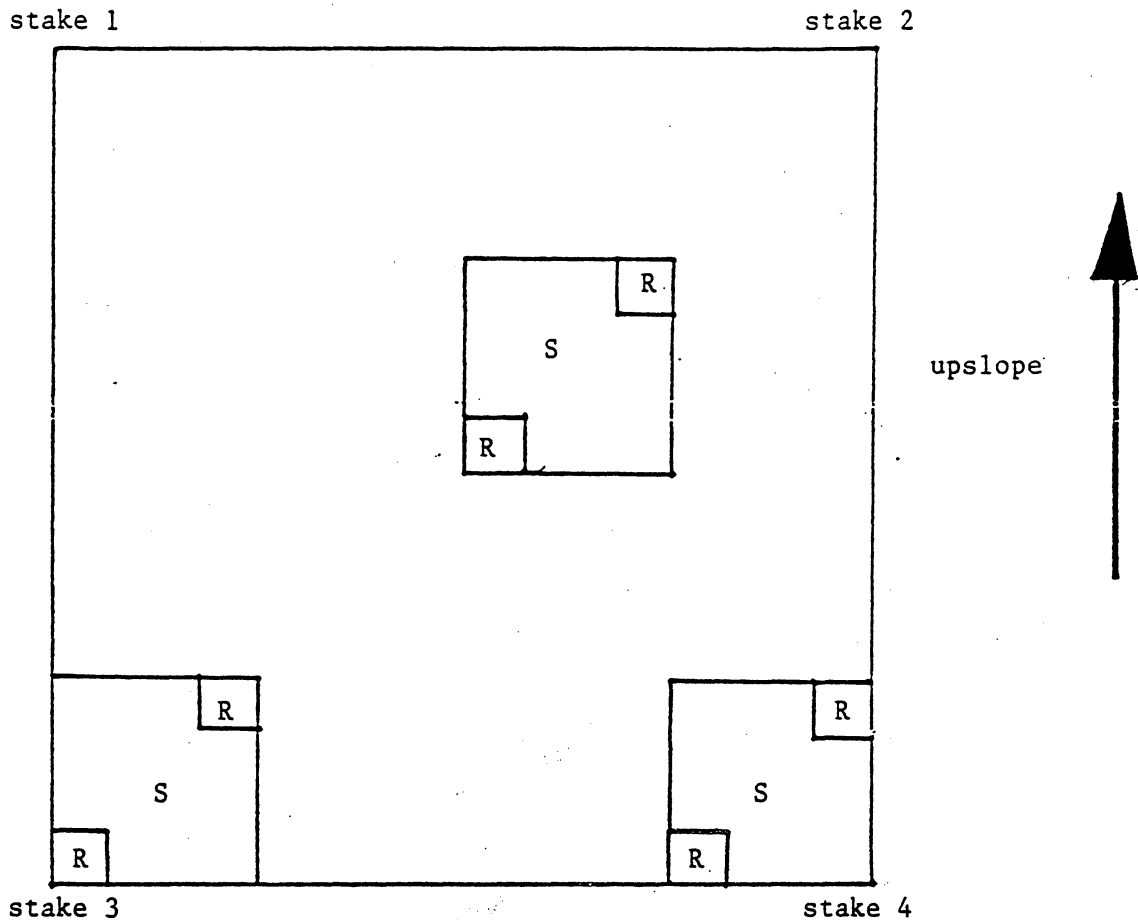
Figure 5. Configuration of permanent vegetation plots. Three such plots are randomly located in each study stand in the South District of Shenandoah National Park, in Rockingham County, Virginia.

Main plot for sampling overstory (woody stems greater than 5 m in height) is 24 m x 24 m (0.0576 ha).
S = 6 m x 6 m (0.0036 ha) shrub strata (woody stems 1 m to 5 m in height) plot.
R = 1 m x 1 m (0.0001 ha) regeneration strata (woody stems less than 1 m in height) plot.

Scale: 2.5 cm = 6 m

horizontal area of 0.0576 ha. Plot corners were marked with 0.66 m

sections of 1.25 cm PVC pipe placed over 0.5 m pieces of 1 cm steel rod

driven into the ground. PVC stakes were labeled with stand number, plot

number and stake number as in Figure 5. Plot corners were located using

a staff compass and cloth tape. Three 6 x 6 m (0.0036 ha) sub-plots were

systematically located in each plot (Figure 5), and two 1 x 1 m (0.0001

ha) sub-sub-plots were systematically located in corners of each

subplot. Systematic location of sub-plots will facilitate future

remeasurements of these plots, which are part of a larger system of

permanent plots.

Slope position was recorded for each plot as shown in Appendix II.

Slope percent was measured along each side of the plot in a downslope

direction using a clinometer and the mean of these two measurements was

recorded as the average slope percent of that plot. Aspect was measured

in the same manner using a staff compass. Elevation of each plot to the

nearest 20 m was estimated using a topographic map. Slope shape

(concave, convex, or planar) was also recorded for each plot. Data

characterizing the topography of the study sites can be found in

Appendix III.

All woody stems greater than 5 m in height present on a plot were

recorded by species, DBH (measured to the nearest 0.1 cm), crown class

(dominant, codominant, intermediate, or suppressed), and status (live,

dead before the fire, or top-killed by the fire). Sampling occurred

early enough after the fire to differentiate between trees killed by the

fire and those previously dead by noting presence or absence of fine

twigs, decayed wood, and peeling bark. Presence or absence of live

basal sprouts (sprouts originating within 0.5 m of groundline) and

height of stem-bark char to the nearest 0.1 m were also recorded for

each stem. Height of stem-bark char was measured from groundline to the

highest point of bark blackening on the uphill face of the tree. Total

tree height was measured on a subsample of free to grow, uninjured,

dominant or codominant trees of the two dominant species on each plot

using a clinometer and tape. In order to use the site index curves most

applicable to the study area it was necessary to determine the total age

for the same trees measured for height by taking increment cores at

breast height (1.4 m) and adding four years to the breast-height age.

All woody stems present on each 6 m x 6 m sub-plot which were

greater than or equal to 1 m and less than or equal to 5 m in height

were tallied by species, height to the nearest 0.1 m, and origin.

Seedling origin stems were defined as those stems which originate from

seed which germinated after the fire. Seedling sprouts were defined as

those stems which originated from stems with a basal diameter less than

5 cm which were top-killed by the fire. Basal sprouts originated from

rootstocks of stems with basal diameters greater than 5 cm which were

top-killed by the fire, and surviving regeneration was composed of stems

which were present before the fire and were not top-killed by the fire.

No excavation of stems to verify origin (seedling or seedling sprout)

was performed within plots. However, many seedling sprouts were readily

recognizable as such, since the remains of the fire-killed stem were

often attached to the newly sprouted stem. A number of stems were

excavated in areas adjacent to the plots in order to ensure accurate assessment of stem origin when the remains of fire-killed stems were not evident. Each clump of a seedling sprout was tallied by species, height of the two tallest stems in the clump (within strata definition), number of stems between 1 and 5 m tall in the clump, and total number of stems in the clump including those less than 1 m tall. Basal sprouts were tallied in the same manner but the crown class and status of the parent tree were also recorded (see above). Often a sprout clump would be located near the plot boundary such that part of the clump was "in" and part of the clump was "out". In this case the height of the two tallest stems in the clump that were within the plot boundaries were recorded along with the number of stems between 1 and 5 m in height within the plot. The total number of stems in the clump included all stems in the clump regardless of height or location relative to the plot boundary.

All woody stems less than 1 m in height were recorded on the 1 m x 1 m sub-sub-plots in the same manner as above except that stem height was not measured.

Data Analysis

The effects of high-intensity and low-intensity fire on canopy mortality were evaluated by comparing absolute and relative basal area/ha and number of stems/ha by species using analysis of variance for a completely randomized design followed by Fisher's Least Significant Difference (LSD) t-test for mean separation. Species richness and

community diversity were analyzed using similar statistical procedures.

Species richness was calculated according to the following formula:

$$R = S-1/\log n \qquad\qquad [5]$$

where:

R = species richness

S = number of species present in a stand

n = number of individuals present in a stand

The Shannon Index of community diversity was calculated based on the relative density and relative basal area of each species according to the following formula:

$$H' = -\sum_{i=1}^{K} (p_i * \log p_i) \qquad\qquad [6]$$

where:

H' = Shannon Index of Community Diversity

p_i = the relative density or basal area of species i

K = the number of species present in a stand

From Pielou (1977)

Simple linear regression was used to determine if the mean height of stem-bark char could be used to predict the percentage of stand basal area and number of stems killed by fire. Analysis of variance and

analysis of covariance followed by Fisher' LSD t-test, and the Wilcoxon Rank Sum Test were used to assess, by species, the effects of parent tree attributes and fire intensity on basal sprouting and basal sprout growth. The effects of fire on the abundance, species composition, and origin of regeneration were evaluated using statistical procedures similar to those used in the analysis of the tree strata. Absolute and relative density and frequency by species and origin, height by species and origin and community diversity and species richness were variables used to determine the effects of fire on the abundance, species composition and origin of regeneration following wildfire.

Logistic regression analysis (Walker and Duncan 1967, Monserud 1976) was used to predict the probability of individual tree mortality (top-kill). Separate regressions were calculated for each species and compared. All individual trees from the six most abundant species occurring in stands burned by the high and low levels of fire intensity were utilized for this analysis, and each tree formed a separate data point. Maximum likelihood was used to estimate the parameters.

Logistic regression has several advantages over other techniques used to develop equations for prediction of event probabilities, such as discriminant analysis. The logistic equation is constrained to give values in the interval [0,1], which is the range in which probabilities can occur; it is less sensitive to violations of the multivariate normality assumption than discriminant analysis; and continuous or nominal independent variables can be utilized to predict binary, ordinal, or nominal dependent variables, which eliminates the need to

arbitrarily assign the data to classes and compute proportions in each class to be used as dependent variables (probabilities) (Walker and Duncan 1967, Daniels et al. 1979, Strub et al 1986). The values predicted by the logistic regression equation may validly be interpreted as probabilities of the occurrence of a certain event, and are modelled as a function of several independent variables which affect the event (Walker and Duncan 1967). The general form of the logistic multiple regression model is as follows:

$$P = [1+ \exp(-(b_0 + b_1 X_1 +...+ b_k X_k))]^{-1} \qquad [7]$$

where:

P = the probability of occurrence of some event

b_k = model coefficients

X_k = independent variables which affect the event

From Walker and Duncan (1967)

Tree DBH and the height of stem-bark char were used as variables to predict the probability of fire-caused tree mortality. DBH was expected to have a negative coefficient, as larger trees should have developed thicker bark which should insulate the cambial layer from heat produced by fire. Height of stem-bark char was expected to have a positive coefficient, as greater height of stem-bark char would indicate a greater amount of heat energy being directed to the tree which should increase the probability of mortality. Various interactions and higher

orders of the above variables were tested for significant contributions

to the model as well.

Logistic regression was also used to develop models for prediction

of the probability of basal sprouting in response to fire. Possible

predictor variables included DBH, height of stem-bark char, and

mortality status of the above-ground stem (live or dead).

Effects of Wildfire on Canopy Mortality

The effect of wildfire on the overstory of mixed oak stands
differed significantly between the high and low levels of fire
intensity. High-intensity fire reduced stand basal area by an average of
67%, from 26 m^2/ha to 8 m^2/ha, and the number of stems/ha was reduced by
an average of 81%, from 1072 stems/ha to 191 stems/ha (Tables 4, 5, and
6). Low-intensity fire resulted in much less canopy mortality, with
basal area being reduced only 8%, from 26 m^2/ha to 24 m^2/ha; the number
of stems/ha was reduced by only 15%, from 1371 stems/ha to 1170 stems/ha
(Tables 4, 5, and 6). The relative basal area and relative number of
stems of chestnut oak was increased, and the relative basal area and
relative number of stems of scarlet oak and northern red oak was
decreased 1-1/2 growing seasons after high-intensity fire (Table 5,
Table 6). The relative basal area and relative number of stems of
blackgum (Nyssa sylvatica Marsh.) was slightly increased by
high-intensity fires, although the magnitude of change was not
statistically significant (Table 5, Table 6). In contrast,
low-intensity fires resulted in no significant shift in species
composition (Table 5, Table 6). High-intensity fires decreased
community diversity, while low-intensity fires did not significantly
affect community diversity (Table 7). Neither level of fire intensity
resulted in significant changes in species richness of the overstory

Table 5. Relative basal area by species for mixed oak stands in Shenandoah National Park, Virginia, before and 1-1/2 growing seasons after being affected by wildfire.

Species	Fire Intensity				
	High		Low		Unburned
	pre-burn	post-burn	pre-burn	post-burn	-----
	------------------------Basal area (%)-------------------				
Chestnut oak	63.8a[1,2]	83.7b	62.7a	65.0a	68.9a
Scarlet oak	13.9a	2.0b	18.8a	17.2a	13.3ab
Red oak	7.5a	1.7b	4.4a	4.6a	7.3a
Pignut hickory	8.1a	4.5a	8.9a	8.9a	4.3a
Black oak	2.1a	0.0b	0.6ab	0.3ab	0.4ab
Blackgum	1.1a	2.0a	1.1a	1.2a	1.6a
Red maple	0.7a	0.5a	2.1a	1.8a	1.5a
Black locust	0.2a	0.0a	0.0a	0.0a	0.7a
Sassafras	0.2a	0.0a	0.0a	0.0a	0.0a
Serviceberry	0.4a	0.1a	0.5a	0.5a	0.4a
Dogwood	0.1a	0.0a	0.4a	0.4a	0.1a
White oak	0.3a	0.0a	0.1a	0.1a	0.6a
Pitch pine	0.6a	3.9a	0.2a	0.2a	0.8a
White pine	1.0a	1.4a	0.0a	0.0a	0.0a
Total	100.0	99.8	99.8	100.2	99.9
	------------------------ Basal area (m^2/ha) --------------				
All species	26.2a	8.4b	26.1a	24.0a	24.2a

[1] Means within species (rows) followed by the same letter do not differ at alpha = 0.05 (One-Way ANOVA for completely randomized design followed by Fisher's Protected LSD t-test).

[2] Statistical analyses were performed on arc-sine transformed percentages.

Table 6. Relative number of stems per hectare by species for mixed oak stands in Shenandoah National Park, Virginia, before, and 1-1/2 growing seasons after a wildfire of variable intensity.

Species	Fire Intensity				
	High		Low		Unburned
	pre-burn	post-burn	pre-burn	post-burn	----
	------------- Number of trees/ha (%) -------------				
Chestnut oak	46.5a[1,2]	68.3b	62.7a	65.0a	68.9a
Scarlet oak	12.4a	2.4b	11.7a	10.8ab	9.1ab
Pignut hickory	22.8a	9.5a	24.5a	23.6a	12.4a
Red oak	4.9a	2.7a	3.0a	3.0a	6.1a
Blackgum	4.5a	12.2a	4.4a	5.3a	6.8a
Serviceberry	2.9a	0.4a	2.5a	2.5a	1.8a
Red maple	2.6a	2.7a	8.9a	6.9a	7.1a
Black oak	1.7a	0.0b	0.3ab	0.2b	0.4ab
Sassafras	0.6a	0.0a	0.1a	0.0a	0.0a
Dogwood	0.6a	0.0a	1.9a	1.9a	0.7a
Black locust	0.1a	0.0a	0.0a	0.0a	0.1a
White oak	0.1a	0.0a	0.1a	0.1a	0.4a
Pitch pine	0.1a	1.3a	0.1a	0.1a	0.3a
White pine	0.1a	0.4a	0.0a	0.0a	0.0a
Total	99.9	99.9	99.9	99.9	99.6
	------------- Number of trees/ha -------------				
All species	1072a	191c	1371a	1170ab	1024b

[1] Means within species (rows) followed by the same letter do not differ at alpha = 0.05 (One-Way ANOVA for completely randomized design followed by Fisher's LSD t-test).

[2] Statistical analyses were performed on arc-sine transformed percentages.

Table 7. Community diversity and species richness of the overstory of mixed oak stands in Shenandoah National Park, Virginia, before, and 1-1/2 growing seasons after a wildfire of variable intensity.

Stand Attribute	Fire Intensity				
	High		Low		Unburned
	pre-burn	post-burn	pre-burn	post-burn	-----
Shannon Diversity Index (H')[1]					
H' Basal area	0.497a[3]	0.255b	0.462a	0.443a	0.440a
H' # stems	0.610ab	0.413b	0.618a	0.601ab	0.59ab
Species Richness (R)[2]					
R	3.75a	3.47a	3.58a	3.36a	3.35a

[1] H'=- SUM (pi*log pi)   (Pielou 1977)

[2] R= (# species-1)/ (log # individuals)

[3] Means within attributes (rows) followed by the same letter do not differ at alpha = 0.05 (One-Way ANOVA followed by Fisher's LSD t-test).

(Table 7).

The mean height of stem-bark char was found to be a good predictor of the percentage of pre-fire stand basal area and number of trees killed by fire (Figures 6 and 7) . The mean height of stem-bark char explained 91% of the variation in fire-killed basal area and 95% of the variation in reduction of the number of trees caused by fire.

Logistic regression models were developed to predict the probability of fire-induced individual tree mortality (top-kill) for chestnut oak, scarlet and northern red oak, pignut hickory, blackgum, and red maple, using all stems of these species in stands burned by both the high-intensity and low-intensity fire as data points. This resulted in an adequate sample size for these species, and also a broad range in DBH and height of stem-bark char for each species (Table 8). The model selected as most appropriate for prediction of the probability of fire-induced tree mortality included DBH and height of stem-bark char as predictor variables. This model was chosen for the following reasons:

1. The models for all species were highly significant (p < 0.0001, Table 9).

2. All individual coefficients were highly significant (p < 0.05, Table 9).

3. The models achieve desirable properties using only two predictor variables, DBH and height of stem-bark char.

4. The biological theory underlying the models is readily understandable. DBH negatively infuences the probability of mortality, and height of stem-bark char positively
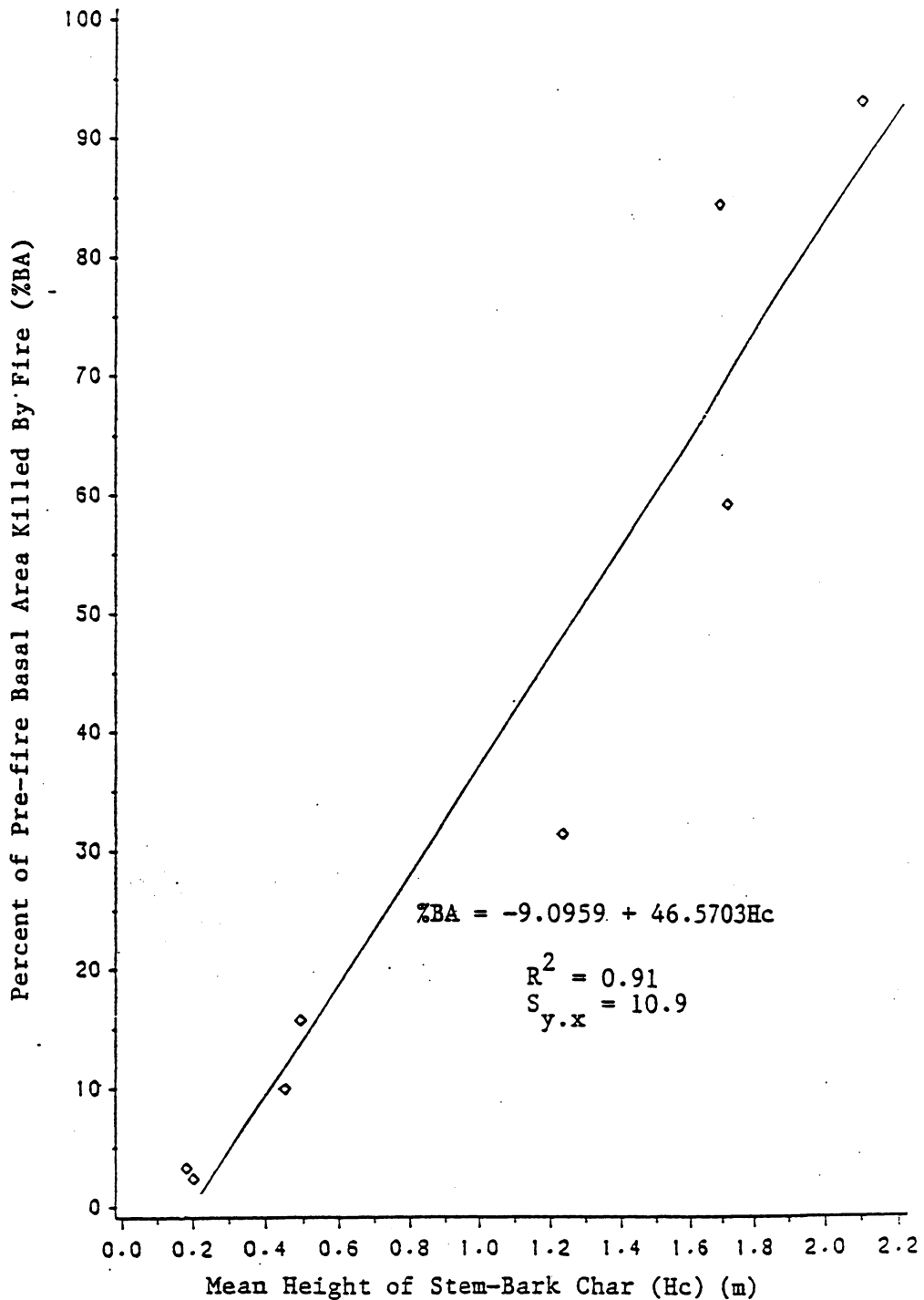
Figure 6. Percent of pre-fire stand basal area top-killed by wildfire in May, 1986, as a function of the mean height of stem-bark char in 55 to 60 year old mixed oak stands in Shenandoah National Park, Virginia. Pre-fire stand basal area ranged from 22 to 29 m^2/ha.
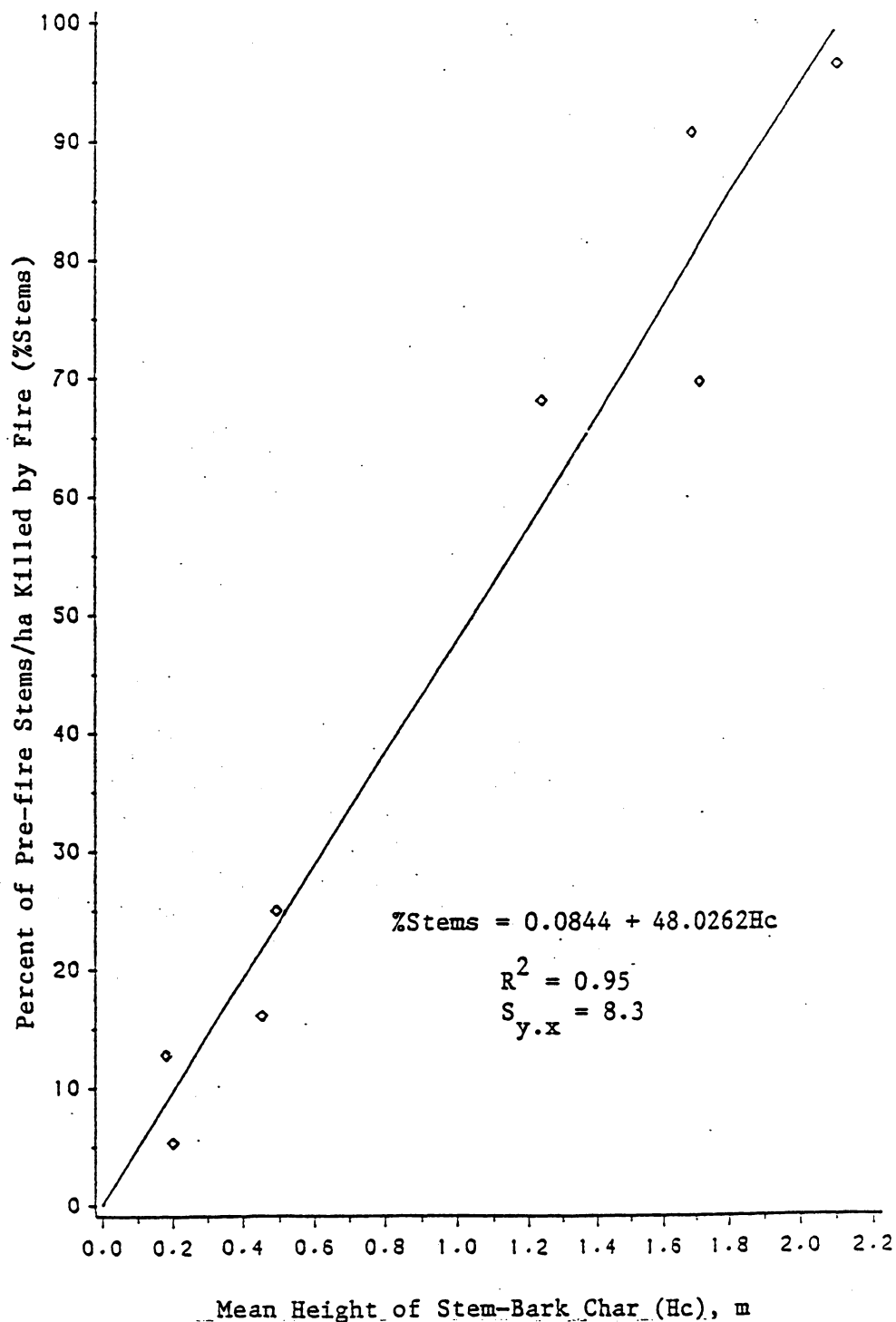
Figure 7. Percent of pre-fire number of stems per hectare top-killed by wildfire in May, 1986, as a function of the mean height of stem-bark char in 55 to 60 year old mixed oak stands in Shenandoah National Park, Virginia. Pre-fire stands had 1000 to 1400 stems per hectare.

Table 8. Number of top-killed and surviving stems, and mean and range of diameter at breast height, and height of stem-bark char[1] by species for eight mixed oak stands[2] burned by wildfire in Shenandoah National Park, Virginia.

Species	Stems		DBH		Height of Stem Char	
	Surviving	Killed	Surviving	Killed	Surviving	Killed
	----- number ---		------ cm -----		------- m ------	
Chestnut oak						
Mean	456	278	19.8a[4]	16.9b	0.57a	2.04b
Range	---	---	6–53	7–42	0–4.8	0–5.1
Red oak group[3]						
Mean	117	148	19.8a	17.8b	0.43a	1.33b
Range	---	---	6–37	6–37	0–1.7	0–5.0
Pignut hickory						
Mean	194	203	9.7a	8.9b	0.19a	1.07b
Range	---	---	4–29	4–28	0–1.4	0–3.4
Blackgum						
Mean	54	22	8.0a	6.3b	0.68a	2.28b
Range	---	---	4–29	4–28	0–3.3	0–3.5
Red maple						
Mean	70	43	7.4a	5.9b	0.10a	0.25b
Range	---	---	3–18	3–12	0–0.5	0–1.8

[1] Height of stem-bark char was measured from groundline to the highest point of bark blackening on the uphill face of the tree.

[2] The area sampled in each stand was 0.1728 ha.

[3] Red oak group includes northern red oak and scarlet oak.

[4] Means within a species and attribute followed by the same letter do not differ at alpha = 0.05 (Wilcoxon Rank Sum Test).

Table 9. Logistic regression models for prediction of the probability of tree mortality (top-kill) following wildfire in mixed oak forests in Shenandoah National Park, Virginia.

Model form:

$$P(\text{mort}) = [1 + \exp(-(b_0 + b_1(\text{dbh}) + b_2(\text{char})))]^{-1}$$

where:

P(mort) = probability of mortality
dbh   = diameter at breast height (cm)
char   = height of stem-bark char[1] (m)

Species	n	b_0	b_1	b_2	LRS[2]	C[3]	R[4]
Chestnut oak	734	1.0229	-0.2646	2.6232	522.49	0.938	0.73
Red oak group[5]	265	0.1899	-0.1318	3.3222	140.74	0.874	0.61
Pignut hickory	397	1.2165	-0.4758	6.0415	313.72	0.941	0.75
Blackgum	76	2.7750	-1.1224	2.8312	56.35	0.956	0.76
Red maple	113	0.8221	-0.4098	8.4682	33.79	0.798	0.45

[1] Height of stem-bark char was measured from groundline to the highest point of bark blackening on the uphill face of the tree.

[2] The Likelihood Ratio Statistic (LRS) is a chi-square statistic that tests model goodness-of-fit (Harrel 1986). The LRS of all models was highly significant ($p < 0.0001$).

[3] The C statistic is the proportion of concordant pairs and is a measure of the predictive ability of the model (Harrel 1986). It can range from 0 to 1; values closer to 1 indicate better predictive ability.

[4] The R statistic measures the proportion of log likelihood explained by the model and is somewhat analogous to the coefficient of determination in Least Squares Regression.

[5] Red oak group includes northern red oak and scarlet oak.

influences probability of mortality (Table 9).

The response surfaces for predicted probability of mortality due to fire for each species as a function of DBH and height of stem-bark char are shown in Figures 8 - 12.

Models for different species were compared by pooling the data for various species groups and using dummy variables to test for the effect of individual species. The difference in the - 2 log likelihood between the full model and the reduced model is the Likelihood Ratio Statistic (LRS), which is a chi-square statistic that can be used to test model homogeneity and goodness-of-fit (Harrel 1986, Greene and Schilling 1987). The degrees of freedom for the test of model homogeneity is the difference in the number of parameters estimated in each equation. On the basis of these chi-square tests, models for scarlet and northern red oak did not differ significantly, therefore the data for these species were pooled. The LRS indicated that all models presented in Table 9 differ significantly ($p < 0.05$). The test for model homogeneity can be implemented with simple calculations using quantities obtainable from the SAS LOGIST procedure output (Harrel 1986) (Appendix IV).

Statistics useful for assessing goodness-of-fit and predictive ability indicate that the models presented in Table 9 are useful models for prediction of the probability of mortality due to fire. The R statistic is the proportion of log likelihood accounted for by the models; the R value ranges from 0.45 for red maple to 0.76 for blackgum (Table 9). The C statistic is the proportion of concordant pairs, and measures the ability of the models to predict probabilities which are
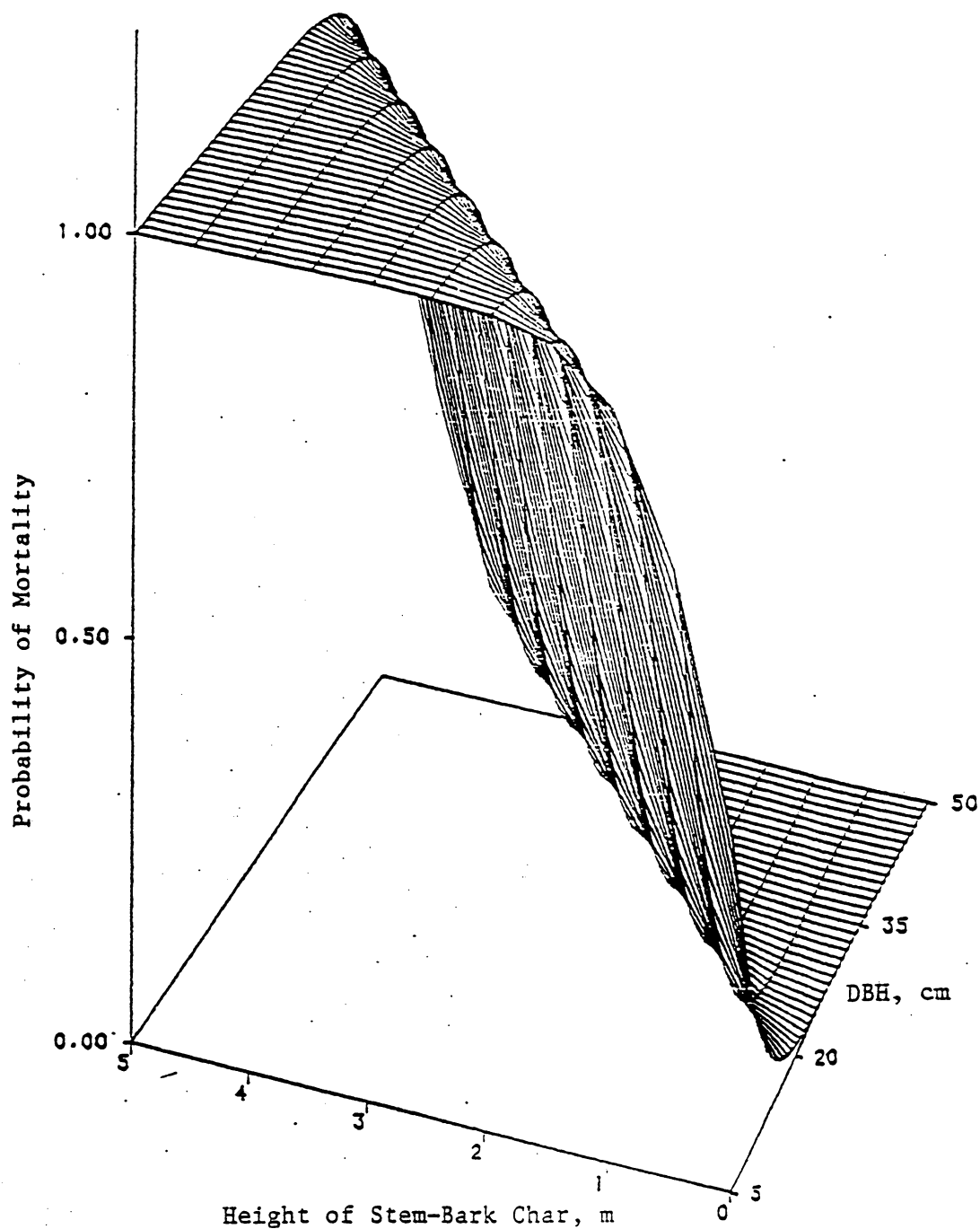
Figure 8. Predicted probability of mortality (top-kill) based on data from 734 chestnut oak trees between 6 and 53 cm DBH in mixed oak stands burned by wildfire in May, 1986, in Shenandoah National Park, Virginia. The study stands were burned by headfires and the height of stem-bark char ranged from 0 to 5 m.
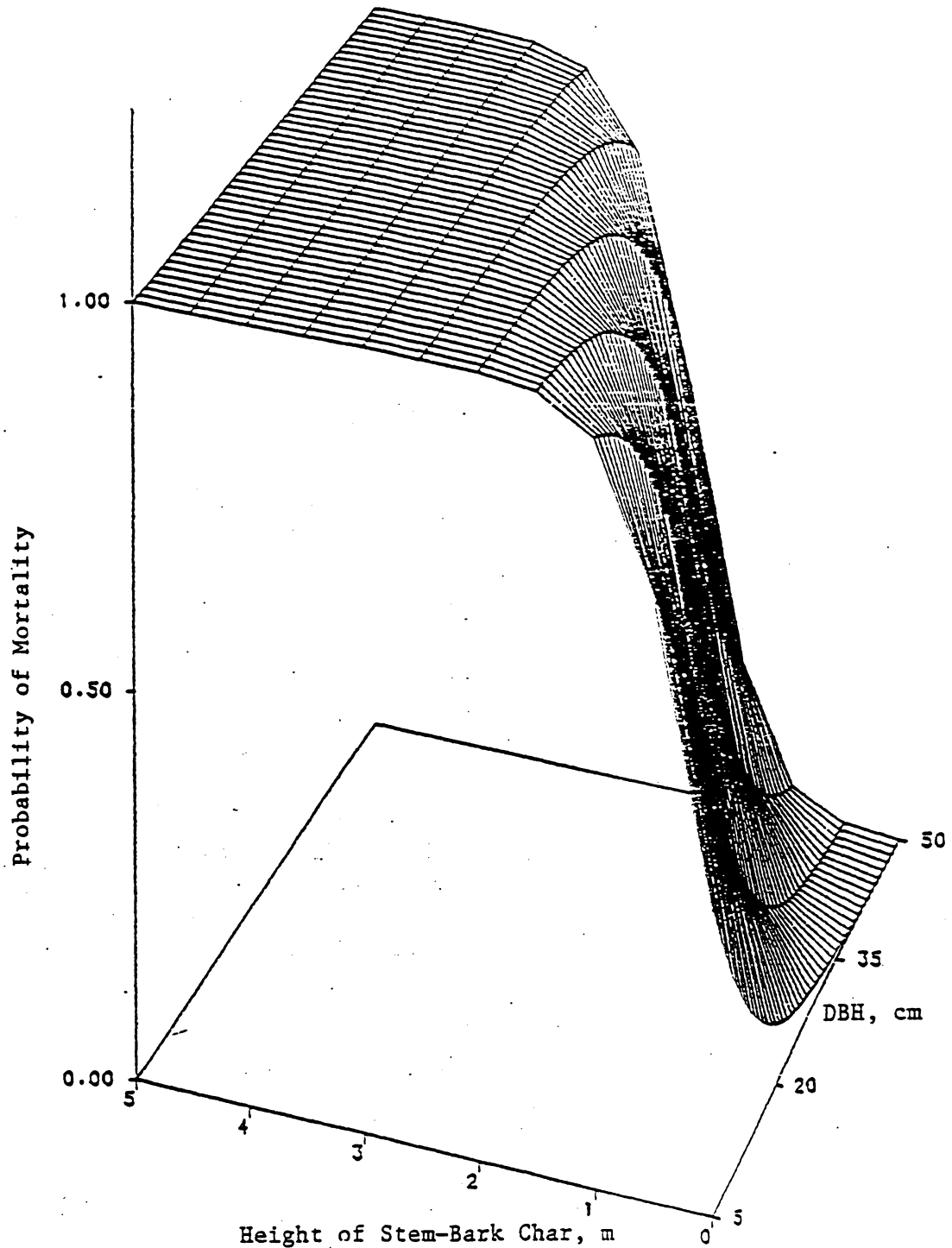
Figure 9. Predicted probability of mortality (top-kill) based on data from 265 scarlet and northern red oaks between 5 and 37 cm DBH in mixed oak stands burned by wildfire in May, 1986, in Shenandoah National Park, Virginia. The study stands were burned by headfires and the height of stem-bark char ranged from 0 to 5 m.
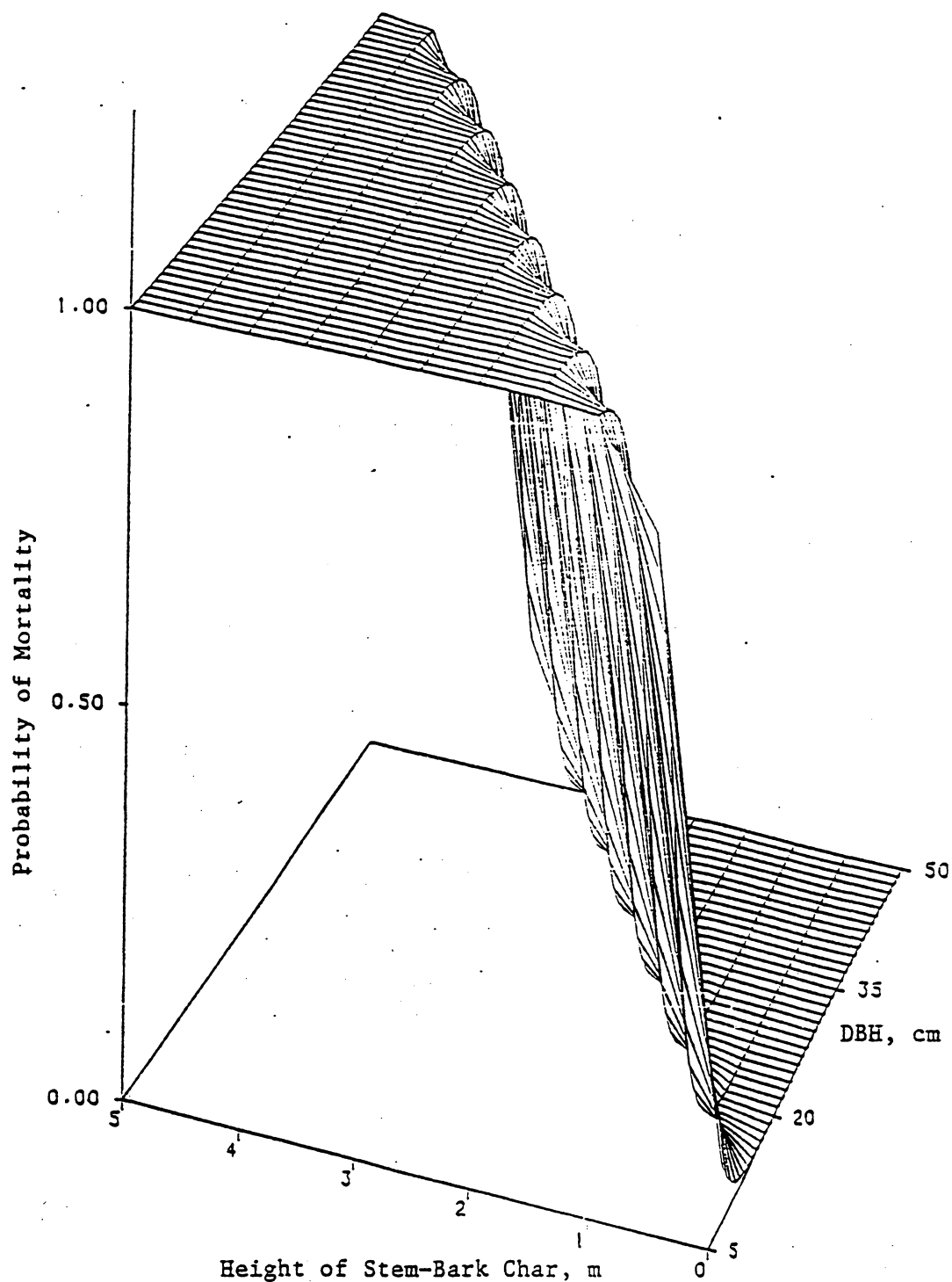
Figure 10. Predicted probability of mortality (top-kill) based on data
from 397 pignut hickory trees between 4 and 29 cm DBH in mixed oak
stands burned by wildfire in May, 1986, in Shenandoah National Park,
Virginia. The study stands were burned by headfires and the height of
stem-bark char ranged from 0 to 3.4 m.
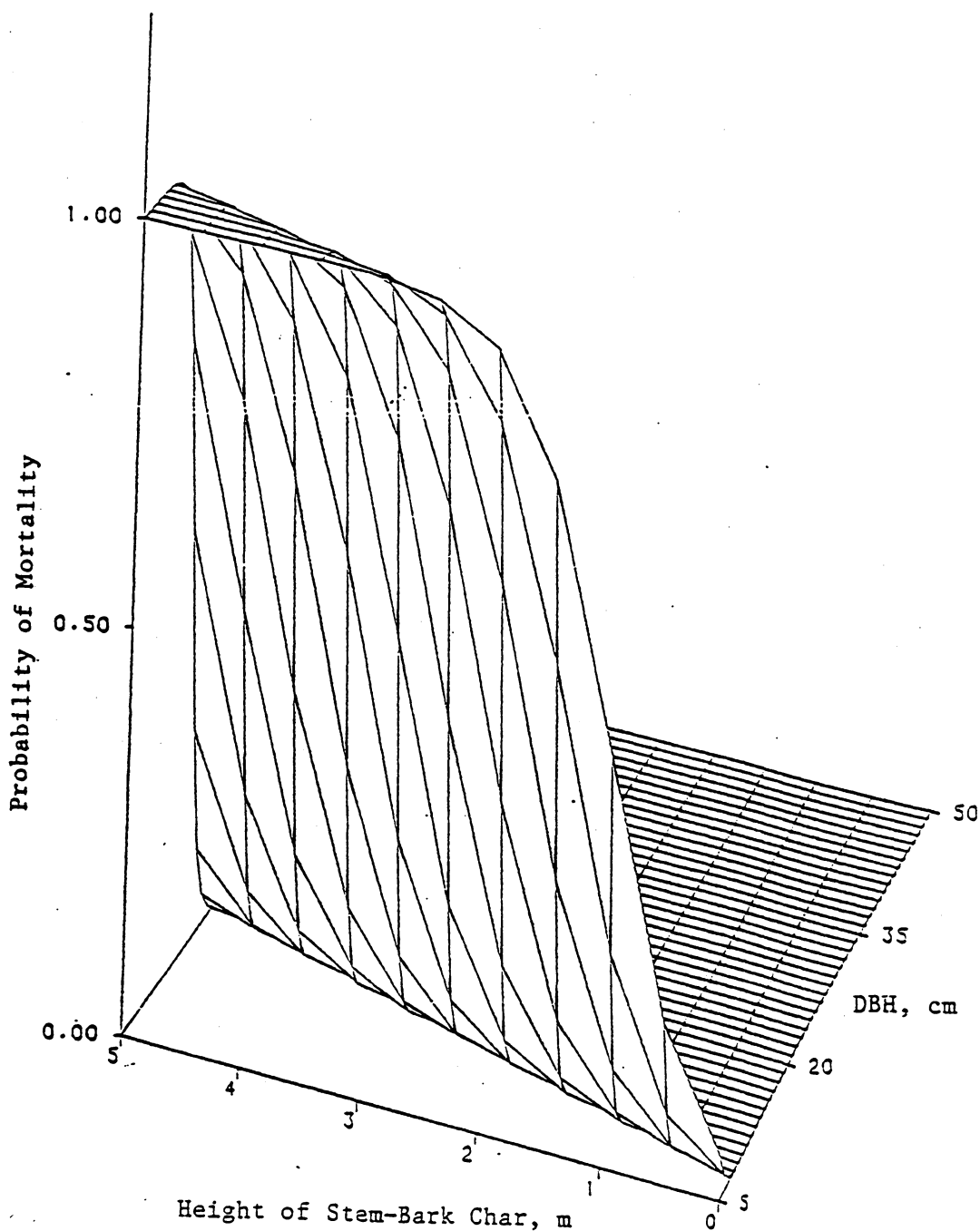
Figure 11.   Predicted probability of mortality (top-kill) based on data from 76 blackgum trees between 3 and 28 cm DBH in mixed oak stands burned by wildfire in May, 1986, in Shenandoah National Park, Virginia. The study stands were burned by headfires and the height of stem-bark char ranged from 0 to 3.5 m.
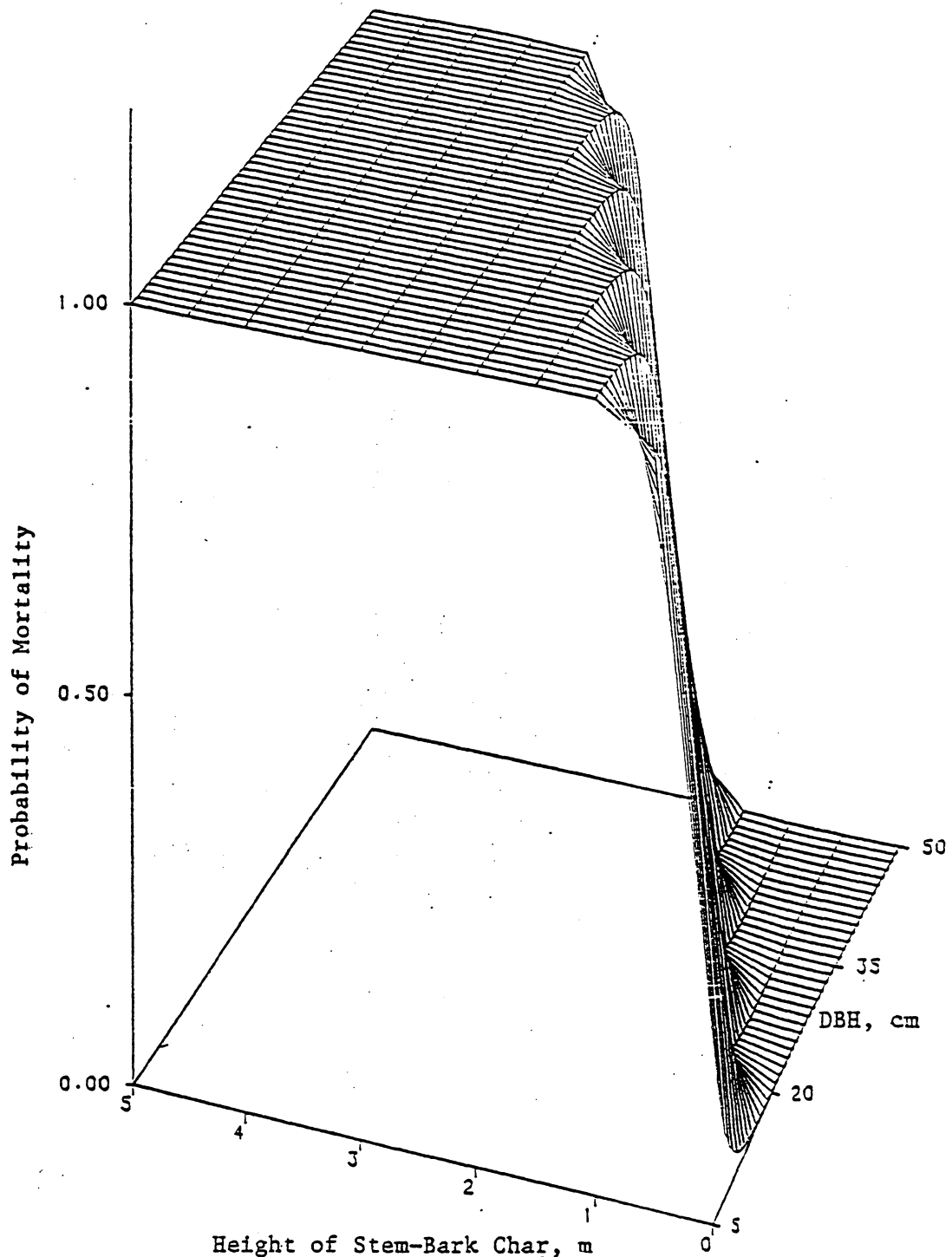
Figure 12. Predicted probability of mortality (top-kill) based on data from 113 red maple trees between 3 and 18 cm DBH in mixed oak stands burned by wildfire in May, 1986, in Shenandoah National Park, Virginia. The study stands were burned by headfires and the height of stem-bark char ranged from 0 to 1.8 m.

consistent with the observed responses of sample trees. Concordant pairs are defined as follows. For a given pair of observations having different values of the response variable, if a tree that died has a greater predicted probability of mortality than a tree that survived, that is a concordant pair of observations. The C statistic for these models ranges from 0.798 for red maple to 0.956 for blackgum, indicating that the models have good predictive ability (Table 9).

Abundance, Species composition, and Origin of Regeneration

Effects of fire on basal sprouting

Most deciduous tree species found in Appalachian oak forests are capable of producing basal sprouts in response to stress, such as heat injury to the cambium, or cutting of the above-ground portion of the tree. For the purposes of this study, basal sprouts are defined as any sprouts arising from dormant buds which originate within 0.5 m of the groundline, and are analogous to stump sprouts in the literature concerned with regeneration following clear-felling, with one important distinction. Stump sprouts arise from cut stumps, and as such, the above-ground portion of the parent tree is not only dead, it is removed. In contrast, when a tree is injured by fire, basal sprouts can be produced by trees whether the above-ground stem is killed or not.

Direct comparisons of basal sprouting frequency, height growth of basal sprouts, and number of stems per basal sprout clump were not made

between the high and low levels of fire intensity, because differences
in these phenomena between the two levels of fire intensity are caused
by varying rates of mortality and amount of sunlight reaching the forest
floor. Instead, comparisons were made between fire-killed and surviving
stems, and data from the two levels of fire intensity were pooled.
Analysis of covariance indicated that top-killed trees had higher
sprouting frequency than did trees that survived the fire, after
accounting for the effect of varying DBH (Table 10). Although scarlet
oak, northern red oak, pignut hickory, and blackgum all exhibited this
trend, the difference in sprouting frequency between fire-killed and
surviving trees was significant at the 5% level for only one species,
chestnut oak (Table 10). Red maple and serviceberry (Amelanchier arborea
Michx.) exhibited less difference in sprouting frequency between
fire-killed and surviving stems than did upland oaks and pignut hickory
(Table 10). Sprouting frequency did not differ significantly between
species after accounting for the effect of varying DBH and status
(top-killed or surviving); sprouting frequency was high for fire-killed
stems of all species, ranging from 75% to 94% (Table 10).

The average DBH of sprouting, fire-killed chestnut oaks was smaller
than the average DBH of fire-killed chestnut oaks that did not sprout
(Table 11). Mean DBH of fire-killed and surviving stems did not differ
between sprouting and nonsprouting stems of all other species (Table
11). Sprouting frequency of fire-killed chestnut oak was greatest for
trees less than 15 cm DBH, and lowest for trees greater than 30 cm DBH,
but other species did not show distinct trends in sprouting frequency by

Table 10. Sprouting frequency of top-killed and surviving trees in mixed oak stands in Shenandoah National Park, Virginia, 1-1/2 growing seasons following wildfire.

Species	Status	
	Top-killed	Surviving
	--- Sprouting Frequency (%) ---	
Chestnut oak	87.9a[1]	61.2b
Scarlet oak	87.4a	59.9a
N. red oak	84.5a	62.2a
Pignut hickory	76.4a	58.8a
Blackgum	93.7a	84.8a
Serviceberry	90.0a	89.2a
Red maple	94.4a	91.9a

[1] Means within species followed by the same letter do not differ at alpha = 0.05 (One-Way ANOVA for completely randomized design followed by Fisher's LSD t-test).

Table 11. Mean diameter and height of stem-bark char by species for sprouting and nonsprouting trees in mixed oak stands in Shenandoah National Park, Virginia, 1-1/2 growing seasons following wildfire.

Species	Sprouting Class	Status		Surviving
		Surviving	Top-killed	Char Ht. (m)[1]
		---- DBH (cm) ----		
Chestnut oak	sprouted	20.4a[2]	14.4a	1.1a[3]
	no sprouts	23.2a	22.5b	0.5a
Scarlet oak	sprouted	18.8a	17.5a	0.4a
	no sprouts	21.6a	15.6a	0.3a
Red oak	sprouted	18.0a	23.1a	0.6a
	no sprouts	18.9a	19.0a	0.4a
Pignut hickory	sprouted	12.4a	7.6a	0.6a
	no sprouts	10.3a	7.1a	0.2b
Blackgum	sprouted	7.7a	6.3a	1.2a[4]
	no sprouts	12.8a	6.2a	0.2a

[1] Char height is the height of stem-bark char, defined as the distance from groundline to the highest point of blackening on the uphill face of the tree.

[2] Means within species and status (columns) followed by the same letter do not differ between sprouting class at alpha = 0.05 (Wilcoxon Rank Sum Test).

[3] Mean heights of char of sprouting and nonsprouting chestnut oaks differ at alpha = 0.07 (Wilcoxon Rank Sum Test).

[4] Mean heights of char of sprouting and nonsprouting blackgums differ at alpha = 0.055 (Wilcoxon Rank Sum Test).

diameter class (Table 12).

These data provide evidence that basal sprouts are produced by surviving stems in response to increasing severity of fire injury (Table 11). The mean height of stem-bark char is greater for surviving stems that produced basal sprouts than for nonsprouting, surviving stems, although the magnitude and statistical significance of the difference varied by species (Table 11). Height of stem-bark char was not compared between sprouting and nonsprouting stems of fire-killed trees because these trees had obviously sustained enough heat injury to cause mortality.

Logistic regression equations were developed to predict the probability of a tree producing basal sprouts in response to fire (Table 13). Basal sprouting probability models differed between chestnut oaks and the red oak group, but not between scarlet and northern red oaks, as was the case with mortality prediction models (Table 13, Table 9). Tree DBH and height of stem-bark char were the best variables for prediction of chestnut oak sprouting probability, and height of stem-bark char and parent tree status (fire-killed or surviving) were the best predictors of sprouting probability of scarlet and northern red oaks (Table 13). The response surface for predicted probability of basal sprouting for chestnut oak is presented in Figure 13.

Data from stands burned by high-intensity fire were used to compare the mean height of the tallest stem per sprout clump and number of stems per sprout clump between fire-killed and surviving stems of chestnut oak, and between chestnut oak and the red oak group (scarlet and

Table 12. Sprouting frequency by species and diameter class for top-killed trees in mixed oak stands in Shenandoah National Park, Virginia, following wildfire.

Species	DBH Class (cm)		
	1 - 14.9	15.0 - 29.9	>30.0
	--------- Sprouting Frequency (%) ---------		
Chestnut oak	92.8a[1]	81.1a	41.7b
Scarlet oak	80.9a	90.4a	100.0a
Red oak	80.0a	100.0a	75.0a
Pignut hickory	75.8a	95.8a	----

[1] Means within species followed by the same letters do not differ at alpha = 0.05 (One-Way ANOVA for completely randomized design followed by Fisher's LSD t-test).

Table 13. Logistic regression models for prediction of the probability of basal sprouting following wildfire in mixed oak forests in Shenandoah National Park, Virginia.

Chestnut oak model

$$P(sprout) = [1 + exp(-(2.0577 - 0.1701(dbh) + 1.9970(char)))]^{-1}$$

where:

P(sprout) = probability of producing basal sprouts
dbh = diameter at breast height (cm)
char = height of stem-bark char (m)

Red oak model[1]

$$P(sprout) = [1 + exp(-(-0.4787 + 0.6770(char) + 1.8836(stat)))]^{-1}$$

where:

P(sprout) = probability of producing basal sprouts
char = height of stem-bark char (m)
stat = 0 if tree stem survived the fire
       1 if tree was top-killed by the fire

Species	n	LRS[2]	p - value	C[3]	R[4]
Chestnut oak	734	341.91	< 0.0001	0.875	0.592
Red oak	265	69.33	< 0.0001	0.82	0.45

[1] Red oaks include scarlet and northern red oak.

[2] The Likelihood Ratio Statistic (LRS) is a chi-square statistic that tests model goodness-of-fit (Harrel, 1986).

[3] The C statistic is the proportion of concordant pairs and is a measure of the predictive ability of the model (Harrel 1986). It can range from 0 to 1; values closer to 1 indicate better predictive ability.

[4] The R statistic measures the proportion of log likelihood explained by the model and is somewhat analogous to the coefficient of determination in Least Squares Regression (Harrel 1986).
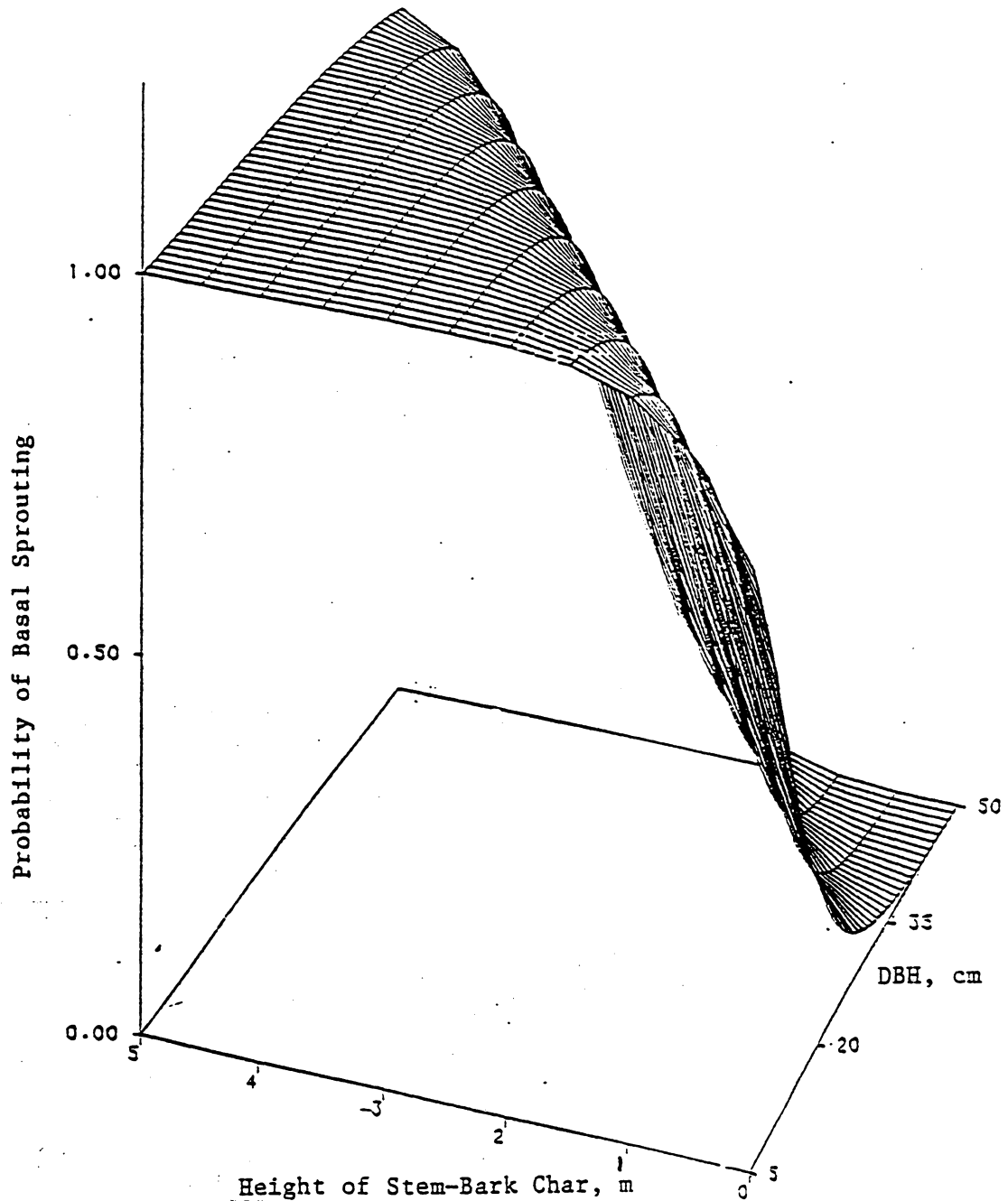
Figure 13. Predicted probability of basal sprouting based on data from 734 chestnut oak trees between 6 and 53 cm DBH in mixed oak stands burned by wildfire in May, 1986, in Shenandoah National Park, Virginia. The study stands were burned by headfires and the height of stem-bark char ranged from 0 to 5 m. Top-killed and surviving stems both produced basal sprouts.

northern red oak). Data from the two levels of fire intensity could not be pooled due to major differences in environmental conditions such as temperature and amount of sunlight reaching the forest floor, resulting from differences in canopy cover between the two levels of fire intensity. Sprout height and clump density could not be compared between levels of fire intensity due to small sample size in the low level of fire intensity.

At the end of the second growing season following burning, the mean height of the tallest chestnut oak sprout per basal sprout clump was 1.70 m, compared to 1.55 m for red oaks, indicating that chestnut oak sprouts grow faster than red oak sprouts, after accounting for the effect of varying parent-tree DBH (p = 0.06, Analysis of covariance). The number of stems per basal sprout clump two growing seasons after high-intensity fire did not differ between chestnut oaks and red oaks, and averaged 19.4 stems and 17.2 stems, respectively.

Basal sprout clumps produced by surviving stems of chestnut oak had lower clump densities and slower rates of height growth than basal sprout clumps produced by fire-killed chestnut oaks (p < 0.055 and p < 0.001, respectively), after accounting for the effect of varying parent tree DBH by the use of analysis of covariance. Sprout clumps arising from fire-killed chestnut oaks had an average of 19.9 stems per clump whereas sprout clumps arising from surviving chestnut oaks averaged 12.3 stems per clump. Mean height of the tallest sprout per clump was 1.74 m for fire-killed chestnut oaks and 1.41 m for surviving chestnut oaks.

Shrub strata

Various levels of fire intensity exerted different effects on the density of stems between 1 and 5 m in height. The greatest density of stems in the shrub strata was 7824 stems/ha, found in stands burned by high-intensity wildfire (Table 14). Unburned stands averaged 981 shrub strata stems/ha and stands burned by low intensity fire averaged only 370 shrub strata stems/ha (Table 14). However, no significant differences occurred between levels of fire intensity in terms of community diversity or species richness (Table 15).

Chestnut oak had the highest relative density of shrub stems in unburned stands and stands burned by high-intensity fires, and had the second greatest relative density in stands burned by low-intensity fires, where pignut hickory formed the greatest proportion of shrub stems (Table 14). Basal sprouts comprised the majority of chestnut oak stems in the shrub strata of stands burned by low-intensity fire, whereas the majority of pignut hickory shrub strata stems were individuals that had survived the fire (Table 16). The data indicate that the relative density of stems in the shrub strata of several species is decreased by high-intensity fire. These species include pignut hickory, red maple, blackgum, serviceberry, dogwood (Cornus florida L.), witch hazel (Hamamelis virginia L.), mountain laurel (Kalmia latifolia L.), and American chestnut (Castanea dentata Marsh.) (Table 14). Pioneer species such as black locust and tree-of-heaven (Ailanthus altissima (Mill.) Swingle) were only found in stands burned

Table 14. Relative density by species of stems in the shrub strata (woody stems 1 m to 5 m in height) of mixed oak stands two growing seasons after burning by wildfire, in Shenandoah National Park, Virginia.

Species	Fire Intensity		
	High	Low	Unburned
	------------------ Density (%) --------------		
Chestnut oak	34.7a[1]A[2,3]	16.2abA	18.5aA
Sassafras	31.7aA	13.5abcA	13.2abcA
Scarlet oak	12.9bA	9.6bcdA	13.0abA
Blackgum	3.9bcA	7.9bcdA	8.9bcA
N. red oak	7.8bcA	0.0dB	0.0cB
Pignut hickory	1.2efA	27.1aA	2.3bcA
Red maple	1.9efA	6.5bcdA	10.5abA
Serviceberry	1.5defA	6.6bcdA	7.3abcA
Black locust	1.7deA	0.0dB	0.0cB
Tree-of-heaven	0.5efA	0.0dA	0.0cA
Dogwood	0.9efA	1.6dA	9.0aAB
Black oak	0.3efA	0.0dA	0.0cA
Witch hazel	0.0fA	7.8cdeA	12.9abA
Mountain laurel	0.0fA	0.0dA	3.8abcA
Grapevine	0.3efA	0.0dA	0.0cA
Blackjack oak	0.2efA	0.0dA	0.0cA
Chinkapin	0.2efA	0.0dA	0.0cA
American chestnut	0.1efA	3.2cdA	6.9abcA
Viburnum spp.	0.0fA	0.0dA	0.4bcA
Bear oak	0.0fA	0.0dA	0.9bcA
	----------- Density (# stems/ha) --------		
All Woody Species	7824 A	370 B	981 B

[1] Means for species within levels of fire intensity (columns) followed by the same lower case letter do not differ at alpha=0.05 (One-Way ANOVA for completely randomized design followed by Fisher's LSD t-test).

[2] Means within species followed by the same upper case letter do not differ between levels of fire intensity at alpha=0.05 (One-Way ANOVA for completely randomized design followed by Fisher's LSD t-test).

[3] Statistical analyses were performed on arc-sine transformed percentages.

Table 15. Community diversity (H')[1] and species richness (R)[2] for the shrub strata (stems 1 - 5 m in height) and the regeneration strata (stems less than 1 m in height) of mixed oak stands, two growing seasons after burning by wildfire, in Shenandoah National Park, Virginia.

	Shrub Strata			Regeneration Strata		
	Fire Intensity					
	High	Low	Unburned	High	Low	Unburned
H'	0.71a[3]	0.67a	0.70a	0.78a	0.73a	0.76a
R	4.03a	4.12a	4.32a	6.14a	6.35a	5.74a

[1] $H' = - \text{SUM} (p_i * \log p_i)$ where p_i = relative density of the ith species. From Pielou (1977).

[2] R = (Number of species - 1)/ log (number of individuals).

[3] Means within a strata and index (rows) followed by the same letter do not differ at alpha = 0.05 (One-Way ANOVA for completely randomized design followed by Fisher's LSD t-test).

Table 16. Relative density by species and origin of stems in the shrub strata (woody stems 1 m to 5 m in height) of mixed oak stands, two growing seasons after burning by wildfire, in Shenandoah National Park, Virginia.

Species	Origin	Fire Intensity	
		High	Low
		------- Density (%) -------	
Chestnut oak	basal sprout	34.4a[1] A[2,3]	16.2a A
|	seedling sprout	0.3ef A	0.0d A
Sassafras	basal sprout	0.7ef A	3.3cd A
|	seedling sprout	31.1a A	10.2ab A
Scarlet oak	basal sprout	12.2b A	7.9bc A
|	seedling sprout	0.7ef A	1.7cd A
Blackgum	basal sprout	2.9d A	0.0d B
|	seedling sprout	0.5ef A	0.0d A
|	surviving stem	0.4ef A	7.9bc A
N. red oak	basal sprout	7.8c A	0.0d B
Pignut hickory	basal sprout	1.2def A	1.7cd A
|	surviving stem	0.0f A	25.4a A
Red maple	basal sprout	1.9def A	4.9bcd A
|	surviving stem	0.0f A	1.6cd A
Serviceberry	basal sprout	1.5de A	1.6cd A
|	surviving stem	0.0f A	5.0cd A
Black locust	basal sprout	0.7ef A	0.0d A
|	seedling sprout	0.9de A	0.0d A
Tree-of-heaven	Seedling	0.5ef A	0.0d AB
Dogwood	basal sprout	0.2ef A	0.0d A
|	seedling sprout	0.7ef A	0.0d A
|	surviving stem	0.0f A	1.6cd A
Black oak	basal sprout	0.2ef A	0.0d A
|	seedling sprout	0.1ef A	0.0d A
Witch hazel	surviving stem	0.0f A	7.8bc A
Grapevine	seedling sprout	0.3ef A	0.0d A
Blackjack oak	seedling sprout	0.2ef A	0.0d A
Chinkapin	seedling sprout	0.2ef A	0.0d A
American chestnut	seedling sprout	0.1ef A	3.2bcd A

	--- Density (# stems/ha) ---	
All Woody Species and Origins	7824 A	370 B

[1] Means for species and origin combinations within levels of fire intensity (columns) followed by the same lower case letter do not differ at alpha=0.05 (One-Way ANOVA followed by Fisher's LSD t-test).
[2] Means within species and origin combinations (rows) followed by the same upper case letter do not differ between levels of fire intensity at alpha=0.05 (One-Way ANOVA followed by Fisher's LSD t-test).
[3] Statistical analyses were performed on arc-sine transformed percentages.

by high-intensity fire (Table 14).

Many species were able to regenerate via both sprouts and seed, but most stems which were able to grow taller than 1 m within two growing seasons after burning were either basal sprouts or seedling sprouts (Table 16). Basal sprouts formed the greatest proportion of shrub strata stems of all upland oak species, blackgum, pignut hickory, red maple, and serviceberry in stands burned by high-intensity fires, and the greatest proportion of upland oaks and red maple in stands burned by fires of low intensity (Table 16). Sassafras and black locust (Robinia pseudoacacia L.) are both capable of producing root sprouts which make it extremely difficult to distinguish which stems come from separate rootstocks. Seedling sprouts and root sprouts of these species were therefore grouped together and referred to as seedling sprouts, which comprised the majority of stems of these two species in stands burned by both levels of fire intensity. Stems of blackgum, pignut hickory, red maple, serviceberry, dogwood, and witch hazel which were between 1 and 5 m tall were able to survive low-intensity fires, and stems of hickory survived low-intensity fires with such high frequency to make it the most abundant species in the shrub strata in stands affected by low-intensity fire (Table 16, Table 14). The only shrub strata stems to survive high-intensity fires were blackgums (Table 16); most of these individuals appeared to have suffered considerable fire injury and all had produced basal sprouts. Trends which were evident in terms of density of stems in the shrub strata were similar with regard to frequency of occurrence of the various species and origin combinations

found in the shrub strata.

The tallest stems in the shrub strata across all levels of fire intensity were those that had been able to survive the fire (Table 17). For species that have stems which survived the fire, the mean height of stems is greater in stands burned by low-intensity fires than in unburned stands (Table 17). Within each level of fire intensity, basal sprouts tend to be taller than seedling sprouts, although the differences are not statistically significant (Table 17). Tree-of-heaven is the only species for which stems of seedling origin can match the height growth of oak basal sprouts: mean height of tree-of-heaven seedlings was 1.92 m and mean height of chestnut oak basal sprouts was 1.72 m two growing seasons after high-intensity fire (Table 17).

Regeneration strata

As might be expected, there was a slight trend of greater numbers of stems/ha in the regeneration strata (stems less 1 m tall) in stands burned by high-intensity fire (Table 18). Species richness and community diversity did not differ significantly between different levels of fire intensity (Table 15). Vaccinium spp. accounted for the greatest proportion of stems in the regeneration strata across all levels of fire intensity (Table 18). The relative density of chestnut oak was greatest in unburned stands, 15.4%, and lowest in stands burned by high-intensity fire, 4.9% (Table 18), in spite of the large number of chestnut oaks in basal sprout clumps. In contrast, sassafras and

Table 17. Mean height by species and origin of stems between 1 and 5 m in height in mixed oak stands in Shenandoah National Park, Virginia, two growing seasons following wildfire.

Species	Origin	Fire Intensity		
		High	Low	Unburned
		--------- Height (m) ------------		
Blackgum	surviving	3.80a[1]	3.78ab	3.51a
Tree-of-heaven	seedling	1.92b	----	----
Chestnut oak	basal sprout	1.72bc	1.17c	----
N. red oak	basal sprout	1.56bc	----	----
Scarlet oak	basal sprout	1.55bc	1.13c	----
Sassafras	seedling sprout	1.25bc	1.20c	----
Pignut hickory	basal sprout	1.21bc	----	----
Blackgum	basal sprout	1.18bc	----	----
Blackgum	seedling sprout	1.17bc	----	----
Serviceberry	surviving	----	----	3.53a
Red maple	surviving	----	----	2.10bcd
Dogwood	surviving	----	----	3.33ab
Pignut hickory	surviving	----	3.42bc	3.24abc
Witch hazel	surviving	----	2.50c	2.06cd
White pine	surviving	----	----	2.30abcd
Am. chestnut	surviving	----	----	2.17bcd
Sassafras	surviving	----	----	1.77d
Chestnut oak	surviving	----	----	1.76d
Scarlet oak	surviving	----	----	1.50d
Mountain laurel	surviving	----	----	1.46d
Azalea spp.	surviving	----	----	1.34d
Vaccinium spp.	surviving	----	----	1.36d
Viburnum spp.	surviving	----	----	1.17d
Bear oak	surviving	----	----	1.10d

[1] Means followed by the same letter do not differ at alpha = 0.05 (Two-Way ANOVA for completely randomized design followed by Fisher's LSD t-test).

Table 18. Relative density by species of stems less than 1 m tall in mixed oak stands, two growing seasons after burning by wildfire, in Shenandoah National Park, Virginia.

Species	Fire Intensity		
	High	Low	Unburned
	------- Number of stems/ha (%) ---------		
Vaccinium spp.	41.0a[1,2]	52.5a	50.0a
Chestnut oak	4.9a	9.7ab	15.4b
Sassafras	12.4a	8.6a	3.3a
Grapevine spp.	15.8a	3.4ab	0.1b
Mountain laurel	7.5a	4.5a	4.9a
Pignut hickory	3.5a	0.8a	0.3a
Azalea spp.	3.1a	4.5a	10.7a
Blackgum	2.3a	3.6a	0.1b
Smilax spp.	2.0a	0.3a	1.2a
Black locust	1.5a	0.0b	0.0b
Bear oak	1.2a	0.2a	1.0a
Black oak	1.0a	0.0a	0.0a
Red maple	0.6a	4.1a	2.8a
N. red oak	0.5a	1.5a	1.7a
Viburnum spp.	0.5a	1.6a	2.8a
Witch hazel	0.5a	1.0a	2.8a
Blackjack oak	0.5a	0.5a	0.0a
Virginia creeper	0.4a	0.1a	0.0a
Scarlet oak	0.3a	0.7a	0.6a
Dogwood	0.1a	0.8a	0.2a
Tree-of-heaven	0.1a	0.0a	0.0a
Pitch pine	0.1a	0.0a	0.0a
Rubus spp.	0.0a	0.8a	0.0a
Serviceberry	0.0a	0.3ab	2.2b
Yellow-poplar	0.0a	0.1a	0.0a
American chestnut	0.0a	0.0a	0.1a
White pine	0.0a	0.0a	0.3a
Striped maple	0.0a	0.0a	0.3a
White ash	0.0a	0.0a	0.1a
	--------- Number of stems/ha ----------		
All species	230,139a	162,639a	155,417a

[1] Means within species followed by the same letter do not differ at alpha = 0.05 (One-Way ANOVA for completely randomized design followed by Fisher's LSD t-test performed on arc-sine transformed percentages).

[2] Statistical analyses were performed on arc-sine transformed percentages.

grapevines formed a larger proportion of the regeneration strata in stands burned by high intensity fire than in stands burned by low-intensity fire or in unburned stands (Table 18).

DISCUSSION

Effects of Wildfire on the Structure and Composition of Mixed Oak Forests

High intensity-fire had a much more profound impact on the structure and composition of mixed oak forests than low-intensity fire. Canopy mortality due to fire-injury was much greater in stands burned by high-intensity fire, in terms of both number of stems and basal area per hectare. Although mean basal area reduction was 67% in stands burned by high-intensity fire, basal area reduction ranged from 31% to 93% in these stands (Table 4). It may appear that the stand with only 31% fire-caused basal area reduction burned with a lower intensity. This is true to a certain extent, when one looks at the range of height of stem-bark char, but this same stand experienced 68% canopy mortality in terms of number of stems per hectare (Table 4). Field observations indicated that this stand contained a disproportionate number of large wolf-trees of chestnut oak and eastern white pine (Table 1). The degree of fire-induced canopy mortality will depend not only on fire intensity, but also on such factors as species composition, diameter distribution, stand age, fire type, season of burning, and environmental parameters such as temperature, wind direction and speed, and relative humidity. The high variability in the structure and composition of Appalachian oak forests can introduce large variation in the effects of wildfire on these forests, even when many of the above factors are held constant.

The post-fire condition of stands burned by high-intensity fires

resembled stands which had been clearcut (Figure 14). Stand basal area was reduced to such a degree that the amount of sunlight reaching the forest floor was nearly 100%, and regeneration development was vigorous in terms of density and height growth; 99.6% of all stems between 1 and 5 m tall were new regenerants (Table 16), with a few scattered surviving blackgums being the only stems less than 5 m tall to survive the fire. Ross and others (1986) reported the density and height growth of regeneration three years after clear-felling of mixed oak stands in the Ridge and Valley of Virginia; the sites examined in that study seem fairly comparable to the sites in the present study in terms of geologic parent materials, soils, site quality, and species composition prior to canopy removal. Stands burned by high-intensity fires averaged 7824 stems/ha (Table 14), and the mean height of basal sprouts that were greater than 1 m tall was 1.7 m for chestnut oak and 1.6 m for scarlet and northern red oaks (Table 17), compared to 35,500 stems/ha, and mean height of oak stump sprouts of 2.6 m reported by Ross and others (1986). However, it would appear that the difference is largely attributable to the fact that the present study reports figures from two growing seasons following canopy removal due to fire, whereas Ross and others (1986) report the condition of stands three years after clear-felling. Since approximately 230,000 stems/ha less than 1 m tall were present in the post-burn stands two growing seasons after the fire (Table 18), it seems likely that many of these stems will grow into the next height class during the third growing season following the fire.

The trends evident with regard to differences between species in

Figure 14. Mixed oak stand burned by high-intensity surface headfire in May, 1986 in Shenandoah National Park, Virginia. Note vigorous regeneration development and absence of canopy cover. Photograph taken by the author in June, 1987.

rate of height growth following high-intensity fire are similar to those reported by Ross and others (1986). With the exception of tree-of-heaven seedlings, which were not present in the stands studied by Ross and others (1986), oak basal sprouts are the tallest stems of new regeneration following high-intensity fires (Table 17). The results of this study also suggest that chestnut oak basal sprouts grow faster than scarlet and northern red oak basal sprouts (Table 17), after accounting for varying parent tree attributes, which concurs with the results of Ross and others (1986). Mean height of the tallest chestnut oak basal sprout per clump two growing seasons after high-intensity fire was 1.72 m (Table 17), and mean density of chestnut oak sprout clumps was 19.2 stems. Similarly, Ross et al. (1986) report a mean height of the tallest sprout per chestnut oak sprout clump of 1.74 m, and mean clump density of 18.8 stems, two growing seasons after clear-felling.

Low-intensity fires caused much less canopy mortality than did high intensity fires (Table 4, Figure 15); this resulted in considerably less vigorous understory devlopment (Tables 14, 16, and 17). The most notable effect of low-intensity fires was the reduction in the number of stems between 1 and 5 m tall (Table 14), and the increase in mean height of surviving stems between 1 and 5 m tall (Table 17), due to mortality of smaller stems within this height class. The number of stems less than 1 m in height was slightly higher for stands burned by low-intensity fires than for unburned stands, this is consistent with the findings of Teuke and Van Lear (1982), McGee (1979), and Thor and Nichols (1974), who reported that low-intensity fires will cause an increase in the density

Figure 15. Mixed oak stand burned by low-intensity surface headfire in May, 1986 in Shenandoah National Park, Virginia. Photograph taken by the author in June, 1987.

of small regeneration. The lower density of stems between 1 and 5 m

tall (Table 14), and the decreased height of new regenerants in stands

burned by low-intensity fire (Table 17) relative to stands burned by

high-intensity fire, provides evidence that the rate of height growth is

slower in stands burned by fires of lower intensity. This is probably

reflective of lower light levels under the canopy of stands burned by

fires of lower intensity, resulting from less canopy mortality in these

stands (Table 4, Figure 15).

One effect of low-intensity fires that was not quantified in the

present study is possible degradation of stem quality due to

fire-induced basal wounding and subsequent invasion by wood-rotting

fungi. The measured heights of stem-bark char, when examined in light

of past work on basal wounding by fire, indicate that a significant

number of stems are likely to develop butt rot (Loomis 1974). This

would hold true for the few surviving trees in stands burned by

high-intensity fire, as well as the greater number of trees which

survived low-intensity fire.

Effect of wildfire on basal sprouting

Sprouting frequency of fire-killed stems of all major tree species

in burned stands ranged from 76% to 94%, and did not differ

significantly between species (Table 10). Sprouting frequencies of

fire-killed oaks ranged from 84% to 88% (Table 10), compared to

sprouting frequencies ranging from 58% to 72% for cut oak stumps

reported by Ross and others (1986). Both studies reported sprouting frequency as the percentage of stems which bore live basal sprouts two growing seasons following top-kill. Sprouting frequencies by diameter class (Table 11) for chestnut and scarlet oak were similar to those reported by Kays and others (1988) following a growing season harvest in the Virginia Piedmont, except that Kays and others (1988) did not report such a drastic decrease in sprouting frequency of large chestnut oaks. Previous work has indicated that trees of larger diameter are less likely to produce basal sprouts following cutting (Ross et al. 1986, Kays et al 1988). The mean DBH of both fire-killed and surviving chestnut oaks that produced basal sprouts was smaller than the mean DBH of chestnut oaks that did not produce basal sprouts, but this trend was not evident for scarlet oak, northern red oak, or pignut hickory (Table 11). Logistic regression analysis also indicated that the probability of sprouting decreased with increasing DBH for chestnut oak, but that DBH did not significantly affect probability of sprouting for scarlet and northern red oak in this study (Table 13, Figure 13). The reasons for this are unclear, but the better fit of the chestnut oak sprouting probability model (Table 13) suggests that the apparent lack of any effect of diameter on sprouting of scarlet and northern red oak in the present study may be related to the smaller sample size and range of DBH for red oaks sampled (Table 8).

The data from this study suggest that trees which survive fires will produce basal sprouts in response to the degree of fire injury sustained, and that the height of stem-bark char can be used as a

measure of the severity of injury. This theory is supported by the greater mean height of stem-bark char for surviving trees which produced basal sprouts than for nonsprouting trees, as well as the positive coefficients and statistical significance of height of stem-bark char in the sprouting probability models for both chestnut oak and the red oak group (Tables 11 and 13, Figure 13). The sprouting of dormant buds is inhibited by auxins produced in the leaves and buds and translocated to the roots. Fire injury, such as partial girdling of the cambium, will disrupt the translocation of these hormones, which will cause sprouting from dormant buds.

Prediction of stand level mortality

The mean height of stem-bark char was found to be a good predictor of the percentage of pre-fire basal area and number of stems per hectare killed by fires of variable intensity (Figures 6 and 7). Theoretically, it would seem that a linear model should not be the best model form for prediction of the percentage of fire-induced canopy mortality, as percent canopy mortality should asymptotically approach 100% as fire intensity increases. Various models, using the inverse of mean char height in order to account for the theorized asymptotic nature of the relationship, and the natural logarithm of percent canopy mortality to account for increased variance in the response variable for larger values of the predictor variable, were fit to the data, but none fit as well as the simple linear model. It appears that the relationship

between height of stem-bark char and percent fire-induced canopy
mortality is linear within the range of the data, and extrapolation
beyond the ranges listed in Table 4 and depicted in Figures 6 and 7 is
not advisable. In fact, the actual regression coefficients are likely to
be useful for prediction of fire-induced canopy mortality only for
conditions quite similar to those used in their development. These
conditions include the following: stands are mixed oak forests dominated
by chestnut, scarlet, and northern red oak averaging about 18 cm DBH;
stand basal area is between 20 and 30 m^2/ha; stand age is approximately
50 to 60 years; upland oak site index is approximately 18 m (base age
50); and stands are burned by headfires during the growing season
(Tables 1, 2, 4, and 8). However, even if the relationship between
height of stem-bark char and canopy mortality in Figures 6 and 7 is only
valid given the above conditions, the strength of the relationship
suggests an important possibility. Few studies in which the
relationship between fire intensity and the resulting degree of canopy
mortality was quantified were found in the literature. Similarly, no
studies that quantified the relationship between fire intensity and
height of stem-bark char were found in the literature, although (Van
Wagner 1973) found a highly significant relationship between fire
intensity and height of crown scorch in conifer stands in Canada, and
Cain (1984) found evidence indicating that height of stem-bark char was
a reasonable measure of relative fire intensity in southern pine stands.
Intuitively, this relationship is very sensible, especially considering
the relationship between flame length and fire intensity (Equation 4)

described by Rothermel and Deeming (1980). Although it is possible that

the height of stem-bark char may be influenced by such factors as

ambient temperature, relative humidity, wind speed, amount and

flammability of fuels, fire type, species-specific bark characteristics,

and season of burning, as well as fire intensity, it would be desirable

to attempt to quantify the relationship between height of char and fire

intensity. If height of stem-bark char was found to be a good predictor

of fire intensity, the value of many post facto wildfire studies would

be greatly improved, since fire intensity could be estimated and related

to fire effects, and comparison of results of different would be greatly

facilitated. Height of stem-bark char is also considerably easier to

measure than fire intensity, and can be measured with greater accuracy

and precision than the rapidly fluctuating flame length, an important

practical consideration. The relationship between mean height of

stem-bark char and percentage of pre-fire basal area and number of stems

per hectare killed by fire depicted in Figures 6 and 7 indicates

considerable potential for height of stem-bark char to be used as an

estimator of fire intensity.

    Prediction of individual tree mortality

    Logistic multiple regression models were developed for prediction

of the probability of fire-induced mortality (top-kill) for individual

trees of chestnut, scarlet, and northern red oaks, pignut hickory, red

maple,and blackgum, as a function of height of stem-bark char and tree

DBH (Table 9, Figures 8 - 12). A chi-square test indicated that the probability of mortality in response to fire did not differ between scarlet and northern red oaks, so data from these species were combined to fit a single red oak group model. The chi-square tests indicated that all models presented in Table 9 and Figures 8 - 12 are significantly different at alpha = 0.05.

The models for prediction of the probability of fire-induced mortality have several very desirable characteristics. The models account for the effects of tree size, amount of heat directed to the bole, and species characteristics, on the probability of fire-induced tree mortality. The coefficients in Table 8 are biologically reasonable; the negative coefficient for DBH indicates decreasing probability of mortality with increasing DBH, and the positive coefficient for char height indicates increasing probability of mortality as more heat is directed to the bole (Table 9). The logistic equation is a highly desirable model form for prediction of mortality probabilities for several reasons. The predicted values can be interpreted not only as the probability of mortality for an individual stem, but also as the proportion of stems which are likely to die in a population of stems having equal values of the predictor variables (Monserud 1976, Daniels et al. 1979, Greene and Schilling 1987). A multiplicative interaction between predictor variables is also implied by the model form, which is logical, as tree mortality in response to fire is affected by the interaction of tree size and amount of heat directed to the tree bole. Due to the strong correlation between DBH and tree height in even-aged

stands of relatively shade-intolerant species, the models can account for mortality due to stem girdling and crown scorch.

Harmon (1984) developed linear and nonlinear models for prediction of the probability of trees surviving low-intensity surface fires (defined as fires which killed less than 25% of the pre-fire basal area, and estimated fire intensity between 67 and 117 kW/m, using the Van Wagner (1973) crown scorch equations) as a function of bark thickness, for chestnut oak, blackgum, red maple, and several other species not found in stands sampled in the present study. The Harmon (1984) models had good fit statistics, and the so-called sigmoidal survival function was also constrained to predict probabilities within the interval [0,1]. Although quite useful, the models had some undesirable characteristics as well. Bark thickness is much more difficult to measure than DBH, and species vary in their rates of bark accumulation (Harmon 1984). It is also more desirable to be able to differentiate the relative resistance of different species to fire-induced mortality on the basis of tree size, rather than bark thickness, and the Harmon (1984) models do not account for the effect of differing levels of fire intensity, which alter the amount of heat directed to individual stems.

The fact that the mortality probability models developed in this study differ between species indicates the existence of species-specific characteristics which influence susceptibility or resistance to fire-induced mortality. These may include bark thickness, in absolute terms or relative to stem diameter, differences in the insulating ability of the bark of various species independent of thickness, or

different relationships between DBH and crown height for different species. The implicit multiplicative interaction between predictor variables in the logistic model form means that one cannot compare relative fire resistance between species simply by comparing the magnitude of coefficients for each predictor variable. The most reliable method for comparison of relative fire resistance among species is to plot the response surfaces for each species and compare them visually. This is why all response surfaces (Figures 8 - 12) are plotted at the same scale, even though the range of predictor variables is less than shown in the response surfaces (Figures 8 - 12, Table 9) for all species except chestnut oak. Examination of response surfaces for probability of fire-induced mortality as a function of DBH and height of stem-bark char (Figures 8 - 12) indicates that blackgum and chestnut oak have the greatest resistance to fire-induced mortality, followed by the red oak group, pignut hickory, and red maple. The indication that blackgum has the greatest fire resistance of all species studied is consistent with the fact that blackgum was the only species for which stems in the 1 to 5 m height class were able to survive high-intensity fire.

The order of species-specific fire resistance is supported by evidence provided by past research. Hare (1965) found differences in the insulating ability of bark were related to bark thickness, but also found differences between species in bark insulating ability that were independent of thickness. Spalt and Reifsnyder (1962) found that the thermal conductivity of bark is linearly related to bulk density, and increased heat resistance of bark is related to an increase in the ratio

of corky outer bark (dead cells) to denser, inner bark. Blackgum and chestnut oak both possess thick, corky bark, which is likely an important factor influencing their relatively high fire resistance. Species with thinner, less corky bark such as red oaks, pignut hickory, and red maple are more likely to die as a result of heat injury than are chestnut oaks and blackgums of similar size and subjected to the same degree of heating.

Van Wagner (1973) stated that crown scorch is much more likely to cause mortality of pine species than cambial girdling, since the thick, corky bark of most pine species makes them relatively resistant to girdling. It is likely that any fire of sufficiently high intensity to cause girdling of pines will also completely scorch their foliage, resulting in mortality, and a significant amount of crown scorch can occur as a result of fires which are not intense enough to cause cambial injury in pines (Van Wagner 1973). Data and field observations from this study suggest that cambial girdling is the primary cause of fire-injury for many deciduous tree species. The ability of deciduous species to produce epicormic sprouts from dormant buds means that deciduous trees can replace scorched foliage, and supply moisture and nutrients to sustain the new foliage, provided that the vascular cambium is not girdled at some point between the root system and the crown. Many trees in stands burned by high-intensity fire were observed to produce epicormic sprouts. The ability of some blackgum stems which were less than 5 m in height to survive high-intensity fire, where stem-bark char reached as high as 5 m, also provides evidence that crown scorch is

likely to be a relatively insignificant cause of mortality for deciduous

trees (Table 17). The fact that mean height and DBH did not vary

between upland oaks (Tables 2 and 8) suggests that the greater

resistance to fire-induced mortality of chestnut oak relative to scarlet

and northern red oaks (Figures 8 and 9) is due to increased resistance

to cambial girdling due to differences in bark properties between these

species. Pitch pine has a characteristic rare for pine species, the

ability to produce epicormic and basal sprouts from dormant buds. Stands

of pitch pine and Table Mountain pine adjacent to the study stands were

burned by crown fires in which the foliage of all tree crowns was

completely consumed, and stem-bark char reached as high as total tree

height. Pitch pines in these stands were observed to produce both basal

and epicormic sprouts (Figure 16), providing additional evidence that

pines do have high resistance to cambial girdling, and that the ability

to sprout from dormant buds decreases the likelihood of mortality due to

crown scorch.

It is interesting to note that the predicted probability of basal

sprouting of chestnut oak (Figure 13) displays a pattern that is quite

similar to the predicted probability of mortality of chestnut oak

(Figure 8), although it appears that chestnut oaks will sprout in

response to less heat injury than is necessary to cause mortality

(top-kill). A question that this raises is whether many of the

surviving parent trees which produced basal sprouts will die as a result

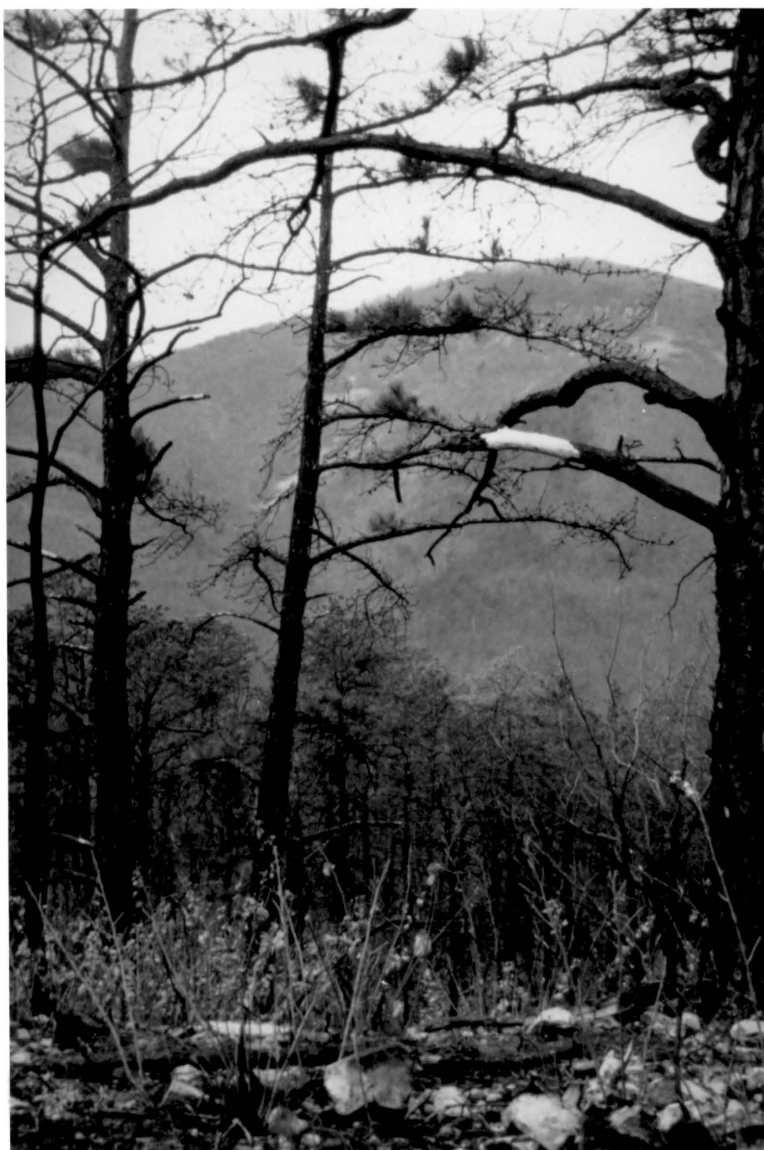of fire-injury after a few more growing seasons pass.

Figure 16. Pitch pine sprouting from dormant buds after suffering 100% crown scorch and stem-bark char from groundline to the tip of the terminal bud in a high intensity crown headfire in May, 1986. Photograph by the author in March, 1987.

CONCLUSIONS

Surface fires of high-intensity have a much greater impact on the structure and composition of mixed oak forests than do low-intensity surface fires. High-intensity fire can result in nearly complete canopy mortality (top-kill), whereas low-intensity fire will leave the overstory of mature oak stands nearly intact.

The height of stem-bark char was found to be a significant predictor of tree mortality, at both the individual tree level and at the whole stand level. As such, data from this study suggest that the height of stem-bark char has potential for use as an estimator of the relative amount of heat directed to individual tree boles, and as an estimator of average fire intensity over larger areas. The logistic models for prediction of the probability of tree mortality have utility in the assessment of damage to mixed oak stands, and provide a quantitative basis for making intelligent decisions regarding the management of stands which have been subjected to fire. It must be remembered that basal wounding and subsequent rot can develop in trees which survive fire injury, unfortunately it was not possible to study the development of rot subsequent to heat injury, as the destructive sampling required to measure cambial wound dimensions within a few growing seasons after fire is not permissible in National Parks. It would be possible to return to the study sites in 5 to 10 years to determine the amount of wounding sustained by trees surviving fire. This

could provide information useful for land managers faced with the decision of whether to fell a stand which burned or let it grow. It is undesirable for most landowners to allow poorly stocked stands, or stands which have many cull trees with severe rot to continue to grow, as stands such as this are undesirable both economically and aesthetically. Such stands are also often difficult to regenerate successfully to desirable species upon final harvest.

The high degree of canopy mortality in oak stands burned by high intensity fire results in the development of a new stand. It seems reasonable to conclude that oak species, especially chestnut oak, are likely to maintain dominance in the developing stand in proportion to their pre-fire dominance, due to prolific sprouting and competitive rate of height growth. Whether or not developing oak basal sprout clumps will be seriously affected by rot is open to question, but studies of oak stump sprouts following clear-felling indicate that sprouts which originate near the groundline have a high likelihood of becoming sound, high quality stems. Observations that most oak basal sprouts in the study stands originated within 0.1 m of the groudline indicate good potential for these stems to be free from rot.

Most stems which are taller than one meter tall two growing seasons after high-intensity fire were either basal sprouts produced by stems of the pre-fire overstory, or seedling sprouts. Field observations indicated that the majority of these stems were free of direct overhead crowding, and that stems of this size formed a nearly continuous cover over the forest floor, which would interfere with the development of

smaller stems. This supports previous research which indicates that seedlings germinating after a major disturbance such as severe fires or clear-felling have little chance of becoming dominant or codominant members of the developing stand.

Data from this study suggest a possible exception to this theory, however. Tree-of-heaven seedlings appear to be capable of matching the early height growth of oak basal sprouts. Tree-of-heaven stems composed a small proportion of the regeneration in stands burned by high-intensity fires, but individuals of this species were found in stands more than two miles from the nearest seed source. The prolific seed production and dissemination, and competitive growth rates characteristic of tree-of-heaven indicate potential for this aggressive pioneer to be an increasing competitive problem in eastern hardwood forests.

The main effect of low-intensity fires appears to be a decrease in the density of stems between 1 and 5 m tall. The greater number of stems less than a meter tall suggest that this is only a temporary condition. Low-intensity fires seemed to decrease the density of oak advance regeneration. However, this may change with time, as the vegetational community responds to the microenvironmental changes resulting from the fire. Low-intensity fires may also cause unacceptable degradation of stem quality in surviving trees.

Results of this study indicate that oak species are capable of successful regeneration following overstory removal on medium quality sites. Although it is possible that time will alter trends indicated by

the data, there is little evidence that low-intensity fires will significantly improve the establishment of oaks in the understory of mixed oak forests on medium quality sites.

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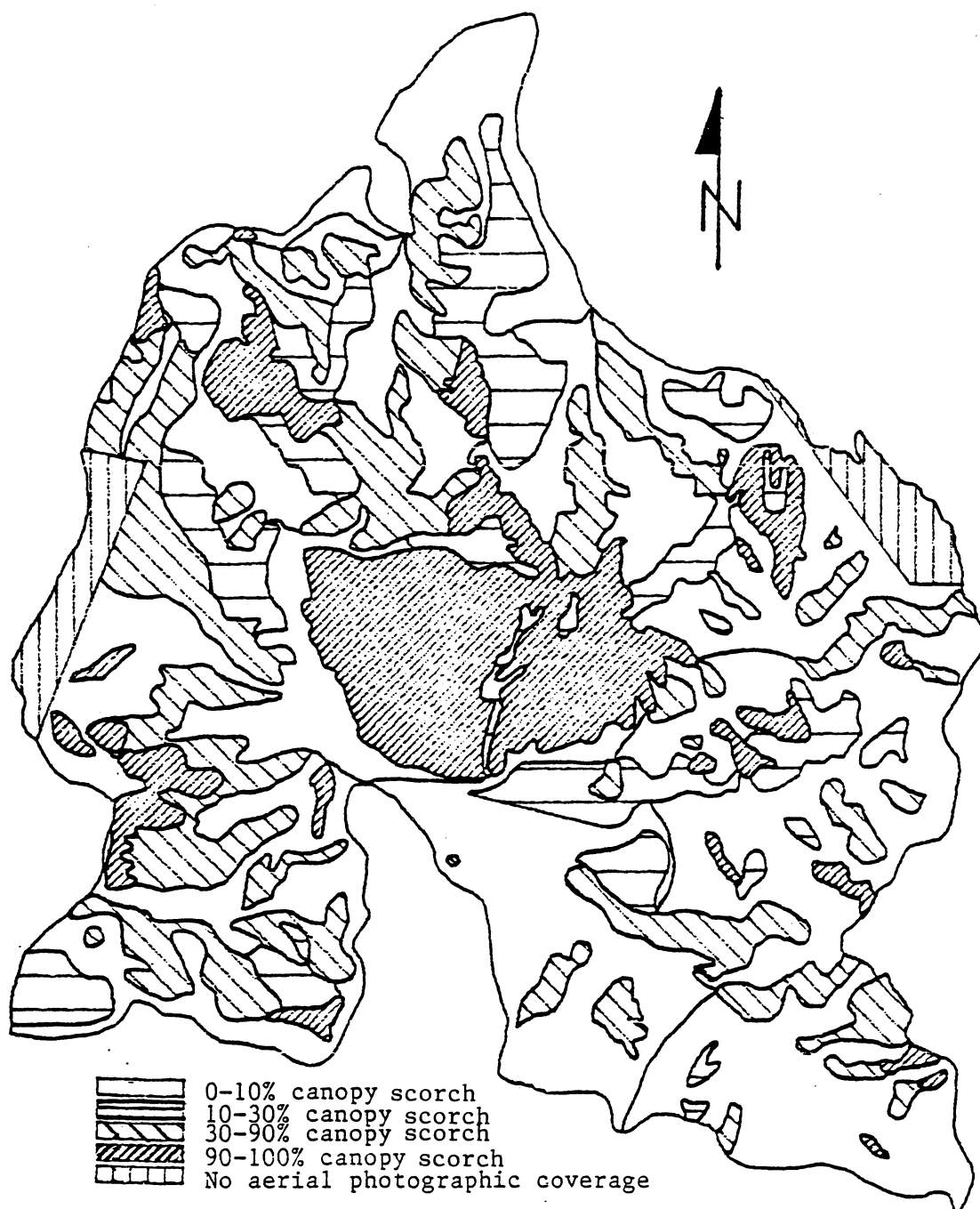Carvell, K.L., and E.H. Tryon. 1961. The effect of environmental

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0-10% canopy scorch
10-30% canopy scorch
30-90% canopy scorch
90-100% canopy scorch
No aerial photographic coverage

Appendix I. Map showing four classes of forest canopy destruction
as a result of the Big Run Fire, May 2-5, 1986, in the South District
of Shenandoah National Park, Rockingham County, Virginia. Map was
prepared from aerial color-infrared slides. Fire burned 1899 ha.
Courtesy of Resource Management Staff, Shenandoah National Park,
USDI National Park Service.

Scale 1: 36,000

Appendix II. Rankings for topographic parameters used for determination of the Forest Site Quality Index (FSQI). The sum of the ranks for aspect, slope inclination, and slope position is the FSQI, which ranges from 3 to 16, with higher values indicating higher site quality. FSQI estimates relative site quality by reflecting a topographic moisture gradient, and was developed in the Ridge and Valley Physiographic Province of Virginia, West Virginia, and Maryland (Wathen 1977, Meiners 1984).

Rank	Aspect Azimuth $^{\circ}$ Range	Slope Position	Slope Percent Range
1	196–260	shoulder	>60%
2	166–195, 261–280	backslope	45–59%
3	146–165, 281–340	summit	30–44%
4	0–20, 341–360	footslope	15–29%
5	81–145	floodplain	0–14%
6	21–84		
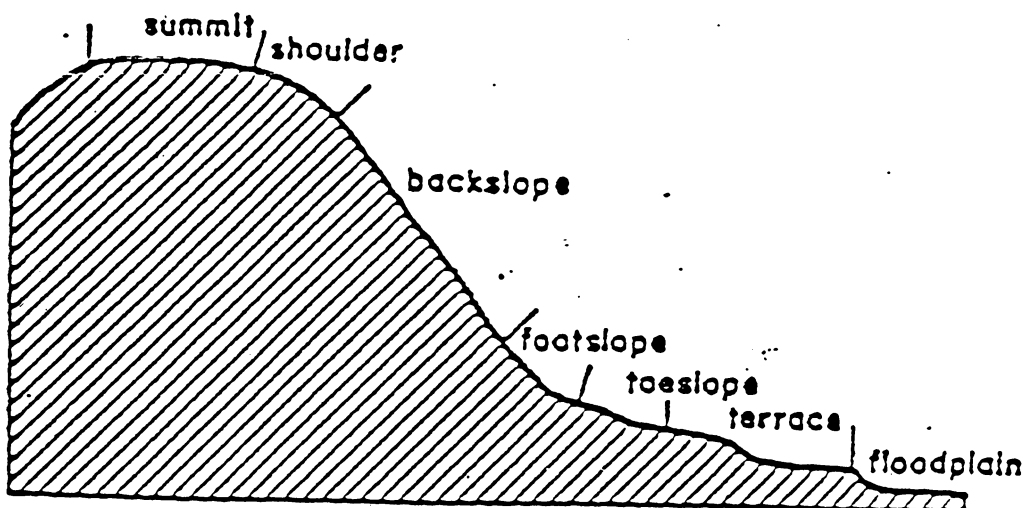
Appendix III. Mean topographic parameters for mixed oak stands burned by wildfire in May, 1986, in Shenandoah National Park, Virginia.

Stand	Fire Intensity	Elevation (m)	Aspect Azimuth	Slope Percent	Slope Position
1	High	700	171	42	backslope
2	High	610	153	54	backslope
3	High	520	144	48	backslope
4	High	560	153	46	backslope
5	Low	560	317	39	footslope
6	Low	610	118	36	backslope
7	Low	580	106	45	backslope
8	Low	580	133	43	backslope
9	Unburned	700	313	46	backslope
10	Unburned	640	117	56	backslope
11	Unburned	670	245	45	footslope
12	Unburned	610	150	52	backslope

Appendix IV. Calculations for implementation of chi-square test for testing goodness-of-fit and homogeneity of logistic regression models for two species using quantities obtainable from the SAS procedure LOGIST (Harrel 1986).

Model 1: Mort = f(DBH, char)

Model 2: Mort = f(DBH, char, d_1, d_1*DBH, d_1*char)

where:

Mort = 0 if tree survived fire

1 if tree killed by fire

DBH = tree diameter at breast height (cm)

char = height of stem-bark char (m)

d_1 = 1 if species = chestnut oak

0 otherwise

Model 1 and Model 2 were fitted to dataset containing 999 observations, of which 734 are chestnut oaks, and 265 are scarlet and northern red oaks.

Results from output of Proc LOGIST (Harrel 1986)

Model 1 :

- 2 log likelihood for model containing intercept only = 1363.2 Convergence in 7 iterations.

- 2 log likelihood = 791.17

Likelihood Ratio Statistic (LRS) = 1363.2 - 791.17 = 572.03

LRS distributes as a chi-square: LRS = 572.03, 2 df: p < 0.0001

R = 0.646 (proportion of log likelihood accounted for)

C = 0.900 (proportion of concordant pairs) (Harrel 1986)

Interpretation: Model 1 is a significant predictor of the

probability of fire-induced mortality for upland oaks.

Model 2 :

- 2 log likelihood for model containing intercept only = 1363.2

Convergence in 7 iterations.

                                - 2 log likelihood =   674.44

LRS = 1363.2 - 674.44 = 688.75, 6 df : p < 0.0001

R = 0.706

C = 0.928

Both models are highly significant, and predicted probabilities are

generally in aggreement with observed outcomes (survival or mortality),

as indicated by the C statistic. Note however, that Model 2 has higher

values for R and C, indicating possibility that Model 2 fits the data

better than Model 1. This suggests a test of the hypothesis that the

two models are homogeneous, that is, that the resistance of chestnut

oaks and scarlet and northern red oaks to fire-induced mortality is

similar. This test is analogous to the use of dummy variables to

perform an F-test to test for coincidence of two regression lines in

simple linear regression using Ordinary Least Squares.

<u>Test performed as follows</u>:

1. Take the difference in model – 2 log likelihood of Model 1 and and Model 2.

2. This gives the LRS for comparing two regression models. Both models must be fit to the same dataset or the test is not valid.

3. The LRS is a chi-square statistic, and the difference in the number of parameters estimated in each model is the degrees of freedom for the test.

LRS = Model 1 (– 2 log likelihood) – Model 2 (– 2 log likelihood)

LRS = 791.17 – 674.44 = 116.73, with 3 df: p < 0.001

<u>Conclusion</u>:

There is strong evidence that Model 1 is different than Model 2, that is, the probability of fire-induced mortality as a function of DBH and height of stem-bark char differs between chestnut oak and scarlet and northern red oaks.

In order to obtain fit statistics (R, C) for the model for each species, it is best to split the data into a chestnut oak dataset and a red oak dataset, and fit Model 1 to each dataset separately. A further test indicated that models for scarlet oak and northern red oak did not differ, so a single model for prediction of the probability of fire-induced mortality for the red oak group was reported.

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