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Perspectives On The Ecology And Silviculture Of Oak-Dominated Forests In The Central And Eastern States

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Perspectives On The Ecology And Silviculture Of Oak-Dominated Forests In The Central And Eastern States

Paul S. Johnson

A harmony between ourselves and nature depends on—indeed, requires—modern technological tools to teach us about the Earth and to help us manage wisely what we realize we have inadvertently begun to unravel.
D. B. Botkin, *Discordant Harmonies (1990).*

Oak (*Quercus*) forests cover an extensive area in the Central and Eastern States. Much of that area is encompassed by the oak-hickory type, which itself accounts for about 114 million acres of oak forest (Sander *et al.* 1983). Included are over 75 tree species, more than 230 species of birds and mammals, and numerous other plants and animals. Oaks also are often a significant component of several associated ecoregions including the northern hardwood, mixed mesophytic, lowland hardwood, and southern pine forests (Braun 1972).

Current directions and strategies for managing publicly owned oak forests are under scrutiny because of competing social and economic agenda. Socially responsible forest management is further complicated by ecological forces that are shrinking the area of oak forests. Moreover, the relative importance of oaks is decreasing within the forests that remain. Oak forests are changing ecologically because of widespread successional displacement of oaks by more shade tolerant species, the absence of fire, and the increased mortality of oaks caused by gypsy moth (*Lymantria dispar* L.) defoliation, and oak dieback and decline. Urban expansion, road construction, conversion of oak forests to pasture and cropland, and accelerated harvesting of oak stands have produced additional losses. The public and in some cases land managers may be unaware of the nature and extent of these losses, especially those induced by long-term ecological changes.

In my view, the rational management and resolution of conflict over the management and use of oak forests will require that we first understand oak ecology. Our knowledge should include an awareness of past human influence on the nature, occurrence, and extent of oak forests, and how that contrasts with their present-day influence. A comprehensive understanding of oak ecology will be needed to provide the foundation for a mix of effective management practices that are ecologically sustainable and compatible with current social and economic values. The objective of this paper is to discuss silvicultural options and opportunities for attaining those objectives from an ecological perspective.

ECOLOGICAL BACKGROUND

Oaks, Fire, and Humans

Depending on locale, the history of uncontrolled burning of the forests in Central and Eastern States ended less than a century ago (the Missouri Ozarks) to several hundred years ago (New England). Before that period, forests throughout the region were frequently burned by Native

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Americans (Beilmann and Brenner 1951, Cooper 1961, Curtis 1959, Day 1953). In southeastern Minnesota, recurring fires shaped the landscape for several thousand years before the arrival of Europeans (Grimm 1984). Although not every acre burned every year, the historical and scientific evidence indicates that the frequency, intensity, and extent of those fires had an enormous impact on the region's ecology (Abrams 1992).

In the prairie-forest transition zone of Minnesota, Wisconsin, and Iowa, wildfires were controlled soon after the area was settled. There, fire was perceived as a threat to life and property during the settlement period of 1860 to 1890 and the settlers responded accordingly. Pammell (1928, p. 13) described the landscape, wildfires, and prevailing attitude about fire in the LaCrosse, Wisconsin, area during the early settlement period of 1870 to 1880:

> "I shall never forget the numerous prairie fires which were annually started in the region of La Crosse. Dozens of these fires could be seen from our house for a number of weeks during the spring season. There were long lines of fire up the hills and down, and these of course, destroyed everything in their path. Consequently, many of the bluffs were not covered with trees, and they were prairies on hillsides, largely due to the fact that these annual fires destroyed the young tree growth. Later when the [human] communities became more stable, these fires did not occur so frequently. It was often a man's job to put out the fires, and I recall many of these that I helped put out on our farm...."

Eventually, wildfires in that area were effectively controlled and the regional ecology was irrevocably affected. Today, the landscape of the LaCrosse area is a mosaic of cultivated fields and pastures in the valley bottoms and flat uplands. Closed-canopy oak forests occupy the intervening slopes. The exception occurs on steep south and southwest slopes where native prairie persists as a witness to the past (Curtis 1959).

Other areas of the central and eastern region experienced a similar history. Beilmann and Brenner (1951) described the "intrusion" of forests into the Ozark Plateau as a phenomenon of the late 19th century. Houck (1908, p. 26) cited Father Vivier's description of the area in 1750:

> "...trees are almost as thinly scattered as in our public promenades. This is partly due to the fact that the savages set fire to the prairies toward the end of autumn, when the grass is dry; the fire spreads everywhere and destroys most of the young trees."

More recent studies also link humans to fire in the Missouri Ozarks. Based on tree ring analysis of a post oak savanna, Guyette and Cutter (1991) found that fire frequency and extent were greatest between 1700 and 1810, a time when the mean fire-free period was 4.3 years. Fires occurred most frequently between 1785 and 1810, which was a period of influx of Native Americans and exploration by Europeans. This period was followed by 40 years of few fires beginning with an exodus of Native Americans from the area about 1815 and lasting until about 1860. A similar fire history characterized a nearby eastern redcedar (*Juniperus virginiana* L.) glade, where the mean fire-free interval was 3.2 years before 1879 and thereafter was 22 years (Guyette and McGinnes 1982). The frequency and extent of presettlement fires in Missouri are further substantiated by oak savannas, which covered an estimated one-third of the State or 13 million acres (Nelson 1985).

By 1898, when many Ozark oak savannas were beginning to form closed-canopy forests, there was growing concern over the timber supply because of its utilization by an expanding railroad system. Reacting to this misplaced concern, Shepard (1898, cited in Beilmann and Brenner) pointed out the "mistaken idea that the supply of timber in this region is decreasing; on the contrary it is largely increasing... It has probably increased one-third within the last forty years."

It is probably not too surprising that fire had a great influence on shaping the vegetation of the relatively dry prairie-forest transition zone.

Perhaps more surprising is the repeated burning of oak forests in the more humid regions to the east (Day 1953, Komarek 1974, Little 1974, Lorimer 1989). The presettlement and early settlement history of the Central and Eastern States thus contradicts the widely held perception of a primeval landscape largely unaffected by humans. The abundance of oak today is consequently an artifact of a disturbance regime that no longer exists. From a historical perspective, it may be more accurate to regard the absence of fire as unnatural rather than natural. But just as burning had its social and ecological consequences, the permanent *cessation* of burning also produced its consequences. For oak, the cessation of burning was not an entirely positive event.

The Regeneration Ecology of Oaks

The more than 30 species of oaks in the region (Little 1979) occur in diverse ecosystems ranging from xeric to hydric (Braun 1972). In xeric ecosystems, oak reproduction tends to accumulate or build up in the understory of the parent stand. Accumulation may occur over several decades, resulting in populations of seedlings and seedling sprouts that originated from several sporadic acorn crops. Seedling sprouts are seedlings whose shoots have died back one or more times. Oak reproduction survives dieback by resprouting from dormant buds near the root collar. Dieback may extend partially or all the way to the root collar. At the root collar there is a concentration of dormant buds capable of producing new sprouts. These are released from dormancy by the death of the terminal buds. Although the physiological mechanisms causing dieback in oak reproduction are poorly understood, several factors can be involved including water stress, insects, pathogens, and fire (Kramer and Kozlowski 1979, Linit *et al.* 1986, Olson and Boyce 1971, Robbins and Croghan 1984).

Whatever its cause, recurrent shoot dieback of oak reproduction appears to be important in the adaptive strategy of oaks, especially among the more xeromorphic species. Seedlings of those species have inherently slow shoot growth, and attaining a competitive rate of growth depends on the development of a large root:shoot ratio and large root mass. Potential shoot growth of oak reproduction (stems less than 2 inches d.b.h.) increases as root mass increases. This relation is reflected in the positive correlation between the basal diameter of reproduction (a correlate of root mass) and the annual shoot growth of oak reproduction after clearcutting (Sander 1971). Shoot growth of sprouts originating from the stumps of cut overstory trees also increases with increasing stump diameter (Johnson 1979). However, growth peaks at some diameter, usually at about 6 to 8 inches depending on species, and then declines. All living oaks from seedlings to mature trees thus have some potential to contribute to the regenerative capacity of a stand.

Oak reproduction grows little in height under the prevailingly low light conditions of the understory (Hanson *et al.* 1987, Johnston 1941). Although shading promotes shoot and leaf growth at the expense of root growth (Kolb and Steiner 1990), recurrent shoot dieback beneath the overstory eventually results in reproduction with large root:shoot ratios. In that environment, the investment in root is hypothetically maximized when stem mass is small, which in turn minimizes the effect of the stem as a carbon sink and source of respiration. This investment in root ultimately facilitates rapid shoot growth (Dickson 1991, Johnson 1979).

Shoot growth of oak reproduction occurs in distinct flushes followed by periods of rest (Cobb *et al.* 1985, Dickson 1991, Johnson 1979, Reich *et al.* 1980, Romberger 1963). Under ideal environmental conditions such as those occurring in the nursery and greenhouse, oak seedlings can produce several flushes in a single growing season (Dickson 1991, Johnson *et al.* 1984). However, under the stresses typical of field conditions, multiple flushing usually occurs only if root systems are large and the reproduction is growing in full or nearly full light. Consequently, rapid shoot growth seldom occurs under field conditions unless the overstory is destroyed or substantially reduced in density (Johnson 1979, Johnston 1941). Such events can result from fire, windthrow, insect- and disease-related mortality and defoliation, drought, and timber harvesting.

The accumulation of oak reproduction under the parent stand is one of the most important aspects of the regeneration ecology of oaks. Oak silviculturists call this "advance reproduction" because, in the even-aged management of oaks, it is present in advance of final harvest. It also largely determines the composition of the next stand. Oaks capitalize on this accumulation process because it facilitates the quick capture of growing space when the overstory is removed or destroyed. This capacity largely depends on the accumulated population of seedling sprouts with large roots. True seedlings and small seedling sprouts usually have root systems that are too small to support rapid shoot growth (Sander 1971). And stump sprouts from overstory trees, by themselves, are usually not numerous enough to capture all of the available growing space (Sander *et al.* 1984).

Nevertheless, it is the number, size, and spatial distribution of all three classes of reproduction that express the total oak "regeneration potential" of a stand (Sander *et al.* 1984). Because this potential is encoded in the advance reproduction and the parent stand itself, the new stand is essentially a "legacy" (*sensu* Franklin *et al.* 1989) of the parent stand. Sustaining oak-dominated forests thus depends on perpetuating preestablished reproduction (propagules) from one generation to the next. Whether overstory destruction occurs in small gaps or over large areas, the resulting spatial units of reproduction are even aged[1]. Although oaks generally are not recognized as gap species (Crow 1988, Ehrenfeld 1980), they have the potential to exploit small gaps if advance reproduction of sufficient size is present. For example, 10-foot-tall advance reproduction of northern red oak captured canopy gaps as small as one-twenty-fifth acre in a mixed sugar maple-oak stand in southwestern Wisconsin (Lorimer 1983).

Recurrent fire promotes the accumulation of oak reproduction in at least three ways: (1) by eliminating or reducing the number of fire-sensitive understory competitors, including shrubs and shade-tolerant trees, (2) by reducing overstory density by killing trees with thin, fire-sensitive bark, and (3) by killing the stems of oak reproduction, which increases the root:shoot ratio of those that survive by resprouting. The first two factors increase light on the forest floor, which in turn creates conditions more conducive to the accumulation of oak reproduction. Although poorly adapted to surviving under shade, oak reproduction is well adapted to surviving fire because of the concentration of dormant buds near the root collar. These buds often remain an inch or more below the soil surface where they are protected from lethally high temperatures (Korstian 1927).

When fires are frequent and intense, oak savannas may result. These are communities comprised of a few large, thick-barked, fire-resistant overstory trees per acre, a unique and diverse grass and forb flora, and abundant oak reproduction largely comprised of seedling sprouts (Curtis 1959, Haney and Apfelbaum 1990). Oak savannas, which once dominated the landscape throughout much of the Central and Eastern States are today rare plant communities (Haney and Apfelbaum 1990). Most became closed-canopy oak forests within a few decades after burning and grazing ceased (Curtis 1959).

Not all oak-dominated ecosystems require fire or disturbance for their sustainment. Many xeric oak forests appear to be relatively stable communities that show little evidence of succession to more shade tolerant or mesophytic species. Probably the largest North American ecosystem of this type occurs in the Ozark Plateau in Missouri where oak reproduction may accumulate in the understory for 50 years or longer[2]. However, even in this ecosystem, the capacity of oak reproduction to accumulate varies with topographic features such as slope position and aspect, which collectively control light, heat, and soil moisture (Sander *et al.* 1984). Oak-dominated ecosystems characterized by such "auto-accumulating" reproduction also occur on the drier sites throughout the central and eastern region including the Ohio Valley (Minckler and

[1]*However, if their age structure were based on age from germination ("age from birth"), they could be uneven aged, depending on the duration of the reproduction accumulation period.*

[2]*Data on file, North Central Forest Experiment Station, Columbia, MO.*

Woerheide 1965), the Appalachians (Ross *et al.* 1986), and the Lake States (Johnson 1966). Although we have not yet formalized the definition of an auto-accumulating ecosystem, four or more cohorts of oak reproduction representing at least two decades of accumulation are usually present.

Oak-dominated ecosystems that are auto-accumulators of oak reproduction are thus often self-sustaining. However, not all oak forests are auto-accumulators and not all auto-accumulators are capable of self-renewal following disturbance (Johnson 1992, Loftis 1990a). Moreover, auto-accumulators do not necessarily accumulate large numbers of oak reproduction. For example, about 1,200 seedlings and seedling sprouts per acre typify Ozark oak stands (Sander 1979b). Only 150 to 300 new oak seedlings per acre may become established even after a good acorn crop (Sander 1979b, McQuilkin 1983). In contrast, 138,000 Nuttall oak (*Quercus nuttallii* Palmer) seedlings per acre were observed after a single bumper acorn crop in a hydric non-accumulator ecosystem in Louisiana (Johnson 1975). Thus, large numbers of seedlings by themselves do not constitute an auto-accumulator ecosystem. These anomalies emphasize the importance of distinguishing among different kinds of oak-dominated ecosystems, as discussed later.

Oak reproduction typically does not accumulate over successive acorn crops on mesic or hydric sites in the absence of disturbance (Carvell and Tryon 1961, Johnson 1975, Will-Wolf 1991). Nevertheless, reported oak reproduction densities in these ecosystems exceeded 50,000 seedlings per acre in a mesic forest (Tryon and Carvell 1958) and 100,000 per acre in a bottomland forest (Johnson 1975). At times, there may be little or no oak reproduction in these ecosystems (Sander 1983). When oak seedlings do occur, they typically represent one or two acorn crops. Because of low survival rates, seedlings from a single cohort may die before the next good acorn crop occurs. The rapid rate of seedling disappearance results largely from the shade-intolerance of oak reproduction and the low light levels on the forest floor of such ecosystems (Hanson *et al.* 1987). In the absence of disturbance, these ecosystems typically possess high overstory basal areas and a dense subcanopy of shade-tolerant non-oaks (Braun 1972, Loftis 1990b). Such vertical stratification, together with high total stand density, occurs in mesic oak forests throughout the central and eastern regions. The oaks are thus eventually replaced by more shade tolerant species. Timber harvesting, regardless of method and intensity, accelerates the replacement process (Abrams and Nowacki 1992).

Micro-sites where oak seedlings become initially established in large numbers may not be where they ultimately survive to form large root systems (Harrison and Werner 1984, Johnson 1966). Although cool moist sites such as those on northeast-facing slopes may favor initial establishment, the more favorable places for long-term survival, root development, and thus accumulation may occur on the more southerly or neutral aspects (Sander *et al.* 1984, Walters 1990, Carvell and Tryon 1961).

There is a general relation between site quality and regeneration success: the better the site the more difficult it is to regenerate oaks (Arend and Scholz 1969, Loftis 1990b, Lorimer 1989, Trimble 1973). Obtaining the accumulation of oak reproduction necessary for successful regeneration on highly productive sites requires recurrent disturbance. Historically, fire created the necessary conditions. Thus, one potential solution to sustaining oak-dominated forests on productive sites is prescribed burning. Based on a review of research results on prescribed burning in eastern hardwoods and southern mixed pine-hardwood stands, Van Lear and Waldrop (1988) concluded that fire, if correctly employed, has the potential to accomplish that objective.

Although the overall area of oak forests is shrinking, numerous localized expansions also are occurring through dispersal of acorns by birds and mammals from oak-dominated communities to other plant communities (Johnson and Adkisson 1986). The observed aggressiveness of the heavy-seeded oak in invading open areas (*e.g.*, Jokela and Sawtelle 1985) thus may be largely related to dispersal by animals. Blue jays (*Cyanocitta cristata*) are especially effective at dispersing acorns from oak forests to open or semi-open environments such as old fields and forest edges, which are favorable for acorn germination and seedling growth (Johnson and Webb

1989). Dispersal may extend up to 2.5 miles, and individual jays may cache in the ground up to 3,000 acorns per year (Johnson and Adkisson 1986). Although only a small proportion of cached acorns survive to germinate and grow, the large number that are cached and the favorable environments they are dispersed to collectively create an effective mechanism for the expansion of oak forests. This mechanism has been proposed as an explanation for the rapid migration of oaks as the climate warmed during early postglacial times (Johnson and Webb 1989). Consequently, acorn dispersal by jays may provide for the maintenance of biodiversity during periods of global warming such as that which may be occurring today (Johnson and Webb 1989) and in landscapes of fragmented oak forests that are common to many areas of the Midwest and East today (Johnson and Adkisson 1985). Florida scrub jays (*Aphelocoma coerulescens coerulescens*) similarly disperse and ground-cache acorns in even greater numbers than those observed for blue jays, but over shorter distances (DeGange *et al.* 1989).

OAK SILVICULTURE: AN ECOLOGICAL PERSPECTIVE

Ecological Classification

The objective of ecological classification is to define vegetation-site (habitat) units within which there is great ecological similarity. Accordingly, all forest communities representing a given unit respond similarly to natural forces and processes and thus to management practices. This does not mean that stands of similar overstory composition and structure are necessarily "ecologically equivalent" units. This feature distinguishes ecological classification from traditional forest cover-type descriptions (*e.g.*, Eyre 1980).

The floristic habitat classification system for northern Wisconsin developed by Kotar and others (1988) is an example of a system that, for each habitat type, integrates basic ecological relationships with management implications. It is based on the assumption that specific, recurrent combinations of understory plants and their percentage cover can be used to define and differentiate among different kinds (classes) of ecosystems. Each defined combination exhibits a unique fidelity to its "system" across a wide range of seral (successional) states. The resulting ecological units also are associated with certain soils, landforms, and a presumed "climax" overstory formation with specified successional pathways.

The defined ecological units thus can be used to distinguish among oak-dominated ecosystems that may be superficially similar but ecologically very different. For example, among ecosystems characterized as auto-accumulators, there may be significant differences in the rate of reproduction accumulation and in the specific mechanisms or factors controlling the accumulation process. Recognizing such differences is fundamental to understanding and managing oak forests. The classification system of Kotar and others (1988) considers such relations. It implicitly specifies the relative ease or difficulty in sustaining oaks as an overstory component in the ecological units (habitat types) in which oaks commonly occur. Ecological classification also can be used to specify other important features characteristic of each defined ecological unit. These include productivity, characteristic overstory diameter distributions, value to wildlife, soils, and nutrient cycling relations (Cleland *et al.* 1985, De Geus 1986, Kotar 1991, Zak and Pregitzer 1990). A comprehensive overview of ecological forest site classification is presented by Barnes and others (1982).

The development and application of ecological classification systems will be essential to enlightened management of oak-dominated forests. We have struggled too long in our management of oak forests without objective definitions of the different kinds of ecosystems in which oaks occur. Ecological classification has the potential to be an effective silvicultural tool.

The History and Application of Clearcutting in Oak Forests

In the 1960's, even-aged management became the *modus operandi* for much of the central and eastern hardwood region. It was enthusiastically accepted for several reasons. It met the ecological requirements of the commercially valuable, shade-intolerant species, including the oaks, white ash (*Fraxinus americana* L.), black cherry

(*Prunus serotina* Ehrh.), and yellow-poplar (*Liriodendron tulipifera* L.). Even-aged management also was economically efficient. Logging, road building, and administrative costs were minimized because large timber volumes could be harvested from relatively small areas. Moreover, the system was simple to implement, in contrast to the relatively complicated timber marking rules used to create the diameter distributions needed in uneven-aged management (Roach 1963, Smith and Lamson 1982, Trimble *et al.* 1974). There also was growing uncertainty about the ability of the commercially valuable intolerant species, including the oaks, to sustain the age distributions required in uneven-aged management (Roach 1963, Schlesinger 1976). Other factors that contributed to the acceptance of clearcutting included its endorsement by wildlife managers, the development of even-aged stocking guides (*e.g.*, Gingrich 1967), an established scientific basis for its application (Roach and Gingrich 1968), and its utility in regulating the distribution of age classes of stands. The many advantages of clearcutting upland hardwood forests have been further discussed by McGee (1987).

Clearcutting became the most widely recommended and applied regeneration method, albeit with mixed success in the management of oak forests. It was most successfully applied in the drier ecosystems where oak reproduction naturally accumulated in the understory. Such forests occurred in the Missouri Ozarks and the oak-hickory forests of the Ohio Valley and Appalachians (Sander 1977, Ross *et al.* 1986). There, as elsewhere, the success of clearcutting hinged on the presence of *adequate* oak advance reproduction (Sander 1971, Sander and Clark 1971).

For some oak-dominated ecosystems such as the Ozark Plateau, methods were developed for determining the adequacy of the oak regeneration potential of stands for meeting timber management objectives (Sander *et al.* 1984). Adequacy, in turn, was judged by the projected future utilization of growing space of oaks that were likely to attain dominance or codominance after final harvest. These projections were based on models for predicting the future stand from an inventory of the overstory (the source of stump sprouts), and the advance reproduction. Their primary purpose was to facilitate decisions on when and how timber harvesting and regeneration were to occur.

Clearcutting was less successful in regenerating oaks in the more mesic ecosystems of the Ohio Valley, Appalachians, and elsewhere (Beck and Hooper 1986, Gammon *et al.* 1960, Johnson 1976, Loftis 1983b). Failures were largely related to the inability of oak reproduction to accumulate in the heavily shaded understories of these forests. But even when clearcutting failed to regenerate oaks, other commercially valuable species often replaced the oaks. In many mesic ecosystems, clearcutting accelerated the succession of oak-dominated forests to mixed-mesophytic forests.

In the first two decades of clearcutting (1960's and 1970's), forest managers were not greatly concerned about a diminishing oak resource even though many were aware of the problem. Apathy was reinforced by the economic acceptability of the species that usually replaced oaks in mesic ecosystems. Although there was some concern over increases in the proportion of the less valuable shade-tolerant hardwoods such as sugar maple (*Acer saccharum* Marsh.), red maple (*A. rubrum* L.), and American beech (*Fagus grandifolia* Ehrh.), clearcutting usually provided an acceptable mix of species even though the oak component was greatly reduced (Beck and Hooper 1986, Johnson 1976, Loftis 1983b). But that apathy diminished during the last decade for two reasons. First, in the late 1970's, there was a large increase in the demand and value of oak logs (Hoover 1985). This spurred management specifically *for* oaks and a concern for the economic losses that accompanied a shrinking oak resource. Second, the rise of environmental activism drew public attention to the perceived negative consequences of clearcutting. These perceptions included decreased aesthetic value, biodiversity, old growth, and certain wildlife values along with increased forest fragmentation.

To counter concerns about the possible adverse impacts of clearcutting oak forests, various modifications of the method were advocated. These included reducing the size of clearcuts; leaving snags, cull trees, and uncut islands of trees; deferring the removal of non-commercial

residual trees; shaping cuts to fit more aesthetically into the landscape; and leaving uncut strips where clearcuts bordered roads, lakes, streams, and other sensitive areas (Evans and Conner 1979; Smith *et al.* 1989; USDA Forest Service 1973, 1980). Although all of these methods were compatible with attaining conventional oak regeneration objectives, they also were consistent with other management objectives such as improving wildlife habitat for some species and enhancing landscape aesthetics and biodiversity (Perry *et al.* 1990). Nevertheless, the use of clearcutting on public lands, especially the National Forests, has further declined in favor of systems that focus less on producing commodities and more on preserving biodiversity and other values (Gale and Cordray 1991, Hansen *et al.* 1991, Kessler 1991, Perry and Maghembe 1989, Salwasser 1990).

Alternatives to Clearcutting

The shelterwood method

The decline of clearcutting required options. One option was the shelterwood method, which frequently has been advocated for oaks (Beck 1991, Hannah 1987, Korstian 1927, Sander 1979a, Scholz 1952, Smith 1986). The method is potentially suited to regenerating oaks in the troublesome mesic ecosystems because it focuses on controlling stand density near the end of the rotation—a critical period for reinforcing the accumulation of oak reproduction.

Recent research in the southern Appalachians demonstrated the potential of the shelterwood method to regenerate northern red oak on highly productive sites (site index 70 to 90 feet at 50 years) (Loftis 1983b, 1990a, 1990b). The resulting prescription requires a single shelterwood cut that reduces stand basal area by 30 to 40 percent of full stocking. Stand density is reduced by removing the subcanopy of tolerant non-oaks using herbicides and leaving the main canopy intact. This reduces the establishment and growth of yellow-poplar advance reproduction, the primary competitor of red oak on these sites. Recommendations are to remove the shelterwood about 10 years after treatment. This two-step method increases the proportion of oak reproduction with sufficient growth potential to compete with yellow-poplar and other fast-growing associates after the shelterwood is removed. However, the method is only successful in obtaining low to moderate numbers of dominant and codominant oaks (*e.g.*, 5 to 20 per acre) in the new stand. Although such numbers may not restore the oak component to its previous level of importance, the method is nevertheless potentially useful in sustaining oak as an overstory component.

The shelterwood method was more effective in regenerating northern red oak in a non-accumulator ecosystem (red oak site index 60-70) in the Driftless Area of southwestern Wisconsin (Johnson *et al.* 1989). Based on two case histories, reduction in overstory density to 70 to 80 feet square of basal area per acre 10 years before shelterwood removal resulted in 200 to 400 dominant and codominant red oaks per acre during the second decade after shelterwood removal. However, early results from a study on a similar site in southeastern Minnesota suggested that the method may be more effective when reduction in overstory density coincides with a good acorn crop and competition is reduced by soil scarification (Bundy 1991). Application of the shelterwood method in mesic oak forests in southern Michigan (Hill and Dickmann 1988) and southern Wisconsin (Lorimer 1989) also produced encouraging results.

The minimum duration of the shelterwood period, *i.e.*, the length of time between the shelterwood cut and shelterwood removal, is uncertain. Results from the Driftless Area suggest that 1 or 2 years may be sufficient if competition is controlled. Results from one case history in southwestern Wisconsin even indicate that northern red oak can be successfully regenerated with little or no oak advance reproduction provided that the final harvest occurs during the dormant season following a good acorn crop (Johnson *et al.* 1989). Although that result challenges the axiom that advance reproduction is required, the safest approach nevertheless would be to obtain adequate advance oak reproduction before the seed source is removed in the final harvest.

The contrasting experiences in the southern Appalachians, the Driftless Area, and elsewhere emphasize the need for shelterwood prescriptions that are both species- and ecosystem-specific.

Xeromorphic oaks such as black oak (*Quercus velutina* Lam.), white oak (*Q. alba* L.), scarlet oak (*Q. coccinea* Muenchh.), and chestnut oak (*Q. prinus* L.) growing in xeric ecosystems may require two or more decades in the understory before they accumulate the root mass necessary for competitive shoot growth after overstory removal (Sander 1979a, Sander and Clark 1971). However, because these species typically auto-accumulate in the xeric ecosystems in which they typically occur, use of the shelterwood method usually will be unnecessary there (Sander and Clark 1971). In contrast, experiences with regenerating northern red oak in the Driftless Area indicate that a shorter establishment period is sufficient. The shelterwood system nevertheless provides flexibility in the timing of residual-tree removal. When consistent with successional processes and management objectives, the shelterwood can even be maintained until a two-aged stand, or "irregular" shelterwood develops (Beck 1991, Smith 1986).

In mesic and hydric ecosystems, application of the shelterwood method can lengthen the accumulation period beyond that which would normally occur. To be effective, this may require control of both overstory and understory density. But because of the less conservative and less root-dependent shoot growth of the reproduction of the oak species that normally grow on these sites, the shelterwood period may not need to be as long as that required in xeric forests. Unfortunately, there are few research results on this aspect of the shelterwood method, especially for bottomland oaks. The shelterwood system also can be used in combination with underplanting (Johnson *et al.* 1986, Johnson 1989). This method provides the opportunity to introduce genetically improved stock into stands. However, there is little theoretical evidence that application of the shelterwood system is dysgenic (Howe 1989). But analysis of population genetics, to date, has not specifically considered the peculiarities of the oak regeneration process and the possible interactions between that process and silviculturally induced genetic selection.

The potential for regenerating oaks using prescribed burning in combination with the shelterwood method has been demonstrated in principle (Nyland *et al.* 1982, Van Lear and Waldrop 1988). However, it may be unnecessary in many auto-accumulator ecosystems. Wherever it is used, its success will depend on matching the frequency, timing, and intensity of burns with other ecosystem characteristics. Based on a review of the literature, Van Lear and Waldrop (1988) concluded that frequent low-intensity back fires with low flame heights would be the most useful for building up oak advance reproduction under a shelterwood. Such burns do not damage the overstory or site. In contrast, high-intensity head fires are likely to cause excessive overstory mortality because the flames produced may reach into tree crowns. Such fires also may cause site damage. Single low-intensity burns are not likely to be effective (Johnson 1974, Nyland *et al.* 1982, Will-Wolf 1991).

The seed tree method

The seed tree method is an even-aged method that usually leaves 10 or fewer seed-producing trees on every acre (Smith 1986). However, it is not recommended for regenerating oaks because the seed trees provide too little reproduction too late, *i.e.*, after the final harvest. In the few reported cases where the method has been applied to oak forests, the seed trees contributed little to regeneration (DeBell *et al.* 1968, Johnson and Krinard 1983). Nevertheless, the method may be useful if sustaining acorn production on harvested areas is important. It can provide for substantial acorn production if good seed producers are retained. However, identifying the seed producers requires long-term records on seed production, which few forest managers are likely to have. The method nevertheless creates more structural diversity and visual appeal than a clearcut.

Sharp (1958) observed that fewer than 30 percent of white oaks produced acorns and many of those were poor producers. If this is typical of most oak species, leaving 10 good acorn producers per acre would retain 40 percent or more of the acorn producing capacity of the original stand. This assumes that there are about 75 oaks in the overstory at the end of the rotation. Moreover, under the open-grown conditions created by the seed tree method, seed tree crowns can expand to their maximum, thus increasing acorn production per tree. The

crowns of some seed trees may degenerate from crown dieback resulting from their sudden exposure (Smith 1986). However, the snags that ultimately develop could provide valuable habitat for cavity nesting birds and the standing dead wood essential to preserving biodiversity (Franklin *et al.* 1989, Hansen *et al.* 1991). Moreover, seed trees could be quickly converted to dead snags by girdling once it was determined they had little or no value for acorn production or other purposes.

The group selection method

The group selection method is an uneven-aged system of forest management potentially suitable for oaks. In this method, small regeneration openings are created in the main canopy. The openings provide the necessary light for intolerant species like the oaks. The method preserves some of the characteristics of a mature forest especially when single-tree selection is used to remove trees between the groups (Marquis and Johnson 1989). Recommended opening size ranges from one-tenth acre to one-half acre (Law and Lorimer 1989, Marquis 1989, Roach 1963). Larger openings should be referred to as patch cuts or clearcuts if the objective is to perpetuate these as even-aged units through successive rotations (Marquis 1989). As in clearcutting, the reproduction that develops in the openings depends on the advance reproduction present at the time the opening is created (Sander and Clark 1971).

A disadvantage of the group selection method where timber quality is important is the large ratio of perimeter to opening area that results from small canopy openings. For example, 10 one-third-acre circular openings (3.3 acres) create 3.2 times more perimeter (forest edge) than a single 3.3-acre circular clearcut. Forest-grown oaks are prone to develop epicormic branches when exposed to the high light intensities occurring along forest edges, which in turn, reduces their bole quality and value (Trimble and Seegrist 1973). However, light intensity along opening edges is a function of opening diameter and other factors (Fischer 1981, Minckler *et al.* 1973). Another disadvantage of the group selection method is that with time it becomes increasingly difficult to fit new openings among older ones (Marquis and Johnson 1989).

A variation of the group selection method has been dubbed the "group shelterwood" method (Coder *et al.* 1987). In this method, groups are removed where oak advance reproduction, either by design or chance, has adequately accumulated. Because adequate oak reproduction occurs irregularly in time and space, especially in non-accumulator ecosystems, the resulting groups are likely to occur irregularly. The method has potential application in sustaining timber production where other values including aesthetics are important and an even flow of wood products is not important. The method is compatible with preserving diversity and aesthetics near sensitive recreation, scenic, and other areas that are not regulated for timber production.

Based on theoretical considerations, Murphy and others (1990) concluded that the proper application of the group selection method is unlikely to produce dysgenic effects. Nevertheless, the method is fraught with the potential for such problems as discussed below for the single tree selection method.

The single tree selection method

The single tree selection method generally is considered inappropriate for managing oak forests (Sander and Clark 1971). Canopy openings the size of individual trees presumably provide inadequate light for the accumulation of oak reproduction. The method also encourages the development of a tolerant understory. The ecological and silvicultural literature generally substantiates these assertions, although virtually all the related studies were in mesic ecosystems where oak reproduction apparently did not auto-accumulate (Della-Bianca and Beck 1985, Schlesinger 1976, Trimble 1970, 1973).

The method has nonetheless been used successfully for 50 years on a large industrial forest in the Missouri Ozarks (personal communication C. Trammel). In applying the method, periodic timber removals are made across all diameter classes to obtain a specified diameter distribution. The diameter distribution of the forest now

approximates the classical reverse J-shaped distribution characteristic of an all-aged forest (personal communication J. Dwyer). The key to the success of this method lies in the intrinsic diameter distribution of the oaks in this ecosystem. The distribution is characterized by numerous small-diameter trees comprised largely of white oaks (Schnur 1937). In contrast, the smaller diameter trees in mesophytic oak forests are likely to be tolerant non-oaks; small diameter oaks are typically absent (Loftis 1983a). Thus, the relatively large pool of smaller overstory oaks plus the accumulated reproduction characteristic of the Ozark forest are potentially available for sustained recruitment into the main canopy. Moreover, individual-tree survival models show that the survival of lower story oaks in the uneven-aged Ozark forest is substantially greater than that for trees of the same size in an even-aged forest at the same overall stocking level (personal communication J. Dwyer). This suggests that the necessary diameter distribution and ingrowth can be sustained. The method deserves further study for application in this and similar auto-accumulator ecosystems.

Unfortunately, some foresters substitute diameter-limit cutting as a surrogate for the correct application of the single tree selection method. The former is regarded as a practice leading to dysgenic consequences (Smith 1986, Zobel and Talbert 1984). Howe (1989) proposed that dysgenic selection can be averted only if: (1) the defined age classes are narrow, (2) the oldest and largest trees within each age class are cut, (3) outstanding phenotypes in younger age classes are not cut, and (4) poor phenotypes in each age class are removed in each cutting cycle. He further emphasized that meeting the above standards essentially requires knowing the age of each tree, but questioned the practicality of determining such. Consequently, selection cutting should not be invoked in the name of biodiversity or other worthy ecological objectives if the actual application of the method, despite intentions, contravenes the long-term maintenance of healthy and productive ecosystems.

Oak-dominated forests nevertheless can be managed as even-aged or uneven-aged stands and regenerated by each of the conventional silvicultural methods. However, success in applying any one method will depend on understanding the oak regeneration processes intrinsic to the ecosystems in which it is applied. Our current inability to differentiate among the different kinds of oak-dominated ecosystems and the regeneration process inherent in each represents a major problem in managing and sustaining oak forests. Nevertheless, we have accumulated significant information and knowledge on how to manage many of these ecosystems for meeting a variety of traditional timber management objectives. We know much less about how to meet objectives associated with other values.

Restoring, Creating, and Sustaining Special Oak-dominated Communities

Managing oak-dominated forests potentially includes plant communities that until recently have not been widely considered in the practice of silviculture. These include oak savannas, mixed oak-conifer forests, oak forests for mast production, old-growth oak forests, and "aesthetic" oak forests. Moreover, the various ecological and anthropocentric values intrinsic to these communities are not necessarily mutually exclusive. For example, managing for oak savannas may be compatible with maximizing biodiversity and acorn production for wildlife while simultaneously producing a community that ranks high aesthetically because of its park-like appearance.

Oak savannas

Oak savannas once covered 27 to 32 million acres in the prairie-forest transition zone extending from Minnesota to Texas and eastward into Ohio (Nuzzo 1986). They also may have been widespread in the East in precolonial times (Day 1953). Today, midwestern oak savannas occupy only 0.02 percent of their estimated presettlement (pre-1840) area (Nuzzo 1986). In southeastern Minnesota and northeastern Iowa, extensive oak savannas and related fire-dependent communities existed for thousands of years before the arrival of Europeans (Chumbley *et al.* 1990, Grimm 1984). Wherever these and other fire-dependent communities occurred, they were associated with humans and their use of fire (Beilmann and Brenner 1951, Cooper 1961, Day 1953, Dorney 1981, Grimm 1984, Komarek 1974, Little 1974, Wilhelm 1972).

Although oak savannas have been variously defined, they are characterized by scattered trees largely consisting of oaks, a sparse shrub layer, and a ground layer rich in grasses and forbs (Haney and Apfelbaum 1990). Nuzzo (1986, p. 11) more specifically defined oak savannas as communities "...dominated by oaks having between 10 and 80 percent canopy, with or without a shrub layer, with an herbaceous, predominantly grassy ground layer composed of species associated with both prairie and forest communities..." Oak savannas thus possess characteristics of both prairies and forests and can occur in habitats ranging from xeric to hydric (Curtis 1959). In the Midwest, five classes of oak savannas have been tentatively defined based on their floristic composition, soils, and disturbance regime (Haney and Apfelbaum 1990). Virtually all of the oak species native to the Central and Eastern States can occur in savannas, although they are not all equally fire-resistant (Lorimer 1985).

In the absence of fire, oak savannas typically form closed canopy forests within two to three decades (Beilmann and Brenner 1951, Bray 1960, Curtis 1959, Grimm 1983, Nuzzo 1986). Many closed canopy forests of today thus may have originated from savannas. The demise of oak savannas was hastened by cultivation and grazing, by elimination of fire, and by building of roads and railroads, which formed effective fire barriers (Nuzzo 1986). In the absence of fire, the invasion of trees and shrubs results in a decline in the total number of plant and bird species that inhabit these communities. For example, the succession of tall grass savanna to a closed-canopy forest in northern Illinois reduced the number of plant species from more than 300 to less than 25 and bird species from 28 to 4 (Haney and Apfelbaum 1990). In the Ozark Plateau, 64 species of birds occur in a community described as "grass and shrub savanna" in contrast to 39 species in a mature oak-hickory forest with little undergrowth (Evans and Kirkman 1981). The great biodiversity and rarity of oak savannas, together with their dependence on human-caused fires, support the emerging land management philosophy that conserving biodiversity will require careful human intervention in ecological processes if such communities are to be restored, created, or sustained (*cf.* Crow 1990, Hansen *et al.* 1991).

Frequency, intensity, and season of burning affect the composition, structure, and biodiversity of oak savannas. When a Minnesota oak savanna was burned at intervals ranging from 1 to 12 years, the largest number of plant species occurred in areas burned every 2 years (Tester 1989). A proposed optimum burning schedule consisted of 2 years of no burning alternating with 2 years of burning. The no-burn years allow fuels to accumulate so that the subsequent burn is hotter and thus more effective in controlling non-prairie species. However, these relations and their effects may differ among different types of oak savannas. Burning in different seasons also may produce different results (Van Lear and Waldrop 1988). Moreover, restoring savannas where succession has advanced to a closed canopy oak forest may require annual burns for several years or decades to reduce the predominance of the forest flora including the shrub layer and small-diameter trees, while allowing the invasion and buildup of prairie flora. Thus, management strategies for restoring savannas may differ from those for sustaining existing savannas (White 1986). There are few tested prescriptions for managing oak savannas, and like oak regeneration problems in closed-canopy forests, prescriptions may need to be tailored to each type of community and management objective.

Sustaining oaks as a component of the overstory in oak savannas poses a separate problem apart from attaining biodiversity objectives. Large-diameter oaks are fire resistant because of the insulating capacity of the thick bark of the lower bole. However, intense headfires (*i.e.*, fires with high flames moving with the wind or uphill) can kill even the largest oak if there is sufficient fuel to produce flames that reach tree crowns (Van Lear and Waldrop 1988). In contrast, low-intensity back fires (*i.e.*, fires with low flames moving against the wind or downhill) move at a slow to moderate rate and are usually lethal only to small stems. The stems of oak reproduction and other small or fire-sensitive woody vegetation are killed in either type of fire. Fire-sensitive species such as maples, birches (*Betula* spp.), elms (*Ulmus* spp.), cherries (*Prunus* spp.), and ashes (*Fraxinus* spp.) are eliminated while the oaks survive by resprouting from the root collar (Brown 1960, Curtis 1959, Niering *et al.* 1970, Thor and Nichols 1973). Over time, recurrent

burning at short intervals leads to the accumulation of oak seedling sprouts. Some develop large roots and the capacity for rapid shoot growth just as they do in many xeric closed-canopy forests. But in an open oak savanna, it is the cessation of burning rather than overstory removal that facilitates the recruitment of reproduction into the overstory. Ten to twenty years without fire may be necessary to obtain the necessary recruitment (Haney and Apfelbaum 1990).

The cycle that sustains an oak savanna, then, is *not* continual burning at short intervals, but rather a series of fires at short intervals interrupted by a prolonged period without fire. This pattern produces an overstory comprised of several age classes of overstory trees that are separated by two or more decades (Haney and Apfelbaum 1990). Before settlement by Europeans, the requisite time intervals occurred probabilistically over an immense, heterogeneous landscape (Grimm 1984). The random but frequent occurrence of fires was superimposed on relatively fixed landscape features such as topography and natural fire barriers (*e.g.*, lakes and streams). This produced a mosaic of diverse plant communities that included prairies, savannas, dense patches of sapling oak reproduction ("oak scrub"), and closed-canopy forests. Grimm (1984) called the resulting landscape mosaic the "fire-probability pattern." The term emphasizes the occurrence of different types of communities, each controlled by a relatively constant annual probability of burning. He concluded that the individual communities comprising the mosaic were relatively stable because each was in equilibrium with its specific disturbance regime. In this context, recurring fire represented a stabilizing effect, *i.e.*, it ensured the absence of change in the existing state (*cf.* Anderson and Brown 1986).

The upper age limit for many oaks (*i.e.*, their above-ground stems) is about 250 years, although individual trees may survive longer (Burns and Honkala 1990). Thus, the eventual mortality of overstory oaks in savannas necessitates their periodic replacement by the recruitment of reproduction. The pool of advance reproduction is likely to be more than adequate in a savanna subjected to frequent (*e.g.*, biennial) burning over many years. The recruitment potential also can be objectively assessed using predictive regeneration models such as those developed for closed canopy forests (*e.g.*, Sander *et al.* 1984).

At the end of the recruitment period, burning must be resumed to restore the desired savanna state. After one or two decades of no burning, the accumulation of fuels may carry an intense fire. Nonetheless, the next burn must permit sufficient crown survival of new recruits while simultaneously killing the crowns of excess stems. In general, fire-caused crown-kill decreases with increasing tree diameter and bark thickness (Niering *et al.* 1970, Tester 1989, White 1983). There is thus a need for models that predict crown survival in relation to tree diameter for fires of different intensities. Such models have been developed for turkey oak (*Quercus laevis* Walt.) in the Florida sand hills (Rebertus *et al.* 1989). General guidelines are available for prescribed burning of oak savannas (Irving 1971, White 1986), and experience from similar communities may provide additional interim guidelines (*e.g.*, Thor and Nichols 1973, Van Lear and Waldrop 1988).

Oak-pine mixtures

Naturally occurring mixtures of oaks and pines are common throughout the central and eastern region. These communities seldom occur by design. In the absence of fire, they usually are successional to oak. Examples include the displacement of pines by oaks in the New Jersey pine barrens (Little 1964), the Ozark Plateau (Brinkman and Smith 1968), and the Piedmont and Coastal Plain (Phillips and Abercrombie 1987). In old fields in the Northeast, a common successional pattern is from old field to birch and cherry to eastern white pine (*Pinus strobus* L.) to mixtures of pines, oaks, and other hardwoods (Cline and Lockard 1925). In the sand hills of Florida, there is an antagonistic relationship between turkey oak and longleaf pine (*Pinus palustris* Mill.) under certain burning regimes. This is expressed by the ability of longleaf pine to "repel" turkey oak regeneration because of the flammability of the pine's leaf litter (Rebertus *et al.* 1989). In the management of southern pines, oaks traditionally have been viewed as weed species. There and elsewhere much research and effort have been expended in eliminating oaks to further the monoculture of pines.

The conscious management of oak-pine mixtures is probably best developed in the Southeast (Phillips and Abercrombie 1987, Waldrop *et al.* 1989). There, a low-cost silvicultural method was developed as an alternative to the high site preparation costs required in establishing pure pine stands. It also appealed to nonindustrial landowners interested in managing their woodlands for a combination of timber products, wildlife, and aesthetics. The method allows the landowner to grow a significant pine component at an affordable investment cost while also providing nontimber values. It is especially useful in cutover pine and mixed oak-pine stands that have not regenerated to pine. The site index for white oak on these sites typically ranges from about 60 to 70.

This regeneration method, called the "fell and burn technique," requires commercial clearcutting followed by the felling of all residual trees greater than 5 ft tall during mid-April to early June. At this time, most of the residual trees are near or at full leaf expansion. The timing of this treatment is critical because: (1) the felled vegetation is needed to carry a summer burn and (2) cutting at the peak of leafout reduces the carbohydrate reserves of the hardwood sprouts, which in turn slows their regrowth. By cutting trees after they have leafed out, they also dry more thoroughly through "transpirational drying" (McMinn 1986). The felled trees are then allowed to dry until early July or later, when turkey and quail nesting is completed. At this time the stand is burned, producing a high-intensity fire over a moist fuel bed. The result is more top kill and carbohydrate reserve depletion of the oaks and other hardwoods. Although burning does not reduce the number of hardwood sprouts, it does reduce their height growth. Burning removes about 80 percent of the forest floor, but about two-thirds of the root mat remains intact to prevent erosion. In late winter or early spring after burning, about 450 genetically improved pine seedlings per acre are hand planted. In the Piedmont, both shortleaf pine (*Pinus echinata* Mill.) and loblolly pine (*Pinus taeda* L.) are planted, and in the Coastal Plain, loblolly pine is preferred. Total costs in applying the method are about half that of using conventional pine regeneration methods.

The method results in high survival and rapid growth of the planted pines because competition from hardwoods is effectively reduced but not eliminated. The method thus produces a diverse mixture of oaks, pine, and other hardwoods. Wildlife benefits include browse for deer and cover for many other species. As the stands mature, the oaks also provide mast. Other benefits include improved insect and disease resistance compared to pine monocultures. Although the prescription calls for removal of all residual stems before burning, it would appear to accommodate retaining some residual trees to reinforce structural diversity without detracting from other objectives.

Oak forests for mast production

Acorns are eaten by numerous species of birds and mammals including deer, bear, squirrels, mice, rabbits, foxes, racoons, grackles, turkey, grouse, quail, blue jays, woodpeckers, and waterfowl (Goodrum *et al.* 1971, Rogers and Lindquist 1991, Van Dersal 1940). In the Central and Eastern States, the population and health of many wildlife species rise and fall with the cyclic production of acorns (Healy 1991, Pfannmuller 1991). Acorns' importance to wildlife is related to several factors including the widespread occurrence of oak forests. Acorns also are palatable, highly nutritious, and available during the critical fall and winter period (Short 1976). It would only seem natural, then, that some oak stands and perhaps extensive forests be managed primarily for acorn production. With the exception of managing some bottomland oak stands as green tree reservoirs for waterfowl (Minckler and Janes 1965), that has seldom been done. Nevertheless, there is the potential for such management. Even though our knowledge of acorn production is incomplete, much information is available to guide us in making reasonable decisions in oak mast management.

Acorn production is relatively unpredictable from year to year. On the average, most species produce a good crop of acorns one year in three or four (Olson 1974). In years of low or moderate acorn production, most acorns are consumed by insects (Christisen and Kearby 1984). Moreover,

the production of acorns differs among species (Beck 1977, Downs and McQuilkin 1944, Goodrum *et al.* 1971). Some species are inherently better acorn producers than others, and different species tend to produce good acorn crops in different years. Other factors being equal, trees of large diameter produce more acorns than trees of small diameter. However, in some species, production declines after reaching a threshold diameter (Downs 1944). Oaks with crowns fully exposed to light, such as dominant and codominant trees, produce more acorns than crowns that are totally or partially shaded (Gysel 1956, Sharp 1958, Verme 1953). Species in the white oak group (subgenus *Lepidobalanus*) tend to produce good crops every other year if not limited by weather factors such as late spring frosts. Acorn production among individuals in the white oak group also tends to be synchronized, *i.e.*, when one tree produces well, all of the potential acorn-producing trees in the population produce well. In contrast, acorn production of species in the red oak group (subgenus *Erythrobalanus*) tends to be asynchronous, *i.e.*, some producers yield well in a given year while others do not (Sharp 1958). In addition, only a relatively small proportion of trees are inherently good seed producers. For example in white oak, only 30 percent of large, healthy trees produced any acorns even in good seed years (Sharp 1958) and an even smaller proportion produced a good crop in those years (Sharp and Sprague 1967).

Collectively, the above information suggests that substantial gains in the acorn production of established stands can be obtained by applying the following guidelines:

1. Before the first thinning, identify and reserve the good acorn producers in each stand. Objective criteria for identifying good producers are presented by Sharp (1958). Reliable identification requires observing and keeping records of individual trees for 5 years or more. If that is impractical, a rough assessment of the acorn-producing capacity of individual trees can be made by observing production during a single year in which a good acorn crop occurs for the major species present. However, in the red oak group, many good producers may be overlooked in a single year because of the asynchronous production in those species. Although there is a tendency for the co-occurrence of good acorn crops in the red oak and white oak groups, the relation is highly variable (Smith and Scarlett 1987).

2. During thinning, retain a mixture of oak species to minimize the impact of the large year-to-year fluctuation in acorn production in any one species. (Favoring retention of species in the red oak group over species in the white oak group will tend to even out year-to-year variation in production but may not increase average annual production.)

3. Thin around the identified acorn producers to expose their crowns to full light on all sides. This facilitates crown expansion and increases branch density. Branch density increases acorn production per unit of crown area because of the increase in numbers (density) of acorn-bearing branches (Verme 1953). Although area-wide thinning may be necessary to meet other objectives, it is not necessary for meeting acorn production objectives *per se* because only 20 or fewer good seed producers are likely to occur per acre. Because these seed producers typically will be dominant and codominant trees, they may account for proportionately more basal area and stocking than their numbers alone indicate.

4. Increase or decrease the rotation (or in uneven-age management, maximum tree diameter) to include the tree diameter of maximum acorn production of the predominant species in each stand. For example, peak production in northern red oak is attained when tree diameters (d.b.h.) reach 20 inches and then declines in larger trees. In contrast, white oak production is maximized at 26 inches (Downs 1944). Many other species, however, do not exhibit well-defined diameter-related peaks in production, at least within the diameter ranges observed (Downs 1944, Goodrum *et al.* 1971).

Although our knowledge of the basic biology of acorn production suggests that the application of these guidelines should produce gains in acorn production, there is no research to directly substantiate that it will. There is even contradictory evidence. For example, thinning in pin oak (*Quercus palustris* Muenchh.) stands in green tree reservoirs actually reduced acorn production in flooded stands and did not significantly increase acorn production in nonflooded stands (McQuilkin and Musbach 1977). However, thinning was not specifically aimed at identifying and retaining acorn producers. The above guidelines nevertheless are similar to those presented by Beck (1989). They also are consistent with the view of managing wildlands as life support systems (Kessler 1991).

Old-growth oak forests

Because of their longevity, oaks have the potential to be important species in old-growth forests. Although there is lack of agreement on how to define old-growth hardwood forests, Parker (1989) set a minimum age criterion of 150 years for the central hardwood region. Others have set lower age limits for other regions or specific areas (Meyer 1986, Rusterholz 1991). However, old-growth forests are more than a collection of old trees. Together with large, old trees, they also include specific features such as numerous large snags and downed logs in various stages of decay, and a specific ground flora, animal life, microorganisms, and unique functional relations associated with these features (Franklin 1989, Rusterholz 1991). Relatively few such hardwood forests occur in the Central and Eastern States (Davis 1990). Rusterholz (1991) even questioned the existence of true old-growth oak forests in the Upper Midwest because of that region's disturbance history and the intolerance of oaks to shade.

It is thus apparent that if old-growth oak forests are to become a more prominent feature of the landscape, substantial numbers of existing older stands must be set aside, protected, and allowed to develop to the necessary age and state. There is a need for such forests as habitat for threatened and endangered species, as ecology "laboratories," and as scientific benchmarks against which the ecological efficacy of other management practices can be gauged. Nevertheless, there are unanswered questions about the objectives and strategies for managing old-growth oak forests. When the management objective is to sustain old-growth oak forests and to exclude disturbance, including fire, the objective may contradict the long-term ecological requirements of oaks.

Perhaps there is a place for a mix of management practices for old-growth oak forests. One option is to allow succession to proceed in the absence of *any* human-made disturbance and to not be concerned about the successional displacement of the oaks. Another option is to intervene with prescribed burning and/or other cultural measures to simulate the disturbance regime of the presettlement era that sustained certain kinds of oak forests. An argument for considering the latter is exemplified by old-growth red oak-sugar maple forests in a mesic ecosystem in southern Wisconsin. These forests are among the most botanically diverse in the region (Peet and Loucks 1977) and provide the best habitat for some uncommon breeding birds (Ambuel and Temple 1983, Mossman and Lange 1982). Yet they are ecologically unstable and successional to sugar maple and other tolerant hardwoods. Recent results of prescribed burning in one such forest suggested that fire alone may not be sufficient to sustain an oak component given the practical constraints in the frequency and intensity of burns that can be used (Will-Wolf 1991). Supplemental measures such as planting and creating canopy gaps by girdling were suggested.

Oak forests under stress

Many of today's oak forests are under stress from various factors including defoliation by gypsy moth, oak dieback and decline, and possibly global warming. All of these stresses influence and potentially limit the flexibility we have in sustaining and managing oak forests. Although a detailed discussion of these problems is beyond the scope of this paper, methods are available for minimizing their negative effects. The gypsy moth is moving westward and may eventually affect the entire oak range (Gottschalk *et al.* 1989). However, silvicultural guidelines are available for minimizing the impact of the gypsy moth on the growth, regeneration, and acorn production of oak forests (Gottschalk 1987, 1988, 1990).

The widespread occurrence of dieback and decline, whose effects on oaks range from partial crown dieback to tree death, is thought to be caused by a complex of stresses initiated by drought and/or defoliation followed by insect and disease attack (Wargo and Haack 1991). Maintaining species diversity is important in minimizing the impact of this stress syndrome (Houston 1987). Global warming and associated summer drought and winter freeze-thaw cycles also have been implicated in the increased occurrence of dieback and decline of forest trees across North America (Auclair *et al.* 1990, 1992). Although oaks are known for their great drought tolerance, recent research has shown that, to the contrary, they are particularly vulnerable to sudden and fatal disruptions in xylem water conduction (embolisms) induced by water stress, especially stress caused by winter desiccation (Cochard and Tyree 1990).

SUMMARY

Oak forests in the Central and Eastern States can be managed for numerous commodity and noncommodity products and ecological values. Nevertheless, sustaining the existing oak resource is a major concern throughout much of the oak range. It is a shrinking resource in the richer ecosystems of the Ohio Valley, Appalachians, and other areas where oaks are successionally displaced by more mesophytic species. Regenerating oaks remains a serious problem in those ecosystems. However, the shelterwood method shows promise for resolving that problem in some mesic ecosystems. In the drier ecosystems such as the Ozark Plateau and elsewhere, oak forests are often self-sustaining.

Comprehending the options, opportunities, and limitations in managing oak forests requires, among other things, an understanding of oak ecology, the reaction of oaks to environmental stress factors, and the historical relation between oaks, fire, and humans. It is also important to recognize ecological differences among the different kinds of oak forests and how these differences are related to silvicultural and management options. Stands that are superficially similar may react differently to a given silvicultural practice. Recognizing such differences may be especially important in identifying and solving regeneration problems. Rigorously developed ecological classification systems can be important silvicultural tools for identifying potential problems and opportunities.

Biodiversity and aesthetic values of existing oak forests can be sustained or increased by making modest adjustments in traditional silvicultural methods and by adopting some new ones. Both even-aged and uneven-aged systems are applicable. Landscape diversity can be increased by restoring or creating special communities associated with high biodiversity, wildlife food production, and rare or endangered habitats. These include oak savannas, mixed oak-conifer stands, forests managed for oak mast, and old-growth oak forests. Although many silvicultural prescriptions remain to be developed, many silvicultural tools are available for increasing biodiversity at both the stand and landscape level. But using these tools effectively will require astute management to sustain the array of ecological states that are possible and desirable.

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Johnson, Paul S.
1992. **Perspectives on the ecology and silviculture of oak-dominated forests in the Central and Eastern States.** Gen. Tech. Rep. NC-153. St. Paul, MN: U.S. Department of Agriculture, Forest Service, North Central Forest Experiment Station. 28 p.

Describes the historical and ecological relations between oaks, fire, and humans and reports the consequent silvicultural options and limitations in managing and sustaining oak-dominated forests.

KEY WORDS: Quercus, fire, regeneration methods, savannas, old-growth.

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Our job at the North Central Forest Experiment Station is discovering and creating new knowledge and technology in the field of natural resources and conveying this information to the people who can use it. As a new generation of forests emerges in our region, managers are confronted with two unique challenges: (1) Dealing with the great diversity in composition, quality, and ownership of the forests, and (2) Reconciling the conflicting demands of the people who use them. Helping the forest manager meet these challenges while protecting the environment is what research at North Central is all about.

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