Available online at www.sciencedirect.com
ScienceDirect
Forest Ecology and Management 254 (2008) 454–462
Forest Ecology and Management
www.elsevier.com/locate/foreco

Oak decline in the Boston Mountains, Arkansas, USA: Spatial and temporal patterns under two fire regimes

Martin A. Spetich [a,*], Hong S. He [b,1]

[a] Arkansas Forestry Sciences Laboratory, USDA Forest Service, Southern Research Station, P.O. Box 1270, Hot Springs, AR 71902, USA
[b] School of Natural Resources, University of Missouri-Columbia, 203 ABNR Building, Columbia, MO 65211, USA

Received 26 January 2007; received in revised form 27 August 2007; accepted 24 September 2007

Abstract

A spatially explicit forest succession and disturbance model is used to delineate the extent and dispersion of oak decline under two fire regimes over a 150-year period. The objectives of this study are to delineate potential current and future oak decline areas using species composition and age structure data in combination with ecological land types, and to investigate how relatively frequent simulated fires and fire suppression affect the dynamics of oak decline. We parameterized LANDIS, a spatially explicit forest succession and disturbance model, for areas in the Boston Mountains of Arkansas, USA. Land type distribution and initial species/age class were parameterized into LANDIS using existing forest data. Tree species were parameterized as five functional groups including white oak (*Quercus alba* L., *Quercus stellata* Wangenh., *Quercus muehlenbergii* Engelm.), red oak (*Qurecus rubra* L., *Quercus marilandica* Muenchh., *Quercus falcata* Michx., *Quercus coccinea* Muenchh.), black oak (*Quercus velutina* Lam.), shortleaf pine (*Pinus echinata* Mill), and maple (*Acer rubrum* L., *Acer saccharum* Marsh.) groups. Two fire regimes were also parameterized: current fire regime with a fire return interval of 300 years and a historic fire regime with an overall average fire return interval of 50 years. The 150-year simulation suggests that white oak and shortleaf pine abundance would increase under the historic fire regime and that the red oak group abundance increases under the current fire regime. The black oak group also shows a strong increasing trend under the current fire regime, and only the maple group remains relatively unchanged under both scenarios. At present, 45% of the sites in the study area are classified as potential oak decline sites (sites where red and black oak are >70 years old). After 150 simulation years, 30% of the sites are classified as potential oak decline sites under the current fire regime whereas 20% of the sites are potential oak decline sites under the historic fire regime. This analysis delineates potential oak decline sites and establishes risk ratings for these areas. This is a further step toward precision management and planning.

Keywords: Arkansas' Boston Mountains; Decline; Fire; LANDIS; Landscape model; Mortality; Oak; Risk; Simulation; Forest health

1. Introduction

"From earliest times, oaks have held a prominent place in human culture" (Johnson et al., 2002). "The oak tree is a symbol of American forests" (Smith, 2006). Oak-dominated forests represent 51% of eastern United States forests (Spetich et al., 2002). In the Boston Mountains of northern Arkansas, these complex ecosystems have been modified by both human and natural factors. These factors include drought, oak decline, and the interaction of humans and fire including fire suppression.

Native Americans have been an active part of the North American landscape for over 12,000 years. In fact, hardwood trees were the most important wood for native dwellings and oak was the most important species for fuelwood (Sabo et al., 2004), so these forests also played an important role in Native American culture. Native American forest and land management included the use of fire for agricultural clearing and driving game (DenUyl, 1954; Campbell, 1989; DeVivo, 1990; Reich et al., 1990; Denevan, 1992). Changes in Native American population likely led to changes in fire across the Boston Mountain landscape. For instance, during periods of relatively low-level population, the relationship of increasing population resulted in increasing fire frequency in the Boston Mountains (Guyette and Spetich, 2003; Guyette et al., 2006). By the mid 1800s, Euro Americans began settling in Arkansas and began clearing forest land for agriculture. They adapted many of the Native American methods, including the use of fire.

* Corresponding author. Tel.: +1 501 623 1180x105; fax: +1 501 623 0186.
E-mail addresses: mspetich@fs.fed.us (M.A. Spetich), HeH@missouri.edu (H.S. He).
[1] Tel.: +1 573 882 7717; fax: +1 573 882 1977.

0378-1127/$ – see front matter
doi:10.1016/j.foreco.2007.09.087

This activity resulted in a mean fire return interval of 2.9 years in the lower Boston Mountains during the period of regional development from 1881 to 1910 (Guyette and Spetich, 2003). However, since 1910, fire suppression efforts have increased the mean fire interval to greater than 80 years.

Fluctuations in historic use of fire and fire suppression have likely influenced species composition in the Boston Mountains (Spetich, 2004). However, successful regeneration of oak has been problematic (Spetich et al., 2002). Throughout Arkansas and parts of Missouri, the complexity of both regeneration and stand development has been further complicated by a recent oak decline event.

In the Boston Mountains, stem densities have nearly tripled since the 1800s and this is likely due, at least in part, to nearly a century of fire suppression (Foti, 2004; Spetich, 2004). These high stem densities, combined with drought, likely led to the recent and extensive oak decline event in Arkansas and surrounding areas (Spetich, 2004). From 1856 to 1986, there have been 57 oak mortality events recorded in the eastern United States (Millers et al., 1989). This included one in 1959 in the Ozark Mountains of Arkansas, one from 1980 to 1981 in northwestern Arkansas, and one from 1980 to 1986 in nearby Missouri. The most recent oak decline event occurred from 1999 to 2005 in Arkansas and Missouri and severely impacted approximately 120,000 ha in Arkansas' Ozark National Forest alone (Starkey et al., 2004).

Oak decline has been studied extensively. Manion (1991) describes oak decline as the outcome of the interaction of three major groups of factors: predisposing factors, inciting factors, and contributing factors. Predisposing factors include physiologic age and oak density; inciting factors include a long drought; and contributing factors include opportunistic insects such as oak borers and diseases like hypoxylon canker. During the most recent oak decline event in Arkansas' Boston Mountains, predisposing factors included forests of high tree density, mature trees, and dry, rocky soils; the inciting factor was a 3-year drought that began in 1998; and contributing factors included armillaria root rot, oak borers, hypoxylon canker, and defoliating insects. White oak is generally more drought and fire tolerant (Abrams, 2003) and has greater longevity (Guyette et al., 2004) than red oak and, therefore, better adapted to sites exhibiting oak decline symptoms. Starkey et al. (2004) notes that species in the red oak group are impacted by oak decline more rapidly and to a greater extent than the white oak group. Most studies of oak decline have been conducted at site scales at which descriptive and experimental approaches were used to identify relationships among declined oaks, site conditions, and insect or disease agents.

In this study, we treat oak decline as a dynamic process occurring at large spatial extents and involving multiple ecological processes from small site scales to large landscape scales. Our premises include: (1) oaks that are susceptible to decline can be delineated by species composition, age structure, and site conditions spatially, and (2) the dynamics of oak succession and mortality can be predicted using simulation models temporally. At site scales, succession of individual tree species and establishment of new species affect the dynamics of species composition and age structure and, thus, affect the dynamics of oak decline, whereas at landscape scales, the shifting mosaic of species composition and age cohorts caused by fire and environmental heterogeneity (land type association) will also affect the dynamics of oak decline.

The objectives of this study were to delineate potential current and future oak decline areas using species composition and age structure in combination with ecological land types, and to investigate how relatively frequent fires and fire suppression affect the dynamics of oak decline. A spatially explicit model (LANDIS) and modelling approach were employed to assess spatial and temporal variation in oak species composition and age structure, and in the spatial pattern of oak decline sites, under two fire regime scenarios.

2. Methods and materials

2.1. Study area

The Boston Mountains are the highest and most southern member of the Ozark Plateau Physiographic Province. These sharply dissected mountains run approximately 320 km east to west, forming a band 48–64 km wide north to south, with elevations from 275 m in valley bottoms to 762 m at the highest point. Most ridges are flat to gently rolling and generally less than 0.8 km wide. Mountainsides consist of alternating steep simple slopes and gently sloping benches. Vegetation across the landscape is a forest matrix with inclusions of non-forest. The forest is oak dominated. The study site consists of a 427,660 ha area within these mountains (Fig. 1). The landscape within the study area is hilly, highly dissected, and includes a variety of land types.

2.2. LANDIS model description

LANDIS is a spatially explicit, raster-based succession and disturbance model (Shifley et al., 1997; Mladenoff and He, 1999; He et al., 2005). In LANDIS, a heterogeneous landscape can be delineated into various forest ecosystems (land types or ecoregions, depending on the study scale). At a given focal resolution, such as within each forest ecosystem, environmental conditions such as climate and soils are assumed to be homogeneous, as is species establishment (Mladenoff and He, 1999; He et al., 1999). Each raster unit or cell is a spatial object that tracks: (1) the presence or absence of age cohorts of individual species parameterized from forest inventory data; (2) the land type (slope, aspect, position) a cell encompasses; (3) the establishment coefficients of all species in this cell; and (4) disturbance and harvest history if simulated. For each cell, non-spatial processes such as vegetation dynamics – including species birth, growth, and death – are simulated using species vital attributes (Table 1). ''Birth'' simulates a new species seeding in from another site, or on-site species establishment. ''Birth'' also simulates the vegetative reproduction for species that can reproduce by sprouting, based on vegetative reproduction probability and minimum age required for such reproduction (Table 1). ''Death'' typically simulates species

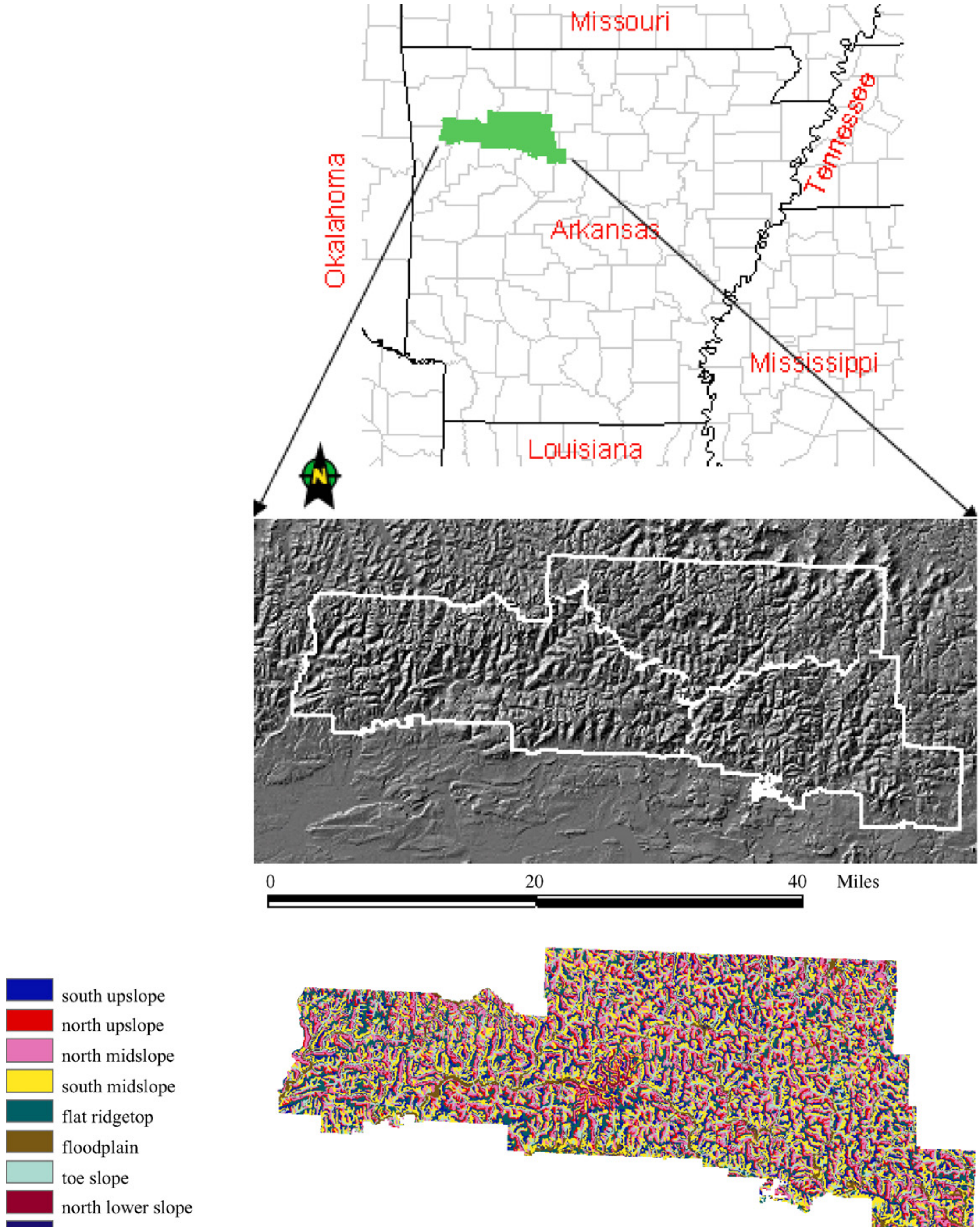

Fig. 1. The 427,660 ha study area is located in the Boston Mountains in northwest Arkansas as indicated by the green area on the top figure. The study area is hilly and dissected, with a variety of land types.

longevity-related mortality, applying only to the age cohort that reaches species longevity. ''Growth'' simulates species age class increments during each model iteration.

At a landscape scale, spatial processes such as seed dispersal, disturbance, and management (harvesting and fuel treatment) are simulated for each time-step (model iteration). The seed dispersal process consists of three distinct steps: seed travel, on-site checking, and seedling establishment. First, seed travels based on the exponential function of the effective and maximum seeding distances for a given species. Seed has a higher probability of reaching a site within the species effective seeding distance than beyond this distance (He and Mladenoff, 1999b). Second, when seed successfully arrives at a given site, the on-site checking procedure determines whether the species is able to establish itself based on other species that occur on the site and the shade tolerance rank of the seeding species relative to the species occupying the site. For example, black oak cannot seed into a site where maple is well established because the

Table 1
LANDIS species life history parameters adapted for Arkansas' Boston Mountains

Species group name	Longevity (years)	Mean maturity (years)	Shade tolerance (class)	Fire tolerance (class)	Effective seeding distance (m)	Maximum seeding distance (m)	Vegetative reproduction probability	MVP (years)
Maple	200	20	5	2	100	200	0.30	20
Shortleaf pine	200	20	3	4	90	120	0.95	10
White oak	250	20	3	4	70	800	0.10	90
Red oak	150	20	3	3	70	800	0.20	90
Black oak	80	20	3	2	70	150	0.10	60

Longevity is shorter on these dry–mesic to xeric sites than is generally reported for higher productivity, more fertile mesic sites in the northeastern U.S. Species included within each species group: Maple (*Acer rubrum* L., *Acer saccharum* Marsh.); Shortleaf pine (*Pinus echinata* Mill); White oak (*Quercus alba* L., *Quercus stellata* Wangenh., *Quercus muehlenbergii* Engelm.); Red oak (*Qurecus rubra* L., *Quercus marilandica* Muenchh., *Quercus falcata* Michx., *Quercus coccinea* Muenchh.); Black oak (*Quercus velutina* Lam.). Vegetative reproduction probability is the probability of stump sprouting up to the MVP year. MVP year is the mean vegetative reproduction probability year, representing the year when vegetative reproduction ends. For example, shortleaf pine vegetative reproduction probability (VRP) is 0.95 until year 10, after year 10 VRP is zero.

latter has a higher shade tolerance class. Finally, once a species is allowed to seed into the site, a uniform random number from 0 to 1 is drawn for comparison with the seed establishment coefficient to decide if seed can become established. A species can establish only when its establishment coefficient is greater than the random number drawn. Therefore, species with high establishment coefficients have higher probabilities of becoming established (Mladenoff and He, 1999).

Fire is simulated as a stochastic process based on the distributions of mean disturbance return intervals and mean disturbance sizes characterized for various land types (He and Mladenoff, 1999a). For example, fire occurs more frequently on south-facing slopes than on north-facing slopes. Fire is also a ''bottom-up'' disturbance in the sense that small, young trees are more susceptible to damage than large, older trees. LANDIS simulates fire intensity as five classes, from low-intensity ground fires to high-intensity crown fires. Correspondingly, tree species are grouped into five fire tolerance classes. Fire severity is the interaction of susceptibility based on species age classes, species fire tolerance, and fire intensity. Fire intensity is determined by the amount and quality of fuels. LANDIS uses vegetation composition and age to derive fine fuels, and derives coarse fuels from stand age and disturbance history. Fire intensity is then determined by the combination of fine and coarse fuels (He et al., 2004).

2.3. LANDIS input data and simulation scenario

The vital attributes presented in Table 1 that drive succession and dispersal processes in LANDIS were compiled based on the data parameterized for a nearby study area in the Missouri Ozarks (Shifley et al., 2000, 2006). Species and age cohorts for each stand within the study area were contained in the Ozark National Forest's GIS resource database. The database contains boundaries of individual stands, and species composition and stand age information for each stand. Tree species in the database were grouped into five species functional groups of interest: white oak (*Quercus alba* L., *Quercus stellata* Wangenh., *Quercus muehlenbergii* Engelm.), red oak (*Qurecus rubra* L., *Quercus marilandica* Muenchh., *Quercus falcata* Michx, *Quercus coccinea* Muenchh.), black oak (*Quercus velutina* Lam.), shortleaf pine (*Pinus echinata* Mill), and maple (*Acer rubrum* L., *Acer saccharum* Marsh.). The initial species composition map for LANDIS was derived by converting the stand database vector data to raster data.

Land type data were derived from the Ozark-St. Francis National Forest Ecological Classification System (ECS) Land Type Associations (LTAs). Within the study area, there are nine LTA classes based on slope position and slope orientation (Fig. 1). Species establishment coefficients for each land type were determined by using a combination of forest inventory analysis (FIA) data by various land types and an existing data set for the Missouri Ozark Highlands (Shifley et al., 2000).

Two fire scenarios were parameterized: one based on current fire intervals and one based on historic fire intervals. Guyette and Spetich (2003) found that the historic fire return interval in the study area was 2.9 years during the Euro American settlement period. This was derived by dating fire scars found on trees in this area. However, these fires occurred under conditions in which the fuels matrix was quite different than today. For instance, analysis of Arkansas' General Land Office surveys indicate that the forest was of much lower tree density which likely included substantial amounts of shrubs and grasses in the fuels matrix (Foti, 2004). Density of the forests of the 1990s, when our simulations begin, was nearly three times greater than that of the 1800s, resulting in heavily shaded conditions that are not conducive to the growth of grasses (Foti, 2004).

Moreover, fire return intervals derived in the 1800s fuels matrix would have recorded fires of a wider range of intensities than those simulated in LANDIS. The LANDIS fire module simulates five fire intensity classes accounting for high-intensity crown fires to low-intensity ground fires, but not fires that mainly impact shrubs and grasses. For example, it is possible for the lowest fire intensity class in LANDIS to kill some trees up to 20% of longevity (e.g., 30-year-old red oak and 16-year-old black oak). The lowest intensity fires simulated in LANDIS may overlap some, but not all shrub and grass fires recorded by fire scars. It is likely that fires that included these shrubs and grasses were prevalent in our study area in the 1800s; therefore, using the fire return interval derived from fire scar studies based on 1800s data would exaggerate fire effects on trees in LANDIS simulations and would not accurately reflect simulation starting conditions of the 1990s.

Since an accurate historic reference value that could be used for LANDIS is unknown, we chose a mean value of 50 years that approximates the higher end of the historic range of variability. Because this value changes by land type, the specific fire return intervals used in our simulations are 30 years for south-facing slopes, 60 years for north-facing slopes, and 50 years for ridges and flat areas.

Guyette and Spetich (2003) found that the fire return interval for the period of 1920–2000 was longer than 80 years. A study in the Missouri Ozarks suggested that the current fire return interval under fire suppression was between 300 and 415 years (Westin, 1992). Thus, we set the fire return interval for the current fire scenario at 300 years with reference to Westin (1992), and Guyette and Spetich (2003).

Having a simulation period of 150 years, while one of the fire return intervals is set to 300 years, bears some explanation. In the LANDIS model, fire ignition and spread involves stochastic components. In such a stochastic environment, some areas can be burned more than once while other areas may never burn. For the 300-year fire return interval, the total burned area during the 150 simulation years is roughly 50% of the study area (excluding water and other non-forest land types).

Species composition and land type maps were rasterized to a 60 m $\times$ 60 m cell size resolution, which yields 2204 columns, 905 rows and nearly 2 million pixels. To conduct the LANDIS (v 4.0) simulations, we started with a forest composition map with species age classes representing the initial configuration in the 1990s. We applied the simulation to the entire study area for 150 years, examining species composition, age structure, and spatial distribution of all major tree species under both current and historic fire scenarios.

Fire is the main stochastic process in this simulation. To ensure the assumptions that the fire return interval for historic and current fire scenario are correctly met, we used the model calibration approach that was published in He and Mladenoff (1999a). Model calibration involves multiple runs. For each run, we analysed model output and calculated the simulated fire return intervals for all land types. When the correct fire return intervals were not simulated, we adjusted the fire coefficient and re-ran the model until a close match was found (90%). This effort needed more than 10 model runs for each scenario but is more efficient than model replication.

2.4. Data analyses

Results of the simulation were summarized as percentage cover (the number of pixels in which a species group occurs divided by the total number of pixels) by each simulation decade and for each functional group. Sites where older black oak and red oak groups existed were considered potential oak decline sites (Starkey et al., 2004). These sites were occupied by cohorts of mature overstory trees at least 70 years old with no mature shortleaf pine group present. Those stands with mature shortleaf pine were not included due to uncertainty of data for those sites. To reflect the difference of oak decline by land type, south slopes and ridgetops with poor water and nutrient conditions were considered high-risk sites, north slopes were considered medium-risk sites, and toe slopes and floodplains were considered low-risk sites (Manion, 1991). Therefore, analysis of oak decline sites and their risk ratings combines both biotic (species and age) and abiotic (land type) factors.

3. Results

3.1. Species abundance in the landscape

The initial distributions indicated that red oak and white oak were the most abundant species in this area, shortleaf pine is abundant in the south, black oak occurs throughout the area but is less abundant, and maple dominates smaller areas sparsely distributed across the study area. Dominant trees of most species in these forest stands are 60–90 years old at the beginning of the simulation. Regeneration of shortleaf pine is obvious with younger age cohorts spreading across the landscape (Fig. 2).

The abundance of white oak will increase under both fire scenarios (Fig. 3). Under the historic fire scenario, white oak abundance is higher than it is under the current fire scenario, and white oak overtakes red oak as the most dominant species. Red oak is currently the most abundant species, occurring on 45% of the landscape in the beginning simulation year. Under the current fire scenario, red oak abundance increases to about 60% of the landscape at simulation year 150. By year 150, Red oak is less abundant under the historic fire scenario. Black oak also shows a strong increasing trend under both fire scenarios by year 150 (Fig. 3).

Shortleaf pine abundance is higher in the historic fire scenario than under the current fire regime. At year 0, shortleaf pine covers a low percentage ($\sim$25%) of the study area. Restoring the historic fire regime slowly increases its abundance, and this species covers about 33% of the landscape at 150 years (Fig. 3).

3.2. Temporal dynamics of species age classes

We classify the age cohorts into seedling ($\leq$10 years), sapling (11–30 years), pole (31–70 years), saw log (71–90 years), and old growth (>90 years). To focus on age groups that are susceptible to oak decline, we combine saw log and old growth into one older age group and seedling, sapling and pole are combined into a younger age group (Fig. 4). The domination of the older age group of red oak at the beginning of the simulation year suggests that the current potential for oak decline is high and will remain high for the next several decades until the older age group dies out and younger age groups dominate the landscape in about 80 years. The older age group will gradually build up again in the landscape toward year 150 as the succession cycle continues.

Under the historic fire regime, however, the proportion of the older age group of red oak decreases more rapidly than under the current fire regime (Fig. 4). This suggests that, although oak

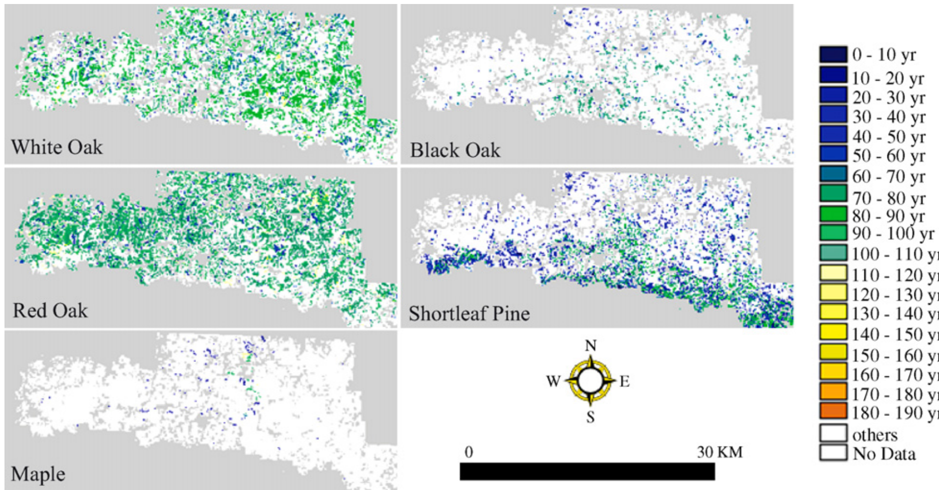

Fig. 2. The initial distributions of red oak, white oak, shortleaf pine, and maple groups in the study area. Most species are 60–90 years old at the beginning of the simulation.

decline will persist in the landscape, oak decline would affect a smaller proportion of the landscape under the historic fire regime. Moreover, oak decline in this area is primarily related to overabundance of the red oak old age groups, since the proportion of black oak in the older age groups is relatively small.

3.3. Oak decline sites under historic and current fire regime

Potential oak decline sites are further delineated spatially (Fig. 5). Results show that potential oak decline sites are scattered throughout the landscape. The distribution pattern of oak decline sites is largely dictated by the dissected land types and the distributions of black and red oak. Oak decline sites that aggregate as large patches (e.g., 4000 ha) are not common in this landscape (Fig. 5). At simulation year 0, about 45% of the landscape areas are classified as potential oak decline sites. These are the sites that have black and red oak present within a 60×60 pixel that are over 70 years old and have no shortleaf pine. Corresponding to the species age class dynamics, at simulation year 150, about 30% of the landscape area remain as potential oak decline sites under the current fire regime. However, under the historic fire regime at simulation year 150, only 20% of the area is classified for oak decline (Fig. 5).

The potential oak decline sites can be further broken down by land type because oak decline occurs more commonly on dry and poor land types than on moist land types (Fig. 5). Thus, at simulation year 0, besides 45% of the area being at risk, about 20% of the study area is coded as high-risk of oak decline (Fig. 5). The medium and low-risk categories of sites each occupy about 15 and 10% of the study area, respectively. At year 150 of the simulation sequence, the proportion of both high-risk and medium-risk oak decline area decreased to 12% under the current fire regime and only 7% under the historic fire regime. Low-risk sites occupy 6% of the study area under each fire scenario (Fig. 5).

Although spatial pattern of oak decline is largely affected by the dissected terrain and the occurrence of red and black oak, areas of concentrated high risk for oak decline can be identified in this spatially explicit study. For example, due to the close proximity of many oak decline sites, the circled areas in Fig. 5 are hot spots of oak decline throughout the 150-year simulation, suggesting that management could effectively be focused on these areas.

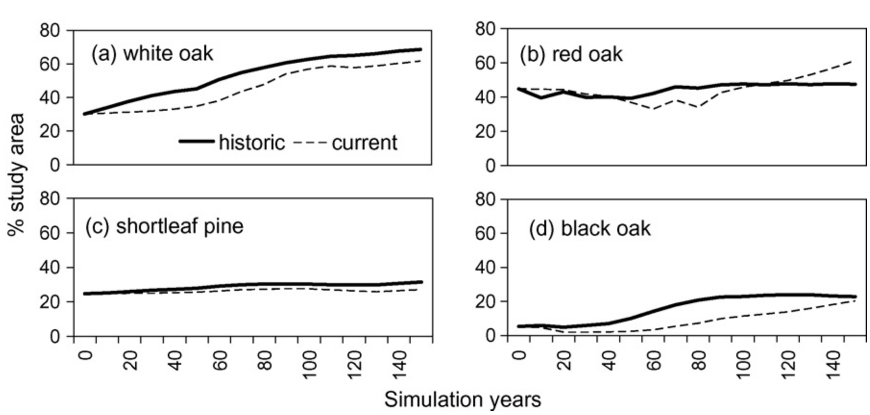

Fig. 3. Simulated species percentage of study area over time on the landscape for (a) white oak, (b) red oak, (c) shortleaf pine, and (d) black oak under the historic and current fire scenario.

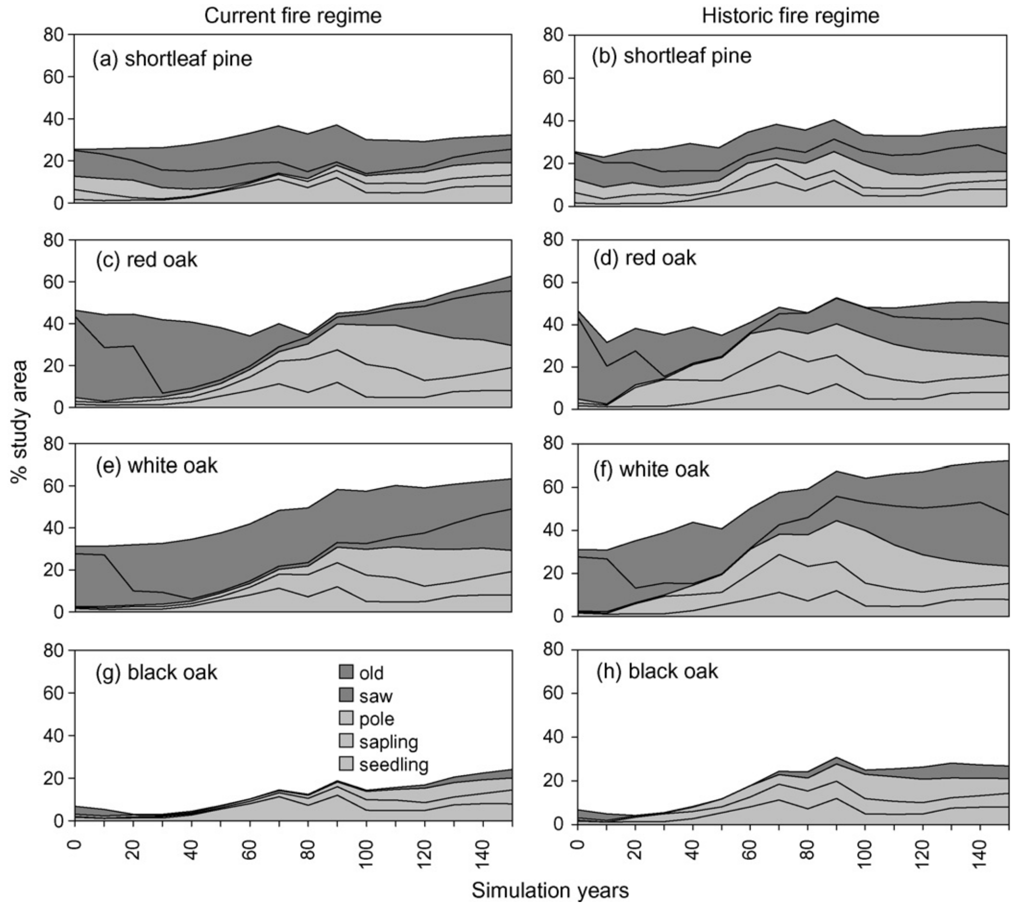

Fig. 4. Age cohort, species group as percent of study area by simulation year. We classify the age cohorts into seedling (≤10 years), sapling (11–30 years), pole (31–70 years), saw (71–90 years), and old (>90 years). To enhance the visual illustration, saw log and old are combined into the dark grey shade and the remaining younger age classes are combined into the light grey shade. Black oak has no old class because it's longevity is modelled at 80 years.

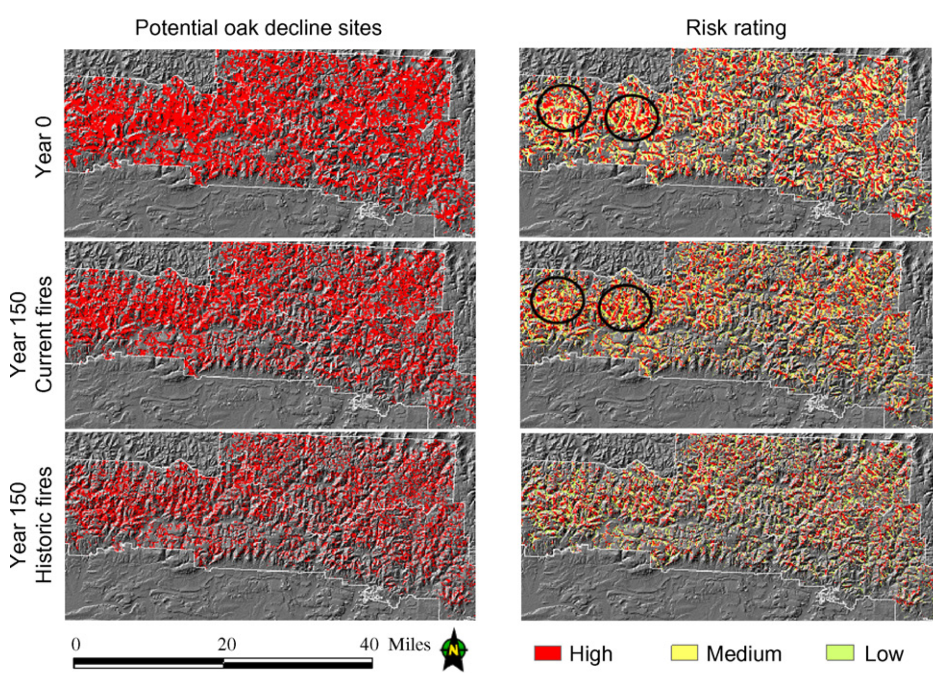

Fig. 5. Left panel shows current potential oak decline sites (year 0) in red color, and the simulated oak decline sites at year 150 under the current and historic fire scenario. Right panel shows oak decline sites risk rating at year 0, and year 150 for the current and historic fire scenario. Black circles indicate concentrated patches of high-risk oak decline. These patches are referred to as hot spots of potential oak decline.

4. Discussion

In this study, we were able to spatially delineate oak decline sites and predict oak succession and mortality dynamics across the 427,660 ha study area. At individual sites, species change occurred primarily through regeneration dynamics in response to fire. As mature old trees generally died from old age, fire primarily altered the composition of smaller trees, impacting regenerating tree composition to the greatest extent. Over time, many of these trees grew into the dominant species. At the landscape scale, the shifting mosaic of regenerating species and developing age cohorts produced by fire and the heterogeneity of land type association resulted in the patterns illustrated in Fig. 5.

Although both red and white oak currently have large proportions of their growing stock in older age classes (>70 years old), species in the red oak group are typically impacted to a greater extent than those in the white oak group during an oak decline event and, thus, are at a relative disadvantage (Starkey et al., 2004). In our simulations, impacts of future oak decline were mitigated by reducing the population of red oak on susceptible sites by altering regeneration and stand dynamics through more frequent fire, resulting in the more decline-resistant white oak species replacing red oak.

The extent of maple remained relatively level and low under both fire scenarios. However, on much more mesic sites, such as those in the northeastern and north central U.S., maple would likely have expanded under the present fire scenario (e.g. Spetich and Parker, 1998). In comparison, response of trees to site conditions of these Ozark oak decline study sites resulted in regeneration dynamics that generally did not favour maple beyond its present extent. This is not surprising considering that maple was initially sparsely distributed across the study area (Fig. 2).

The extent of the increase in white oak was quite impressive. Under both the current and historic fire scenarios, white oak will more than double its area from year 0 to 150, and it will overtake red oak in area covered across the study landscape. Under the historic fire scenario, white oak would occupy 53% of the study area and red oak 36%. This is consistent with a finding that mortality rates were much higher for red oak than for white oak in nearby Missouri during 1991–2002 (Shifley et al., 2006). When we take into consideration that forest management in the east is often successful at regenerating oak on only a few percent of the landscape, the differences that resulted from the historic versus the current fire scenarios are important. White oak's longer longevity, lower vulnerability to oak decline, and ability to survive under dryer conditions compared to red oak would all be advantages during stand development.

On dryer sites, shortleaf pine could gain the advantage, but based on our simulations, this would probably occur over a longer period of time than our 150-year simulation and most likely where periodic fire occurred. The starting landscape is the result of more than 80 years of fire suppression, so even when the historic fire regime is restored, the 150-year simulation does not result in substantial increase in shortleaf pine. However, in areas where fire is now being combined with harvesting and understory control, as in shortleaf pine-bluestem restoration projects (e.g. Liechty et al., 2004), the additional investment in energy results in a more immediate change to dominant pine structure.

Currently, red oak is the most abundant species in the landscape, and under the current fire scenario, the abundance of both red and black oak shows an increasing trend. Red oak remains the more abundant of the two species through the 150-year simulation. However, the predominance of older age groups of red oak at the beginning of the simulation suggests that there is potential for widespread oak decline and that episodes of decline could recur for several decades.

In these simulations, fire was a practical and effective way to modify the effects of oak decline and the more frequent historic fire scenario resulted in the smallest total area of potential oak decline. The high initial proportion of older red oak decreases more rapidly under the historic fire scenario than under the current fire regime, with the result that a smaller proportion of the future landscape is vulnerable to oak decline under the historic fire scenario. This suggests that prescribed fire may be a useful tool in the management of oak decline.

One difficulty involved in managing oak decline in this landscape is the scattered distribution of vulnerable sites. The distribution pattern is largely dictated by the dissected land types and the distributions of black and red oak. These scattered patterns present challenges to traditional silvicultural treatments. For instance, removing susceptible age cohorts may be difficult because oak decline sites that aggregate as large contiguous patches are not common in this landscape. However, areas with concentrated patches of high-risk oak decline can be delineated. For example, large contiguous hot spots of potential oak decline can be identified, so management might be effectively focused on these areas. Prescriptions such as historic landscape scale fire by itself, or in combination with other silvicultural practices, could be applied efficiently under these circumstances. Based on our simulations, fire mainly alters regenerating species composition, eventually resulting in overstory species that are less vulnerable to oak decline. Through inclusion of other silvicultural practices, such as improvement harvesting, this process would likely be accelerated.

From a broader perspective, if climatic conditions continue to become more severe, then relatively moister areas (such as north slopes, coves, and inner benches) that were effectively untouched by oak decline during and after past droughts could be affected by oak decline in the future. If this occurs, white oak could gain additional advantage over red oak. Restoration through the use of historic landscape scale fire applied now could promote site changes that would allow advance regeneration of white oak to become established in advance of climate impacts, thus mitigating some of the future forest health impacts of climate change.

5. Conclusions

Application of the LANDIS forest succession model indicates that oak decline is a process associated with forest succession, which operates at long temporal scales. This analysis delineates potential future oak decline sites and determines the risk ratings of those areas. This represents a move toward precision management and planning that is aimed at mitigating the effects of oak decline. Concentrated areas of high-risk patches for oak decline can be identified in a spatially explicit manner. These hot spot areas may be given high-priority status in landscape scale management strategies.

Acknowledgements

Funding support for this study came from U.S. Forest Service Southern Research Station and from the University of Missouri GIS Mission Enhancement Program.

Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at doi:10.1016/j.foreco.2007.09.087.

References

Abrams, M.D., 2003. Where has all the white oak gone? BioScience 53 (10), 927–939.

Campbell, J.J.N., 1989. Historical evidence of forest composition in the blue-grass region of Kentucky. Proceedings of the Seventh Central Hardwood Conference, General Technical Report NC-132, USDA Forest Service, pp. 231–246.

Denevan, W.M., 1992. The pristine myth: the landscape of the Americas in 1492. Ann. Assoc. Am. Geogr. 82, 369–385.

DenUyl, D., 1954. Indiana's old-growth forests. Proc. Ind. Acad. Sci. U.S.A. 63, 73–79.

DeVivo, M.S., 1990. Indian use of fire and land clearance in the southern Appalachians. Fire and the Environment: Ecological and Cultural Perspectives – Proceedings of an International Symposium. General Technical Report SE-69, USDA Forest Service, pp. 306–310.

Foti, T.L., 2004. Upland hardwood forests and related communities of the Arkansas Ozarks in the early 19th century. In: Spetich, M.A. (Ed.), Proceedings of the Upland Oak Ecology Symposium: History, Current Conditions, and Sustainability. General Technical Report SRS-73. USDA Forest Service, Southern Research Station, pp. 3–9.

Guyette, R.P., Spetich, M.A., 2003. Fire history of oak-pine forests in the lower Boston Mountains, Arkansas, USA. For. Ecol. Manage. 180, 463–474.

Guyette, R.P., Muzika, R.M., Kabrick, J., Stambaugh M.C., 2004. A perspective on quercus life history characteristics and forest disturbance. In: Spetich, M.A. (Ed.), Proceedings of the Upland Oak Ecology Symposium: History, Current Conditions, and Sustainability. General Technical Report SRS-73. USDA Forest Service, Southern Research Station, pp. 138–142.

Guyette, R.P., Spetich, M.A., Stambaugh, M.C., 2006. Historic fire regime dynamics and forcing factors in the Boston Mountains, Arkansas, USA. For. Ecol. Manage. 234, 293–304.

He, H.S., Li, W., Sturtevant, B. R., Yang, J., Shang, Z.B., Gustafson, E.J., and Mladenoff, D.J., 2005. LANDIS, a spatially explicit model of forest landscape disturbance, management, and succession—LANDIS 4.0 User's guide. North Central Research Station General Technical Report NC-263. USDA Forest Service, North Central Research Station, 93 pp.

He, H.S., Mladenoff, D.J., 1999a. Spatially explicit and stochastic simulation of forest landscape fire disturbance and succession. Ecology 80, 81–99.

He, H.S., Mladenoff, D.J., 1999b. Effects of seed dispersal in the simulation of long-term forest landscape change. Ecosystems 2, 308–319.

He, H.S., Mladenoff, D.J., Crow, T.R., 1999. Linking an ecosystem model and a landscape model to study forest species response to climate warming. Ecol. Model. 114, 213–233.

He, H.S., Shang, Z.B., Crow, T.R., Gustafson, E.J., Shifley, R.S., 2004. Simulating forest fuel and fire risk dynamics across landscapes—LANDIS fuel module design. Ecol. Model. 180, 135–151.

Johnson, P.S., Shifley, S.R., Rogers, R., 2002. The Ecology and Silviculture of Oaks. Wallingford, Oxon, New York.

Liechty, H.O., Luckow, K.R., Daniel, J.S., Marion, D.A., Spetich, M.A., Guldin, J.M., 2004. Shortleaf pine ecosystem restoration: impacts on soils and woody debris in the Ouachita Mountains of the southern United States. In: 16th International conference, August 24–26, Victoria, Canada, Society for Ecological Restoration, pp. 1–5.

Manion, P.D., 1991. Tree Disease Concepts. Prentice Hall, Inc., Englewood Cliffs, NJ.

Millers, I., Shriner, D.S., Rizzo D., 1989. History of hardwood decline in the eastern United States. General Technical Report NE-126. USDA, Forest Service, Northeastern Forest Experiment Station, 75 pp.

Mladenoff, D.J., He, H.S., 1999. Design and behavior of LANDIS, an object-oriented model of forest landscape disturbance and succession. In: Mladenoff, D.J., Baker, W.L. (Eds.), Advances in Spatial Modeling of Forest Landscape Change: Approaches and Applications. Cambridge University Press, Cambridge, UK.

Reich, P.B., Abrams, M.D., Ellsworth, D.S., Kruger, E.L., Tabone, T.J., 1990. Fire affects ecophysiology and community dynamics of central Wisconsin oak forest regeneration. Ecology 71, 2179–2190.

Sabo, III, G., Lockhart, J.J., Hilliard, J.E., 2004. The forest as a resource: from prehistory to history in the Arkansas Ozarks. In: Spetich, M.A. (Ed.), Proceedings of the Upland Oak Ecology Symposium: History, Current Conditions, and Sustainability. General Technical Report SRS-73. USDA Forest Service, pp. 33–35.

Shifley, S.R., Fan, Z., Kabrick, J.M., Jensen, R.G., 2006. Oak mortality risk factors and mortality estimation. For. Ecol. Manage. 229, 16–26.

Shifley, S.R., Thompson, F.R., Larsen, D.R., Mladenoff, D.J., 1997. Modeling forest landscape change in the Ozarks: guiding principles and preliminary implementation. General Technical Report NC-188. USDA Forest Service, North Central Experiment Station, St. Paul, Minnesota, pp. 231–241.

Shifley, S.R., Thompson III, F.R., Larsen, D.R., Dijak, W.D., 2000. Modeling forest landscape change in the Missouri Ozarks under alternative management practices. Comp. Electron. Agric. 27, 7–27.

Smith, D.W., 2006. Why sustain oak forests? In: Dickinson, M.B. (Ed.), Fire in eastern oak forests: delivering science to land managers, proceedings of a conference: General Technical Report NRS-P-1. USDA Forest Service, Northern Research Station, Newtown Square, PA, pp. 62–71

Spetich, M.A., Parker, G.R., 1998. Distribution of biomass in an Indiana old-growth forest from 1926 to 1992. Am. Mid. Nat. 139, 90–107.

Spetich, M.A., Dey, D.C., Johnson, P.S., Graney, D.L., 2002. Competitive capacity of *Quercus rubra* L. planted in Arkansas' Boston Mountains. For. Sci. 48, 504–517.

Spetich, M.A., 2004. Upland oak ecology—a synthesis. In: Spetich, M.A. (Ed.), Proceedings of the Upland Oak Ecology Symposium: History, Current Conditions, and Sustainability. General Technical Report SRS-73. USDA Forest Service, Southern Research Station, Asheville, NC, pp. 3–9.

Starkey, D.A., Oliveria, F., Mangini, A., Mielke, M., 2004. Oak decline and red oak borer in the Interior Highlands of Arkansas and Missouri: natural phenomena, severe occurrences. In: Spetich, M.A. (Ed.), Proceedings of the Upland Oak Ecology Symposium: History, Current Conditions, and Sustainability. General Technical Report SRS-73. USDA Forest Service, Southern Research Station, Asheville, NC, pp. 217–222.

Westin, S., 1992. Wildfire in Missouri. Missouri Department of Conservation, Jefferson City, MO, p. 161.