
Restoration of Midwestern Oak Woodlands and Savannas

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20.1 Introduction

There are various definitions for savanna and woodland in the ecological literature. Characteristic elements of each community are broadly defined and often overlap according to the authorities (Curtis 1959; Nuzzo 1986; Nelson 2010). Some confusion is inevitable when categorizing what is in reality a continuum of states from prairie to forest in which there can be much variation. Additional variation arises within each of these community types producing unique associations where composition and structure are further modified by site factors such as soil texture, depth, drainage, and parent materials, which control water and nutrient availability. Nonetheless, given sufficient distance between two points along the continuum, distinct communities are recognizable.

Ground flora composition and species dominance, overstory tree density, and number of canopy layers are used to distinguish between savannas and woodlands. Community structure and composition are inter-related because overhead tree canopies and shrub layers modify the microenvironment, in particular available light, at the ground, which influences floral diversity and species dominance. Historically, fire is considered to have been

the driving disturbance that determined the organization of plants, community structure, resource availability, and thus, defined the natural community. Today, restoration and management of savannas and woodlands is focused on directly managing the structure to influence composition and competitive dynamics that favor desired species or plant functional groups, conserve native biodiversity and mitigate problems from invasive species.

20.1.1 Savannas

Savannas are often characterized as being dominated by grass cover with widely spaced trees (Figure 20.1) (McPherson 1997; Anderson et al. 1999; Nelson 2010). However, there is variation in understory composition and forbs can be prominent in some Midwestern savannas (Anderson et al. 1999; Leach and Givnish 1999). In the past, savannas commonly bordered large prairies on gently rolling hills. Their juxtaposition to prairies, low tree density and high fire frequency, favored ground flora that was a mixture of heliophytic grasses, sedges, and forbs common to prairies, especially at the lower range of overstory crown cover (i.e., 5%–10%) for savannas (Curtis 1959; Nuzzo 1986; Haney and Apfelbaum 1990; Nelson 2010). However, increasing tree density and cover creates a more diverse range of light environments compared to prairies. This leads to increasing species richness in savanna ground flora as species that prosper under the partial shade of the oaks cooccur with prairie species growing in the more open areas. In fact, Packard (1988) has identified species that are endemic to tallgrass savannas, and he argues that savannas are natural communities in their own right and not just transitional, ecotonal communities between prairies and forests (Packard 1993). Tree diversity in Midwestern savannas is relatively low, and is dominated by oak species. In contrast, ground flora diversity is high with, for example, 300–500 species being present over a hundred hectares or so in Missouri and Wisconsin savannas/woodlands, respectively (Leach and Givnish 1999; Nelson 2010).

In the continuum from prairie to forest, the number of vegetation canopy layers increases. In savannas, shrubs, and tree reproduction may be present in isolated patches associated



FIGURE 20.1

Ground flora composition and woody structure help to define (left to right) savanna, open woodland, and closed woodland ecosystems. Savanna is from Ha Ha Tonka State Park, Missouri, where frequent fire has been used for over 30 years. Open woodland is from Bennett Spring State Park, Missouri, where annual burning has been done for 40 years. Closed woodland is from Chilton Creek, Missouri, that has been burned periodically (four times) in 10 years. (Photos by Dan Dey [savanna and closed woodland] and Paul Nelson [open woodland].)

with refuges from fire created by topographic breaks or hydrologic features, but typically, savannas have two main canopy layers, that is, ground flora and overstory trees (Nelson 2004). Woody vegetation is largely oak sprouts and shrubs typical of prairies. Prairie grasses are often dominant in full sunlight and are diminished under increasing tree shade. The threshold of tree crown cover that suppresses dominance by prairie grasses becomes a distinguishing characteristic between savanna from woodland. However, some ecologists set the upper limit of tree cover in savannas at 30% ($<7 \text{ m}^2/\text{ha}$ basal area) and classify systems with 30%–50% crown cover ($7\text{--}11 \text{ m}^2/\text{ha}$ basal area) as open woodlands (e.g., MTNF 2005; Nelson 2010). Much of the nuances in differences between savanna and woodland depend on the composition of the ground flora and the associated range of overstory density and light environments that promote the desired composition. Bader (2001) and Packard and Mutel (1997) have developed lists of shrubs, grasses, sedges, and forbs that are characteristic of oak savannas and woodlands. Other conservation organizations and state agencies have produced similar lists for their states and regions. These lists can be used to assess the potential for restoration, define desired future conditions, and monitor treatment effectiveness.

20.1.2 Woodlands

Woodlands have greater tree cover than savannas, and the dominance of grasses decreases with increasing overstory density, yet plant diversity is still high in the ground layer (Figure 20.1). In woodlands, a midstory canopy is sparse or lacking resulting in a fairly open understory and sufficient light at ground-level to support a diverse plant assemblage. As crown cover exceeds 50%, forbs and woody species begin to increase in dominance, as do species that are more tolerant of shaded conditions. With increasing tree crown cover (up to 80%–100%, Nelson 2010), the ground flora of woodlands begins to resemble that of a forest. Shrubs and tree seedlings/saplings may increase in abundance along the transition from savannas to woodlands and forests. Development of a shade-tolerant midstory canopy can result in three strata of vegetation.

20.1.3 Important Drivers in Savannas and Woodlands

Savannas and woodlands were largely fire-mediated ecosystems that were locally modified by grazing and browsing, and by site factors including soils, hydrology, geology, landform, and climate (Anderson et al. 1999). Fire was arguably the single most influential driver and the character of savannas and woodlands at any one time was defined by the nature of the fire regime: intensity, season, extent, severity, and frequency. The fire regime was in turn influenced by human culture and land use because historic fire in eastern North America was largely an anthropogenic phenomenon, as it still is today (Gleason 1913; Pyne 1982; Williams 1989; Abrams 1992; Delcourt and Delcourt 1997, 1998). Guyette et al. (2002) established a strong linkage between fire history and human population density, culture, and land use effects on fuels over the past 300 years in the oak/pine dominated Ozark Highlands of Missouri. In the oak-dominated landscapes of the Wichita Mountains, Oklahoma, Stambaugh et al. (2014) observed that fire regimes were temporally and spatially dynamic, progressing through stages corresponding to cultural and land use changes. They concluded that humans were the progenitors of frequent fire regimes that favored open oak ecosystems. The effects of human population and culture on fire regimes over the past 300 years have been documented at various areas in the eastern U.S. (McEwan et al. 2007; Aldrich et al. 2010; Stambaugh et al. 2011; Brose et al. 2013b).

Woodlands and savannas occurred where fire was more frequent. In recent history (circa 1650–1850), fire occurred most frequently in the southern and Midwestern regions of the United States before the onset of landscape transformation by European settlement and agriculture (see Guyette et al. 2012). Variations in topography, human population density and land use, soils, and hydrology modified the fire regime and resulted in an intricate mosaic of natural communities across the landscape (Grimm 1984; Nowacki and Abrams 2008; Hanberry et al. 2012, 2014a,c,d).

In eastern North America, the climate and soils are suitable for development of forests practically everywhere. Fire was the main disturbance that sustained the extensive eastern tallgrass prairies and promoted the development of the Prairie Peninsula region (Transeau 1935) by preventing the encroachment and dominance of trees. Plants indicative of prairies, savannas, and open woodlands are adapted to frequent and even annual burning, and fire promotes their flowering, reproduction, and growth by ensuring adequate light, increasing nutrient availability, removing excessive litter, and retarding woody competitors (Packard and Mutel 1997). Historically, savannas occurred along the edges of prairies, representing a compositional and structural transition from prairie to woodland and forest. Savannas occurred where fires burned often enough to sustain only a very low density of overstory trees, which is necessary for grasses to dominate in the understory (Grimm 1984; Anderson et al. 1999; Batek et al. 1999; Nelson 2010; Mayer and Khalyani 2011; Starver et al. 2011). On more protected sites (north to east aspects and coves) in topographically rough areas, woodlands and forests burned less frequently.

Grazing and browsing animals such as bison (*Bison bison*), elk (*Cervus canadensis*), and white-tailed deer (*Odocoileus virginianus*) were part of savannas and woodlands in the Midwest. Where local population densities were large, they affected grass and shrub densities at landscape levels (Rooney 2001). Large ungulates modified fire regimes through their use of these habitats, removing fuels by consumption, thus, lessening the occurrence and intensity of the next fire. Freshly burned areas attracted large ungulates because of the abundance of nutritious, highly palatable and available forage and browse. This spatially and temporally dynamic interaction between grazers/browsers and fire at a landscape scale created a shifting mosaic and increased heterogeneity of habitats that supported relatively high biodiversity in flora and fauna. The fire–grazer interaction has been termed “pyric herbivory” (fire driven grazing) by Fuhlendorf et al. (2008).

20.1.4 Historic and Modern Extent of Savannas and Woodlands

Savannas and woodlands are major world terrestrial biomes and were once prominent in North America before European settlement (Beerling and Osborne 2006). Nuzzo (1986) estimated that about 12 million hectares of oak savannas occurred throughout the Midwestern United States. Oak woodlands were just as prominent if not more, though there are few estimates of their former extent. Hanberry et al. (2014b) estimated that 65% of the historic forests (circa 1812–1840) in the Missouri Ozark Highlands (about 5.7 million ha) were woodlands (55%–75% stocking, Gingrich 1967; 175–250 tph; for trees ≥ 12.7 cm dbh). The distribution of oak forests today may be an indication of the former dominance of oak woodlands because historic oak woodlands succeeded to forests in the absence of fire that resulted from modern fire suppression efforts (Cottam 1949; Heikens and Robertson 1994; Nelson 2010; Hanberry et al. 2014a, d). Now, oak forest types represent 51% of all forestland (78.5 million ha) in the eastern United States (Smith et al. 2009).

In contrast to widespread occurrence of oak forests today, Nuzzo (1986) estimated that <0.02% of the original savannas remain, and untold woodlands have succeeded to

forests. Noss et al. (1995) have identified oak savannas in the Midwest as critically endangered ecosystems, and restoration of oak woodlands and savannas are increasingly becoming major management goals of public agencies and conservation organizations (e.g., MTNF 2005; OSFNF 2005; Upper Mississippi River and Great Lakes Region Joint Venture 2007; Eastern Tallgrass Prairie and Big Rivers Landscape Scale Conservation Cooperative 2013).

In areas of intermediate rainfall (i.e., 1000–2500 mm per year), transitional states between savanna and forest are ecologically unstable and are rare in nature (Mayer and Khalyani 2011; Starver et al. 2011). The greatest loss of oak savannas occurred when they were converted to agriculture production. Woodlands that escaped deforestation and conversion to pasture or crop fields were openly grazed, burned, and timbered. Following efforts to suppress wildfires and control open range grazing, remnant savannas and woodlands rapidly succeeded to forests. Because of their adaptations to fire, drought, and browsing, oak seedlings with large, well-developed root systems had accumulated in savanna and woodland understories so that when a sufficient fire-free period occurred, these sprouts grew rapidly to form closed tree canopies in about 20 years (Cottam 1949; Curtis 1959; Nelson 2010). If they were able to grow large enough and develop thick bark before another fire, the trees were able to avoid being top killed by fire and recruit into the overstory, thus contributing to the development of closed forest structure.

Much of the eastern forest has been made more homogeneous in composition and structure from the extensive logging that took place over a relatively short period circa 1850–1920 (Williams 1989). Oak species initially benefitted from changes in land use during European settlement, but more recently Fei et al. (2011) quantified a decline in oak density and importance in a spatially explicit analysis of the eastern United States. Also, Fei and Steiner (2007) reported a concurrent rise in red maple (*Acer rubrum*) dominance throughout its range in the East. Likewise, Hanberry (2013) documented a decline of 7%–9% in the proportion (trees ≥ 12.7 cm dbh) each of northern red oak (*Quercus rubra*), black oak (*Q. velutina*), and white oak (*Q. alba*) in forests of the eastern United States since 1968, which was accompanied by a 7% increase in proportion for each red maple and sugar maple (*Acer saccharum*).

Schulte et al. (2007) reported a loss of diversity in canopy species and large-sized trees, and simplified landscape compositions and structures in the Great Lakes States Region, which included losses of oak in areas that were in the northern tier of the Lake States on mesic and fine-textured soils. Hanberry et al. (2012, 2014a, c, d) reported similar changes in Missouri, where average tree size has decreased, tree density has increased, oak dominance has declined, and landscape diversity of savannas, woodlands, and forests has been diminished by widespread development of closed forests since the mid-1800s. As a consequence of widespread changes in land use, the age structure of oak ecosystems has been simplified and compressed in the Central Hardwood Region (Shifley and Thompson 2011) and the northern United States (Shifley et al. 2012) such that 60%–70% of the forests are between 40 and 100 years old.

20.1.5 Importance of Savannas and Woodlands

In the eastern United States, concerns for the sustainability of the oak resource arise because oaks are a dominant cover type and have such high ecological and economic value (McShea and Healy 2002; Logan 2006; Johnson et al. 2009). Oak savannas and woodlands are aesthetically pleasing and they support a high level of native floral and faunal diversity (Curtis 1959; Bader 2001; Nelson 2010). However, substantial declines in the

amount of oak woodlands and savannas, and the decreasing dominance of oak forests may have significant negative impacts on wildlife populations (Rodewald 2003; McShea et al. 2007; Fox et al. 2010).

Oak savannas often have higher plant diversity at a variety of scales (α , β , and γ diversity) than prairies or forests (Leach and Givnish 1999; Peterson and Reich 2008). The irregular distribution of overstory trees in savannas at variable but low density creates high spatial heterogeneity in environmental resources over the range from fully open to canopy closure conditions at spatially local scales, and this promotes increased compositional and structural diversity in vegetation. Increasing tree canopy cover in woodlands and forests reduces this heterogeneity and thus plant diversity. At the other extreme of the gradient in tree canopy cover and density, C_4 grasses suppress forb abundance and diversity (Collins et al. 1998; Leach and Givnish 1999). But in savannas, increasing overstory shade reduces C_4 grasses locally, giving forbs the opportunity to flourish and add to overall diversity (Peterson et al. 2007). The dense shade of closed-canopy forests decreases grass and forb diversity, abundance, and reproductive capacity (Taft et al. 1995; Bowles and McBride 1998; Peterson and Reich 2008). Diversity in plant composition and structure is also influenced by variation in soil characteristics that affect plant productivity, and disturbances such as fire, grazing, and browsing. The interaction of overstory density, soil condition, and disturbance that promotes spatial heterogeneity in environment and selective pressure against species capable of dominance (e.g., C_4 grasses) will most likely result in higher levels of diversity. Savannas provide habitat for rare and endangered plant species by offering a multitude of microenvironments at local spatial scales, and they may support viable populations of more than 25% of all plant species within a region (Leach and Givnish 1999). High diversity in plant species richness and vertical structure (high environmental heterogeneity) in savanna flora is often associated with increasing species diversity in invertebrates, small mammals, birds, and herptofauna (Huston 1994; Haddad et al. 2001). In addition, savannas are often good habitat for rare species such as the Karner blue butterfly (*Lycaeides melissa*) (Leach and Givnish 1999).

Wildlife species often prefer key structural and compositional features of oak woodlands and savannas (Anderson et al. 1999; Davis et al. 2000; Nelson 2010). Starbuck (2013) reported that big brown bat (*Eptesicus fuscus*), eastern red bat (*Lasiurus borealis*), evening bat (*Nycticeius humeralis*), and tricolored bat (*Perimyotis subflavus*) preferred savanna and woodland habitats over closed canopy forests in the Missouri Ozarks. Thompson et al. (2012) observed that restored savannas and woodlands in the Ozark Highlands provided habitat for a diverse mix of grassland and canopy nesting bird species that are of high conservation concern. Blue-winged warbler (*Vermivora cyanoptera*), eastern towhee (*Pipilo erythrophthalmus*), eastern wood-pewee (*Contopus virens*), field sparrow (*Spizella pusilla*), prairie warbler (*Dendroica discolor*), and summer tanager (*Piranga rubra*) were more abundant in savannas and woodlands than in closed canopy forests. Reidy et al. (2014) found that large-scale savanna and woodland restoration in the Missouri Ozarks provided additional habitat for woodland generalists and early successional species, some of which are of conservation concern. In the managed restorations, most of the focal bird species they studied responded positively to a history of fire over the past 20 years. In this largely forested landscape, fire increased the diversity of habitats available to songbirds, with corresponding increases in bird species richness, diversity, and density. Others have demonstrated the importance of having savannas and woodlands on the landscape for the conservation of rare and declining bird species that rely on disturbance and early successional habitats (Davis et al. 2000; Brawn et al. 2001; Brawn 2006; Grundel and Pavlovic 2007; Au et al. 2008). Even bird species that are known to prefer mature, closed-canopied

interior forests benefit from early successional habitat in the nearby landscape because juvenile birds forage for food and use the habitat as a refuge from predators (King and Schlossberg 2014).

A lack of early successional habitat characteristic of regenerating forests, woodlands, and savannas is a major wildlife conservation concern (Hunter et al. 2001; Greenberg et al. 2011; King and Schlossberg 2014). For example, the greatest numbers of bird species of conservation concern that are suffering declining populations rely on grasslands, early successional habitats that promote grass and herbaceous vegetation, and savanna and open woodlands where grasses and forbs are abundant in structurally open environments. Several factors act in concert to limit creation of early successional habitat or restoration of savannas and woodlands. Fire suppression has effectively eliminated fire from eastern woodlands and savannas, thus, promoting their densification and transition to forest structure (Hanberry et al. 2014a, d), with subsequent loss of a rich abundance of grasses, legumes, and forbs in the understory, thus altering the fuel dynamics of historic fire regimes. Forest succession in the absence of fire results in system changes (i.e., fuel loading, fuel type, fuel structure, flammability, etc.) that make it harder to restore savanna and woodland character by prescribed burning, and it may necessitate additional silvicultural interventions to initiate the restoration process (Nowacki and Abrams 2008; Kreye et al. 2013). A decline in forest harvesting on public lands, pressure from interest groups who oppose even-aged regeneration methods, and harvesting by high grading, diameter-limit cutting or uneven-aged silviculture on private lands has led to a sharp decline in early successional habitat in the East. Opposition to savanna and woodland restoration has risen to national political interest recently in the Missouri Ozarks due to concern for damage to timber from prescribed burning and to reductions in timber supply by managing for low-density savannas and woodlands (Vaughn 2014). The potential adverse impacts of prescribed fire smoke emissions on human health complicate or may constrain the use of fire in restoration (Fowler 2003; Ryan et al. 2013). All these sociopolitical factors collectively have contributed to the imbalance in forest age structure and a loss of landscape heterogeneity by the reduction and delay in savanna and woodland restoration.

Despite these challenges, the intent of many public agencies and organizations is to promote the conservation of native flora and provide quality habitat for fauna that desire structurally open environments with an abundant, diverse herbaceous ground flora through restoration of fire-dependent savannas and woodlands. If the current trend toward aging forests and lack of open-environments such as savannas and woodlands is not corrected, then there will be serious ramifications for biodiversity due to declining acorn production in old growth oak forests and lack of quality habitat for species that rely on early successional and open-environments at broad scales in the next century.

Loss of acorn production will decrease the ability of oak to regenerate naturally, promote succession toward other species that may have less habitat value for wildlife, and negatively impact the myriad of wildlife species that rely on acorns. Mast production is foundational to food pyramids and ecosystem function because so many species use acorns (McShea and Healy 2002) and they are interrelated through predator-prey relationships (e.g., Clotfelter et al. 2007). Significant increases in forest regeneration by even-aged methods are needed to bring forest age-structure into a more sustainable balance (see Section 20.2.3) and to increase early successional habitat for species of conservation concern. Modifications of traditional even-aged methods in forest management, for example, shelterwood method, are needed to restore savannas and woodlands. Large-scale restoration of savannas and woodlands would also contribute to landscape diversity, provide quality habitat, and promote the conservation of native flora and fauna. Increases

in early successional habitat through large-scale restorations of woodlands and savannas are being implemented especially in the prairie-forest transition zone from Texas north to Minnesota, and eastward toward Wisconsin, Missouri, and Arkansas, where hundreds of thousands of hectares are planned for restoration (e.g., MTNF 2005; OSFNF 2005).

Loss of landscape diversity (e.g., Hanberry 2012, 2014a, c, d) poses major conservation, forest health, productivity, and resilience issues and challenges. For example, Schulte et al. (2007) stated that current landscape composition and structure is more simplified in the Great Lakes Region and is associated with lower species richness, functional diversity, structural complexity, and level of ecosystem goods and services compared to preEuropean settlement according to analysis of public land surveys. Loss of mixed conifer-hardwood forests, woodlands, and savannas due to changes in land use including conversion to agriculture, extensive logging, urban development, and fire suppression have contributed to the simplification of the landscape. A more homogeneous landscape is subject to widespread and catastrophic losses and degradation following invasive species introductions, endemic insect and disease outbreaks, and increasing environmental stresses from changing climates. In the past, abiotic stresses and biotic attacks on natural communities were limited in extent because of the diversity in composition and structure of vegetation across the landscape. A landscape that is low in natural community diversity and has a shortage of early successional stages is limited in the biodiversity it can support (Hunter and Schmiegelow 2011; Shifley and Thompson 2011; Shifley et al. 2014) and it has low resilience in a future of inevitable and increasing threats to ecosystem function, health, and productivity.

The presence of oak in savannas, woodlands, and forests has important implications to ecosystem function and productivity. For example, oak foliage in forest canopies and oak litter provide important inputs to terrestrial and aquatic ecosystems and both enhance productivity by supporting a greater diversity and abundance of organisms involved in energy and nutrient cycles than those supported by forested landscapes without oaks (Hansen 2000; Rubbo and Kiesecker 2004; Tallamy and Shropshire 2009; Stoler and Relyea 2011). Certain oak species are considered pyrophytic because their litter facilitates burning in oak and oak/pine ecosystems. Kane et al. (2008) noted that turkey oak (*Quercus laevis*) and post oak (*Q. stellata*) burned with the greatest intensity, sustainability, and consumability of eight southeastern oak species. Hiers et al. (2014) recognized the facilitative role pyrophytic oaks such as blackjack oak (*Q. marilandica*) and post oak play by providing flammable fuel capable of carrying prescribed fire needed for the restoration and management of fire-dependent ecosystems. The litter from these species has fast leaf drying rates and leaf curling habits, which is important to the disturbance processes in fire-dependent ecosystems. In open woodlands and savannas, higher levels of fine fuels from increased diversity and production in ground flora (i.e., grasses and forbs) may add to the flammability of ground fuels necessary to propagate surface fires, especially when pyrophytic oak leaves are suspended in the grass/forb matrix (Loudermilk et al. 2012; Hiers et al. 2014). Also, restoration of historic fuel conditions may act to extend the annual “burning window,” that is, the time when conditions are good for conducting prescribed fires.

Managing for disturbance adapted ecosystems and increasing biodiversity at all scales are considered key management strategies to address anticipated impacts due to climate changes (Janowiak et al. 2011, 2014; Brandt et al. 2014). Restoration of savannas and woodlands would contribute to both these mitigation strategies for the range of future climate scenarios predicted. Tree species common to Midwestern savannas and woodlands such as post oak, blackjack oak, and bur oak (*Quercus macrocarpa* Michx.) are expected to be favored by predicted changes in temperature and precipitation, modifications in their seasonal patterns, and frequency of extreme weather events (Brandt et al. 2014). Communities

such as these, with high species richness, are considered more resilient to climate change, better able to recover from disturbance such as drought, less vulnerable to environmental stress and biotic threats, and less susceptible to high-severity wildfires (Brandt et al. 2014).

20.2 Restoration and Management of Savannas and Woodlands

20.2.1 Site Assessment and Selection

Selecting sites for savanna and woodland restoration is an important first step that, once made, has a large bearing on the cost of restoration and the realization of conservation and biodiversity benefits. A list of indicator plants for savannas or woodlands can be used to assess the presence of remnant populations in surveys of the candidate areas and the potential for a positive floral response following the reintroduction of fire (e.g., Bader 2001; Farrington 2010). A more systematic floristic inventory can be made using the Floristic Quality Assessment method (Swink and Wilhelm 1994; Taft et al. 1997) to evaluate the quality of the current flora, integrity of the natural community, and potential for restoration. The method is also useful for monitoring change, assessing treatment effectiveness, and providing input for adaptive management.

Stand structure and characteristics of individual trees may indicate previous savanna or woodland conditions. The presence of large “wolf” trees with low and wide spreading crowns and large lateral branches indicate that previously the trees had grown in the open in a savanna or open woodland. Sometimes these trees are also surrounded by dense thickets of smaller diameter saplings or pole-sized trees that invaded the savanna or woodland following fire suppression. Oaks often dominate the initial recruitment of trees in savannas and woodlands during an extended fire-free period following a long-term history of frequent fire (Stambaugh et al. 2014). Encroachment of mesophytic tree species such as red maple requires longer periods of fire suppression and initiates in closed-canopy oak forests, setting the stage for replacement of the oak (Fei and Steiner 2007; Nowacki and Abrams 2008; Johnson et al. 2009).

Variation in topography affected historic fire regimes and hence landscape setting can be indicative of the distribution of remnant savannas and woodlands. Stambaugh and Guyette (2008) found that topographic *roughness* was inversely correlated to fire frequency, that is, plains burned more frequently than did highly dissected, topographically rough areas. Hence, prairies were most common on plains, savannas occurred at the edges of prairies on gently rolling terrain, and closed woodlands and forests were located in heavily dissected areas, on steeper hillslopes (Heikens and Robertson 1994; Robertson et al. 1997; Batek et al. 1999). Mesic forests were most common in *fire shadows* on eastern sides of streams and lakes where fire burned least frequently (Grimm 1984; Batek et al. 1999). A study of the witness tree data from General Land Office (GLO) survey notes that were made in the early 1800s in the Midwest can provide insight into the past locations of prairie, savanna, woodland, and forest, and establish historic baselines to assess change. Others have used GLO surveys of Midwestern states to reconstruct historic vegetation conditions, which can help to assess the potential for restoration (e.g., Rodgers and Anderson 1979; Radeloff et al. 1998; Batek et al. 1999; Schulte et al. 2002; Kilburn et al. 2009). Historic photos or journals can supplement information from GLO survey data, provide anecdotal confirmation of natural community types, and give insights on fire history and land use practices.

Consideration of soils can be helpful to evaluate the likely occurrence of savannas and woodlands. Soil conditions that promote seasonal drought are more prone to burning and consequently have the potential for higher fire frequency given a nearby ignition and fire spread. Savannas and woodlands are more likely to have occurred on droughty sites. Any condition that limits the volume of soil available for roots and available water content such as shallow depth to bedrock, claypans, or fragipans can lead to summer drought conditions. Soil texture affects available water content. Sandy soils are often well-drained and have low capacity to store water. Conversely, clayey soils can store water but much of it may be unavailable to plants as it is bound by the clay. Post oak flatwoods are savanna/woodland systems that occur on soils underlain by a claypan on poorly drained level areas (Anderson 1983; Nelson 2010). Sites that experience seasonal drought and are of low productivity limit the development of tree/shrub crown cover, stand density, and multiple canopy layers. Less-dense woody cover permits the higher light levels in the understory that are needed to support savanna and woodland ground flora. This increases the likelihood that species of high conservatism and floristic index may be present on the site, if it has not been damaged by grazing, erosion, or other factors that would remove the seed-bank and remnant populations of plants.

Knowledge of local fire history can provide insight on the occurrence of savannas and woodlands. Site-specific fire histories are increasingly being developed throughout eastern North America (Dey and Guyette 2000; Guyette and Spetich 2003; Guyette et al. 2003; Stambaugh et al. 2006a, 2011; Hoss et al. 2008; Hart 2012; Brose et al. 2013b). A landscape model predicting historic fire frequency for the continental United States has been produced by Guyette et al. (2012) and is a useful guide for areas that lack local fire history data. Annual and biennial fire regimes are more closely associated with prairies and savannas (Anderson et al. 1999; Anderson 2006; Nelson 2010). Less-frequent fire return intervals favor woodland (>5 years) and forest (>10 years) development. Variation in fire occurrence at a given site is equally important as average fire frequency. To support a woodland structure, infrequent but extended fire-free periods of 10–30 years are needed to permit recruitment of tree saplings into the overstory (Arthur et al. 2012; Dey 2014). Fire regimes that lack variability in fire-free periods trend toward producing savanna or prairie communities. Knowledge of land use history is useful for considering how degraded sites may be. Severely degraded sites include those that were cultivated annually or overgrazed for decades, allowed to erode severely, or developed dramatically altered structure and composition resulting from long-term fire suppression. These sites are typically of low floristic quality having lost native seedbank and remnant vegetation, and are dominated by weedy, invasive, exotic or native-generalist species. Historical photos or local journals provide anecdotal insights on historical conditions and land uses.

20.2.2 Defining Desired Future Conditions

In restoration as in any type of management, it is essential to define the desired future condition. This guides development of reasonable alternative prescriptions and selection of the preferred method for moving the current vegetation toward the desired state. It is important that the desired future condition be specified in as quantitative and measurable of terms as possible, and that indicators of success are ecologically and socially meaningful. Intermediate thresholds for indicators at key stages are helpful to aid in monitoring treatment effectiveness and progress toward the desired state, and to identify when management needs to be modified to correct successional trajectories leading to undesirable conditions (Dey and Schweitzer 2014).

Restoration of savannas and woodlands inherently implies that a former, historic condition be recovered to some extent. Analysis of GLO survey data can help guide tree species composition and stand structure, and used to model landscape patterns of natural communities as they vary along environmental gradients. Ecological knowledge of species adaptations to disturbance, physiological requirements, and competitive dynamics can be used to hypothesize a desired future state. Understanding of historic disturbance regimes helps in the setting of reasonable targets. These considerations are normally useful for establishing desired natural community types and defining broad ranges in composition and structural characteristics of high-quality natural areas, but more quantitative measures of key ecosystem structure or function require study of modern examples and analogs. There are usually isolated modern day examples, nondegraded reference sites considered to be high-quality savannas and woodlands. They can be studied to help quantify composition and structural metrics. A number of reference sites are preferable to just one because they help capture the range of natural variation that can be expected in restoration. Caution must be exercised when using reference sites to guide restoration. They may be isolated, atypical examples, or small relic areas that have been modified by invasive species, overbrowsing by white-tailed deer, or have altered disturbance regimes that are deviant from historic scenarios. Thus, they may not be good examples to emulate or represent desirable future outcomes.

Methods of analyzing historic vegetation surveys such as the GLO data to model vegetation composition and structure are becoming more sophisticated and better able to account for surveyor bias in selection of witness trees (Hanberry et al. 2012, 2014a,c,d). These models are spatially explicit, and based on many standard physical and ecological variables related to the distribution and character of natural communities. They are available in spatially geo-referenced inventories that can be integrated in a GIS and used in decision making in site selection and characterization of natural communities. For example, Hanberry (2012, 2014a,c,d) developed spatially explicit models of the probability of occurrence by tree species and estimated tree density, average diameter, basal area, and stocking for the state of Missouri by ecological subsections based on GLO survey data. Estimates of these metrics can be used to quantify the historic composition and structure. Modern forest stocking charts such as that developed by Gingrich (1967) for upland *Quercus-Carya* forests in the Central Hardwood Region (Figure 20.2) or by Larsen et al. (2010) for floodplain *Populus deltoides*-*Acer saccharinum*-*Platanus occidentalis* forests in the Midwest can be used to define desired tree structure and to set structural thresholds that would prompt management action in restoration.

20.2.3 Managing Stand Density

Restoration of oak savannas and woodlands entails reducing the density of trees and reintroducing fire. Higher light levels than exist in most forest understories (~3%–5% of full sunlight) are needed to stimulate germination, promote growth, and encourage seed production of the herbaceous species commonly found in savannas and woodlands. For example, for prairie grasses such as big bluestem (*Andropogon gerardii*) and little bluestem (*Schizachyrium scoparium*), to proliferate, tree canopy cover needs to be <50% (Anderson et al. 1999; Nelson 2010; Mayer and Khalyani 2011; Starver et al. 2011). Reducing overstory density to about 6.9 m² per ha, 30%–40% stocking, or 50% crown cover produces about 50% of full sunlight at the ground level in oak forests (Figure 20.3) (Parker and Dey 2008; Blizzard et al. 2013).

Low-intensity dormant season fires, which are commonly used in restoration, have limited capacity to reduce stand density because they can only reliably top kill hardwood stems

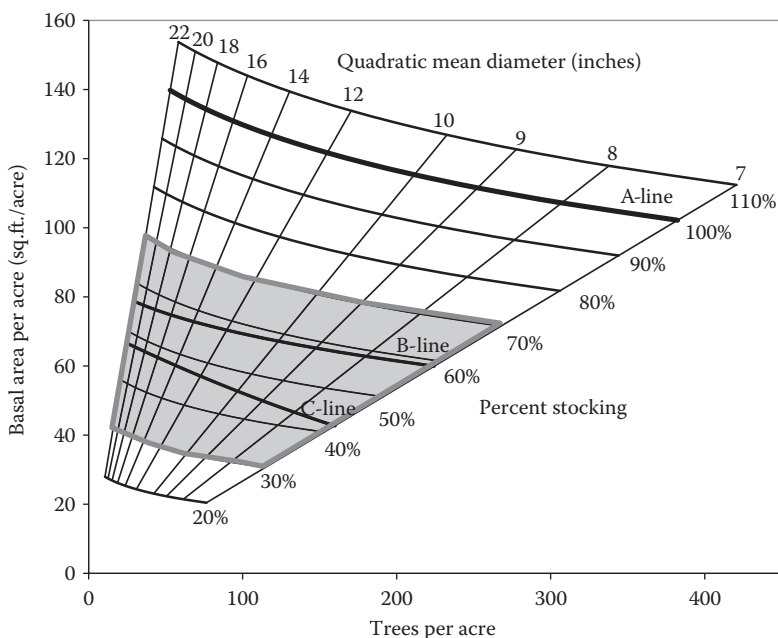


FIGURE 20.2

Stocking chart for upland *Quercus-Carya* in the Central Hardwoods Region (Gingrich 1967) used to manage savanna and woodlands. Stocking in woodlands is maintained between about 30% and 70% stocking. Closed woodlands have higher stocking (70%) compared to open woodlands (30%–40%) and savannas (<30%). When regenerating closed woodlands, stocking is reduced to below B-level stocking.

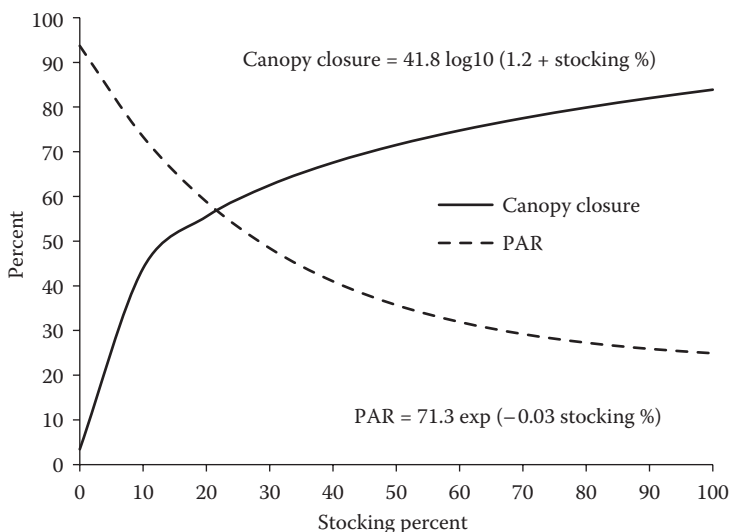


FIGURE 20.3

Relationships between stocking, canopy closure, and photosynthetically active radiation (PAR) in stands that have been thinned from below in the Missouri Ozarks. (Adapted from Blizzard, E.M. et al. 2013. *General Technical Report SRS 75*. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station, pp. 73–79.)

less than about 12 cm dbh (Figure 20.4) (Arthur et al. 2012). Thus, they can only increase understory light in a closed-canopy forest up to about 15% of full sunlight by reduction of the midstory canopy/sapling layer (Lorimer et al. 1994; Ostrom and Loewenstein 2006; Motsinger et al. 2010). Moderate- to high-intensity fires are capable of killing larger trees as well as increasing the proportion of smaller stems that do not sprout postfire (Brose and Van Lear 1998; Brose et al. 2013a). Killing of larger, older trees often leads to complete mortality because they are less likely to sprout after death of the stem than are saplings and pole-sized trees (Dey et al. 1996). This can be accomplished at a local scale by placement of downed tree tops that are cured near the boles of live trees, or by broadcast burning when fuel moisture and relative humidity are low, and fuels are sufficient to generate high fire temperatures for the duration needed to kill cambial tissue. Managing the reduction of larger trees with fire can be problematic because burning under weather and fuel conditions to achieve high-intensity fires is risky from a fire escape and safety standpoint. Fire is more indiscriminant in what trees are killed than are alternative methods for reducing stand density. Generally, other practices such as timber harvesting or chemical/mechanical thinning must be used with fire to complete the restoration process in a timely, safe and affordable manner.

Timber harvesting to reduce the density of large trees is an efficient alternative method of large tree removal, and affords a high degree of control of the spatial arrangement and composition of remaining trees. It also provides needed income to pay for other costs of restoration. Mechanical cutting of large diameter trees before burning is an effective way of controlling overstory density. If larger diameter trees produce sprouts after mechanical cutting, their sprouts are then vulnerable to being top killed by a subsequent fire due to their small diameter and thin bark. If large trees are unmerchantable or timber harvesting is not a management option, another effective method is the stem injection of herbicides. When applied correctly, herbicides can kill trees completely preventing them from sprouting. Dead trees can be left standing to fall apart over time, if public safety is not an issue.



FIGURE 20.4

Annual and periodic low-intensity spring prescribed burns in mature *Quercus-Carya* forests are effective in reducing the midstory canopy of woody species, but there has been little effect on the overstory canopy. At the Chilton Creek woodland restoration area, annual burning (left to right) has been conducted for 10 years and has substantially reduced the mid- and understory vegetation. Two years after the last fire in the periodic burning (4 times in 10 years) treatment, ground cover is higher than in the annual or the no burn treatment. Unburned stands have greater vertical woody structure and less diverse understory. (Photos by Dan Dey.)

In the process of decaying, snags can provide critical habitat for a wide variety of wildlife. Methods of mechanically girdling trees are also available for killing larger trees, though they are not as effective as herbicides.

20.2.4 Managing Ground Flora

Reducing tree density in combination with fire will produce the fastest improvements in ground flora diversity and dominance in most cases where the sites were not previously degraded by livestock overgrazing and soil erosion. Prescribed burning alone generally increases herbaceous species coverage and richness when trying to restore oak savannas from mature forests conditions, but improvements are most dramatic where the overstory is opened as a consequence of burning or thinning (Hutchinson et al. 2005; Hutchinson 2006; Waldrop et al. 2008; Kinkead et al. 2013). Even without fire, positive responses in herbaceous richness and coverage have been achieved by mechanical thinning or timber harvesting (Hutchinson 2006; Zenner et al. 2006; Waldrop et al. 2008; Kinkead et al. 2013). However, these gains in diversity are ephemeral as an abundance of woody sprouts grow rapidly to form canopy closure and shade out the ground flora.

Continued use of fire is crucial to control the growth of woody sprouts that can develop quickly following burning or harvesting. Maintaining higher levels of overstory canopy cover (e.g., >70%) or density (e.g., >13.8 m² per ha) can suppress the growth of woody sprouts in the understory (Dey and Hartman 2005; Kinkead et al. 2013), but would also delay the restoration of savanna ground flora, which requires more sunlight. Law et al. (1994) published a chart useful for managing stand density to produce desired crown cover in upland oak savannas depending on overstory tree size and density. Blizzard et al. (2013) developed models that are useful for estimating available light in the understory from overstory crown cover, density, basal area or stocking for upland oak-hickory forests in the Missouri Ozarks. These can be used to manage stand density to provide sufficient light to promote desired ground flora compositions.

Managing overstory crown cover between 5% and 50% influences the development of the ground flora. Tree crown cover below 30% is needed to promote grass domination and proliferation of heliophytic forbs commonly found in prairies (Nelson 2010). Greater tree cover begins to favor the abundance and diversity of more shade tolerant forbs characteristic of woodlands as the grasses decrease in dominance. Maintaining tree cover above 50% inhibits domination of grasses in the ground layer (Mayer and Khalyani 2011; Starver et al. 2011).

In addition to using fire to restore and maintain the woody structure of savanna ecosystems, the restoration of fire as a disturbance that shapes the ground flora community is important. Fire promotes germination and growth of herbaceous plants by (1) removing litter that acts as a physical barrier to germination and seedling establishment, (2) preparing a receptive seedbed for colonization by wind and animal dispersed seed, (3) breaking chemical and thermal seed dormancy by producing heat and smoke, (4) releasing nutrients tied up in the litter, and (5) increasing light and temperature at ground-level. Frequent fire, that is, every 1–3 years for example (Anderson et al. 1999; Nelson 2010), can be managed to favor dominance of grasses, legumes or forbs by varying the season, intensity and frequency of prescribed burning. Simple recommendations cannot be given here because vegetation response to fire is complicated by its modification by differences in climate, soils, hydrology, grazing and browsing, and overstory interactions. However, Peterson and Reich (2008) observed that annual burning promoted C₄ grasses in open-environments, forb richness was greatest with biennial fires, and shrubs, trees and vines increased in

dominance with longer fire-free periods on a sand plain in east-central Minnesota. Spring fires when grasses are still dormant, favor grass domination in general while summer burns favor forb diversity (Nelson 2010). Dormant season fires have the least impact on the herbaceous plant community (Hutchinson 2006). Consistent application of prescribed fire tends to create homogeneity in the vegetation community, therefore it is recommended to vary the frequency, intensity and season of burning when maintaining savanna ecosystems (Nelson 2010). Restoration of severely degraded sites may require artificial regeneration of the ground flora by seeding and planting. Packard and Mutel (1997) published guidelines for actively managing the recovery of ground flora through artificial regeneration.

20.2.5 Invasive Species, Fire and Restoration

Fire, like many disturbances, increases resource (light, moisture, nutrients) availability and can create receptive environments for nonnative invasive species (NNIS) establishment by removing litter and exposing mineral soil for the colonization and expansion of NNIS. Seeds or propagules onsite or nearby are a prerequisite to invasion. It is incumbent therefore to have a good inventory of the proposed restoration site and nearby areas to know if NNIS may be a problem that needs to be factored into restoration plans. Since repeated burning is standard practice in restoration and maintenance of oak woodlands and savannas, vigilance through monitoring is essential for detecting NNIS problems early on. The list of current NNIS that threaten the integrity of woodlands and savannas is large and varies by region (Grace et al. 2001). Some common problem species in oak savannas and woodlands include smooth brome grass (*Bromus inermis*), Canada thistle (*Cirsium arvense*), musk thistle (*Carduus nutans*), sericia lespedeza (*Lepedeza cuneata*), autumn olive (*Elaeagnus umbellata*), multiflora rose (*Rosa multiflora*), teasel (*Dipsacus* spp.), crown vetch (*Coronilla varia*), white sweetclover (*Melilotus alba*), yellow sweetclover (*Melilotus officinalis*), spotted knapweed (*Centaurea biebersteinii*), European buckthorn (*Rhamnus cathartica*), and Japanese honeysuckle (*Lonicera japonica*). The spread and dominance of NNIS over native species may increase with decreasing tree canopy cover in woodlands and savannas as many of them are adapted to high light environments and fire.

It is impossible to generalize about fire and NNIS because there is so much variation in site conditions, fire behavior, fire regime, NNIS species traits, initial floristic composition and structure, competitive dynamics between NNIS and native species, and a myriad of other factors that affect plant survival, growth, spread and rise to dominance. Zouhar et al. (2008) provided a thorough overview of fire effects on NNIS in natural communities and DiTomaso et al. (2006) discussed the use of fire to control NNIS. Each restoration project area must be evaluated on a case-by-case basis to integrate site conditions, planned disturbances, and reproductive modes and phenology of native and NNIS species in deciding what the potential problems may be and how they can be mitigated by modifying the timing, type and combination of management practices. There are some basic things to consider when evaluating the potential for fire to create or exacerbate a NNIS problem.

First consider the modes of regeneration of NNIS. Plants may reproduce by sexual (seed) or asexual (vegetative) mechanisms. Most species use both modes though one may be more prevalent especially under certain disturbance regimes. An array of sexual reproductive traits of NNIS that enable them to exploit postfire environments include

- Mature rapidly in an annual or biennial cycle
 - Increasing the probability of completing seed production and dispersal in between fires

- Able to self-pollinate
 - Can reproduce in low density or sparse populations
- Prolific production of wind and bird dispersed seeds
 - Maximizes numbers of seeds disseminated for opportunistic colonization of ephemerally favorable sites
- Seeds remain viable in seed bank for years or decades
 - Seed can accumulate to high densities far exceeding annual crops and are in place for release by an appropriate disturbance
- Chemical or thermal induced seed dormancy
 - Increases the probability of surviving burning
 - Fire stimulates germination and synchronizes it with creation of favorable seedbed conditions
- Rapid early growth of plants adapted to high light environments
 - Promotes early dominance and acquisition of resources for development toward maturity and completion of life cycle

NNIS can persist in a regime of fire by either surviving the fire intact or by reproducing asexually. The cambium of perennial woody species (e.g., tree of heaven (*Ailanthus altissima*)) is protected from the heat of fire by bark. The cambium's ability to avoid fire damage increases exponentially with increasing bark thickness (Hare 1965). The location of meristematic tissues in the form of vegetative and reproductive buds influences whether they will survive a fire or not. Buds located in soil are in most cases insulated from the heat of fire as most surface fires do not increase temperatures to lethal levels in the top 5 cm of soil (Iverson and Hutchinson 2002; Iverson et al. 2004). Buds at the extremities of branches in tall trees and shrubs are distanced from the direct heat of many surface fires, especially dormant season low intensity burns. Species with root-centric growth, who preferentially store carbohydrates in their roots, may build large energy reserves between fires. This stored energy is used to fuel rapid shoot growth from adventitious buds following a fire that top kills the parent stem.

Fire effects on plants depend, in part, on the specific fire regime, plant life cycle and reproductive strategies, plant phenology at the time of burn, and population demographics. Fire type (ground, surface, or crown), severity and frequency affect the persistence of plants, and determine if populations will be reduced, or spread to dominate after burning. Fire size and inclusion of unburned areas within the restoration area influence NNIS colonization and spread. Larger burns and more completely burned areas limit NNIS invasion from outside seed sources and from previously established individuals and colonies. Season of burning in relation to phenology influences damage and mortality. Fire severity is greater when plants are actively growing at the time of burning. Burning plants before they set seed, or when seed is vulnerable to direct fire injury reduces NNIS populations more effectively. Plants with below ground reproductive structures such as rhizomes, caudices, bulbs, corms or root crown buds are well-protected from fire. Many plants with these traits sprout prolifically after fire and prosper in an environment of readily available resources. Plants that store viable seed for years or decades in organic or soil layers are well-suited to persist as a species after fire. In the winter and spring, moist or frozen organic layers protect seed stored within or beneath that horizon.

The timing and severity of fires is key to controlling NNIS, and determines the fate of native species as well. The easiest NNIS to control are annuals that produce seed after the fire season, whose seed is readily exposed directly to fire's flames and does not persist in the seedbank. Late spring to early summer fires are most likely to control NNIS annuals that set seed later in the summer. Biennial and perennial NNIS are more difficult to control. More severe fires are needed to kill reproductive structures in organic or mineral soil layers. Few NNIS are controlled with a single fire. It takes consecutive, repeated fires to stop seed production by killing existing individuals and to eliminate plants that arise from the seed bank or from vegetative structures, which often are stimulated by the initial fire. Scheduling fires several years apart only allows NNIS to add seed to the seedbank, or build energy reserves in belowground structures.

Burning followed by herbicide application is an effective alternative method (DiTomaso et al. 2006). The fire kills current vegetation, stimulates germination, converts large plants into small concentrated sprout clumps through top kill and sprouting, and removes debris that facilitates herbicide application. Herbicides are effective at killing plants that sprout prolifically from large underground bud banks and stored energy reserves. The succulent growth of seedling sprouts and germinants readily absorbs herbicides, increasing the efficacy of the herbicide. Fire effect on native species must also be considered in planning the prescribed fire regime to ensure they are not adversely impacted and that their response to fire is vigorous. Dominance of native species after fire can help to suppress NNIS establishment or recovery.

20.2.6 The Role of Grazing in Restoration

In the past, bison, elk and deer were free to roam across the landscape, and fires burned without suppression. Our challenge today, is to develop modern analogs of these critical interactions for smaller landscapes. There are few examples in research or management where wild or domestic grazers/browsers have been integrated with fire and other vegetation management to restore savannas and woodlands at the scale of landscapes. However, Collins et al. (1998) found that bison grazing or mowing in late June increased total species richness by promoting C_3 grasses, forbs and woody species in prairie ecosystems that are dominated by C_4 grasses under a regime of annual to frequent fire. Also, Hartnett et al. (1996) demonstrated that bison grazing in combination with various fire frequency burn treatments (i.e., annual to 4-year cycles) significantly increased plant species diversity by preferentially grazing C_4 grasses and creating greater spatial heterogeneity. Similar results may be expected in savanna and woodland ecosystems but this hypothesis needs to be tested.

Restoration of savannas and woodlands is a relatively new management goal, having increased in application over the past 30 years in the Midwest. Initial efforts focused on simply reintroducing fire to reclaim floristic quality. Fire was prescribed fairly often, for example, every 3 years, based on local fire history knowledge. Before long, it became obvious that burning alone was not having its desired effect on regulating stand density or woody sprouts. Prescribed fires were typically low intensity and conducted in the late winter to spring seasons. Only recently have we begun to understand and manage stand density and ground flora interactions, and realize the positive benefits of variability in the fire regime (season, intensity, frequency, severity, size) in creating heterogeneity in habitats and increasing biodiversity at the community and landscape level. Introduction of native grazers and browsers is still in the future in research and management of savannas and woodlands in the Midwest. There is resistance by some managers to the idea of

introducing grazing in restoration projects because of the long history of ecosystem degradation caused by overgrazing by domestic cattle, which led to soil erosion and compaction, loss of native floral diversity, and proliferation of exotic species. But managed grazing as part of a suite of restoration practices may have a positive role to play in managing vegetation and achieving natural community, biodiversity, and conservation goals (Bronny 1989; Harrington and Kathol 2008). Certainly, there are differences in the grazing habits and behavior of domestic cattle compared to bison and elk that affect structure, composition and diversity of flora (Hartnett et al. 1996), and these must be studied in the future. Integrating grazing may be difficult to nigh impossible to implement on small parcels, but on larger landscape restoration projects, it could contribute to restoring ecosystem function, and promoting heterogeneity in habitats that favor biodiversity for a wide array of native flora and fauna.

20.2.7 Sustaining Savannas and Woodlands

Once the structure, composition, and function have been restored to savannas and woodlands, it eventually becomes necessary to plan for replacing some of the overstory trees to sustain desired stocking. This need arises because some of the trees will succumb to competition-induced mortality, and others will die in old age of physiological stress from injuries suffered through management and other agents causing physical damage and decay. Post oak is one of the longer lived oak species and individuals may live to be over 400 years old, but even they need to be replaced to sustain savanna or woodland overstories.

Harvesting mature trees to regenerate and recruit the next generation of overstory can provide financial benefits. If managed properly, mature oak trees are capable of producing forest products, and the periodic harvest and sale of these can be used to offset some of the management costs of restoration, though potential forest product values are limited in savannas due to low tree density. It is not something that is commonly thought of in restoration since the emphasis has been on the diversity and quality of the ground flora. But, prescribed fire can be a major inciting agent of tree value loss in savanna and woodland restoration, as fire-wounding provides entry to wood decaying fungi. The key is to achieve ecological benefits through prescribed burning without causing unnecessary damage to trees that are capable of producing valuable products.

A comprehensive management system that includes a plan for regenerating trees is recommended for all restoration prescriptions to ensure the sustainability of the ecosystem.

Presently, there are no well-defined silvicultural systems that include a planned series of treatments for regenerating and tending savannas and woodlands. Nonetheless, there are many relevant silvicultural methods potentially applicable for restoration. Most of the regeneration methods used in forest management can be applied to savannas and woodlands. For example, trees can be regenerated with the clear-cut or shelterwood methods (each with reserves) and tended with thinning and prescribed burning in even-aged systems, or regenerated with the uneven-aged group selection method and tended with thinning by mechanical or chemical methods. Application of prescribed fire with the group selection method is complicated because the matrix area may need burning but fire must be excluded from the individual group openings to permit recruitment, and they are scattered throughout the area. With time the overall restoration area becomes an integrated mosaic of different aged group openings that need protecting from fire.

However, these regeneration and tending methods may be applied differently in savannas and woodlands than in forests. For example, retaining residual stocking with reserve trees may be more preferable for regenerating savannas and woodlands than forests. This

residual overstory provides habitat for wildlife and provides partial shade to reduce the growth of woody advance regeneration that is released by harvesting. It also influences the composition of ground flora based largely on the shade tolerance of the underlying grasses, sedges, forbs and legumes. Applying a method to develop two-aged stands comprised of a partial overstory (>20%–30% stocking of dominant/codominant trees) and a regenerating subcanopy may reduce intense shading of ground flora by woody vegetation developing in the regeneration layer that would occur at lower overstory stocking.

During the recruitment phase in savannas and woodlands, when tree seedling sprouts grow into larger size classes advancing into the overstory, prescribed fire should be excluded until a portion of the recruiting cohort is sufficiently large to escape being top killed by fire's reintroduction. Here it is important to recognize that in mature woodlands there will only be about 74–99 canopy dominant or codominant trees per hectare and <27 trees per hectare (>50 cm dbh) in a savanna. Thus, managing trees in savannas or woodlands is analogous to the silvicultural practice of crop-tree management in which a small number of trees is selected at an early age as the "crop" trees destined to become the future dominant trees at maturity, which can be harvested for profit when it is time to replace the overstory trees. Savanna and woodland crop trees are carefully cultured from a young age while the vast majority of trees in the stand are given no special attention or protection from fire. Thus, the noncrop trees are subject to removal arbitrarily by burning or deliberately by mechanical thinning, in fact they may need to be removed to maintain desired stocking for ground flora development.

Recruitment of seedling sprouts and grubs into the overstory requires a sufficiently long fire-free period for trees to grow large enough in size to gain resistance to being top killed by the next fire. In general, this may require a 20–30-year fire-free period depending on tree growth rates and source of reproduction (Johnson et al. 2009; Wakeling et al. 2011; Arthur et al. 2012). Oak stump sprouts are the fastest growing source of oak reproduction. One of the fastest growing oak species in the uplands is scarlet oak, and its stump sprouts can achieve diameters averaging 7.6 cm in 10 years when growing in the open (Dey et al. 2008). White oak, bur oak and post oak are slower growing species. Increasing overstory density reduces the growth of oak sprouts and lengthens the time needed for them to achieve a diameter, that is, bark thickness to become resistant to top kill by fire. These longer fire-free periods are not atypical in historical fire history records for the period before European settlement (Guyette et al. 2002; Guyette and Spetich 2003; Stambaugh et al. 2006a,b).

If producing marketable timber is also an objective, the fire-free interval may need to be 30 years or longer to allow a critical number of trees to become large enough to not be severely damaged by prescribed fire. If individual crop trees are given protection, for example, fuels are reduced around trees to minimize scarring or fires are ignited in a way to reduce severity in the immediate vicinity, then the greater area being renewed may be burned for other ecological reasons. Harvested trees can be directly felled away from trees to be retained to maintain overstory structure, thus placing tree crowns that may be left as slash on the site at a distance from retention trees. Felled-trees can also be whole-tree skidded to concentrate slash at the landing where it can be isolated and burned. Slash near retention trees can be cut to lie flat on the ground to minimize fire intensity near the bole of the retention tree, and to promote more rapid decay of the slash.

Fire damage to mature oaks was studied by Marschall et al. (2014) who found that there was a 10% loss in value 14 years after red oak trees (≥ 28 cm dbh at time of first scar) were fire scarred. They suggested that pole-sized trees (13–28 cm dbh) were at high risk of value loss if fire-scarred due to the length of time decay would have to develop before the trees reached maturity. After the recruitment phase, when sufficient trees >28 cm dbh are in

the overstory, care must be practiced when reintroducing prescribed burning to prevent mortality of those trees or to minimize damage to them.

In woodlands, reducing the stand stocking to approximately 20%–30% (Figure 20.2) is important for ensuring that the advance reproduction can recruit rapidly into the overstory, to minimize the time that fire must be withheld from the immediate area. Overstory stocking in savannas may be low enough to promote overstory recruitment if prescribed burning ceases for the requisite length of time for trees to develop resistance to fire induced top kill. Maintaining greater residual stocking levels will substantially reduce the growth rate of the recruiting trees (Dey and Hartman 2005; Dey et al. 2008), increasing the duration of the fire-free interval needed for allowing sufficient numbers of trees to grow larger than the threshold diameters identified above.

In large landscape burns there is less control over local fire behavior, in fact a goal may be to have variability in fire intensity and severity across the restoration area. It is also impractical to protect individual trees over thousands of acres at a time and fire is more indiscriminate in which trees are killed or damaged. In these situations, even-aged regeneration methods with area regulation are better suited for managing savannas and woodlands when it comes time to replace overstory trees. With area regulation, specific stands or land units are selected for regeneration and tending to replace the overstory. In selected areas, prescribed fire can be excluded from stands or land units with fire lines, roads, or natural fire breaks to protect the seedlings and allow for recruitment into the overstory. After a sufficient number of trees have recruited and are no longer in danger of being top killed or severely damaged, fire can be reintroduced along with other tending methods. Because fuels may accumulate for up to 12–15 years in the Central Hardwood Region in the absence of fire before a balance between fuel input and decomposition is achieved (Stambaugh et al. 2006b), care must be exercised when reintroducing fire and the initial burn should be low intensity to begin reducing fuel loading. This is especially important if activities such as tree thinning have contributed additional fuels.

Single-tree selection has not been a successful method for regenerating oak forests, except perhaps in the most xeric regions where shade tolerant competitors are few (Johnson et al. 2009). Also, it may be exceptionally difficult to manage woodlands using single-tree selection because this method requires the continuous establishment and recruitment of seedlings and small trees that are vulnerable to top kill and damage that can lead to substantial value loss by periodic fire. In general, the fire-free interval will need to increase as target stocking increases, since higher stocking will slow the growth of the recruiting trees; thus, lengthening the time it takes for desirable numbers to recruit into size classes less vulnerable to fire damage or top kill.

Longer rotations may be desired in savannas and woodlands than in forests. Rotations of 100 years are commonly used in hardwood forest management for optimizing the sustained production of timber, somewhat shorter (70–80 years) for red oak species to mitigate for oak decline losses. However, a longer rotation can be used for managing long-lived species where timber production is not a primary objective. For example, in the Ozark-St. Francis National Forest (OSFNF 2005), rotations for oak woodlands were extended to 140–160 years and for oak savannas to 180–200 years. Extending the rotation means that savannas and woodlands can be maintained in a mature state and tended with prescribed fire for a longer proportion of the rotation, especially if the overstory is dominated by white oaks. White oak, bur oak and post oak are longer-lived than red oak species, not as susceptible to oak decline, and are better able at compartmentalizing decay arising from fire injury (Fan et al. 2008; Dey and Schweitzer 2015). Lengthening the rotation decreases the size of the area that needs to be regenerated in each harvest period.

20.3 Conclusion

Open-structured oak savannas and woodlands were once prominent natural communities in eastern North America. They existed because of a long-history of frequent fire. Their distribution changed over time with changing climates, and human populations and cultures. With the advent of fire suppression, these communities succeeded to closed forests. Today, they are rare throughout the East. Restoration of oak savannas and woodlands has become a focus of land managers for a myriad of reasons including conservation of native biodiversity, quality habitat for game species and those of conservation concern, diversification of habitats at the landscape scale, for aesthetic and recreational purposes, and to restore ecosystem function. Restoration and maintenance of these systems requires active management. Reintroducing fire is fundamental to restoration, but other silvicultural practices are needed to efficiently manage vegetation composition and structure, and achieve desired future conditions. Research is needed to

- Identify desired future conditions.
- Assess innovative treatment combinations.
- Quantify threshold resource requirements for ground flora species.
- Understand relationships between woody structure and resource availability to ground flora.
- Predict vegetation response to variability in fire regime.
- Develop silviculture methods to prevent or mitigate invasive species.
- Model soil-topography influences on disturbance regimes and community composition and structure.
- Evaluate the interactive effects of fire-grazers/browsers-vegetation.

Management efforts to restore oak savannas and woodlands often precede research, provide early tests of innovative treatment combinations, and help to identify key questions. Monitoring to inform adaptive management is an important source of knowledge and part of the learning process. Restoring oak savannas and woodlands will help to expand the distribution of rare natural communities, conserve native biodiversity, create a more diverse landscape, provide habitat for wildlife species of concern, and should increase our options for responding to uncertain futures due to increasing human population, climate change, and invasive species.

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