

Utah State University

DigitalCommons@USU

---

The Bark Beetles, Fuels, and Fire Bibliography

Quinney Natural Resources Research Library,  
S.J. and Jessie E.

---

1981

## Fire in Ecosystem Distribution and Structure : Western Forests and Scrublands

Bruce M. Kilgore

Follow this and additional works at: <https://digitalcommons.usu.edu/barkbeetles>



Part of the [Ecology and Evolutionary Biology Commons](#), [Entomology Commons](#), [Forest Biology Commons](#), [Forest Management Commons](#), and the [Wood Science and Pulp, Paper Technology Commons](#)

---

### Recommended Citation

Kilgore, B. (1981). Fire in ecosystem distribution and structure : western forests and scrublands. In: HA Mooney, TM Bonnicksen, and NL Christensen (tech.cord) Proceedings of the Conference: Fire Regimes and Ecosystem Properties, pp. 58-89. USDA Forest Service, General Technical Report WO-GTR-26.

This Contribution to Book is brought to you for free and open access by the Quinney Natural Resources Research Library, S.J. and Jessie E. at DigitalCommons@USU. It has been accepted for inclusion in The Bark Beetles, Fuels, and Fire Bibliography by an authorized administrator of DigitalCommons@USU. For more information, please contact [digitalcommons@usu.edu](mailto:digitalcommons@usu.edu).



## FIRE IN ECOSYSTEM DISTRIBUTION AND STRUCTURE: WESTERN FORESTS AND SCRUBLANDS

Bruce M. Kilgore

Associate Regional Director, Resource Management and Planning,  
Western Region, National Park Service, San Francisco, Calif.

### ABSTRACT

Fire plays an important role in determining structure of forests and scrublands throughout the West. Distribution and structure of vegetation depends upon topography, climatic regime, and fire regime. Six fire regimes are defined based on fire frequency and intensity, varying from frequent, low-intensity surface fires to very long return interval, stand replacement fires. In certain western forests and scrublands fire suppression for the past 50 to 100 years has led to longer intervals between fires, increases in surface and crown fuels, changes in forest structure, and sequential impacts on fire intensity, postfire age structure, species composition, fuel accumulation, and both horizontal and vertical pattern. Better understanding of fire regimes is basic to our management of western ecosystems.

KEYWORDS: western forests, fire regimes, fire frequency, fire intensity, scrublands.

### INTRODUCTION

Fire plays a major role in the function of most of the forests and scrubland ecosystems of the western United States. In many ecosystems, fire controls the age structure and species composition of the vegetation and creates a mosaic of vegetation types on the landscape (Heinselman 1978). Fire acts with different frequencies and intensities, depending upon the vegetation and topography involved, as well as the climatic regimes which determine the coincidence of ignitions with given burning conditions. Thus vegetation composition and structure depend on climate, fire frequency, and intensity, while fire frequency and intensity in turn depend on vegetation structure, topography, and climatic regimes.

Because of almost annual coincidence of ignitions with suitable burning conditions, western forests, such as those found in the Sierra Nevada, have frequent fires. Because such fires are frequent, they are of low intensity. By contrast even though ignitions are equally as frequent in certain Rocky Mountain forests, they do not coincide as frequently with dry fuel conditions. Thus these fires have a longer recurrence interval and are either more intense surface fires that kill all nonresistant species or are crown fires with shorter or longer return intervals.

Forest and scrublands in the West can be grouped into the following four broad formations adapted from Oosting's (1956) climax formations of North America: (1) subalpine forest, (2) montane forest, (3) woodland, and (4) scrubland. These formations--based on climatic regime and topography--range from vegetation found in moist, higher elevation, north exposures to that found in drier, lower elevation, south exposures. These formations occur in altitudinal zones on the principal mountain ranges in the West: the Rocky Mountain, Cascades, Sierra Nevada, and Coast Ranges as well as on Great Basin and Intermountain ranges. Distribution of plant communities or habitat types in the western United States has been covered broadly by Oosting (1956) and in more regional detail by Daubenmire and Daubenmire (1968), Franklin and Dyrness (1973), Munz and Keck (1959), and Pfister and others (1977).

#### Impact of Fire Frequency on Species Composition

Frequency of fire is critical in selecting those species that will continue as part of the vegetation of a given area. A species cannot survive if fire is introduced too often or too early or deferred too long in its life cycle (Hendrickson 1972). For example, with nonsprouting species, survival in a given area may be threatened by fires which occur before there has been time for a seed pool to accumulate or after the plant's longevity has been exceeded and the store of seed is lost (Gill 1979). The significance of fire frequency in determining the species composition aspect of forest structure through time is illustrated when fire burns often enough to prevent a fire-dependent Douglas-fir (Pseudotsuga menziesii) or lodgepole pine (Pinus contorta) forest from changing to a nonfire-dependent, shade tolerant fir, cedar-hemlock, or spruce-fir forest.

#### Impact of Fire Frequency/Intensity on Vegetation Mosaic

The horizontal pattern or mosaic of vegetation in a given area is made up of contrasting age classes, successional stages, and vegetation types created by recurring fire over long- or short-term rotation periods (Heinselman 1978). The scale of the vegetative mosaic is influenced by frequency and intensity of fire as well as the topographic base on which fire occurs. In relatively flat or gently rolling country where long-term intense crown fires or stand-replacing surface fires are characteristic, patches of the mosaic may cover thousands of acres. In steep, broken terrain, such as the Sierra Nevada, where low-intensity fires burn frequently, the patches may be less than 0.25 acres (0.1 ha) in size (Bonnicksen 1975). Whatever the size of patches or aggregations involved, the overall mosaic changes little, as long as the frequency and intensity of fire characteristic of that vegetation and topography continues; the individual aggregations are periodically rearranged by fire and succession, like pieces of a kaleidoscope (Bonnicksen and Stone 1978, Heinselman 1978).

#### Fire Regimes

Fire is important in so many ecosystems that it is becoming less meaningful to refer to fire-dependent vs. fire-independent systems. Instead we need to speak of systems with varying "fire regimes" made up of such factors as fire frequency and intensity (Heinselman 1978, Sando 1978), season (Gill 1973), pattern (Keeley 1979), and depth of burn (Methven 1978). In order to emphasize the wide range of fire characteristics found in the West and to make comparisons between them, I have organized what is known about the impact of fire frequency and intensity on vegetative structure around a modification of the fire regimes described by Heinselman (1978):

- (1) frequent, low-intensity surface fires (1- to 25-year return interval);
- (2) infrequent, low-intensity surface fires (more than 25-year return interval);

- (3) infrequent, high-intensity surface fires (more than 25-year return interval);
- (4) short return interval, stand-replacement fires (25- to 100-year return interval);
- (5) variable regime: frequent, low-intensity surface fires and long return interval, stand-replacement fires (100- to 300-year return intervals); and
- (6) very long return interval, stand replacement fires (more than 300-year return intervals).

Certain major forest types are covered only briefly because little is known at present about the relationship between forest structure and fire frequency and intensity in those types. More thorough treatment has been given here to certain other forest types--particularly the sequoia-mixed conifer forest--because information is available, even though the type covers a more restricted geographic range.

An intense fire can either burn through the tops of crowns of trees, thus killing trees as a "crown fire," or it can simply girdle or heat-kill fire-sensitive trees by intense burning at ground or surface level. In either case, the stand of trees or segment of the forest is partially or totally killed. However, only in the most short-term sense can this be called simply a "stand-destroying" fire. Because at the same time it is killing the existing group of trees, the fire is providing a set of conditions--sometimes involving the seedbed, opening of closed cones, or stimulation of sprouting species--which lead to replacement of the old forest or scrubland by a new vegetation. In this paper, therefore, a high-intensity surface fire or crown fire which kills most of the existing vegetation is also referred to as a "stand-replacement" fire as used by Gabriel (1976), Habeck (1976), Antos (1977), Sneek (1977), Van Wagner (1978), and Arno (1980).

Fire frequencies are difficult to compare in a meaningful way between different studies of various forest types and geographic regions unless similar-sized units are being compared. Kilgore and Taylor (1979) used "frequency" to mean "interval between fires on the same piece of ground...calculated from records found on a single tree or a cluster of trees growing fairly close together." C. E. Van Wagner (personal communication) feels there is only one basic parameter, namely, "the average number of years between fires at a point." He notes, however, that "fire cycle" is the same concept if it is defined as the number of years to burn an area equal to the whole area in question (see also Methvan 1978). This is similar to Heinselman's (1973) "natural fire rotation," and "fire return interval" (Tande 1977) embodies the same idea. "Fire frequency" or "fire cycle" can be contrasted with "fire incidence" which has been defined as "the interval between fires which burned someplace in a particular-sized unit of the forest, such as a drainage, but not necessarily involving the same point" (Kilgore and Taylor 1979). The importance of this distinction is that "frequency" reflects fuel, climatic, and ignition factors inherent in the ecosystem or forest type, while "fire incidence" also is affected by the size of the area being observed. Thus in a study of a giant sequoia (*Sequoiadendron giganteum*)--mixed conifer forest, the patchy fire regime was recorded every 17-23 years on individual trees, every 9-17 years in 1- to 2-acre (0.4- to 0.8-ha) clusters, every 5-9 years somewhere in 40-acre (16-ha) study sites, and every 2 years somewhere in 2,500-acre (1 000-ha) drainage (Kilgore and Taylor 1979). Obviously, the larger the unit size, the greater the numbers of fires and the shorter the intervals between them without any basic change in ecosystem function. By contrast, where fires are generally extensive, as in chaparral, recurrence intervals would often be similar for both single-acre and 100-acre or larger units. Hence, fire incidence alone is an incomplete description of the fire regime of a forest and a far less useful measure than either "fire frequency" or "fire cycle."

In discussing fire's role in the evolution of plant attributes, Keeley (1979) has pointed out that for that purpose, a "natural" fire frequency should include only lightning ignitions. For other purposes, such a distinction may be unnecessary. In any case, with the exception of sequoia-mixed conifer forest (Kilgore and Taylor 1979), there is little quantitative data upon which to base judgment of what proportion of presettlement ignitions were started by lightning or aboriginal sources. Hence, in this paper, I will not attempt to separate these two sources of ignition, although aboriginal burning was clearly significant in certain ponderosa pine and giant sequoia forests, and perhaps in other vegetation types as well.

Most studies of fire history and fire ecology of certain forest types specify intensity (if at all) in subjective and nebulous terminology such as "cool/light" or "hot/severe." Use of categories such as (1) low-intensity surface fires, (2) medium to high-intensity surface fires (torching), and (3) high-intensity crown fires is preferable, provided a range of fire intensity is noted for each class. Sando (1978) has suggested use of intensities greater than 1200 Btu/ft/sec to separate high-intensity fires from low- to moderate-intensity fires. Ecologists often appear to lack a clear understanding of fire behavior and its importance to the sound interpretations of fire effects (Van Wagner and Methven 1978). So it is not yet possible to relate quantitative intensity measurements, such as heat production per second per unit length of fire front (Byram 1959, Van Wagner 1965, Albini 1976), to specific changes in the ecosystem. Part of the past disagreements about the role of fire in a given forest or habitat type can probably be attributed to this lack of data refinement and documentation (Kilgore 1973b). Future field and laboratory studies must strive to resolve this problem by correlating fire effects with fuel, fire behavior, and environmental parameters.

#### FREQUENT, LOW-INTENSITY SURFACE FIRES

Frequent, low-intensity surface fires were characteristic of presettlement ponderosa pine (*Pinus ponderosa*) and giant sequoia-mixed conifer forests (Weaver 1951, Cooper 1960, Biswell 1967, Kilgore and Taylor 1979). Even certain Douglas-fir and western larch (*Larix occidentalis*) forests in the Rocky Mountains were kept open for centuries by such frequent natural burning (Arno 1980) along with insect epidemics and other mortality factors.

In his summary of the effects of fire on western forests, Weaver (1974) quoted a number of early explorers who, in describing the forest structure of Sierra Nevada montane forests, referred to "the inviting openness of the Sierra woods...trees of all species standing more or less apart in groves, or in small, irregular groups...with openings that have a smooth, parklike surface, strewn with brown needles and burs" (Muir 1894). One even spoke of being able to ride their horses "at a free gallup through the forest" which seemed like a vast covered hall where "in stately groups, stand tall shafts of pine" (King 1871). Although these statements reflect what we think sizable portions of these patchy forests were like, we should be cautious in extending these specific comments to imply that all areas of these forests were this open at all times.

One of these same early-day observers (Muir 1901) described the intensity of a fire he saw in 1875 in what is now Sequoia National Park, Calif., which resulted from this open type of structure and contrasted it with the intensity found in the dense chaparral scrubland and in certain dense forests of the Rocky Mountains and the Cascades:

The fire came racing up the steep, chaparral-covered slopes... in a broad cataract of flames.... But as soon as the deep forest was reached, the ungovernable flood became calm, like a torrent entering a lake, creeping and spreading beneath the trees.... There was no danger of being chased and hemmed in, for in the main forest

belt of the Sierra, even when swift winds are blowing, fires seldom or never sweep over the trees in broad, all-embracing sheets as they do in the dense Rocky Mountain woods and in those of the Cascade Mountains of Oregon and Washington. Here they creep from tree to tree ...allowing close observation....

The presettlement structure of pine and mixed conifer forests in the Sierra Nevada and intensity of fire which is possible because of that structure was described by Show and Kotok in 1924:

The virgin forest is uneven-aged, or at best even-aged by small groups, and is patchy and broken; hence it is fairly immune from extensive, devastating crown fires. Extensive crown fires, though common in the forests of the western white pine region, are almost unknown in the California pine region. Local crown fires may extend over a few hundred acres, but the stands in general are so uneven-aged and broken and have such a varied cover type that a continuous crown fire is practically impossible.

#### Ponderosa Pine Forests

Average fire frequencies in presettlement ponderosa pine forest varied from 6 to 19 years in different parts of its range (Wright 1978), with 5 to 12 years found in Arizona and New Mexico (Weaver 1951), 8 to 10 years being characteristic of ponderosa pine in California (Show and Kotok 1924, Wagner 1961, McBride and Laven 1976); 8 to 18 years in various parts of Oregon and Washington (Keen 1940; Weaver 1955, 1959; Soeriaatmadja 1966; Hall 1976); and 6 to 10 years reported from the Bitterroot National Forest in Idaho and Montana (Arno 1980). In forests containing a mixture of ponderosa pine and white fir (Abies concolor) in southern Oregon, McNeil and Zobel (1979) found mean intervals between fires that varied from 9 to 42 years at given sites.

Early studies of stand development in ponderosa pine by Weaver (1943, 1967) and Cooper (1960, 1961) help interpret structural changes with fire exclusion. In Arizona ponderosa pine forests, Cooper (1960) noted that under presettlement conditions, low-intensity surface fires, set by lightning or Indians, burned through the forest at regular intervals of 3 to 10 years. These fires served as thinning agents and played an important role in the cyclical process of stand development.

In describing a simplified form of this process, Weaver (1967) stated: "Periodic burning causes development of uneven-aged stands, comprised of even-aged groups of trees of various age classes." The system operated because low-intensity fire killed more small pines under canopies of larger trees than in openings. It did so because a surface fire would burn with greater intensity through heavy accumulations of flammable needles, cones, and bark scales which build up under larger trees than in the lighter and often less continuous fuels in the openings. But at various sites throughout the forest, even-aged groups of trees were killed by insects, disease, lightning, or windthrow. These groups of dead trees were gradually reduced to ashes in subsequent fires, leaving openings in the forest canopy within which pine seeds could germinate and young pines survive. (Another possibility would be that fire by chance would miss certain sites and allow enough fuel buildup that overstory trees would be killed in subsequent fires, leading to an opening.) Within such openings, the small accumulation of needles from these newly established young pines would not support a surface fire, although dry grasses and forbs in the openings would. Hence, until the pines were large enough to build up fuels beneath their own canopy, fires were not intense enough to kill them; and by the time they did contribute such heavier fuels, many of them were large enough to survive the low-intensity surface fires.

Cooper (1961) found that ponderosa pine forests in northern Arizona have a relatively simple species composition, yet a complex spatial structure. Several types of pattern or "scales of pattern" were identified, ranging from (1) a large-scale pattern of differences in density, growth, and species composition induced by local variations in topography and soils, through (2) a patchy mosaic pattern of even-aged groups averaging about 0.2 acre (0.1 ha) in size and maintained largely by fire, to (3) variations in stand density within a single even-aged group, primarily due to chance factors in early stand development. Cooper (1961) concluded that, "The clearly identifiable pattern in ponderosa pine forest is the result of the intolerance of the species to shade, the harsh environment which restricts the number of species present, and periodic natural fires...."

Since settlement of much of the West between 1850 and 1900 and the developing policy of fire exclusion, major changes in fire frequency and intensity have taken place. Cooper (1960, 1961) reported that in the absence of frequent, low-intensity fires, dense sapling thickets of young pine have covered the region. This has caused major changes in vegetation structure from the all-aged forest made up of even-aged groups with no understory saplings found under presettlement conditions. By comparing an area typical of presettlement conditions (where natural fires have continued over the years) with two areas typical of present conditions (fire exclusion), Cooper (1960) found that the major decrease in fire frequency had resulted in (1) diameter distributions in young even-aged stands which are skewed toward smaller trees; (2) stagnation of growth in the young pine in stands where density exceeds 6,000 stems per acre (14 800 stems per ha); (3) a less distinct patchwork pattern of reproduction than found in burned forests; (4) many more trees per acre; and (5) more saplings beneath mature trees. In turn, this change in structure as a result of long continued fire exclusion has increased the likelihood of high-intensity wildfire "by permitting excessive buildup of dry fuel on the forest floor, by leaving the tree crown level near the ground where ignition of live needles is more likely, and by permitting the formation of dense uninterrupted sapling stands over wide areas" (Cooper 1960).

Weaver (1947) made a similar comparison between two 40-year-old ponderosa pine stands, one burned and one unburned on the Colville Indian Reservation in Washington. He found the burned area was composed of 1,100 young trees per acre (2 717/ha), on the area stocked) and the trees averaged 7.4 inches (18.8 cm) diameter and 32.2 ft (9.8 m) in height. The unburned area had 14,800 young trees per acre (36 556/ha) averaging 1.7 inches (4.3 cm) diameter and 12.3 ft (3.7 m) in height.

#### Giant Sequoia-Mixed Conifer Forests

Similar changes were taking place in the sequoia-mixed conifer forests. In presettlement times, low- to moderate-intensity surface fires burned through giant sequoia-mixed conifer forests every 9 to 16 years, killing many young white fir while they were still part of the understory level of vegetation and fuels (Kilgore and Taylor 1979). Intense stand-replacing fires were apparently not part of the history of sequoias in Kings Canyon National Park, Calif., during the past 400 to 2,000 years.

With the elimination of aboriginal burning in the late 1800's and ever more effective fire suppression in the early 1900's, an increasingly dense understory of shade tolerant species such as white fir and incense-cedar (*Calocedrus decurrens*) developed beneath the canopy of Sierra Nevada mixed conifer forests (Kilgore and Taylor 1979, Parsons and DeBenedetti 1979). This altered considerably the nearly crown-fireproof structure of the presettlement forest (Kilgore and Sando 1975). Even in 1924, Show and Kotok found that structural changes had occurred which affected the intensity of fire likely in these forests:

Existing second-growth stands are typically even-aged and fully stocked, have a continuous, unbroken canopy, and are consequently susceptible to the most destructive type of forest fire. To those familiar with the occurrence of crown fires in the dense, even-aged forest of western white pine and Douglas-fir in the Pacific Northwest, a field examination of the second-growth western yellow pine forests of California would be immediately convincing that these stands also represent an exceedingly high inherent hazard and that this hazard differs from that of the virgin forest.

Since that time, increasing numbers of studies have been undertaken aimed at understanding the role of fire in montane vegetation of the Sierra Nevada and particularly the giant sequoia-mixed conifer forest (Biswell 1961, 1967; Hartesveldt 1964; Hartesveldt and Harvey 1967; van Wagtendonk 1972, 1974; Agee 1973; Kilgore 1973a, 1973b; Weaver 1974; Bonnicksen 1975; Kilgore and Sando 1975; Parsons 1976, 1977, 1978; Agee and others 1978; Bonnicksen and Stone 1978; Harvey and others 1979; Kilgore and Taylor 1979). Although few of these studies have dealt in a comprehensive way with the impact of fire on the various aspects of forest structure, a number of them yield valuable interpretations of the impact of changes in frequency and intensity of fire on species composition, age structure, horizontal arrangement, and vertical stratification.

The recent studies by Bonnicksen (1975) and Bonnicksen and Stone (1978) are the most specific efforts to understand the relationship between fire frequency and detailed forest structure in a giant sequoia-mixed conifer ecosystem. Bonnicksen and Stone (1978) have sought to (1) characterize quantitatively the structural properties of the presettlement giant sequoia-mixed conifer forest using both early descriptions and an analysis of existing vegetation; (2) characterize the dynamic properties of the forest and the processes which brought about changes in forest structure prior to the influence of European man; (3) document quantitatively the structural changes which have occurred as a result of the fire exclusion policy; and (4) develop quantitative guidelines for reconstructing alternative approximations of the presettlement forest structure.

Like Cooper's (1960) and Weaver's (1943) perception of ponderosa pine forests, and Stone (1978) perceive the giant sequoia-mixed conifer forest as a "mosaic of aggregations." And as in the case of ponderosa pine forests, disturbances such as fire, lightning, windthrow, and insects and disease infestations occur in small areas and produce openings or "gaps" in the forest. This occurs in part because fire often burns in a highly variable pattern of intensities in these forests, as well as in ponderosa pine and most other Sierra Nevada forests. It may burn with high intensity in one site, with low intensity nearby, and not at all in another site, with great variations in temperature and energy output over short horizontal and vertical distances, often related to localized accumulations of fuels. Surface temperatures, as an example, can vary from 204° to 648°C or more with no uniformity of distribution in a given burn (Kilgore 1973a, 1973b).

Once a gap has been created, species which can successfully seed into the opening and survive do so, forming an even-aged patch. If there is no further disturbance, shade tolerant species will begin to grow in the understory, producing several layers, each relatively even-aged. "The result of the gap formation process is a forest that can best be described as a mosaic of groups of trees, or aggregations, which can differ from one another in their age, the number of their vertical layers or tiers, and their species composition" (Bonnicksen and Stone 1978). A primary concern of this paper is the impact of fire frequency and intensity on these three variables and the role of fire in maintaining the distinction between adjacent aggregations.



The main structural changes in the giant sequoia-mixed conifer forests in the absence of surface fires have been (1) a large increase in the younger age classes of shade tolerant white fir in all parts of the forest (Kilgore 1973a) with a corresponding decrease in young, fire-dependent ponderosa pine, black oak (Quercus kelloggii), and giant sequoia (Parsons and DeBenedetti 1979); (2) a smaller increase in young sugar pine (Pinus lambertiana); (3) the survival of one or more vertical layers or tiers of white fir beneath (or perhaps in aggregations adjacent to) the overstory canopy of sequoia and pine; (4) a denser forest generally, particularly of young white fir (Parsons and DeBenedetti 1979), with 1,500 stems of various mixed-conifer species per acre found in some areas (Kilgore 1973a); and (5) a blending of what had been discrete patchy units into a more uniform forest, with more uniform burning intensities, "... gradually destroying the identity of the individual aggregations of trees and shrubs that were so readily seen in presettlement times" (Bonnicksen and Stone 1978).

The most conspicuous change found by Bonnicksen and Stone (1978) in overstory structure of a giant sequoia-mixed conifer forest was the general increase in area dominated by aggregations of pole-sized and mature conifers and a decrease in area occupied by aggregations of sapling- and seedling-sized trees, shrubs, grasses, and forbs. In the 87 years between 1890 and 1977, they found the acreage covered by these older age classes of white fir had more than doubled and there was a corresponding decrease in younger age classes as well as a decline in aggregations of black oak and shrubs. The decline of aggregations dominated by small fir, black oak, and shrubs can be considered a logical result of small fir growing larger combined with fewer gaps in the forest and hence fewer opportunities for young trees and shade intolerant shrubs to become established and survive.

Changes in understory vegetation have also occurred. In presettlement times, according to Bonnicksen and Stone (1978), nearly half of the understory vegetation in the watershed was made up of seedlings, shrubs, and forbs or there was bare soil and rock with no understory. When this was combined with mature and pole-sized aggregations lacking any understory and with aggregations of low stature vegetation, nearly 75 percent of the watershed apparently had the fairly open appearance described in historic accounts. In the absence of fire, however, there has been more than a 300 percent increase in aggregations of pole-sized conifers with an understory of saplings or seedlings.

These changes in structure of both top tier and understory forest aggregations also have an impact on the quantity and arrangement of fuels that accumulate and, as such, on the potential intensity of fire. Without frequent low-intensity surface fires, shade tolerant white fir, with its low-hanging branches from the various age classes of seedling, sapling, and pole-sized trees, forms an almost continuous fuel supply from ground level to their tops 10 to 100 ft (3 to 30 m) or more in height (Kilgore and Sando 1975). The larger fir, in turn, frequently reach the lower crowns of mature trees. This stairstep structure of age classes in the forest forms a ladder-like arrangement of fuels with a high potential for fire to pass from surface fuels into giant sequoia crowns.

Kilgore and Sando (1975) concluded that low to moderate-intensity prescribed burning in a particular giant sequoia-mixed conifer forest modified the structure of the lower layers of the forest in such a way that the potential for high-intensity surface fires was substantially reduced. This was accomplished by reducing surface fuels and crown fuels in the lower 55 ft (16.8 m) of the forest and by increasing the mean height of the base of the crown from 3 to 16 ft (0.9 to 4.9 m).

Localized, high-intensity surface fires, however, were probably also both a natural part of presettlement forests (Bonnicksen 1975) and necessary for survival of giant sequoia seedlings (Harvey and others 1979). As such, fire intensity can have

considerable impact on postfire species composition. Following another prescribed burn, Kilgore and Biswell (1971) found the greatest number of giant sequoia seedlings on plots with the most intense burning conditions. They believed that the intensity of fire created a rising convection column of heat which not only dried out and killed needles more than 100 ft (30 m) up in several giant sequoia trees, but also caused drying out and opening of giant sequoia cones and thus contributed to very heavy seed fall in the area of the hottest burn.

Whereas the lightweight giant sequoia seeds germinate readily in the soft, friable, ashy soil which is provided after most fires, Harvey and others (1979) found the survival rate of seedlings in soils subjected to high-intensity burns was 10 times that of seedlings on other substrates. Because of its great longevity, however, few surviving giant sequoia seedlings are required in each fire cycle to perpetuate the natural structure of the ecosystem.

Bonnicksen and Stone (1978) believed that prescribed burning alone would not restore presettlement conditions to the giant sequoia-mixed conifer forest. They contended that because fuels have accumulated to such an extent, the numbers of sites subjected to high-intensity fire will be larger than would be expected under presettlement conditions. Hence, understory saplings would be reduced in all aggregations, even those in openings needed to start new age classes of white fir. In addition, they felt that the high intensities reached by many prescribed fires will lead to relatively uniform site conditions and to reproduction that may also be more uniformly dispersed than might be desired.

Compensating for the present structural changes in the forest will now be particularly difficult because the larger white fir (with thick bark) are much less likely to be killed by moderate fires, and any higher intensity fires may also kill both sugar pine and giant sequoia (Kilgore 1973b). Some fairly sophisticated and expensive forest manipulation techniques--perhaps including cutting some sizable fir and protecting certain aggregations of small fir--may need to be used to restore the forest to what we believe were more natural environmental conditions.

#### INFREQUENT, LOW-INTENSITY SURFACE FIRES

##### Sierra Nevada Subalpine Forests

Fires in subalpine forests of whitebark pine (*Pinus albicaulis*), red fir (*Abies magnifica*), and lodgepole pine in the Sierra Nevada behave differently from fires burning in lower elevations or in areas with different topography, forest composition, fuels, and rockiness. They commonly spread slowly or not at all, they rarely burn the crowns and kill overstory trees, and they are generally less intense than fires in montane forests of the Rocky Mountains or Cascades (Kilgore and Briggs 1972). Such high-elevation areas often have many natural firebreaks such as sparsely vegetated ridges, barren rocky areas, and streams and draws with relatively fire-resistant riparian vegetation.

In studies of a prescribed burn in 1968 in a forest of red fir and lodgepole pine, many mature lodgepole pine were readily killed by the fire, while germination of lodgepole seedlings was stimulated (Kilgore 1971). Few older red fir were affected by the fire, but many fir saplings and seedlings were killed, resulting in decreased coverage by fir thickets. The intensity of the fire was considerably lower than fire in the drier, heavier fuel conditions which are presently found in lower-elevation, postsettlement mixed conifer forest. There was little evidence of fire entering the crowns; whereas the crowns of some individual trees with sufficient fuels accumulated at their base did ignite, the fire did not spread to adjacent trees.

During the course of its program of allowing natural fires to burn, the National Park Service has monitored a number of fires in these subalpine zones and found that most lightning fires are less than an acre in size and burn only a few days. Several fires, however, burned considerable acreage over several months' time, but none burned intensely over large areas or became a control problem (van Wagtenonk 1977; Parsons 1977, 1979).

We are not sure just how frequent these fires were in presettlement times. With the shorter growing season in these higher elevations, it is estimated that fuel accumulation would be small enough each year that fire frequencies and intensities would be much less than that found in lower elevation montane forests (Kilgore 1971). Despite their relatively low intensity and infrequent occurrence, fires in these subalpine types played important roles. Because of long life spans and slow growth, however, they need not occur often to be a controlling factor.

#### Rocky Mountain Subalpine Forests

Whereas high-intensity fires have clearly been part of the spectrum of fires found in lower subalpine forests in the Rocky Mountains, Arno (1976) concluded that because of moist, sparse fuels in the upper subalpine forest, fires are much less frequent and intense. Sneek (1977) quoted a report from Glacier National Park, Mont. indicating that fires at higher elevations (over 5,000 ft or 2 000 m) are rarely extensive and are infrequent, perhaps every 300 to 600 years. Tande (1977) reported a mean fire return interval of 74 years for 124-acre (50-ha) blocks of subalpine forest in Jasper National Park, Alberta.

Gabriel (1976) found that fires in the subalpine fir zone occurred in a random pattern, with scars on adjacent trees often from different years. This apparently resulted from a high incidence of lightning fires with a low volume of fuel and open stands. Romme (1977) pointed out that a reduced frequency of fire in high-elevation spruce-fir forests of the Medicine Bow Mountains of Wyoming may result from the long-lying snowdrifts and frequent summer rain showers found in these Rocky Mountain subalpine forests.

#### INFREQUENT, HIGH-INTENSITY SURFACE FIRES

##### Redwood Forests

The natural fires in coast redwoods (Sequoia sempervirens) were probably infrequent, intense surface fires (Heinselman 1978). Few studies, however, have been done on fire frequency in this type. In work with more than 100 stumps on a 30-acre (12-ha) site in Humboldt County, Calif., Fritz (1932) concluded that there had been at least 45 severe fires in that locality during the past 1,100 years or about four fires per century.

McBride and Jacobs (1978) found intervals of 21.7 and 27.3 years between fires on 42 stumps found in two sites near Muir Woods National Monument in Marin County, Calif., and "intervals as short as 4 or 5 years on some stumps were not uncommon." In the absence of fire since the early 19th century, a buildup of both surface and ladder-like vertical fuels has occurred, which results in: (1) low-to-medium fire hazard in surface fuels, with some higher hazards in young redwood and Douglas-fir/hardwood types; and (2) fairly high fire hazard in ladder fuels, particularly in the Douglas-fir/brush and hardwood types and the redwood/hardwood types (McBride and Jacobs 1978).

Stone and others (1972) believe redwoods are seral and strongly dependent on fire and flooding for their perpetuation. Franklin and Dyrness (1973) concur in this, based on massive fire scars on almost all large redwoods in one area. Others feel the species is climax because of its high shade tolerance (Waring, personal communication, as noted in Franklin and Dyrness 1973). In his work in the northern range of the redwoods, Steve Veirs (personal communication) at Redwood National Park has identified

patterns of establishment of Douglas-fir (Pseudotsuga menziesii) in redwood stands at 200- to 500-year intervals on mesic sites, presumably related to infrequent, high-intensity surface fires. Inland, on higher, drier sites, redwood dominance is reduced and Douglas-fir age classes suggest a return interval of 50 to 100 years for fires which open the forest canopy to the extent necessary for fir establishment. In its northern range, redwood appears to reproduce at replacement rates (about 1 tree/acre every 50 years) in the absence of fire. With the exception of Douglas-fir, associated trees such as tanoak (Lithocarpus densiflora), hemlock (Tsuga heterophylla), and grand fir (Abies grandis), like redwood, appear to reproduce successfully both with and without ground fires.

#### SHORT RETURN INTERVAL, STAND-REPLACEMENT FIRES

Chaparral and quaking aspen (Populus tremuloides) are two vegetation types which illustrate a short return interval between stand replacement fires. Both forms have all aboveground vegetation killed by periodic, intense burning--either crown fire or high-intensity surface fires, but each survives by sprouting from root crowns or by reseeding. In addition, several types of woodland and scrub vegetation are prevented from invading grasslands by periodic crown fires. Pinyon-juniper and sagebrush scrub illustrate this response.

#### Chaparral

Large sections of chaparral are found in California and Arizona, often occurring as a band of vegetation between grasslands below and forests above. In California alone, it covers 11 million acres (4.5 million ha) (Bentley 1967). Chaparral is best developed or most extensively distributed on the steep slopes and shallow soils of southern California (Menke and Villasenor 1977).

Slope aspect is an important factor in the chaparral environment, in part determining which species make up a local chaparral community and even succession following fire (Hanes 1971). Chamise (Adenostema fasciculatum) is a prominent member of the climax chaparral, often forming almost pure stands in some areas. Other sites are dominated by various species of manzanita (Arctostaphylos spp.), ceanothus (Ceanothus spp.), and scrub oak (Quercus spp.). Species of the latter three genera also make up much of the patches of chaparral that occur within conifer forests (Biswell 1974).

Recent work by Byrne (1978) provides evidence that wildfires were an important part of the California chaparral environment long before European man arrived on the scene. Using data gathered from ancient charcoal fragments deposited in marine sediments between 1400 and 1550 A.D., Byrne's (1978) preliminary conclusions were that prehistoric fires in chaparral and woodland of the Los Padres National Forest area of southern California "occurred less frequently than during the modern period, but those that did occur were of greater intensity and aerial extent. We estimate the recurrence interval of these fires to be anywhere from 20 to 40 years."

Fire has likely been an important factor in chaparral for a long time. The plant species involved have developed two main strategies for surviving burning: many species exhibit vigorous sprouting ability following fire; others have seedcoats that are altered by fire, and thus germination of seeds is favored.

Present frequencies of fire in this type of scrubland are reported to vary from less than 10 years in sections of the south coastal region of California through 20 to 25 years in the north coastal regions to as much as 50 to 100 years in manzanita chaparral growing at 4,000 ft (1 220 m) elevations on northern slopes.<sup>1/</sup> Under fire

---

<sup>1/</sup> Material in preparation for a textbook on fire ecology by Henry A. Wright and Arthur W. Bailey.

suppression conditions today, many areas have not burned for 60 to 100 years. Based on work in Sequoia National Park, Calif., Parsons (1976) concluded that chamise regained much of its prefire structure within 14 years, and that by 35 years, "...it shows all the characteristics of a mature stand, most of the successional subshrub forms having disappeared."

The dramatic impact of interaction between fire frequencies and intensities on the one hand and vegetation structure and reproductive strategies on the other is illustrated by recent work in southern California chaparral. Keeley and Zedler (1978) believe chaparral is adapted to both short and long fire-free intervals, a reflection of the unpredictability of fire in that environment. They suggest a model which includes both (1) a short fire cycle which favors sprouting shrubs over those reproducing entirely from seed; and (2) a long fire cycle in which "sprouters" and "seeders" coexist. In the short cycle (25 years), there are fewer dead shrubs before the fire, more potential resprouts, less intense fires, lower fire-caused mortality of sprouting shrubs, and smaller openings for seedlings. In the long cycle (100 years), however, there are more dead shrubs before the fire, fewer potential resprouts, higher intensity fires, more fire-caused mortality of sprouting shrubs, and larger openings for seedlings. They predict that with the present fire cycle of every 20 to 30 years, both sprouting and seedling species regenerate but, over long periods of time, sprouting species may gain an advantage.

Keeley (1977) believes that southern California chaparral may have evolved under relatively long fire-free intervals. Similar conclusions were reached by Sauer (1977) who suggested that chaparral of the Santa Monica Mountains evidently "was adapted to a regime of infrequent, large, intense, nonselective fires, which has not been substantially altered by man." Evidence of this was presented by Byrne and others (1977).

In the case of the chaparral adjacent to conifer forest in Sequoia National Park, Parsons (1976) believed that fire prevention and suppression have led to the "formation of an unnatural, highly flammable, nearly uniform, old-aged...brush community." He was concerned that with the known high frequency of lightning fires in the area and increasing recreational use, an unnaturally intense fire would both destroy the vegetation mosaic that currently exists and threaten the nearby groves of giant sequoia.

Parsons (1976) indicated that existing fuel accumulations need to be reduced so that more natural fires could be allowed to burn at frequent intervals. Once fire has restored a natural mosaic of age classes and community types, "these age-class boundaries would then function, along with topographic features, as natural fuel breaks for limiting the size of future fires and thus perpetuating the mosaic" (Parsons 1976). Philpot (1977) shares the same concern that without fire for periods of 50 years or longer, age-class boundaries become less distinct and chances increase for larger, more catastrophic fires. Byrne and others (1977), however, believe that some presettlement fires were large and catastrophic.

#### Quaking Aspen

Studies by Loope and Gruell (1973) indicate that in presettlement times successful regeneration of aspen stands was stimulated by fire. Hendricksen (1972) hypothesized fire frequencies of 50 to 100 years for aspen. Loope (1971) noted that while occasional aspen may live to 200 years in the absence of intense fire, they tend to deteriorate after 80 to 100 years and be replaced by conifers. This is true because aspen is fairly short-lived, is susceptible to many diseases, and cannot survive or reproduce in the shade of competing conifers such as Douglas-fir (Loope 1971, Gruell and Loope 1974). Although the aboveground portion of aspen is easily killed by fire, its root system is rarely damaged; hence aspen clones are perpetuated by fire, and aspen gets a headstart on competing conifers by root sprouting.

Clements (1910) found that aspen sprouts grew more rapidly after an intense fire than after a light fire. Horton and Hopkins (1966) also found that a fire intense enough to kill all aspen stems and remove litter and duff is desirable to produce prolific suckering, whereas low-intensity fire does not achieve this result. Gruell and Loope (1974) concluded that fire suppression during the past 50 to 70 years is a major factor contributing to the recent decline of aspen in Jackson Hole, Wyo.

#### Pinyon-Juniper Woodland

The pinyon-juniper association covers more than 43 million acres (17 million ha) in western North America (Wright and others 1979). Before livestock grazing was involved, fire, drought, and competition with grass jointly limited the distribution of juniper (Juniperus occidentalis).

Buckhardt and Tisdale (1976) report presettlement intervals between fires in western juniper stands in southwest Idaho comparable to the 13- to 18-year interval found by Keen (1940) for ponderosa pine forests. Such frequent fires kept junipers restricted to shallow, rocky soils and rough topography in many parts of the West. With initiation of livestock grazing, reduced competition from grasses has permitted pinyon (Pinus edulis) and juniper to invade adjacent grass and sagebrush communities; with reduced numbers of fires and lower intensity burning (without grass fuels), the juniper invasion has continued unchecked (Martin 1978, Wright and others 1979). In a study of two species of mistletoe that infest pinyon and juniper in Grand Canyon National Park, Ariz., Hreha (1978) found that fire was the most limiting factor on the mistletoe, and that the mistletoes and their hosts appeared to be in equilibrium.

Along the uplands in Mesa Verda National Park, Erdman (1970) found that recurrent fires had maintained a Gambel oak (Quercus gambelii)-serviceberry (Amelanchier utahensis) mountain brush vegetation as a fire climax. But under fire suppression policies, pinyon-juniper forest is gradually replacing the extensive shrub vegetation. Prescribed fire will kill pinyon and juniper less than 4 ft (1.2 m) tall (Wright and others 1979), but taller trees are more difficult to kill. Hence, to prevent invasion of grass and shrublands by pinyon and juniper, reburns are needed at least every 20 to 40 years.

#### Sagebrush Scrub

Sagebrush-grass vegetation covers at least 96 million acres (40 million ha) in the western United States, mostly below the pinyon-juniper zone (Wright and others 1979). Big sagebrush (Artemisia tridentata) is the principal fuel for free-running fires in the broad Great Basin Desert ecosystem which covers much of Nevada, southern Oregon, southern Idaho, and parts of Wyoming, Utah, and Arizona (Humphrey 1974). In presettlement times, minimum fire frequency in this community in northern Yellowstone National Park was 32 to 70 years (Houston 1973), and Wright and others (1979) felt 50 years is likely.

There has been little research on effects of fire in the sagebrush desert, perhaps because fires are rarely extensive in the ecosystem (Humphrey 1974). However, this high, cold desert or sagebrush desert is more subject to burning than the hotter, drier, more southerly Mojave, Sonoran, and Chihuahuan Deserts. This is true in part because the higher elevations and greater moisture permit growth of fire-supporting grasses and forbs beneath and between the sagebrush (Humphrey 1974). In addition, there is considerable fuel from adjacent juniper, pinyon-juniper, or grassland vegetation types that carry fires from these areas into the sagebrush type.

Fires have a major impact on such nonsprouting shrubs as Artemisia tridentata within the sagebrush-grass community, but specific responses vary with seasonal timing, intensity, and frequency of burning (Wright and others 1979). In degraded

big sagebrush/Thurber needlegrass (Stipa thurberiana) rangeland communities, north of Reno, Nev., wildfires killed the sagebrush, stimulated sprouting of two native shrubs, and allowed downy brome (Bromus tectorum), an "alien" annual grass, to dominate the study areas (Young and Evans 1978). The annual grass appears to provide an accumulation of fine fuel that readily ignites and carries fire to the widely spaced sagebrush plants, thus setting in motion a cyclic phenomenon leading toward conversion to an annual grassland.

Artemisia tridentata produces a growth ring that can be counted to determine age of plant or plant community (Ferguson 1960). In a study of shrub age structures on a mule deer winter range in Colorado, Roughton (1972) found an unusually large number of Artemisia individuals in the 55- to 59-year age class. This either suggested that some major mortality factor such as fire had occurred at about the turn of the century and "the population had not yet returned to equilibrium" (Roughton 1972) or there have been 60 years of poor conditions for reproduction since then. Future studies using this shrub age-dating technique may allow determination of age structures for a shrub population and in turn an interpretation of successional status and variations from predicted sequences which may result in part from fire impacts.

From a land management standpoint, to keep sagebrush in check, "prescribed fire can be a useful tool in many big sagebrush...communities if the fires are carefully planned...fires should not be too frequent and should be planned in early spring or after late summer" (Wright and others 1979). Particular species responses to fire must be understood to achieve whatever objective the land manager may have.

VARIABLE REGIME: FREQUENT, LOW-INTENSITY SURFACE FIRES  
AND LONG RETURN INTERVAL, STAND-REPLACING FIRES

High-intensity, stand-replacing crown fires occur in various forest types in the West. In certain types, however, the fire regime varies between relatively frequent, low- to moderate-intensity surface fires and long return interval crown fires, depending upon weather conditions and ignition factors in particular sites. When wind and other weather conditions are favorable for intense burning, the likelihood of an ignition leading to a high-intensity crown fire depends on such forest structural factors as: (1) the spacing of trees; (2) the quantity of crowns per unit area; (3) the amount and arrangement of surface fuels; and (4) how high the crown canopy is above the surface fuel heat sources (Brown 1975). Such intense fires have occurred periodically at more than 100-year intervals in lodgepole pine, western white pine (Pinus monticola), and western larch forests of the Rocky Mountains and in Douglas-fir and lodgepole pine forests in the Cascades (Fahnestock 1977).

Wellner (1970) felt that the age structure of most Northern Rocky Mountain forests supported the conclusion that fire in this region tends to be catastrophic, killing most trees in the forest. Recent studies (based on fire-scar and stand-age data) summarized by Arno (1980), however, indicate that "historic fires occurred with vastly different frequencies and had markedly different effects in this region's diverse forest environments." In some habitats, fires maintained many-aged, open stands of seral trees, while in others, fires destroyed and replaced stands. This was true, Arno (1980) points out, because fires burned at variable intensities "...resulting in a mosaic of stands that differed in tree species composition and age-class structure."

One fairly consistent conclusion by scientists and resource managers working in most areas of the Northern Rocky Mountains seems to be that there are few areas in which shade tolerant, climax forests of spruce (Picea sp.), fir (Abies sp.), hemlock (Tsuga sp.), and redcedar (Thuja plicata) have been allowed to develop. Instead, fire--whether frequent and low intensity or less frequent and of higher intensity--has repeatedly favored the development of stands of intolerant, fire-dependent species

such as lodgepole pine, aspen, western larch, western white pine, ponderosa pine, and Douglas-fir (Wellner 1970, Habeck and Mutch 1973, Weaver 1974, and Arno 1980).

Most fires in the moist, north Pacific coastal forest were probably tiny and put out by rain. The larger fires, however, occurred infrequently, and with few exceptions were stand-replacing fires (Lotan and others 1981). Many fires in the Douglas-fir region of the Cascades were also of this type (Weaver 1974), although there is abundant evidence of low-intensity surface burning wherever ponderosa pine are found.

### Lodgepole Pine Forests

The major vegetation pattern in lodgepole pine is caused by stand-replacement fires, although many uneven-aged lodgepole pine stands result from some lower intensity surface fires (Brown 1975). In a recent review of fire history in the Northern Rockies, Arno (1980) concluded that historic fire was rather frequent in communities dominated by lodgepole pine (25- to 50-year mean intervals in 100- to 200-acre (40- to 80-ha) stands where dry summers occur and 50- to 100-year intervals in such stands with wet summers). Brown (1975) concluded that average intervals between fires "for given localities" in this forest type vary from 60 to 500 years. Lodgepole, however, has a lifespan which varies with site and geographic location from 100 to 300 years or more, and tends to be eliminated from sites having a long interval between fires (Antos 1977).

Arno (1976) found evidence of low-intensity, creeping surface fires in lodgepole pine forests in some of his study areas in the Bitterroot National Forest of Montana. Loope and Gruell (1973) found similar evidence in lodgepole pine forests of north-western Wyoming. Gabriel (1976) also documented low-intensity surface fires at 20- to 40-year intervals in the southern part of his study area in the Bob Marshall Wilderness Area of Montana. He contrasted these with larger, stand-replacing fires that had burned much of the northern half of his study area. Even where these intense fires had burned, there were fire-scarred lodgepole pine and Douglas-fir, suggesting a sporadic pattern of lower intensity burning during relatively high-intensity fires.

The intensity and frequency of fires in lodgepole pine largely determine establishment of seedlings and subsequent development of stand density, age structure, and composition (Brown 1975). Higher intensity fires tend to favor lodgepole pine over such species as Douglas-fir and western larch, depending upon the local winds, topography, and fuels. In some instances, frequent fire may prevent lodgepole from invading aspen where the two grow in mixed stands (Brown 1975), except where intense burns kill aspen roots. The timing of the fire also made a difference. Burns that preceded a good larch seed crop must have favored larch. It is harder to predict the species composition of a lodgepole pine-mixed conifer forest after a low-intensity fire (Brown 1975). In areas with low- to moderate-intensity fires, Ayres (1900) found the composition of regeneration was similar to that of the overstory.

The structure in a lodgepole pine forest is affected by such mortality factors as needle miner, bark beetle epidemics, mistletoe infestations, and death of suppressed trees in the understory of a dense forest. These factors provide for a buildup of surface fuels that increase the probability of large, high-intensity fires (Brown 1975). Conversely, Gabriel (1976) found areas in the Bob Marshall Wilderness of Montana where a series of intense fires had modified structure and species composition in a way which prevented additional intense fires for a number of years. Muraro (1971) reported a similar situation where "a natural process of fuel modification by fire intensity may discourage second fires of high intensity on severely burned areas ...." This apparently happened because fires had consumed the heavy fuel loads and broken the continuity of fuels from ground to overstory crown in a way that resulted in relatively clean stands of healthy, thinned lodgepole pine; it will be many years before these stands will reach mortality age, break up, and change structure so that high intensity fires can be carried through again.



When aspen is not present, a second burn in less than 25 years may not allow time for lodgepole pine seed to be produced; hence, too frequent fires can create brushfields which remain for long periods (Ayers 1900, Wellner 1970). Low-intensity surface fires on the other hand sometimes remove spruce and fir from the understory and enable lodgepole pine to remain in nearly pure stands even when intermediate in succession (Gabriel 1976).

### Douglas-fir Forests

Douglas-fir is widely distributed in the West, ranging from relatively dry forest with ponderosa pine, where it assumes a more shade tolerant climax role, to more moist forests with hemlock, cedar, and fir, where it assumes a more intolerant seral role. In the Pacific Northwest, it is a particularly prominent component of the timbered ecosystems of western Oregon and Washington between the subalpine forests of fir and mountain hemlock (Tsuga mertensiana) near the Cascade crest to the narrow band of western hemlock (Tsuga heterophylla) and Sitka spruce (Picea sitchensis) along the coast (Weaver 1974). In much of this region, Douglas-fir is replaced by hemlock, cedar, and fir in the absence of fire. In the ponderosa pine region, however, Douglas-fir is the more shade tolerant species which dominates in the absence of fire (Weaver 1974). This is true because Douglas-fir is moderately tolerant, enduring more shade than ponderosa pine, sugar pine, western white pine, and lodgepole pine but less than western hemlock, western redcedar, and white and subalpine fir (Abies lasiocarpa). Douglas-fir needs mineral soil for germination of seeds and development of seedlings. While seedlings can become established under a closed canopy, they usually do not survive to become saplings without sunlight.

Frequencies and intensities of fires reported from Douglas-fir forests vary from (1) low- to medium-intensity surface fires at frequencies of 7 to 20 years in certain drier types in the Northern Rocky mountains (Houston 1973, Loope and Gruell 1973, Arno 1976, Tande 1977), through (2) moderate-intensity fires every 117 to 146 years in a more mesic situation in northern Montana (Sneck 1977), to (3) high-intensity crown fires reported in more mesic forests in many parts of the Rockies and the Pacific Northwest. The presence of Douglas-fir forest on a given site in the Pacific Northwest is considered by some authors to indicate periodic timber-killing fire (Martin and others 1976) at frequencies of 50 to 400 years. Evidence for such high-intensity crown fires in presettlement forest in the more mesic Douglas-fir region comes from reports by Issac and Muir, as quoted in Weaver (1974), that the first white men to arrive in these areas found extensive, immature, even-aged stands of Douglas-fir of the type that follow intensive fires. From his observations in the Puget Sound region in the late 1800's, Muir assumed these high-intensity crown fires had burned during rare drought conditions when such intense burning was possible. Under severe burning conditions, with strong winds, fires crowned in Douglas-fir and covered large areas, but otherwise were relatively low-intensity surface fires.

A history of repeated fires was common to all habitats supporting high populations of Douglas-fir in the coastal forests of British Columbia studied by Schmidt (1960). On Vancouver Island, nearly 2 million acres (0.8 million ha) of forest were burned between 300 and 400 years ago. Most of the mature Douglas-fir on the island owed their origin to very large fires which burned 310 and 410 years before studies by Schmidt (1970). Eight major forest fires burned this area between 150 and 1,100 years ago. The geographic range of coastal Douglas-fir, according to Schmidt (1960) is controlled "by low summer precipitation, high summer temperatures, and the frequency of severe summer lightning storms." He estimates that every 200 to 300 years for most coastal areas, these elements combine to set the stage for intense stand-replacing fires. The distribution of the species itself seems to be limited by incidence of forest fire; Douglas-fir are found on slopes of a U-shaped valley denuded by repeated fires in the past and on south-facing slopes with higher frequency of fires, while uneven-aged forests of more shade tolerant species are found on the more level plateau country and north-facing slopes where fires burn much less frequently.

Arno (1980) states that the major fires in the Northern Rocky Mountain Douglas-fir habitat types "...burned at different intensities in reaction to changes in stand structure, fuel loadings, topography, and weather. The result was a mosaic of fire treatments on the landscape, which tended to be self-perpetuating." Thus, frequent, low-intensity fires tend to keep stands relatively open, and the open structure, in turn, leads to low-intensity fires. On the other hand, high-intensity stand-replacing fires tended to lead to dense, pole-sized stands which are more susceptible to high-intensity crown fires.

Although small-sized, low- to moderate-intensity fires were characteristic of most of the Douglas-fir stands studied by Sneek (1977) in a small experimental forest in the Northern Rockies, she also found that intense, stand-replacing fires had killed much overstory vegetation in the mid-1800's on certain relatively dry, west-facing slopes. This led to regeneration by both Douglas-fir and lodgepole pine. Douglas-fir saplings are gradually replacing decadent members of the lodgepole pine overstory. In so doing, these saplings are creating a new, vertical forest structure which is ladder-like and hence more flammable. When combined with heavy standing and fallen lodgepole pine fuels, this modified vertical structure will support a stand-replacing fire.

Repeated burning or intense fires can eliminate Douglas-fir from some areas. Gabriel (1976) found that extensive areas in the drier southern half of his study area in the Bob Marshall Wilderness of Montana were changed from Douglas-fir to lodgepole pine by fires which burned the same sites in 1809, 1844-47, and 1889. On more moist sites in the northern half of his study area, he found that species composition changed from Douglas-fir and larch to lodgepole pine following particularly intense fires in 1889.

The decrease in fire frequency, which accompanied the beginning of effective fire suppression in the Bitterroot National Forest of Montana in 1920, has brought about major changes in the structure of many parts of the montane and subalpine forest. In drier habitat types, where Douglas-fir is the potential climax tree, dense understories have developed, "making a ladder of fuels that now endangers even the fire-resistant, old-growth overstory of ponderosa pine, Douglas-fir and western larch" (Arno 1976). Forest structure and fuel arrangement in the forest have also been altered by severe infestations of dwarf mistletoe (*Arceuthobium* spp.) in Douglas-fir, western larch, and lodgepole pine, at least partially the result of the decrease in fire frequency (Alexander and Hawksworth 1975). In some forest types, the consequent greater accumulation of fuels and the continuity of fuels over the landscape favor the development of high-intensity fires in areas where low- to moderate-intensity fires were found between 1735 and 1900 (Arno 1976).

In studies of the Selway-Bitterroot Wilderness in Montana, Habeck (1972) concluded that fire exclusion in the Douglas-fir zone has caused some major changes in the vegetational mosaic found prior to 1900. In those presettlement times, he feels, "the landscape would no doubt have exhibited a patchiness composed of contrasting life forms (blackened acres, brushfields, young conifer reproduction, and various seral and climax stands)." With fire suppression, however, diversity has been reduced and the forest has taken on a uniform appearance.

#### Other Types

The role of fire frequency and intensity is less clear for other species involved in the variable low-intensity surface fire/long return interval, crown-fire regime, such as the western larch and western white pine. Larch often occurs with Douglas-fir and ponderosa pine in certain sections of the Rockies and Cascades in Montana, Idaho, Washington, and Oregon (Sudworth 1908). Weaver (1974) indicated that larch is often associated with Douglas-fir in a mixed stand referred to as the "fir-larch" type, although nearly pure stands do sometimes occur. Unlike Douglas-fir,

larch is extremely intolerant of shade, and as such, its principal associate and competitor seems to be lodgepole pine.

Weaver (1974) quotes Sudworth as indicating that the composition of regeneration after burning depends upon which species releases its seeds first--larch or lodgepole pine. Gabriel (1976) and Antos (1977) note that within the shorter lifespan of lodgepole pine (100 to 250 years), stand-replacing fires increase the proportion of lodgepole pine in an area, whereas moderate-intensity fires favor western larch. However, on sites where lodgepole pine has been eliminated as a result of the long interval between fires, larch forms the dominant layer after intense fires as well (Antos 1977). This is true because even after 300 years without fire, large numbers of old-age larch remain which can serve as seed sources after an intense fire.

Daubenmire and Daubenmire (1968) reported that the proportion of larch commonly increases when fires follow one another within a few decades because this species has such thick bark that it is less susceptible than its associates to injury by surface fires. Gabriel (1976), on the other hand, found that too frequent fire around 1900 decimated larch populations within a portion of the Bob Marshall Wilderness. Antos (1977) noted that, "If lodgepole pine or western larch fail to establish after a fire, a mixture of Douglas-fir, grand fir (Abies grandis), western white pine, and spruce slowly form a new stand."

#### VERY LONG RETURN INTERVAL CROWN FIRES

The longer a forest stand goes without fire, the more the shade tolerant, climax species are favored over fire-dependent seral species. While some species like Douglas-fir can be seral in one forest type and climax in another, other species are nearly always climax, and seem to achieve dominance with very long intervals between fires. These include most of the spruces, true firs, western redcedar, and western hemlock. "As a general rule, the more shade tolerant the conifer, the greater is its susceptibility to damage and mortality from fire" (Lotan and others 1981). These species tend to occur on more moist sites, either (1) in higher elevations, as in the case of certain spruces and true firs; or (2) in sites more moist because of local topographic position; or (3) in sites which are more moist because of climatic conditions, such as the Pacific coastal forests of redcedar, hemlock, and certain other spruces and firs, or on the west slopes of the Northern Rockies in northern Idaho and western Montana, which are influenced by the prevailing winds and the Pacific maritime climate from the west.

In the absence of fire, the successional trend is toward increased dominance of true firs such as subalpine fir, grand fir, and white fir and a decrease in the spruces, Douglas-fir, and lodgepole pine. Because of the long natural interval between fires in these types, the suppression of fires for the last 50 to 75 years has probably had less impact on their structure than in areas of more frequent surface fires (Fahnestock 1976).

#### Spruce-Fir Forests

These forests are usually so damp they seldom burn (Weaver 1974) except during extreme droughts. During those periods, fires burn very intensely through these forests because their highly flammable green limbs are close to the ground. Intense burning in 350-year-old spruce-fir forests in Yellowstone National Park was apparently related to such forest structural factors as compact growth form, with numerous small branches close together, lower branches that do not self-prune, a fruticose lichen which had accumulated on the branches, and heavy accumulations of duff and rotten wood (Despain and Sellers 1977). Such an abundance of surface and ladder fuels led to a high probability of fire brands falling on fuels which would in turn produce flames reaching into flammable spruce-fir crowns.

Oosting and Reed (1952) quote Hansen's evidence that high-intensity forest fires have been a recurrent factor in the spruce-fir forests of the Medicine Bow Range of Wyoming during presettlement time. In the Bob Marshall Wilderness of Montana, Gabriel (1976) noted that periodic low-intensity fire prevented the growth of spruce and subalpine fir in stands of lodgepole pine.

In the Northern Rocky Mountains, Arno (1980) reported that moist subalpine fir-Engelmann spruce (*Picea engelmannii*) forests had fire-free intervals of 150 years or more, and that when fire did come, it was usually a high-intensity, stand-replacing fire. Sneck (1977) found spruce stands in moist draws that were 250 to 370 years old, with no evidence of fire.

Studies by Romme (1977) in the Medicine Bow Mountains of Wyoming indicate that topographic and microclimatic differences between canyon bottoms and adjacent slopes and ridges may result in differences in fire frequency and successional patterns following fire. Fires were less frequent in spruce-fir forests found along canyon bottoms than in adjacent upland lodgepole pine forest, and when fire did occur, succession to spruce-fir was more rapid and direct. "Thus, the pattern of canyon bottom spruce-fir forests surrounded by upland lodgepole pine forest is a result both of fewer fires and more rapid succession in the canyon bottom sites" (Romme 1977).

#### Hemlock-Cedar Forests

The extremely moist forest of western hemlock-western redcedar of the Northern Rocky Mountains and Sitka spruce-hemlock-redcedar of the Pacific coastal forests are discussed together here. Relatively little information on fire frequency and intensity is available on either of these types. In studies in the Selway-Bitterroot Wilderness of Idaho and Montana, Habeck (1976) noted that, "The oldest forest communities (over 400 years old), those burned least often, are those dominated by redcedar on the moist streamsides and in ravines." Hemlock-redcedar forests generally occur in sites with the dampest climate found in the Rocky Mountains, but extreme summer drought from time to time has set the stage for occasional, spectacular crown fires (Arno 1980). The most recent large conflagration was the 56,000-acre (22 672-ha) Sundance Fire in northern Idaho in 1967 (Anderson 1968), but earlier, high-intensity crown fires burned several million acres in 1889, 1910, 1919, 1926, and 1934, much of it in the hemlock and redcedar type of northern Idaho (Arno 1980). In some cases, the fuels generated by the first fires appeared to lead to a second burn which resulted in shrubfields taking over the site for decades.

The presence of cedar-hemlock forest on the western slope of the Olympics in Washington is interpreted by Martin and others (1976) as an indication of very long intervals between fires, more than 150 years. Lotan and others (1981) concluded that the wettest forest in this type may have burned at intervals of 500 years or longer.

#### True Fir Forests

High-intensity, stand-replacement fires at intervals of about 150 years were apparently more prevalent than surface fires in grand fir (*Abies grandis*) forests of Swan Valley, Mont. (Antos 1977). When such a fire occurs, grand fir and western redcedar, which developed below the canopy, are killed; the survival of a few seed trees or of viable seeds in burned cones is critical when large areas are burned (Lotan and others 1981). "Which species will attain the highest density is a function of seed source and the timing of good seed crops" (Antos 1977).

In the absence of fire, grand fir gains dominance over all other conifers in its habitat types as succession proceeds toward climax (Pfister and others 1977). Following fire, Douglas-fir, larch, spruce, and lodgepole pine on some sites invade along with grand fir, but by the time a pole-sized stand has developed, "*Abies grandis* is generally

the only species that continues to reproduce beneath the forest canopy" (Pfister and others 1977), with increased shade and soil moisture often playing important roles here. In the Selway-Bitterroot Wilderness of Montana, Habeck (1976) found that grand fir regeneration was spreading in all directions, into wetter, drier, and higher elevation sites. He concluded that expansion of the fire-sensitive grand fir may be a relatively new event related to fire suppression.

#### DISCUSSION

Gill (1979) noted that it is unfortunate when someone implies that either a wildfire or management fire will not harm vegetation because the flora is "adapted to fire." He points out rightly that this is an incomplete concept, and a better statement would be that, "the vegetation is adapted to a certain fire regime." The significance of this point is driven home by noting the broad range of frequencies and intensities which were found in presettlement forests and scrublands of the western United States, ranging from the frequent low-intensity surface fires of ponderosa pine to the very infrequent and high-intensity crown fires found in many spruce-fir forests. Frequent low-intensity burning would be just as out of place in spruce-fir forests as infrequent high-intensity crown fires are in naturally operating ponderosa pine ecosystems.

Yet, with fire having been suppressed in many western forests for the past 50 to 100 years, unnatural changes have been evolving slowly--even imperceptibly in some cases. This is leading to a sequence of events which may be developing for the first time ever. A conceptual model of this sequence of a sequoia-mixed conifer forest would show that suppression leading to longer intervals between fires would initially lead to increases in surface and crown fuels and to changes in the forest structure such that there would be (1) older age classes; (2) a denser forest (less openings); (3) a multilayered vertical structure; (4) an increase in ladder fuels; and (5) more shade tolerant species. This would next lead to increased probability of higher intensity surface fires with some individual tree crowns burning, and to sequential impacts on postfire forest structure, species composition, fuel accumulation, and both horizontal and vertical pattern. In turn, this would bring about changes in fire frequency and intensity. The unknown in such a model is how a new balance would be arrived at, as long as man puts substantial energy into fire suppression.

Such changes in the absence of fire during the past century or less are greatest in those forests which historically had a short return interval between fires and least in forests which had a very long return interval. For example, if the normal interval between fires is 300 years or more, the recent absence of fire for 50 to 100 years would be of little significance.

In certain northern forests, however, it appears that fire intensity is not a pure function of fuel buildup with age. Work by Van Wagner (1977) and Fahnestock (1977) suggests that crown fire potential is probably greatest at young and moderate age, then decreases as the stand matures. "Certainly the role of fire spread depends more on the quantity and arrangement of fire fuels than on the accumulation of downed logs" (Van Wagner 1978). Once breakup of the forest stand begins through death of certain older or weaker trees, there is a major increase in dead materials added to surface fuels and surface fire intensity might take a surge upward. Because of new openings in the crown canopy, however, continuous crown fires would then be less likely unless a well-defined conifer understory had arisen.

#### Plant Succession

Plant succession following fire does not necessarily follow a single pathway. As Cattellino and others (1979) point out, various species-specific attributes related to reproduction and survival determine the successional outcome when a given ecosystem is disturbed by fire. As an example, in the Northern Rocky Mountains, a community made

up of aspen, lodgepole pine, and western larch will have different outcomes depending upon the intervals between fires as they relate to the following life-history characteristics: (1) aspen's lifespan of 130 years and its vegetative reproduction; (2) lodgepole pine's lifespan of 250 years and its need for 20 years before mature cones are produced; and (3) larch's 300- to 400-year lifespan and its ability to disperse seeds widely from surrounding forests after a fire. If a community with aspen and lodgepole pine is burned less than 130 years after the last fire, both species will be present in the postfire succession. If the stand burns after 250 years, both species will be lost and larch will take over by seeding in from surrounding forests. If the stand burns in less than 20 years, lodgepole pine will be lost, but both aspen and larch would be part of postfire succession (Cattelino and others 1979). Thus, man-caused changes in fire frequency--through suppression or prescribed burning--can have major impact on the species composition and structure of a forest.

#### Insects, Parasites, and Disease

Brown (1975) and others have emphasized the importance of insects, parasites, and disease in modifying the structure of many western forests and, in turn, the frequency and intensity of subsequent fires. Relatively little quantitative work has been done on the effects of fire on plant diseases (Alexander and Hawksworth 1976, Parmeter 1977). Available evidence--particularly concerning fire and dwarf mistletoe (Arceuthobium spp.)--has been summarized by Alexander and Hawksworth (1975, 1976), Hardison (1976a, 1976b), Harvey and others (1976), Wicker and Leaphart (1976), and Parmeter (1977, 1978). The relationship between fire, insects, and forest structure has been discussed by Miller and Keen (1960) and Roe and Amman (1970).

In an overview of the impact of fire on pathogens, Parmeter (1977) concluded that, "Fire can alter disease activity in forests and scrublands directly by affecting the survival and development of pathogens or indirectly by affecting characteristics of plant communities, individual plants, or physical and microbial environments that influence pathogens." The relationship between fire and disease is complicated both by the variability of fire intensity and frequency and by the complexity of possible interactions between fire and disease.

In the absence of fire, numbers of trees infected, intensity of infection, and degree of damage increases with age of trees or stands and size of trees. The rate of spread through multistoried stands is also more rapid than through single-storied stands (Parmeter 1978).

Intense fires tend to have a sanitizing effect on infected stands giving lodgepole pine the edge over mistletoe, while partial burns create ideal conditions for rapid spread of mistletoe in even young stands (Alexander and Hawksworth 1976). Mistletoe, on the other hand, kills and stunts trees, causes spike tops and witches' brooms, which modify vertical structure of the forest, providing "ladders" for fire to ascend and consume tree crowns. This obviously increases fire hazard, flammability, and fire intensity. Fire may encourage spread of the mistletoe parasite by conversion of nonsusceptible climax spruce-fir forests to mistletoe-susceptible nearly pure stands of lodgepole pine (Alexander and Hawksworth 1976). At the same time, fire may limit development of natural genetic resistance to mistletoe by continually preventing survival of any trees that may have become somewhat resistant (Roth 1974).

Prescribed burning in pine plantations in the South reduced pine mortality and total infection by Heterobasidion annosum (Fomes annosus) root rot (Froelich and others 1978). The impacts of fire on the fungus were greatest where the disease was most serious. Although confirming field studies need to be carried out in western forests, Parmeter and Uhrenholdt (1975) have found that spore germination or mycelial growth of several fungi, including Fomes annosus, was reduced by exposure to smoke. By contrast, Trichoderma spp., common fungal competitors of Fomes annosus, increased in soil after

burning (Froelich and others 1978) and germination of Trichoderma spores was increased when exposed to smoke (Parmeter and Uhrenholdt 1975).

Age structure of lodgepole pine forests relates very strongly to potential for attack by mountain pine beetle (Dendroctonus ponderosae Hopk.); beetles first attack larger (14-inch diameter or more) trees which have thick phloem and a better food supply (Roe and Amman 1970). In addition, high-elevation stands show less infestation by pine beetles and greater infection by dwarf mistletoe (Amman 1969). Beetle activity in the absence of fire in the Northern Rocky Mountains leads to replacement of lodgepole pine by Douglas-fir at lower elevations and by subalpine fir and spruce at higher elevations. In the presence of fire, seedling lodgepole pine come in.

As an example, the 1961 Sleeping Child Fire burned more than 25,000 acres of lodgepole pine in the Bitterroot National Forest of Montana which had previously been heavily damaged by mountain pine beetle between 1928 and 1932 (Roe and Amman 1970). Lodgepole pine seedlings became established on about 15,000 acres of the burned area; within 80 to 90 years, these newly established pine trees will simultaneously reach sizes attractive to beetles over sizable areas. Then a widespread infestation of beetles will lead to heavy fuels which in time will lead to intense burning which may be followed by another restocking of lodgepole pine seedlings which will start the sequence again. In areas where mistletoe infection is heavy, tree growth is reduced, thus reducing the likelihood of beetle infestation. However, in that instance, the previously described changes in vertical structure which mistletoe brings about also lead to fire hazards and burning.

In ponderosa pine forests, Miller and Keen (1960) found that western pine beetle (Dendroctonus brevicomis) populations concentrate in fire-injured trees which survive a fire. They noted that unburned stands of ponderosa pine had larger numbers of small trees than burned areas, but that burned areas had larger trees and larger volume per acre. Thus, they concluded that in general, "...the long-term effect of fire is to lessen western pine beetle damage rather than increase it."

It would appear that man's efforts to decrease fire frequency often lead to conditions favoring growth of forest insects or disease organisms. Such growth in turn affects forest structure and fuel buildup in a way which insures the forest will burn, sometimes with greater intensity than would have been the case otherwise.

#### Stability, Diversity, and Resilience

Bonnicksen and Stone (1978) note that in the absence of frequent fire in sequoia-mixed conifer forests, there is not a steady state mosaic. Instead, younger aggregations are being replaced by older ones. There is no fire to make new openings and no opportunity for seedling and sapling white fir to become the top tier dominant in such new aggregations. Keeley (this volume) has suggested that frequent fires are an important part of the fire regime, because they preserve seed trees by preventing widespread crown fires among species such as ponderosa pine which have no means of storing seed on trees or in the soil. Under a periodic fire regime in a mixed-conifer ecosystem, Agee and others (1978) point out that ecosystem dynamics appeared to be more stable. Low decomposition rates and the flammable nature of the forest floor in this system assured that frequent surface fires would occur, periodically releasing stored energy.

"If stability is defined as the ability to resist change, then...vegetative cycles maintained and driven by fire must be considered to be stable" (Vogl 1970). As such, lodgepole pine communities and other fire types would be considered very stable, because fires in such types result in a replacement community of similar structure to that originally found there, while fire in climax types results in extreme change (Brown 1975). On the other hand, when natural fires are suppressed in a mixed conifer forest, biomass and energy accumulate, species composition shifts, the new system is

less flammable, ladder fuels build up in understory trees, and a wildfire which does occur is much more likely to become a high-intensity crown fire, killing mature trees and seriously disrupting the system (Kilgore and Sando 1975, Agee and others 1978).

Loucks (1970) believed that community stability, species diversity, and annual productivity are positively related. He offered the hypothesis that "...evolution in ecosystems has brought about not only adaptation to heterogenous environments, but adaptation to a repeating pattern of changing environments, a stationary process that represents a composite of time intervals over which replacement of species is repeated over and over again." As a part of their study of fire-dependent forests in the Northern Rocky Mountains, Habeck and Mutch (1973) offered a similar hypothesis that, "...the diversity of community life forms engendered some sort of ecosystem equilibrium or a kind of biologic 'check and balances' system that governed the magnitude of the effects accompanying a given forest fire." Taking this a step further, one could also assume that simplifying the community life forms by eliminating fire may modify forest structure in a way that brings imbalance to the natural system. Taylor (1973) found that elimination of fires in lodgepole pine communities in Yellowstone National Park, Wyo., limited ecological diversity by reducing or eliminating those plants and animals found only in successional communities present before closure of the forest canopy.

A different point of view is found elsewhere in the ecological literature. "Resilience" has recently been defined as "the ability of a natural ecosystem to restore its structure following acute or chronic disturbance..." (Westman 1978). This set of properties was included under the term "stability" in earlier work by May (1973), Holling (1973), and Orians (1975). Westman (1978) suggests that "stability" be limited to the "pattern of fluctuations in a relatively unimpacted ecosystem over time" and cites work by Whittaker (1975) to support this. The ability of a system to resist displacement in structure and function when subjected to a disturbing force (such as fire) has been referred to as "inertia" by Orians (1975) and Westman (1978). Holling (1973) had termed this same property "resilience," while Vogl (1970) had called it "stability."

Botkin and Sobel (1975) have pointed out that "stability" in the anecdotal literature has often been implicit and vague. Where it was defined, the concept was similar to the "static stability" found in a classical mechanics system that returns to equilibrium after being disturbed. Such definitions are found in Odum (1971), Krebs (1972), and Smith (1974). Holling (1973) and Botkin and Sobel (1975) contend that such a static stability concept, borrowed from classical physics, may be inappropriate for the analysis of ecosystems in that, "Natural undisturbed systems are likely to be continually in a transient state" (Holling 1973).

Using the example from Heinselman's (1973) history of the 100-year return interval for fire in the forest of the Boundary Waters Canoe Area of Minnesota, Botkin and Sobel (1975) maintained that the concept of an equilibrium state for a natural forest has been contradicted by history and that static stability is a concept with little ecological value. They ask the question: "What does it mean to 'stabilize' or 'preserve' a forest when the natural undisturbed forest is changing through time, and when fire appears to be an intrinsic event?" They feel that the analysis of stability and its dependence on ecosystem complexity must be broadened to include "notions of stability besides the definition of static stability borrowed from statistical mechanics."

May (1973) contends that in the natural world, "it is not true that population stability is uniformly associated with trophic complexity and faunal and floral diversity." On the contrary, a number of natural monocultures, such as the march grass (*Spartina alterniflora*), are very stable; the instability of man-made monocultures results not from their simplicity, but from their unnaturalness. As a mathematical generality, May (1973) contends that increasing diversity and complexity enhance



community instability. But he also notes that we need much better understanding of principles which govern natural associations of plants and animals. This can be primarily gained by studies in pristine ecosystems such as those found in National Parks and Wildernesses.

#### CONCLUSIONS

Fire suppression has caused an increase in intervals between fires in many areas. This decrease in frequency is leading to an increase in surface fuels in many forests and a modification of forest structure.

This structural change is manifested as a shift in the distribution of ages and species composition in various forests and scrublands of the West. In turn, these changes in age classes and species composition affect both the horizontal pattern and vertical structure of the forest. Such structural changes lead to significant modification in the vertical and horizontal fuel arrangement of the forest--including an increase in ladder fuels--and hence to changes in probable intensity of future fires. In many cases, these changes appear to lead toward higher intensity fires in forests which previously had frequent, low-intensity fires as a normal fire regime. Changes are less noticeable so far where long return interval, high-intensity fire regimes are involved.

Such shifts in frequency of burning--which lead to changes in structure--also modify the intensity of subsequent fires, and in turn can lead to substantial changes in postfire forest structure. This new and different forest structure can in turn lead to changes in both frequency and intensity of subsequent fires.

The complexities of these interactions in any given forest are further compounded by (1) the very large number of possible species combinations in various geographic zones and (2) the variations which topography and weather can bring to the impact of a given frequency and intensity of fire on various vegetation types.

There is great need for additional knowledge about natural fire regimes--involving both frequency and intensity--in many forest types of the West. But there is also need to take full advantage of what we already know and to apply this knowledge in management of our forest and scrubland ecosystems. In summary, as has been pointed out elsewhere (Kilgore 1976), we need better understanding of fire as a process and as a tool, but we also need greater commitment by land managers to use on the land the best of what we already know. To carry out an effective resources management program in western forests and scrublands, we will also need a well-trained cadre of fire managers who understand fire behavior under a variety of vegetation and weather conditions. To be fully successful as scientists, laymen, and environmentally concerned citizens, we must also work to gain greater public understanding of and involvement in developing and approving our land management practices.

Just as the fire process is basic to the operation of many ecosystems in the West, so our understanding of fire regimes is basic to our management of such ecosystems in National Parks, Wilderness Areas, and other public lands. With firmer knowledge and insight into how fire frequency and intensity relate to the structure of these systems, we can begin to use fire as a tool to best simulate its natural role in these areas.

#### ACKNOWLEDGMENTS

I wish to thank the following for critical review of the manuscript: M. Alexander, S. Arno, H. Biswell, T. Bonnicksen, N. Christensen, D. Despain, D. Fenn, T. Harvey, D. Houston, J. Keeley, C. Van Wagner, and R. Vogl. Valuable suggestions concerning material to consider for particular vegetation types were offered by J. Dieterich, J. Franklin, M. Gill, M. Heinselman, J. Parmeter, D. Parsons, J. van Wagtenonk, and D. Wood.

PUBLICATIONS CITED

- Agee, James K.  
 1973. Prescribed fire effects on physical and hydrologic properties of mixed-conifer forest floor and soil. Contr. Rep. 143, 57 p. Water Res. Center, Univ. Calif., Davis.
- Agee, James K., R. H. Wakimoto, and H. H. Biswell.  
 1978. Fire and fuel dynamics of Sierra Nevada conifers. For. Ecol. Manage. 1:255-265.
- Albini, Frank A.  
 1976. Estimating wildfire behavior and effects. USDA For. Serv. Gen. Tech. Rep. INT-30, 92 p. Intermt. For. and Range Exp. Stn., Ogden, Utah.
- Alexander, Martin E., and F. G. Hawksworth.  
 1975. Wildland fires and dwarf mistletoes: a literature review of ecology and prescribed burning. USDA For. Serv. Gen. Tech. Rep. RM-14, 12 p. Rocky Mt. For. and Range Exp. Stn., Fort Collins, Colo.
- Alexander, Martin E., and F. G. Hawksworth.  
 1976. Fire and dwarf mistletoes in North American coniferous forest. J. For. 74(7):446-449.
- Amman, Gene D.  
 1969. Mountain pine beetle emergence in relation to depth of lodgepole pine bark. USDA For. Serv. Res. Note INT-96, 8 p. Intermt. For. and Range Exp. Stn., Ogden, Utah.
- Anderson, Hal E.  
 1968. Sundance fire: an analysis of fire phenomena. USDA For. Serv. Res. Pap. INT-56, 39 p. Intermt. For. and Range Exp. Stn., Ogden, Utah.
- Antos, Joseph Avery.  
 1977. Grand fir (*Abies grandis* (Dougl.) Forbes) forests of the Swan Valley, Montana. M.A. thesis. Univ. Mont., Missoula.
- Arno, Stephen F.  
 1976. The historical role of fire on the Bitterroot National Forest. USDA For. Serv. Res. Pap. INT-187, 29 p. Intermt. For. and Range Exp. Stn., Ogden, Utah.
- Arno, Stephen F.  
 1980. Forest fire history of the northern Rockies. J. For. 78(8):460-465.
- Ayres, H. B.  
 1900. Flathead Forest Reserve, Montana. In 20th Annu. Rep. USGS, 1898-1899. Part V--Forest Reserves. p. 245-316.
- Bentley, J. R.  
 1967. Conversion of chaparral areas to grassland: techniques used in California. USDA Agric. Handb. 328, 35 p.
- Biswell, Harold H.  
 1961. The big trees and fires. Natl. Parks Mag. 35:11-14.
- Biswell, Harold H.  
 1967. The use of fire in wildland management in California. In Natural resources: quality and quantity. p. 71-87. S. V. Ciracy-Wantrup and James Parsons, eds. Univ. Calif. Press, Berkeley.
- Biswell, Harold H.  
 1974. Effects of fire on chaparral. In Fire and ecosystems. p. 321-364. T. T. Kozlowski and C. E. Ahlgren, eds. Academic Press, New York.
- Bonnicksen, Thomas Madsen.  
 1975. Spatial pattern and succession within a mixed conifer-giant sequoia forest ecosystem. M.S. thesis. Univ. Calif., Berkeley.
- Bonnicksen, Thomas M., and E. C. Stone.  
 1978. An analysis of vegetation management to restore the structure and function of presettlement giant sequoia-mixed conifer forest mosaics. Final Contr. Rep. for Natl. Park Serv., San Francisco, Calif. 159 p.
- Botkin, Daniel B., and M. J. Sobel.  
 1975. Stability in time-varying ecosystems. Am. Nat. 109(907):625-646.

- Brown, James K.  
1975. Fire cycles and community dynamics in lodgepole pine forests. In Management of Lodgepole Pine Ecosystems Symposium. p. 429-456. D. M. Baumgartner, ed. Wash. State Univ. Coop. Ext. Serv., Pullman.
- Byram, George M.  
1959. Combustion of forest fuels. In Forest fire: control and use. p. 61-89. Kenneth P. Davis, ed. McGraw-Hill, New York.
- Byrne, Roger.  
1978. Fossil charcoal as a measure of wildfire frequency in southern California: a preliminary analysis. In Environmental consequences of fire and fuel management in Mediterranean ecosystems. p. 361-367. H. A. Mooney and C. E. Conrad, eds. USDA For. Serv. Gen. Tech. Rep. WO-3. Washington, D.C.
- Cattelino, Peter J., I. R. Noble, R. O. Slatyer, and S. R. Kessell.  
1979. Predicting the multiple pathways of plant succession. Environ. Manage. 3(1):41-50.
- Clements, F. E.  
1910. The life history of lodgepole burn forest. USDA For. Serv. Bull. 79, 56 p.
- Cooper, Charles F.  
1960. Changes in vegetation, structure, and growth of southwestern pine forests since white settlement. Ecol. Monogr. 30:129-164.
- Cooper, Charles F.  
1961. Pattern in ponderosa pine forest. Ecology 42:493-499.
- Daubenmire, R., and J. B. Daubenmire.  
1968. Forest vegetation of eastern Washington and northern Idaho. Wash. Agric. Exp. Stn. Tech. Bull. 60, 104 p.
- Despain, Don G., and R. E. Sellers.  
1977. Natural fire in Yellowstone National Park. West. Wildlands 4(1):20-24.
- Erdman, James A.  
1970. Pinyon-juniper succession after natural fires on residual soils of Mesa Verde, Colorado. Brigham Young Univ., Science Bull. Biol. Ser. 11(2), 26 p. Provo, Utah.
- Fahnestock, George R.  
1976. Fires, fuels, and flora as factors in wilderness management: The Pasayten case. In Proc. Tall Timbers Fire Ecol. Conf. 15:33-69.
- Fahnestock, George R.  
1977. Interactions of forest fire, flora, and fuels in two Cascade Range wilderness areas. Ph.D. thesis. Univ. Wash., Seattle. [Diss. Abstr. Int. 38:2451-B.]
- Ferguson, Charles Wesley.  
1960. Annual rings in big sagebrush, Artemisia tridentata. Ph.D. thesis. Univ. Ariz., Tucson. [Diss. Abstr. Int. 20:4225, 1960.]
- Franklin, Jerry F., and C. T. Dyrness.  
1973. Natural vegetation of Oregon and Washington. USDA For. Serv. Gen. Tech. Rep. PNW-8, 417 p. Pac. Northwest For. and Range Exp. Stn., Portland, Oreg.
- Fritz, Emanuel.  
1932. The role of fire in the redwood region. Agric. Exp. Stn. Circ. 323, 23 p. Univ. Calif., Berkeley.
- Froelich, R. C., C. S. Hodges, Jr., and S. S. Sackett.  
1978. Prescribed burning reduces severity of Annosus root rot in the South. For. Sci. 24(1):93-100.
- Gabriel, Herman W., III.  
1976. Wilderness ecology: the Danaher Creek drainage, Bob Marshall Wilderness, Montana. Ph.D. thesis. Univ. Mont., Missoula.
- Gill, A. Malcolm.  
1973. Effects of fire on Australia's native vegetation. CSIRO Div. Plant Ind. Annu. Rep.
- Gill, A. Malcolm.  
1979. The role of species characteristics of plants in fire climates as a guide to management. In Fire and Fuel management problems in Mediterranean-climate ecosystems: research priorities and programs. J. K. Agee, ed. Rep. on Res. Workshop, Fire Symp., Stanford Univ., 1977. [Submitted as an MAB Tech. Note.]

- Gruell, G. E., and L. L. Loope.  
1974. Relationships among aspen, fire, and ungulate browsing in Jackson Hole, Wyoming. USDA For. Serv., Intermt. Reg. and U.S. Dep. Interior, Natl. Park Serv., Rocky Mt. Reg. 33 p.
- Habeck, James R.  
1972. Fire ecology investigations in Selway-Bitterroot Wilderness: historical considerations and current observations. Univ. Mont.--USDA For. Serv. Publ. R1-72-001, 119 p. Missoula, Mont.
- Habeck, James R.  
1976. Forests, fuels, and fire in the Selway-Bitterroot Wilderness, Idaho. In Proc. Tall Timbers Fire Ecol. Conf. 14:305-353.
- Habeck, James R., and R. W. Mutch.  
1973. Fire-dependent forests in the northern Rocky Mountains. Quat. Res. 3(3): 408-424.
- Hall, Frederick C.  
1976. Fire and vegetation in the Blue Mountains--implications for land managers. In Proc. Tall Timbers Fire Ecol. Conf. 15:155-170.
- Hanes, T. L.  
1971. Succession after fire in the chaparral of southern California. Ecol. Monogr. 41:27-52.
- Hardison, J. R.  
1976a. Fire and flame for plant disease control. Annu. Rev. Phytopathol. 14:355-379.
- Hardison, John R.  
1976b. Fire and disease. In Proc. Tall Timbers Fire Ecol. Conf. 15:223-234.
- Hartesveldt, Richard J.  
1964. Fire ecology of the giant sequoias: controlled fire may be one solution to survival of the species. Nat. Hist. Mag. 73(10):12-19.
- Hartesveldt, R. J., and H. T. Harvey.  
1967. The fire ecology of sequoia regeneration. In Proc. Tall Timbers Fire Ecol. Conf. 7:65-77.
- Harvey, A. E., M. F. Jurgensen, and M. J. Larsen.  
1976. Intensive fiber utilization and prescribed fire: effects on the microbial ecology of forests. USDA For. Serv. Gen. Tech. Rep. INT-28, 46 p. Intermt. For. and Range Exp. Stn., Ogden, Utah.
- Harvey, H. Thomas, H. S. Shellhammer, and R. E. Stecker.  
[In press.] Giant sequoia ecology: fire and reproduction. U.S. Dep. Interior, Natl. Park Serv. Monogr. Series.
- Heinselman, Miron L.  
1973. Fire in the virgin forests of the Boundary Waters Canoe Area, Minnesota. Quat. Res. 3(3):329-382.
- Heinselman, Miron L.  
1978. Fire in wilderness ecosystems. In Wilderness management. p. 248-278. John C. Hendee, George H. Stankey, and Robert C. Lucas, eds. USDA For. Serv. Misc. Publ. 1365. Washington, D.C.
- Hendrickson, William H.  
1972. Perspective on fire and ecosystems in the United States. In Fire in the Environment Symp. p. 29-33. USDA For. Serv., Washington, D.C.
- Holling, C. S.  
1973. Resilience and stability of ecological systems. In Annu. Rev. Ecol. and Systematics 4:1-23.
- Horton, K. W., and E. J. Hopkins.  
1966. Influences of fire on aspen suckering. Dep. For. Can. Publ. 1095, 19 p.
- Houston, Douglas B.  
1973. Wildfires in northern Yellowstone National Park. Ecology 54(5):1111-1117.
- Hreha, Alyce M.  
1978. A comparative distribution of two mistletoes: Arceuthobium divaricatum and Phoradendron juniperinum (South Rim, Grand Canyon National Park, Arizona). M.S. thesis. Brigham Young Univ., Provo, Utah.

- Humphrey, Robert R.  
1974. Fire in the deserts and desert grassland of North America. In Fire and ecosystems. p. 365-400. T. T. Kozlowski and C. E. Ahlgren, eds. Academic Press, New York.
- Keeley, Jon E.  
1977. Fire-dependent reproductive strategies in Arctostaphylos and Ceanothus. In Environmental consequences of fire and fuel management in Mediterranean ecosystems. p. 391-396. H. A. Mooney and C. E. Conrad, eds. USDA For. Serv. Gen. Tech. Rep. WO-3. Washington, D.C.
- Keeley, Jon E., and Paul H. Zedler.  
1978. Reproduction of chaparral shrubs after fire: a comparison of sprouting and seeding strategies. *Am. Midl. Nat.* 99(1):142-161.
- Keen, F. P.  
1940. Longevity of ponderosa pine. *J. For.* 38:597-598.
- Kilgore, Bruce M.  
1971. The role of fire in managing red fir forests. *Trans. North Am. Wildl. Nat. Resour. Conf.* 36:405-416.
- Kilgore, Bruce M.  
1973a. Impact of prescribed burning on a sequoia-mixed conifer forest. In Proc. Tall Timbers Fire Ecol. Conf. 12:345-375.
- Kilgore, Bruce M.  
1973b. The ecological role of fire in Sierran conifer forests: its application to national park management. *Quat. Res.* 3(3):496-513.
- Kilgore, Bruce M.  
1976. From fire control to fire management: an ecological basis for policies. *Trans. North Am. Wildl. Nat. Resour. Conf.* 41:477-493.
- Kilgore, Bruce M., and H. H. Biswell.  
1971. Seedling germination following fire in a giant sequoia forest. *Calif. Agric.* 25(2):8-10.
- Kilgore, Bruce M., and G. S. Briggs.  
1972. Restoring fire to high elevation forests in California. *J. For.* 70(5):266-271
- Kilgore, Bruce M., and R. W. Sando.  
1975. Crown-fire potential in a sequoia forest after prescribed burning. *For. Sci.* 21(1):83-87.
- Kilgore, Bruce M., and D. Taylor.  
1979. Fire history of a sequoia-mixed conifer forest. *Ecology* 60:129-142.
- King, Clarence.  
1871. *Mountaineering in the Sierra Nevada*. Reprinted by Norton, New York. 320 p.
- Krebs, C. J.  
1972. *Ecology*. Harper and Row, New York. 694 p.
- Loope, Lloyd L.  
1971. Dynamics of forest communities in Grand Teton National Park. *Naturalist* 22(1):39-47.
- Loope, Lloyd L., and G. E. Gruell.  
1973. The ecological role of fire in the Jackson Hole Area, northwestern Wyoming. *Quat. Res.* 3(3):425-443.
- Lotan, James E., Martin E. Alexander, Stephen F. Arno, and others.  
1981. Effects of fire on flora, a state-of-knowledge review. USDA For. Serv. Gen. Tech. Rep. WO-16. Washington, D.C.
- Loucks, Orie L.  
1970. Evolution of diversity, efficiency, and community stability. *Am. Zool.* 10:17-25.
- Martin, Robert E.  
1978. Fire manipulation and effects in Western Juniper (Juniperus occidentalis Hook). In Western Juniper Ecology and Management Workshop. p. 121-136. R. E. Martin, J. E. Dealy, and D. L. Caraher, eds. USDA For. Serv., Pac. Northwest For. and Range Exp. Stn., Portland, Oreg.

- Martin, Robert E., D. D. Robinson, and W. E. Schaeffer.  
1976. Fire in the Pacific Northwest--perspectives and problems. In Proc. Tall Timbers Fire Ecol. Conf. 15:1-23.
- May, R. M.  
1973. Stability and complexity in model ecosystems. Princeton Univ. Press, N.J. 235 p.
- McBride, Joe, and D. Jacobs.  
1978. The history of the vegetation of Muir Woods National Monument. Final contr. rep. U.S. Dep. Interior, Natl. Park Serv., San Francisco, Calif.
- McBride, Joe R., and R. D. Laven.  
1976. Scars as an indicator of fire frequency in the San Bernardino Mountains, California. J. For. 74(7):439-442.
- McNeil, Robert C., and D. B. Zobel.  
[In press.] Vegetation and fire history of a ponderosa pine-white fir forest in Crater Lake National Park. Northwest Sci.
- Menke, John W., and R. Villasenor.  
1977. The California Mediterranean ecosystem and its management. In Environmental consequences of fire and fuel management in Mediterranean ecosystems. p. 257-270. H. A. Mooney and C. E. Conrad, eds. USDA For. Serv. Gen. Tech. Rep. WO-3. Washington, D. C.
- Methven, Ian R.  
1978. Fire research at the Petawawa Forest Experiment Station: the integration of fire behavior and forest ecology for management purposes. In Fire Ecology in Resource Management Workshop Proc. [Dec. 1977]. p. 23-27. D. E. Dube', Inf. Rep. NOR-X-210. North For. Res. Cent., Can. For. Serv., Edmonton, Alta.
- Miller, J. M., and F. P. Keen.  
1960. Biology and control of the western pine beetle: a summary of the first fifty years of research. U.S. Dep. Agric. Misc. Publ. 800, 381 p.
- Muir, John.  
1894. The mountains of California. Houghton, Boston, Mass.
- Muir, John.  
1901. Our national parks. Houghton, Boston, Mass.
- Munz, P. A., and D. D. Keck.  
1959. A California flora. Univ. Calif. Press, Berkeley. 1681 p.
- Muraro, S. J.  
1971. The lodgepole pine fuel complex. Dep. Fish and For., Can. For. Serv. Inf. Rep. BC-X-53, 35 p. For. Res. Lab., Victoria, B.C.
- Odum, Eugene P.  
1971. Fundamentals of ecology. Saunders, Philadelphia, Pa. 574 p.
- Oosting, Henry J.  
1956. The study of plant communities. (2nd ed.) W. H. Freeman and Co., San Francisco Calif. 440 p.
- Oosting, Henry J., and J. F. Reed.  
1952. Virgin spruce-fir of the Medicine Bow Mountains, Wyoming. Ecol. Monogr. 22(2):69-91.
- Orians, G. H.  
1975. Diversity, stability, and maturity in natural ecosystems. In Unifying concepts in ecology. p. 64-65. W. H. van Dobben and R. H. Lowe-McConnell, eds. Junk, The Hague.
- Parmeter, John R., Jr.  
1977. Effects of fire on pathogens. In Environmental consequences of fire and fuel management in Mediterranean ecosystems. p. 58-64. H. A. Mooney and C. E. Conrad, eds. USDA For. Serv. Gen. Tech. Rep. WO-3. Washington, D.C.
- Parmeter, J. R., Jr.  
1978. Forest stand dynamics and ecological factors in relation to dwarf mistletoe spread, impact, and control. In Dwarf mistletoe control through forest management. p. 16-30. R. F. Scharpf and J. R. Parmeter, Jr., eds. USDA For. Serv., Pac. Southwest For. and Range Exp. Stn., Berkeley, Calif.

- Parmeter, J. R., Jr., and B. Uhrenholdt.  
1975. Some effects of pine-needle or grass smoke on fungi. *Phytopathology* 65(1): 28-31.
- Parsons, David J.  
1976. The role of fire in natural communities: an example from the southern Sierra Nevada, California. *Environ. Conserv.* 3:91-99.
- Parsons, David J.  
1977. Preservation in fire-type ecosystems. In *Environmental consequences of fire and fuel management in Mediterranean ecosystems*. p. 172-182. H. A. Mooney and C. E. Conrad, eds. USDA For. Serv. Gen. Tech. Rep. WO-3. Washington, D.C.
- Parsons, David J.  
1978. Fire and fuel accumulation in a giant sequoia forest. *J. For.* 76:104-105.
- Parsons, David J., and S. H. DeBenedettii.  
1979. Impact of fire suppression on a mixed-conifer forest. *For. Ecol. Manage. Manage.* 2:21-33.
- Pfister, Robert D., B. L. Kovalchik, S. F. Arno, and R. C. Presby.  
1977. Forest habitat types of Montana. USDA For. Serv. Gen. Tech. Rep. INT-34, 174 p. Intermt. For. and Range Exp. Stn., Ogden, Utah.
- Philpot, Charles W.  
1977. Vegetative features as determinants of fire frequency and intensity. In *Environmental consequences of fire and fuel management in Mediterranean ecosystems*. p. 12-16. H. A. Mooney and C. E. Conrad, eds. USDA For. Serv. Gen. Tech Rep. WO-3. Washington, D.C.
- Roe, Arthur L., and G. D. Amman.  
1970. The mountain pine beetle in lodgepole pine forest. USDA For. Serv. Res. Pap. INT-71, 23 p. Intermt. For. and Range Exp. Stn., Ogden, Utah.
- Romme, William H.  
1977. Vegetation in relation to elevation, topography, and fire history in a Wyoming montane watershed. M.S. thesis. Univ. Wyo., Laramie.
- Roth, L. F.  
1974. Resistance of ponderosa pine to dwarf mistletoe. *Silvae Genet.* 23:116-120.
- Roughton, Robert D.  
1972. Shrub age structure on a mule deer winter range in Colorado. *Ecology* 53(4): 615-625.
- Sando, Rodney W.  
1978. Natural fire regimes and fire management--foundations for direction. *West. Wildlands.* 4(4):34-42.
- Sauer, Jonathan D.  
1977. Fire history, environmental patterns, and species patterns in Santa Monica Mountain chaparral. In *Environmental consequences of fire and fuel management in Mediterranean ecosystems*. p. 383-386. H. A. Mooney and C. E. Conrad, eds. USDA For. Serv. Gen. Tech. Rep. WO-3. Washington, D.C.
- Schmidt, R. L.  
1960. Factors controlling the distribution of Douglas-fir in coastal British Columbia. *Q. J. For.* 54:156-160.
- Schmidt, R. L.  
1970. A history of presettlement fires on Vancouver Island as determined from Douglas-fir ages. In *Tree-ring analysis with special reference to North America*. p. 107-108. J. Harry G. Smith and John Worrall, eds. Univ. British Columbia Fac. For. Bull. 7.
- Show, S. B., and E. I. Kotok.  
1924. The role of fire in the California pine forest. U.S. Dep. Agric. Bull. 1294, 80 p.
- Smith, J. M.  
1974. *Models in ecology*. Cambridge Univ. Press, Cambridge, Mass. 146 p.
- Sneck, Kathleen M. Davis.  
1977. The fire history of Coram Experimental Forest. M.S. thesis. Univ. Mont., Missoula.

- Soeriaatmadja, R. E.  
1966. Fire history of the ponderosa pine forest of the Warm Springs Indian Reservation, Oregon. Ph.D. thesis. Oreg. State Univ., Corvallis. [Diss. Abstr. Int. 27:2612-B.]
- Stone, E. C., R. F. Grah, and P. J. Zinke.  
1972. Preservation of the primeval redwoods in the Redwood National Park, part I. Am. For. 78:50-55.
- Sudworth, G. B.  
1908. Forest trees of the Pacific Slope. USDA For. Serv
- Tande, Gerald F.  
1977. Forest fire history around Jasper Townsite, Jasper National Park, Alberta. M.S. thesis. Univ. Alberta, Edmonton.
- Taylor, Dale L.  
1973. Some ecological implications of forest fire control in Yellowstone National Park, Wyoming. Ecology 54(6):1394-1396.
- Van Wagner, C. E.  
1965. Describing forest fires--old ways and new. For. Chron. 41(3):301-305.
- Van Wagner, C. E.  
1977. Conditions for the start and spread of crown fire. Can. J. For. Res. 7:23-34.
- Van Wagner, C. E.  
1978. Age-class distribution and the forest fire cycle. Can. J. For. Res. 8:220-227.
- Van Wagner, C. E., and I. R. Methven.  
1978. Discussion: two recent articles on fire ecology. Can. J. For. Res. 8:491-492.
- van Wagtendonk, Jan Willem.  
1972. Fire and fuel relationships in mixed conifer ecosystems of Yosemite National Park. Ph.D. thesis. Univ. Calif., Berkeley.
- van Wagtendonk, Jan W.  
1974. Refined burning prescriptions for Yosemite National Park. U.S. Dep. Interior, Natl. Park Serv. Occas. Pap. 2, 21 p.
- van Wagtendonk, Jan W.  
1977. Fire management in the Yosemite mixed-conifer ecosystem. In Environmental consequences of fire and fuel management in Mediterranean ecosystems. p. 459-463. H. A. Mooney and C. E. Conrad, eds. USDA For. Serv. Gen. Tech. Rep. WO-3. Washington, D.C.
- Vogl, Richard J.  
1970. Fire and plant succession. In Role of fire in the Intermountain West. p. 65-75. Intermt. Fire Res. Council., Missoula, Mont.
- Wagener, Willis W.  
1961. Past fire incidence in Sierra Nevada forest. J. For. 59(10):739-748.
- Weaver, Harold.  
1943. Fire as an ecological and silvicultural factor in the ponderosa pine region of the Pacific slope. J. For. 41:7-15.
- Weaver, Harold.  
1947. Fire--nature's thinning agent in ponderosa pine stands. J. For. 45:437-444.
- Weaver, Harold.  
1951. Fire as an ecological factor in southwestern ponderosa pine forests. J. For. 49:93-98.
- Weaver, Harold.  
1955. Fire as an enemy, friend and tool in forest management. J. For. 53:499-504.
- Weaver, Harold.  
1959. Ecological changes in the ponderosa pine forest of Cedar Valley in southern Washington. J. For. 57:12-20.
- Weaver, Harold.  
1967. Fire and its relationship to ponderosa pine. In Proc. Tall Timbers Fire Ecol. Conf. 7:127-149.



- Weaver, Harold.  
1974. Effects of fire on temperate forests: western United States. In Fire and ecosystems. p. 279-319. T. T. Kozlowski and C. E. Ahlgren, eds. Academic Press, New York.
- Wellner, Charles A.  
1970. Fire history in the northern Rocky Mountains. In Role of fire in the Intermountain West. p. 42-64. Intermt. Fire Res. Counc., Missoula, Mont.
- Westman, Walter E.  
1978. Measuring the inertia and resilience of ecosystems. Bioscience 28(11): 705-710.
- Wicker, Ed F., and Charles D. Leaphart.  
1976. Fire and dwarf mistletoe (Arceuthobium spp.) relationships in the northern Rocky Mountains. In Proc. Tall Timber Fire Ecol. Conf. 14:279-298.
- Whittaker, R. H.  
1975. The design and stability of plant communities. In Unifying concepts in ecology. p. 169-181. W. H. van Dobben and R. H. Lowe-McConnell, eds. Junk, The Hague.
- Wright, Henry A.  
1978. The effect of fire on vegetation in ponderosa pine forests, a state-of-the-art review. Texas Tech. Univ. Range and Wildl. Inf. Ser. 2, 21.
- Wright, Henry A., L. F. Neuenschwander, and C. M. Britton.  
1979. The role and use of fire in sagebrush-grass and pinyon-juniper plant communities, a state-of-the-art review. USDA For. Serv. Gen. Tech. Rep. INT-58, 48 p. Intermt. For. and Range Exp. Stn., Ogden, Utah.
- Young, James A., and Raymond A. Evans.  
1978. Population dynamics after wildfires in sagebrush grasslands. J. Range Manage. 31(4):283-289.