Developing a multiscale modeling framework to assess forest health risk based on the FIA data

by

Sunil Nepal

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Approved by

Zhaofei Fan, Chair, Associate Professor, College of Forestry, Wildlife and Environment
Edward F. Loewenstein, Associate Professor, College of Forestry, Wildlife and Environment
John S. Kush, Research Fellow IV, College of Forestry, Wildlife and Environment
Yaoqi Zhang, Professor, College of Forestry, Wildlife and Environment
J. Morgan Varner, Director of Research, Tall Timbers Research Station

Abstract

Maintaining tree vigor and forest health is critical as forest health risks increase over time. Eastern US forests have been losing quality in recent decades due to forest health risks such as tree mortality and nonnative invasive plant species (NNIPS) under extreme climate and disturbance regimes. These risks are often dynamic in space and time, taking specific spatiotemporal patterns. As such, there is a great need to integrate statistical and mathematical models to develop a multiscale geospatial modeling framework to address forest health risks affecting forest ecosystem sustainability. This study focused on two major leading forest health risks: 1) NNIPS in the Southern forestlands and 2) oak decline and mortality in the Ozarks in the Central US.

First, this study developed methods to quantify invasion severity indices and invasive stages, identified the best modeling scale by evaluating spatial autocorrelation, and investigated influencing factors associated with invasive severity and the invasibility of forests. Invasion severity increased over time and space in the region. County-level modeling unit best explained spatial autocorrelation and was chosen as the best modeling unit for the spatial analysis. The spatial lag model showed positive and statistically significant lag coefficients, implying neighbors' importance in explaining the invasion severity. Road length, neighbors' population density, and the number of households are likely to increase the invasion severity.

Second, this study explored species composition change, evaluated the effect of biotic and abiotic stresses on oak mortality, and examined species' response to extreme drought in terms of resistance, recovery, and resilience. The tree species composition in the Ozarks has changed; oak and pine dominance has reduced, and less commercially valuable species such as hickory, maple, and other eastern hardwood proportions have increased. The red oak group was at serious risk because of the higher mortality and lower ingrowth. Further, species responses to drought were significantly different among species groups. Fast-growing species, such as red oaks, had a lower resistance, recovery, and resilience; in contrast, slow-growing species, such as white oaks and hickories, had a higher resistance, recovery, and resilience.

In summary, this study prescribed tools to quantify and classify the severities of NNIPS; those will help develop and implement policies to control and mitigate the negative consequences of NNIPS. This study also provides insight into species' response to severe drought, which will help better understand potential species composition changes in the future and develop the adaptive mechanism to increase stand productivity.

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List of Abbreviations

BAI	Basal area increment
BAL	Basal area of larger tree
CART	Classification and regression tree
CDIEBKCD	Crown dieback code
CHF	Central hardwood forest
CR	Crown ratio
DBH	Diameter at breast height
Dr	During Drought
ECDF	Empirical cumulative distribution function
ECOSUBCD	Ecological subsection codes
ERC	Eastern red cedar
EVT	Existing vegetation type
FHM	Forest health monitoring
FIA	Forest Inventory and Analysis
GLO	General Land Office
HI	Hickories
HT	Height
LFH	Leading forest health
NRSC	Natural Resources Conservation Service
NNIPS	Nonnative invasive plant species
NOAA	National Oceanic and Atmospheric Administration
OES	Other eastern softwood
OESH	Other eastern soft hardwood
OT	Others
PDSI	Palmer Drought Severity Index
PLS	Public land survey
PreDr	Pre-drought
PostDr	Post-drought
RO	Red oaks
RRR	Resistance, recovery and resilience
SAF	Society of American Foresters
SP	Shortleaf pine
SPCD	Species code
SPGRPCD	Species group code
SRM	Society of Range Management
USDA	United States Department of Agriculture
WBD	Watershed Boundary Dataset
WO	White oaks

1 Introduction

1.1 Background and literature review

Forestlands in the Southern US have experienced various disturbances: Native forest species were harvested for settlement, agriculture, and timber between the 17th and 19th centuries and continue to be harvested as development continues in the region (Wear 2002). In the past, forest degradation and conversion were primary threats to forests; but in recent decades, forest health has become a primary concern (Woodall et al. 2011). The valuable forest resources are depleting their quality due to forest health risks. The United States Department of Agriculture (USDA) Forest Service identified leading forest health (LFH) indicators, including tree mortality, invasive species, and severe climatic events (USDA Forest Service 2009). These indicators show the threatening condition of US forest health and sustainability.

The forestlands in the Southern US are one of the major suppliers of raw forest products to the world's leading forest product industries. The region is considered a 'wood basket' of the US as it provides more than 18% of the world's pulpwood for paper and paper-related products and a significant amount of sawtimber (Hanson et al. 2010; Boby et al. 2014). The Southern forestlands' importance is limited to timber production and ecosystem services, including wildlife habitat, clean water, and carbon storage. Over 215 million forested acres, nearly 30% of the total forestland in the United States, are located in the South (Conner and Hartsell 2002). Diverse species, genetic resources, and high productivity characterize those forests. As elsewhere in the country, Southern forests face threats from climate change, biological invasion of nonnative species, and associated environmental stresses such as widespread drought, increasingly warmer temperatures, and extreme climatic events. Climate change, biological invasion, and environmental stresses can trigger interrelated health problems such as nonnative species invasion, increased wildfire risk, Southern pine beetle (Dendroctonus frontalis Zimmermann) outbreak, oak wilt (Bretziella fagacearum), and forest decline. These directly result in rapid and large-scale shifts in composition, ecosystem function and structure, and productivity (Boisvenue and Running 2006).

Oak decline is significant in the upland hardwood of the Southern US and Ozarks (Hoffard et al. 1995). The impact of oak decline in both regions, including the Ozarks, the most significant highlands region in the central US, is profound because it causes timber loss, low mast production for wildlife, and declines of scenic beauty (Hoffard et al. 1995). Oak decline often involves the interaction of predisposing factors such as climate, site quality, and tree age (Hoffard et al. 1995; Starkey et al. 2004). Extreme climatic events such as severe and long-lasting droughts are also linked with oak decline or mortality events in the Ozarks. Drought is expected to influence Southern forests' future forest health, productivity, and resilience (Dinon et al. 2013).

The USDA Forest Service, starting in 1929, has been collecting and reporting information on America's forests via Forest Inventory and Analysis (FIA) program periodically or annually. The FIA program has designed a protocol to evaluate the status, changes, and trends in indicators of forest condition on an annual basis by using data from ground plots and surveys, aerial surveys, and other biotic and abiotic data sources. FIA data coupled with remotely sensed vegetation and climate data provide a platform to monitor forest health under changing climate scenarios and disturbances and develop a climate-smart forest protection and planning system. A great need exists to integrate the USDA Forest Service protocol with statistical and mathematical models to develop a multiscale, hierarchical (from tree species/groups to stand, to landscape/ecosystem, and the entire region) modeling framework to address forest health risks that affect the sustainability of forest ecosystems. Therefore, this study uses the FIA data to study two crucial regional/sub-regional forest health risks: biological invasion of nonnative invasive trees and forest decline. This study is expected to fill technical and information gaps in climate change and forest health studies.

Forest cover is changing rapidly due to anthropogenic conversion of the land, while pest and pathogen outbreaks, nonnative invasive species, fire, and climate change are among the direct drivers that change Southern forest quality (Hanson et al. 2010). Nonnative invasive species have become one of the primary forest health risks and the most significant challenges to the natural plant community in Southern forestlands (Wear and Greis 2002; Miller et al. 2013). Despite the effort made by the USDA to prevent the introduction of nonnative invasive species, nonnative invasive species keep increasing throughout the landscape. Forest decline and resulting in high tree mortality are other important forest health risks that recurrently plague Eastern forests regionally or sub-regionally (e.g., the Ozarks ecoregion of Missouri and Arkansas)(Fan et al. 2012). Further, the etiology of oak decline is imperfectly understood, but oak crown dieback and tree mortality are usually attributed to an interrelated series of predisposing, inciting, and contributing factors (Manion 1991). Drought and its association with biotic and abiotic stressors are assumed to be responsible for tree mortality in several forest types in the US (Ambrose 2017).

The study adopts a multiscale modeling approach to address the three interrelated objectives (status, changes, and trends) to evaluate forest health conditions under a changing climate and to quantify the effect of multiscale factors on forest health indicators (Fig. 1.1).

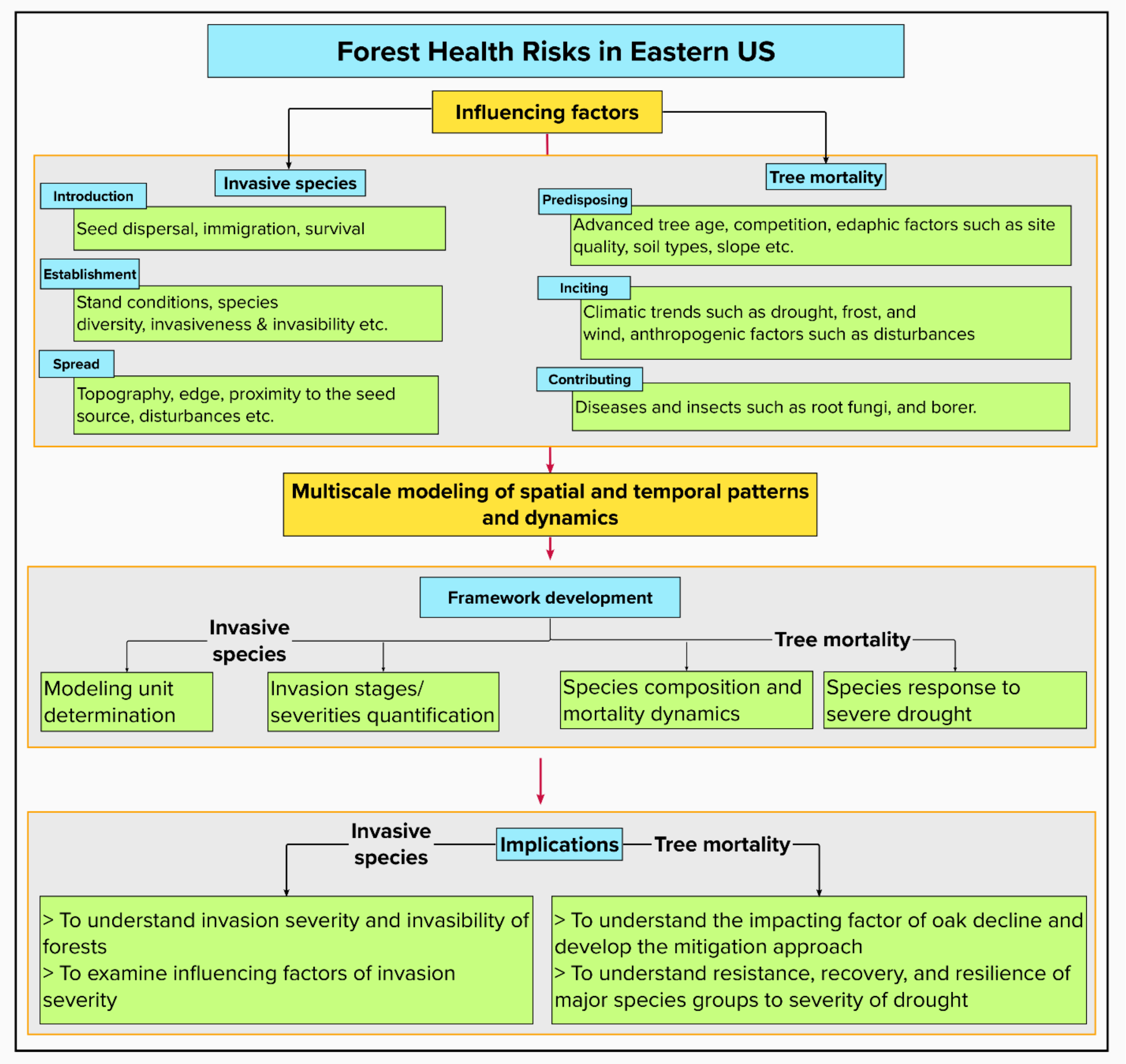

Figure 1.1. Multiscale geospatial modeling framework. Note: Factors responsible for tree mortality are adapted and modified from Manion (1991) & Starkey et al. (2004).

1.2 Objectives

This study has four primary objectives.

1. to develop a modeling framework to identify influencing factors of the abundance and presence of the major nonnative invasive species and to quantify their impacts on Southern forest,

2. to understand the invasion stages and invasiveness of the major nonnative invasive tree species by using a quantitative and analytical approach to map the invasibility of recipient ecosystems,
3. to evaluate the effect of biotic and abiotic stressors on oak mortality to identify mitigation methods, and
4. to assess trees' resistance, recovery, and resilience under severe climatic stress.

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2 Determining spatial units for modeling nonnative invasive plant species spread and factors in Southern forestlands: Alabama as an example

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2.1 Introduction

Nonnative invasive plant species (NNIPS) significantly impact the U.S. economy and native ecosystems (Lodge et al. 2006). Despite control efforts, invasions have increased noticeably throughout the landscape. Southern forest ecosystems are particularly vulnerable to invasions because invasive species can quickly alter native habitats, causing a loss of forest productivity and diversity (Miller et al. 2013). Southern US forest ecosystems are experiencing increasing threats from invasive plant species, which displace native species, degrade fundamental forest structure and functionality (Moser et al. 2009; Wang et al. 2015), and damage the environment and local economies in the southern U.S. (Hussain et al. 2008; Zhai et al. 2018). Moreover, invasive species are expected to increase in geographic range over time, causing large-scale ecological instability of native forests and making control and mitigation measures more costly and challenging (Camarillo et al. 2015).

Most vascular plant communities are susceptible to invasion (Sax et al. 2005). Nonnative invasive species are increasing at an alarming rate throughout southern US forests; however, only partial monitoring and a few invasive control practices are being implemented (Miller 2003). Miller (2003) documented 33 NNIPS rapidly growing in the southern US, which includes Japanese honeysuckle (*Lonicera japonica* Thunb), kudzu (*Pueraria montana* (Lour.) Merr.), privet (*Ligustrum* L.), tree-of-heaven (*Ailanthus altissima* (Mill.) SwingleCh), silk-tree (*Albizia julibrissin* Durazz.), and Chinese tallow tree (*Triadica sebifera* (L.) Small). Most of these invasive species were introduced into the United States because of their ornamental or multipurpose use (Bruce et al. 1997; Miller 2003; Hodges et al. 2016). These species are more vigorous in the introduced habitat than in their native habitats (Siemann and Rogers 2001; Taylor et al. 2016), are tolerant to multiple adverse conditions, have various means for seed dispersal and propagation, and grow more rapidly than most native species (Bruce et al. 1997; Miller 2003; Hodges et al. 2016). Two driving factors make invasive species more vigorous in introduced areas: advantageous competitive capacity and lack of natural enemies in newly introduced areas (Callaway and Aschehoug 2000). Further, disturbed habitats, such as ecosystem edges, including transportation networks, are susceptible to invasion (Simberloff 2013).

Alabama is one of the most forested states in the U.S., with more than 68 % covered with diverse and highly productive forests (Hartsell 2018). As such, Alabama depends on its forests for its economy and the well-being of its residents (AFC 2014). As elsewhere in the U.S., Alabama’s forests are experiencing increasing threats due to the invasion of NNIPS. The Alabama Invasive Plant Council (ALIPC) has identified these seven NNIPS as extended and densely infested species in the state’s managed forested lands: Chinese tallow tree, privet, Japanese honeysuckle, Japanese climbing fern, kudzu, cogongrass (*Imperata cylindrica* (L.) Beauv.), and Nepalese brown top (*Microstegium vimineum* (Trin.) A. Camus) (ALIPC 2012). These invasive species have plagued vast forestlands and become a severe problem for forest landowners in Alabama. The invasion and spread of NNIPS in forestlands are driven by multiple factors interacting with one another across different spatial and temporal scales (Theoharides and Dukes 2007). Accordingly, the distribution of invasive species is non-stationary and varies dramatically in space and across ecosystems. In addition to climatic and geographic factors (e.g., temperature, precipitation, site productivity, forest types), socioeconomic factors (e.g., ownership, land-use change, road density, and resource management intensity and history) have also been shown to be significant determinants of invasive incidence (Hedman et al. 2000; Hussain et al. 2008). It has often been observed that native and invasive species abundance follows a negative relationship at fine scales but roughly a positive relationship at broad scales (e.g., a forest landscape/watershed) (Fridley et al. 2007).

Hydrologic units are spatially homogeneous in mass movement and energy exchange. They are better determinants of significant hydrological, ecological and socioeconomic processes influencing the invasion and spread of invasive species (Mirchi et al. 2010; Band et al. 2012). Ecological units such as provinces, sections, and subsections are generally defined based on climate, vegetation, terrain, and elevation (Bailey 2016). In contrast, counties are mainly political and administrative boundaries. Our primary goal was to identify the best modeling units and develop the geospatial model to examine invasion severity in Alabama. Thus, in this study, we 1) identified the best hierarchical geospatial modeling unit to map the extent and spread of NNIPS; and 2) identified and quantified land uses that significantly affect the invasion and spread of NNIPS in Alabama’s forestlands. These analyses provide baseline information on invasive species modeling and suitable management units for developing better prevention and management strategies to control or mitigate the negative impact of invasive species on Alabama’s forestlands.

2.2 Methods

2.2.1 FIA data

We obtained data from more than 5,000 FIA plots/subplots for the state of Alabama (FIA DataMart 2019). The FIA data were downloaded from the USDA Forest Service’s Forest Inventory and Analysis (FIA) DataMart (Forest Inventory and Analysis Program 2018). Alabama has 5,657 FIA plots, approximately 3 × 3 mile spacing throughout the state, that is, one plot for roughly every 6,000 acres, to collect forest information (McRoberts et al. 2005, Hartsell 2018). We used plots measured three times between 2001-2019 on accessible forested lands. Each FIA plot has four nested subplots; there were more than 22,000 subplots in each measurement throughout the state. Some of these were not accessible, and some were not remeasured; as such, we found that 15,240 subplots were both accessible and remeasured three times. Invasive species information such as presence and cover percent was obtained from the "AL_INVASIVE_SUBPLOT_SPP" table, and plot level information was obtained from the "AL_PLOT" and "AL_COND" tables. Publicly available FIA data provide the approximate latitude and longitude due to a privacy provision. Most annual plots were within +/- half a mile of the approximate locations, and some plots were swapped (Lister et al. 2005). Thus, we obtained actual FIA plot locations from the USDA Forest Service and used them for this analysis.

2.2.2 Land-use, forest types and demographic data in Alabama

We used the LANDFIRE (Landscape Fire and Resource Management Planning Tools) dataset for this analysis. The 2016 existing vegetation type (EVT) data were downloaded from https://landfire.gov/viewer/viewer.html. The EVT provides information about the existing distribution of plant communities (Ryan and Opperman 2013). Data come in 30m × 30m pixels and are available for the conterminous US. We used SAF_SRM classes, the crosswalk to Society of American Foresters (SAF), and the society for Range Management (SRM) cover types. For the state of Alabama, LANDFIRE classified 48 SAF/SRM classes. We reclassified these into six major categories (Fig. 2.1).

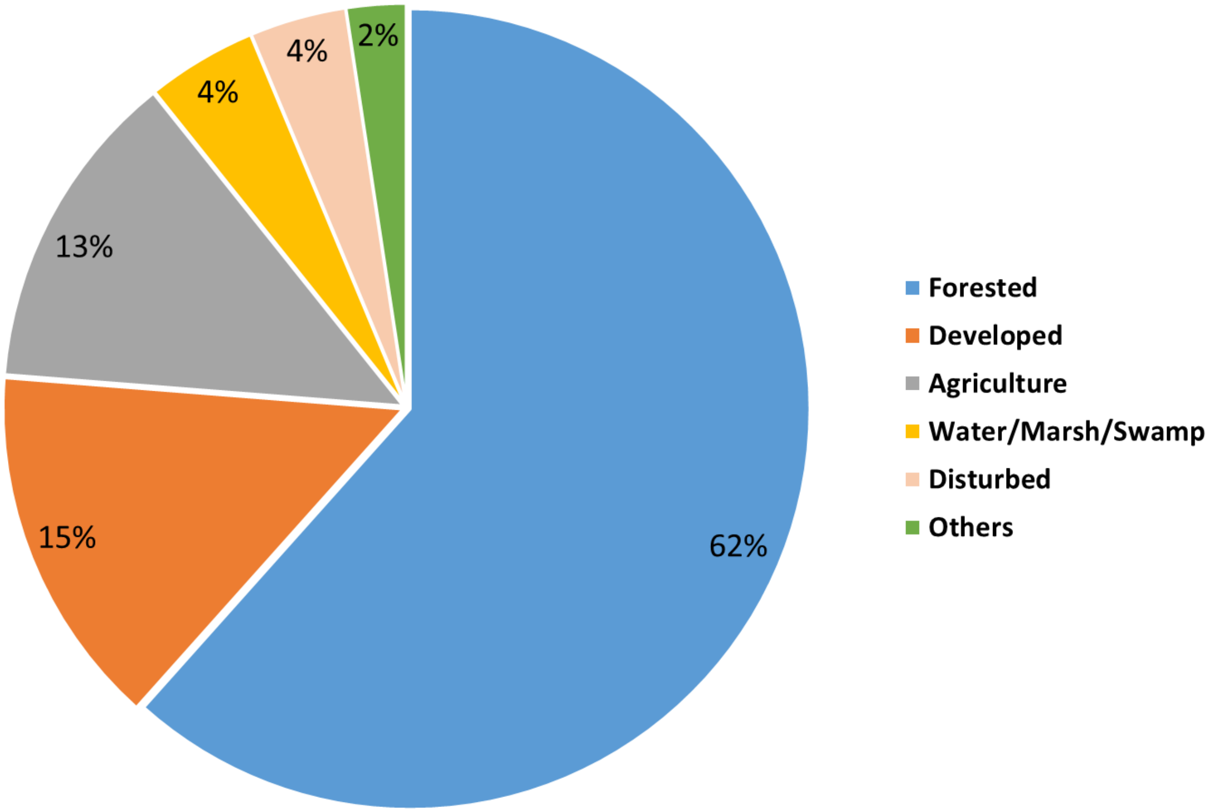

Figure 2.1 Land-use in Alabama based on LANDFIRE data (SAF/SRM classes in 2016)

Forest ownership per modeling unit was generated using FIA data. Additionally, the USDA Forest Service's forest type raster imagery (https://data.fs.usda.gov/geodata/rastergateway/forest_type/index.phpv), which was based on 2002 and 2003 inventory and prepared by Ruefenacht et al. (2008) The geoprocessing was done in ArcGIS Pro (version 2.5.0); the identity tool was used to overlap modeling units with source data (i.e., land-use and forest types) and the area on each land-use class and forest type within the modeling unit was obtained. The cover percent by each land-use class and forest type was calculated and used as an independent variable in the spatial lag model. Demographic data per modeling unit were prepared using 2016 U.S. Census data. Further, road density and length per unit were made based on the interstate, U.S. Highway, and state Highway information available on the ESRI website(https://www.arcgis.com/home/item.html?id=fc870766a3994111bce4a083413988e4).

Details of these variables can be found in Table 2.1.

Table 2.1: Variables used in the spatial lag model in this analysis

Variable	**Variable definition**	**Data types**	**Data description and source**
public_own_pct	percent of publicly owned forest (0-100)	Ownership	FIA DataMart (https://apps.fs.usda.gov/fia/datamart/CSV/datamart_csv.html)
rd_length	total length of major roads (interstate, U.S., and state highways) (miles)	Roads	Esri (https://www.arcgis.com/home/item.html?id=fc870766a3994111bce4a083413988e4)
rd_density	road density in each county (miles/sq.mile)		
elm_ash_cot	Elm/Ash/Cottonwood Group area in percent (0-100)	Forest groups	USDA Forest Service (forest types/groups are based 2002 & 2003 data) https://data.fs.usda.gov/geodata/rastergateway/forest_type/index.php
lob_short	Loblolly/Shortleaf Pine Group area in percent (0-100)		
long_slash	Longleaf/Slash Pine Group area in percent (0-100)		
oak_gum_cypress	Oak/Gum/Cypress Group area in percent (0-100)		
oak_hickory	Oak/Hickory Group area in percent (0-100)		
oak_pine	Oak/Pine Group area in percent (0-100)		
lob	Loblolly Pine area in percent (0-100)	Forest types	
Lob_hard	Loblolly Pine/Hardwood area in percent (0-100)		
long	Longleaf Pine area in percent (0-100)		
mix_hard	Mixed Upland Hardwoods area in percent (0-100)		
sw_no_wo	Sweetgum/Nuttall Oak/Willow Oak area in percent (0-100)		
wo_ro_hi	White Oak/Red Oak/Hickory area in percent (0-100)		
pop_2010	population in 2010	Demographic	2010 U.S. Census demographic information. Downloaded from Esri (https://hub.arcgis.com/datasets/esri::usa-counties/about)
pop_den_2010	population density in 2010 (population/sq.mile)		
households	number of households in 2010		
pop_2010_nbh	avg. population of neighborhood counties		
pop_den_2010_nbg	avg. population density of neighborhood counties (population/sq.mile)		
ag	agriculture lands/sq.mile	Land-use	Land-use in 2016 from downloaded from LandFire (https://landfire.gov/viewer/viewer.html)
developed	developed lands/sq.mile		
disturbed	disturbed lands (recently logged or burn)/sq.mile		
forest	forest lands/sq.mile		
others	others lands/sq.mile		
water	water/sq.mile		
area	total county area (sq.mile)		

2.2.3 Multiscale modeling units

The FIA data were aggregated into multiple spatial units: five levels of hydrological units, three levels of ecological units, and a county level. Watershed Boundary Dataset (WBD) was downloaded from the USDA Natural Resources Conservation Service (https://datagateway.nrcs.usda.gov/). WBD classifies watersheds into six levels based on the size of the watershed. Three-level ecological units' shapefiles: province, section, and subsection, were obtained from the USDA Forest Service website (https://data.fs.usda.gov/geodata/edw/datasets.php).

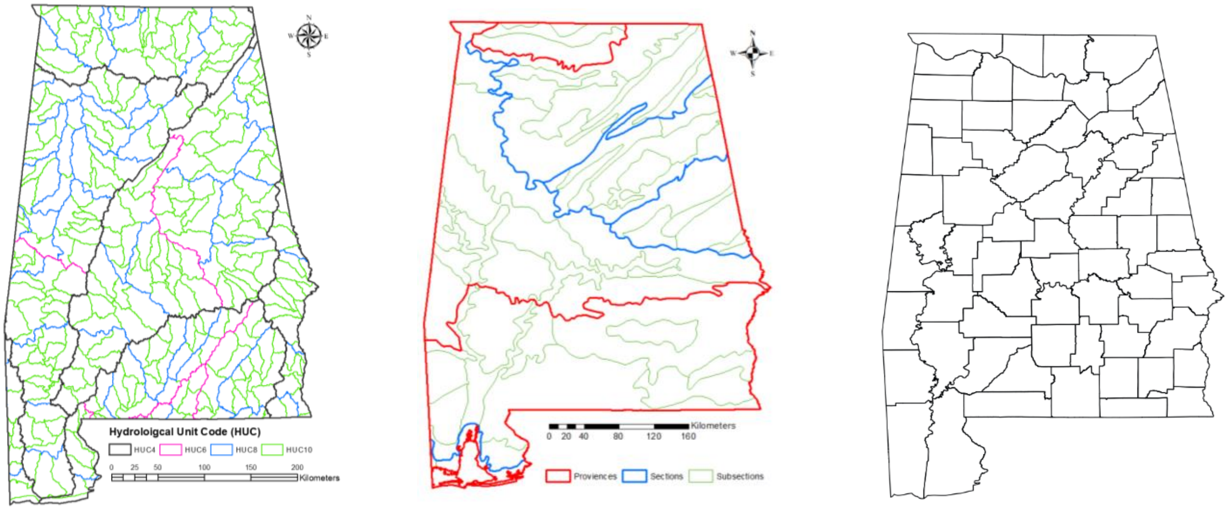

Figure 2.2 Maps of levels of hydrological units (left), three levels of ecoregions (middle), and counties (right) in Alabama. Five levels of hydrological units include: HUC4 to HUC12 (coarse to fine scale); three levels of ecological units include: Province, sections, and subsections (coarse to fine scale).

2.2.4 Data preparation

Based on FIA data, we identified seven top NNIPS in Alabama: Japanese honeysuckle, privet, Japanese climbing fern, sericea (*Lespedeza cuneata* (Dum. Cours.)), silk-tree, Chinese tallow tree, and rose (*Rosa* L.). These species were found at least in 200 subplots (i.e. >1.3% of all subplots) measured between 2013 and 2019. A binary variable was created to represent the presence or absence of invasive species from each subplot. To capture temporal changes, we traced the 15,240 subplots remeasured three times. Alabama's FIA plots measured between 2001 and 2005 (i.e., in cycle 8) were treated as first-time measurement (i.e., T1), 2006-2012 (i.e., cycle 9) as second-time measurement (i.e., T2), and 2013-2019 (i.e., cycle 10) as third-time measurement (i.e., T3). The number of infested subplots by each species and per measurement time is summarized in Table. 2. Using all major nonnative invasive plant species, presence probability (eq.1) and average cover percent (eq.2) were calculated for each modeling unit. Presence probability and cover percent were used to calculate the invasion index (eq.3).

$$\mathrm{P_{ij}} = \frac{\mathrm{S_{ij}}}{\mathrm{N_{ij}}} \qquad (1)$$

Here, $\mathrm{P_{ij}}$ is the presence probability of NNIPS for a modeling unit *i* at measurement *j*. $\mathrm{S_{ij}}$ is the number of infested subplots in a modeling unit *i* and measurement time *j*, and $\mathrm{N_{ij}}$ is the total number of subplots in a modeling unit *i* at measurement time *j*. Here, measurement time *j* = T1, T2, and T3, and modeling unit *i* = Hydrological units (HUC4, HUC6, HUC8, HUC10, and HUC12), county, and ecological units (province, section, and subsection)

$$\mathrm{C_{ij}} = \frac{\sum \mathrm{c_{ij}}}{\mathrm{N_{ij}}} \qquad (2)$$

Here, $\mathrm{C_{ij}}$ is the average cover of nonnative invasive species for a modeling unit *i* at measurement time *j*. $\sum \mathrm{c_{ij}}$ is sum of cover percent of all invasive species from all subplots found in a modeling unit *i* at measurement time *j*, and $\mathrm{N_{ij}}$ is the total number of subplots in a modeling unit *i* at measurement *j*. The invasion index, measuring the invasion severity will be calculated as,

$$\text{Invasion index} = \mathrm{P_{ij}} \times \mathrm{C_{ij}} \qquad (3)$$

If any polygons in the selected modeling units had missing values, those missing values were adjusted with imputed values based on the Inverse Distance Weighted (IDW) imputation method. Addressing missing values is critical because some modeling units are small- as such, no FIA plots fall under them, resulting in NULL or missing values.

2.2.5 Data analysis

Spatial and temporal trends of nonnative invasive species were examined and graphically illustrated. Invasive species' occurrence and severity often follow spatial patterns (With 2002; Fan et al. 2018). Moran's I test statistic, proposed by Moran (1950), can be used to find such a pattern. A positive Moran's I statistic suggests clustering, meaning the data have a positive spatial autocorrelation (Bivand et al. 2008). In this analysis, we chose the modeling unit with the highest Moran's I statistic, suggesting a wider spatial variation in the data. In such a case, we need a geospatial model to account for those spatial autocorrelations. Thus, influencing factors for invasion indices were modeled using the spatially lag model- spatial autoregressive (SAR) (eq. 4). The SAR model assumes lag effect on the dependent variable (i.e., invasion index) by neighbors. For example, nearby counties' invasion indices affect a county's invasion index.

$$\mathrm{Y} = \rho\mathrm{WY} + \mathrm{X}\beta + \varepsilon \ldots\ldots\ldots\ldots..(4)$$

Here, Y is the dependent variable (invasion index), X are independent variables (land-use, forest types, road density, population density, etc.), ρ is a parameter of spatial lag coefficient, $\boldsymbol{W}$ is the spatial weight matrix, and ϵ is residuals.

Geoprocessing, data preparation, and visualization were done in Esri ArcGIS Pro 2.5.0 software and R (R Development Core Team, 2014). Moran's I and spatial lag tests were conducted using the "spdep" package (Bivand et al. 2015) in R. The best fitted spatial lag model was identified by choosing the lowest Akaike information criterion (AIC).

2.3 Results

2.3.1 Invasive species increased overtime

The distinct species count of NNIPS in Alabama's forestlands increased over time. In total, 25, 26, and 33 unique NNIPS were recorded in T1, T2, and T3, respectively. Infestation % increased over time; during the first measurement, only 41.1% of total remeasured subplots were infested, but during the third measurement, the infestation rate had risen to 54.8% (Table 2.2). Alabama forestlands are not only increasing in infestation %, but also adding unique invasive species over time. For instance, in T1, there was an average of 1.32 unique NNIPS in the infested subplots. However, in T3, nearly 27% more (i.e., 1.68 unique nonnative invasive species) were found in Alabama forestlands.

Table 2.2 Invasive presence percent overtime in Alabama forested lands

Measurement	Year	Total subplots	Infested subplots	Infestation %	Invasive species count	Avg species count per subplot (in infested subplots)
T1	2001-2005	15240	6268	41.1%	8251	1.32
T2	2006-2012	15240	7744	50.8%	11405	1.47
T3	2013-2019	15240	8347	54.8%	14020	1.68

The number of subplots by invasive species type in Alabama in each cycle is given in Table 2.3. The abundance of all major nonnative invasive species increased over time. Japanese honeysuckle was the most abundant species, followed by privet and Japanese climbing fern. The change in Japanese honeysuckle abundance over time was nearly 19% between T1 and T2 and 5% between T2 and T3. The second most abundant species, privet, increased by almost 80% between T1 and T2 and 36% between T2 and T3. The rate of increment on less abundant species increased at a higher rate. In general, the average cover percent of these species increased over time. However, the average cover percent of Japanese honeysuckle decreased between T2 and T3. Detail of abundance and cover percent can be seen in Fig. 2.3 and Table 2.3.

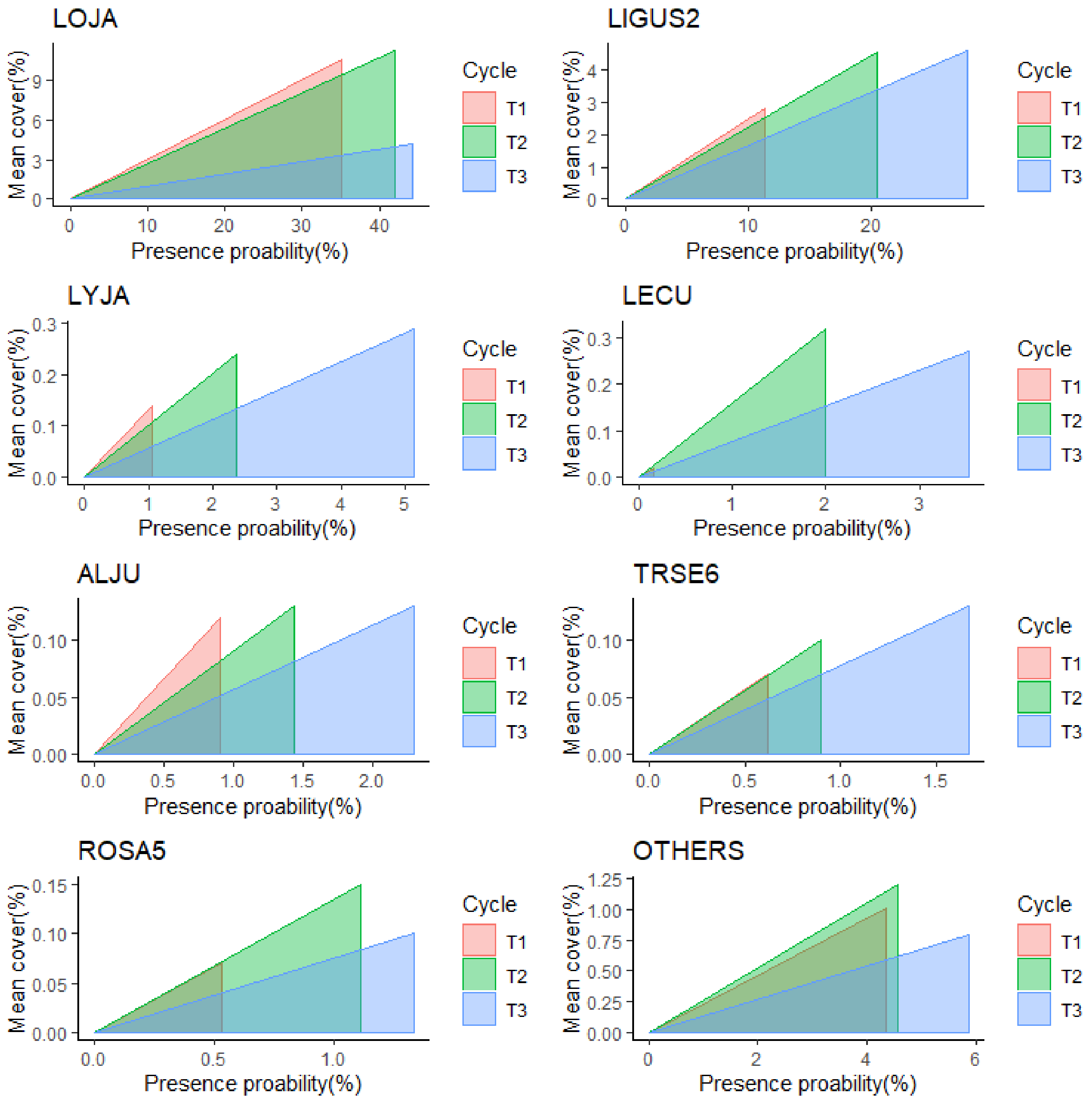

Figure 2.3 Presence probability and cover percent of major NNIPS in Alabama. NNIPS names in this figure based on FIA Vegetation Species Code (VEG SPCD); LOJA (Japanese honeysuckle), LIGUS2 (Privet), LYJA (Japanese climbing fern), LECU (Chinese lespedeza), ALJU (Silk-tree), TRSE6 (Chinese tallow tree), ROSA5 (Rose), and OTHERS (all other nonnative invasive plant species).

Table 2.3: Number of infested subplots by major nonnative invasive plant species in Alabama's forestlands

FIA VEG_SPCD*	Common name	Latin name	Form	Infested subplot count			Presence probability (%)			Mean cover (%)		
				T1	T2	T3	T1	T2	T3	T1	T2	T3
LOJA	Japanese honeysuckle	*Lonicera japonica* Thunb	Vine	5348	6400	6751	35.09	41.99	44.30	10.58	11.18	4.20
LIGUS2	Privet	*Ligustrum* L.	Shrub	1740	3122	4248	11.42	20.49	27.87	2.80	4.55	4.59
LYJA	Japanese climbing fern	*Lygodium japonicum* (Thunb.)	Fern	162	360	781	1.06	2.36	5.12	0.14	0.24	0.29
LECU	Sericea	*Lespedeza cuneata* (Dum. Cours.)	Forb	23	303	537	0.15	1.99	3.52	0.02	0.32	0.27
ALJU	Silk tree	*Albizia julibrissin* Durazz.	Tree	139	219	351	0.91	1.44	2.30	0.12	0.13	0.13
TRSE6	Chinese tallow tree	*Triadica sebifera* (L.) Small	Tree	95	137	255	0.62	0.90	1.67	0.07	0.10	0.13
ROSA5	Rose	*Rosa* L.	Shrub	81	169	203	0.53	1.11	1.33	0.07	0.15	0.10
Others				663	695	894	4.35	4.56	5.87	1.01	1.21	0.80

*Vegetation species code (VEG_SPCD).

2.3.2 Spatial and temporal pattern of major invasive species in Alabama

The spatial and temporal visualization of the presence probability of major nonnative invasive species in Alabama is in Figs. 2.4 and 2.5. Figure 2.4 represents aggregated presence probability of all species over time across space. The presence probability of individual species can be seen in Fig. 2.5, which shows the presence probability of individual species based on T3 (i.e., 2013-2019) records.

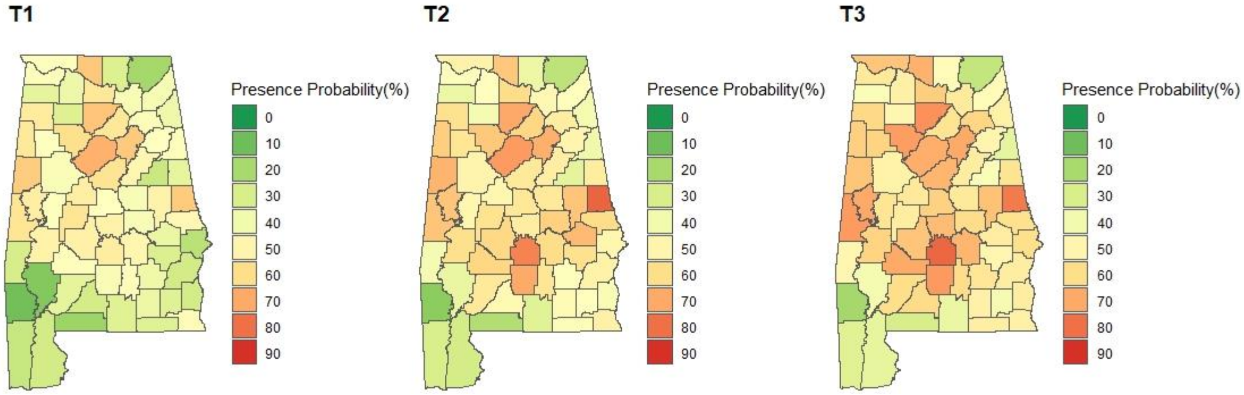

Figure 2.4: Presence probability of all nonnative invasive plant species overtime in Alabama

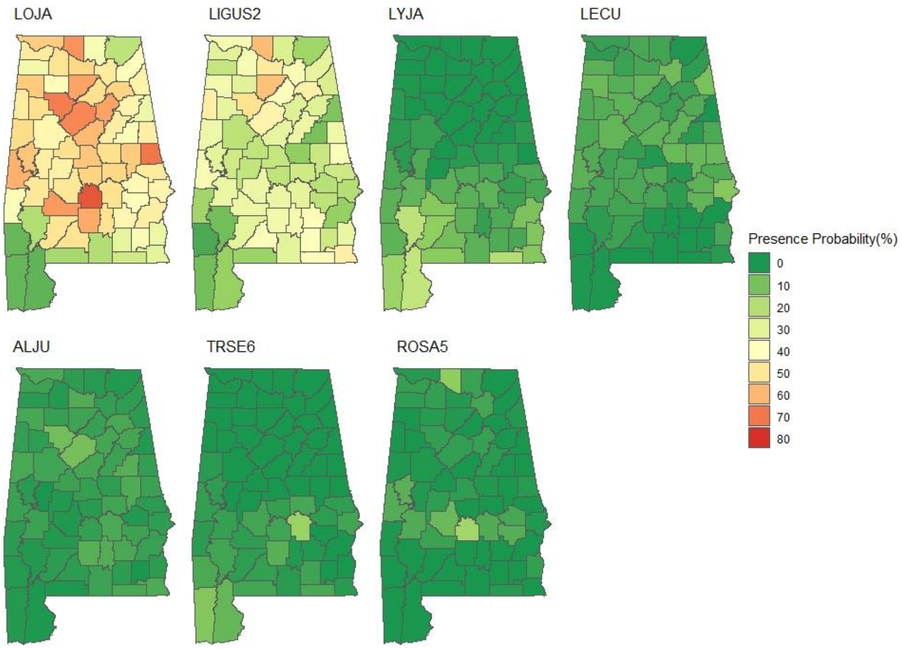

Figure 2.5: Presence probability (%) of individual nonnative invasive plant species measured between 2013 and 2019

2.3.3 Best modeling unit

Among ten modeling units, Moran's I statistics varied over time. County-level modeling units gave the highest Moran's I statistics for T1 and T3, and the ecological unit- section gave the highest for T2. All hydrological units had a lower Moran's I than county-level modeling units (Table 2.4). Thus, we chose county-level modeling units for this analysis.

Table 2.4 Moran's Test based on invasion index of all nonnative invasive plant species

Modeling unit	Product in T1			Product in T2			Product in T3		
	Moran I stat	SD	P-value	Moran I stat	SD	P-value	Moran I stat	SD	P-value
HUC 4	-0.24	-0.323	0.627	0.015	0.091	0.182	-0.18	-0.084	0.534
HUC 6	0.137	1.36	0.087	0.32	2.376	0.009	0.18	1.64	0.051
HUC 8	0.27	3.19	<0.001	0.35	3.98	<0.001	0.23	2.684	<0.001
HUC 10	0.36	10.49	<0.001	0.36	10.36	<0.001	0.26	7.35	<0.001
HUC 12	0.24	15.08	<0.001	0.18	11.59	<0.001	0.18	11.29	<0.001
COUNTY	0.4	5.55	<0.001	0.38	2.25	<0.001	0.37	5.23	<0.001
Ecoregion (province)									
Ecoregion (section)	0.37	2.14	0.016	0.51	2.44	0.007	0.27	2	0.02
Ecoregion (subsection)	0.06	0.82	0.205	0.01	0.37	0.35	0.003	0.28	0.39

2.3.4 Factors influencing nonnative invasive spread in Alabama

We fitted observed invasion indices using the spatial lag model that utilized various independent variables (Table 2.1). Best-fitted spatial lag models were identified for each measurement based on the lowest AIC. Table 2.5 shows selected variables and their coefficients, AIC, lag coefficient, and test values of residual autocorrelation. All models had positive and significant lag coefficients. Observed invasion indices, model-predicted values, and residuals of the fitted models are graphically illustrated (Figs. 2.6, 2.7, and 2.8).

Table 2.5: Estimated regression coefficients and summary statistics of the fitted spatial lag models

Measurement	Model statistics	Variable	Coefficient	Probability	Residual autocorrelation test value	p-value
T1	lag coefficient (ρ) = 0.65 (p-value < 0.000002) AIC = 389.91	intercept	8.68860	0.001**	0.101	0.459
		Area	-0.00974	0.002**		
		households	0.00004	0.007**		
		rd_length	0.01796	0.149		
		lob_hard	-0.28253	0.073.		
		mix_hard	-0.11484	0.098.		
		public_own_pct	-0.11673	0.024*		
T2	lag coefficient (ρ) = 0.57 (p value<0.0002) AIC = 407.1	intercept	22.87208	0.011*	0.059	0.808
		Area	-0.03588	0.0004**		
		pop_den_2010_nbg	0.04485	0.002**		
		rd_lenght	0.12899	0.003**		
		rd_density	-83.19226	0.017*		
		Lob	0.09729	0.016*		
		lob_hard	-0.63702	0.003**		
		sw_no_wo	0.31599	0.04*		
		public_own_pct	-0.13555	0.03*		
T3	lag coefficient (ρ) = 0.55(p value< 0.0002) AIC = 364.52	intercept	19.45200	0.002**	0.819	0.206
		area	-0.02156	0.003**		
		pop_2010_nbh	0.00003	0.011*		
		rd_length	0.08060	0.012*		
		rd_density	-54.84900	0.032*		
		lob_hard	-0.49774	0.0003**		
		long	-0.14223	0.049*		
		wo_ro_hi	-0.11396	0.054.		
		public_own_pct	-0.08586	0.079.		

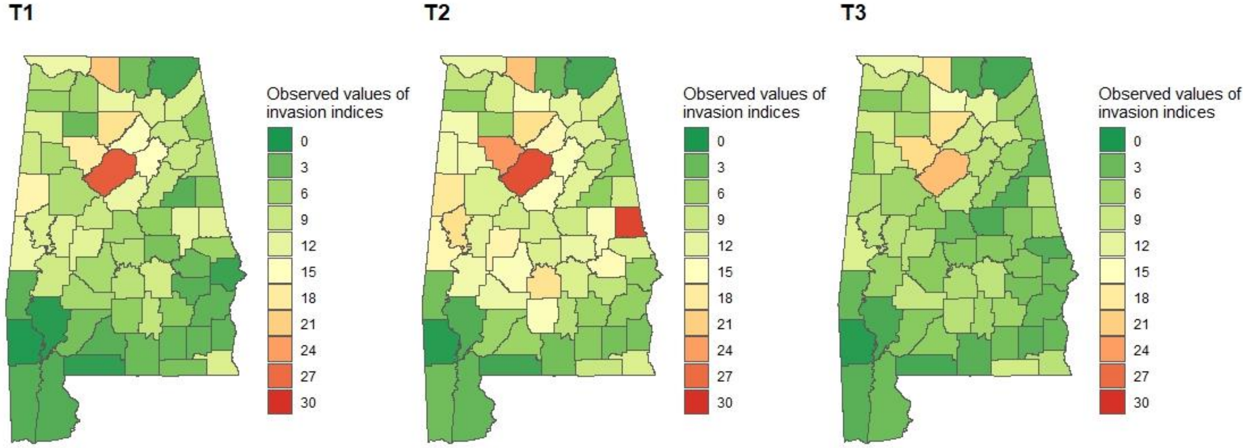

Figure 2.6. Observed invasion indices over time in Alabama. Dark green color represents the lowest and red represents the highest level of invasion

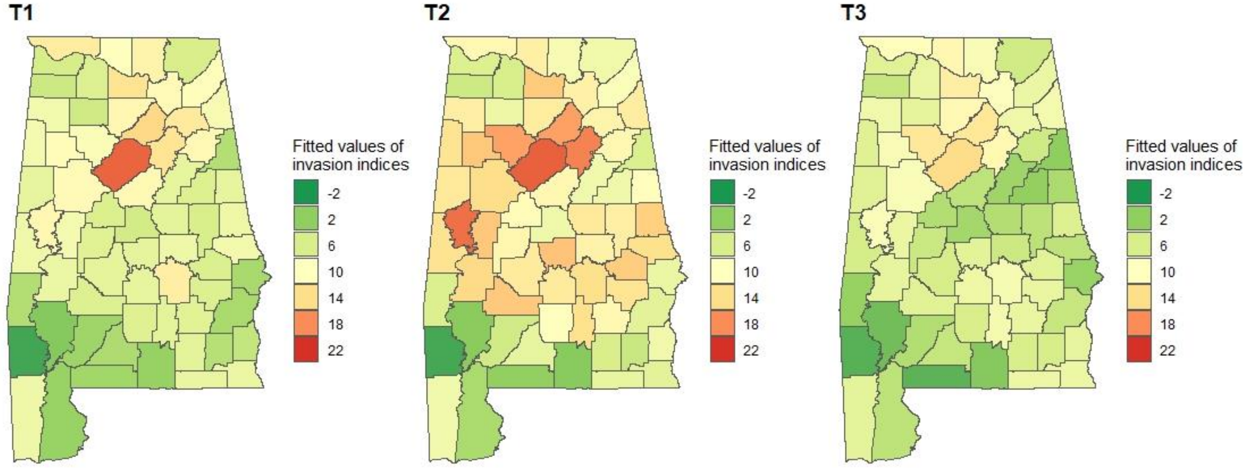

Figure 2.7: Spatial lag fitted values of invasion indices over time

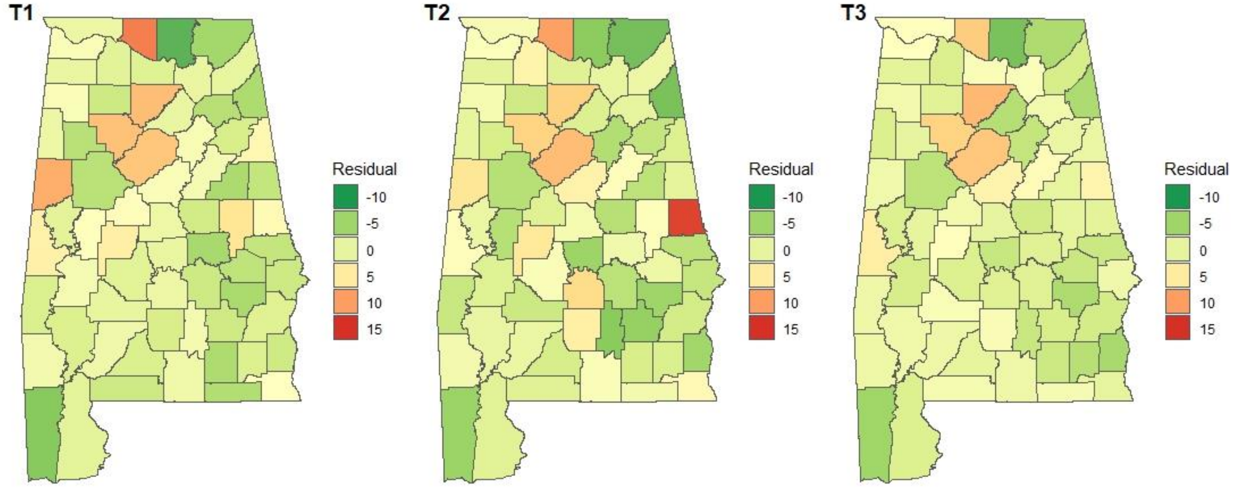

Figure 2.8: Spatial lag's residuals over time

2.4 Discussions

In Alabama, NNIPS are spreading both spatially and increasing in number temporally. This study examined the distribution of major NNIPS, quantified the invasion indices, selected the best modeling units, and developed a model considering the spatial lag effect of invasion indices. We aggregated all major NNIPS and quantified invasion indices using presence probability and cover percent. Hussain et al. (2008) used invasive species count data to explore the ecological and economic aspects of invasive species in Alabama. We believe the invasion indices we used in this study are more meaningful in exploring the invasion severity because it considers two dimensions of the invasion. Nepal et al. (2021) used a similar quantification approach to quantify the invasion severity of Chinese tallow trees.

Our analysis proved that Japanese honeysuckle had been the most prevalent invasive species in Alabama. Millar et al. (2010) also described this species as the most frequent and dense, especially in eastern-central Alabama. Its presence probability was higher than all others' combined values in all measurement periods (Table 2.4). Similarly, its cover percentage was higher than all others' combined values in T1 and T2. Thus, it has contributed significantly to the overall invasion index in T1 and T2. As we noted, the cover percent of Japanese honeysuckle was meager in T3 compared to other measurement cycles (Fig.2.3); thus, the overall invasion index was lower in T3. Japanese honeysuckle is likely to be deciduous in response to drought or cold, even though it is an evergreen or semi-evergreen species (Schierenbeck 2004). It is uncertain if the abrupt drop in cover percent observed in T3 was due to this species' deciduous nature. Future research should focus on why Japanese Honeysuckle's cover percentage was lower in T3. Japanese honeysuckle is normally constant across the landscape, with little increase or decrease; the giant swing in the data are most likely attributable to the change in field guide protocols (Personal communication with Alabama forest commission).

The number of nonnative invasive species has been increasing over time in Alabama. We found 25 NNIPS in T1, 26 in T2, and 33 in T3. The current not completed cycle (2020 and 2021; i.e., T4)- has already reported 26 distinct species. In T3 alone, FIA data revealed seven new species than during the T2 measurement. The increase during the third measurement indicates that nonnative species are spreading across Alabama. As new invasive species establish themselves in Alabama, existing invasive species continue to spread. It leads to widespread invasion presence but at a relatively lower rate of increase in cover percentage.

Choosing appropriate modeling units is vital because spatial aggregation impacts spatial autocorrelation and estimated coefficients (Jacobs-Crisioni et al 2014). Using Moran's I value as our guide, we selected the county-level modeling unit as the most meaningful. County-level modeling units perform better than other modeling units for aggregated invasion indices. However, modeling units might be different for individual species level invasion index. For example, the Chinese tallow invasion index performed better with the hydrological unit HUC8. It may be related to dispersal factors associated with Chinese tallow. Birds and water currents mainly influence its dispersal following flooding (Pile et al 2017, Yang et al 2019).

Hussain et al. (2008) found that invasive species occurrence and abundance were positively impacted by the forest ownership and proximity to densely populated areas in Alabama. They modeled occurrence and abundance separately based on zero-inflated negative binomial regression. To account for the neighboring effect, we developed an invasion index accounting for both occurrence and abundance in the spatial lag model. Thus, our model can better explain the influencing factors of invasions in Alabama. Further, we ran the model across three different measurement data sets. Our results show that all three spatial lag models (Table 2.5) had a positive and significant lag coefficient (ρ). These indicate positive spatial feedback. A higher invasion index in a county also raises the neighboring counties' predicated invasion index.

We found that the area of the modeling unit and the percentage of publicly owned forest cover negatively impacted invasion indices across measurement periods. Zhai et al. (2018) and Hussain et al. (2008) found similar outcomes in the southern states, with areas under private ownership more likely to have more invasives. We also found that the total road length inside the modeling unit positively impacted the invasion index. Even though total road length positively impacted the invasion index, we found a negative relation between road density and the invasion index. The positive relation between the invasion index and road length is likely due to roads facilitating the dispersal of NNIPS (Mortensen et al. 2009). A positive relationship between invasive species richness and total road length in Alabama was also observed by Chen (2012).

The amount and type of forest cover also impact the invasion index. For instance, increasing forest cover of some forest types such as loblolly/hardwoods, mixed hardwoods, longleaf pine, and white oak/red oak/hickory are likely to decrease the invasion index in Alabama. In contrast, increasing the cover of loblolly pine, sweetgum/Nuttall oak/willow oak are likely to increase the invasion index (Table 2.5). Similarly, the human population of neighboring counties is expected to increase the invasion index in the state.

2.5 Conclusions

This study obtained NNIPS data from more than 5,000 remeasured FIA plots across Alabama. We mapped major nonnative invasive species in both spatial and temporal domains. We observed that nonnative invasive species spread across the state and that invasion severity increased over time. Japanese honeysuckle was the most widespread species across the state. We quantified the invasion index/severity based on cover percent and presence probability. Invasion indices were quantified individually for all multiple modeling units: five levels of hydrological units, three levels of ecological units, and county. Moran’s I test showed that the county-level modeling units had the highest spatial autocorrelation; thus, we chose the county-level modeling unit best suited for the spatial lag model. The spatial lag model suggests that forested area, area of modeling units, road length, road density, household numbers, forest ownership, and neighbor’s population density significantly impact the invasion severity in Alabama. The model also suggests that neighboring counties significantly impacted invasion severity. We suggest invasive species controlling practices should focus both within the county and surrounding counties.

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3 Spatiotemporal invasion severity of Chinese tallow (*Triadica sebifera*) and invasibility of forest types in the Southern USA forestlands

(Published in Forest Science, Volume 67, Issue 5, October 2021, Pages 491–500)

3.1 Introduction

The Southern region of the US is often known as the “wood basket” of the nation (Oswalt et al. 2019). Both forest biodiversity and productivity are high in the Southern states, although not always in the same location. Southern forests are an important economic driver in the region because they are a substantial contributor to the economy and to five million private forest landowners of the region. The future benefits of these forests can only be assured if those forests are sustainably managed. However, the future of the forests in this region is severely threatened by the invasion of nonnative invasive plant species (NNIPS). NNIPS can cause significant ecological and economic damage to a forest. For example, they can compete and displace native and desirable species, degrade timber production and recreational values, lower carbon sequestration and biodiversity, change stand structure, and alter fire and natural disturbance regimes (Macdonald 1992, Pimentel et al. 2005, Moser et al. 2009a, Eviner et al. 2012). The degradation of ecosystem functions and services is related to the abundance and invasion severity of NNIPS as well as the invasibility (susceptibility) of an ecosystem to an invasive species. More than 300 potentially invasive plants have been identified in the Southern states, and some are deemed serious threats to the local ecosystem. Miller et al. (2013) noted that more than 9% of the forested land in the South was infested with one or more NNIPS. Further, NNIPS are expanding in extent and affecting forest productivity and ecosystem function (Moser et al. 2009b, Miller et al. 2013).

Chinese tallow *(Triadica sebifera*, hereafter, tallow) tree, an NNIP of considerable concern, is increasingly characterized as a threat to Southern forestlands (Wear and Greis 2013). Tallow has been spreading at an unprecedented rate in the region since it was initially introduced into coastal areas of South Carolina and Georgia in the late 1700s and was planted widely by the US Department of Agriculture in southeast Texas in the 1900s for the soap industry (Howes 1949, Miller et al. 2013, Pile et al. 2017b). Tallow cover has been expanding at an increasing rate (Fan et al. 2012, Suriyamongkol et al. 2016) and has the capability to replace native forest species in the native forest stand (Gan et al. 2009). In eastern Texas alone, the estimated tallow cover exceeded 160,000 acres between 2001 and 2003 (Rudis et al. 2006). Scientists expect the rate of spread to continue in many heavily infested areas and beyond (Gan et al. 2009, Miller et al. 2013).

As the severity and extent of NNIPS has increased across the region, species-specific decision support tools such as spread models and maps have assumed increasing importance (Fan et al. 2018). The increasing economic damage and resulting cost of control suggest that efforts to control tallow should target the establishment phase (Webster et al. 2006, Moser et al. 2009a). The invasion severity of NNIPS describes the likelihood of invasion and the potential harm to native forest ecosystems (Venette et al. 2010). Several approaches have been tested to quantify invasion severity. For example, Bazzichetto et al. (2018) used nonnative invasive species distribution models to predict invasion severity. Invasion severity typically reports risk as the relative likelihood of invasive species presence (Yemshanov et al. 2009, Bazzichetto et al. 2018); however, this term fails to express the impact on the ecosystem. Venette et al. (2010) suggested including impact of invasion on the forested area. Invasion severity may be described as the potential area of infestation and the seriousness of invasion. Rudis et al. (2006) used invasive species cover percentage data to classify FIA plots into different severity classes. A higher percentage cover of an invasive species suggests greater levels of severity. Using this protocol, Rudis et al. (2006) used total cover to describe the severity of tallow invasion in eastern Texas.

Invasion severity based on the presence probability and coverage of NNIPS is one of the recommended measures to quantify invasion severity and potential threats to native ecosystems (Catford et al. 2012). An index of invasion severity should not only aid in the assessment of the extent of NNIPS, reveal spatial and temporal trends, and act as an early warning sign of ecological degradation, but also help to guide management efforts. Bradley et al. (2018) also suggested that a robust assessment of the invasion risk or invasibility of an ecosystem can be achieved using both presence and coverage information of an invasive species. Many distribution maps have been created based on the presence or absence of NNIPS or some specific invasion indicators such as alien species richness, abundance, and relative abundance at the regional level (Gan et al. 2009, Fan et al. 2012, Iannone et al. 2015).

In this study, we had access to a comprehensive data source the USDA Forest Service's Forest Inventory and Analysis (FIA) database, which provides extensive spatial and temporal data for NNIPS with presence and cover percentage at the plot level. These variables can be used to classify invasion severity (Catford et al. 2012, Bradley et al. 2018, Fan et al. 2018). In this study, we intend to evaluate county-level invasion risk and severity of tallow across southeastern US forestlands using repeatedly measured FIA plots during the most recent two inventory cycles. Our specific objectives in this study are to quantify and classify invasion severity and examine the trends over time for tallow and to compare the invasibility of selected forest type groups to tallow invasion by identified severity classes.

3.2 Materials and Methods

3.2.1 Study Area

For this study we chose seven heavily forested coastal states of the Southern US: eastern Texas, Louisiana, Mississippi, Alabama, Florida, Georgia, and South Carolina. The forests of the Southern coastal states are highly diverse, ranging from upland oak/hickory forest to lowland gum/cypress swamp (Wear and Greis 2012). Six major forest type groups that were identified in the regions are elm/ash/cottonwood, loblolly/shortleaf pine, longleaf/slash pine, oak/gum/cypress, oak/ hickory, and oak/pine. Upland oak forests are dominant in the northern part of the region, especially in the Appalachian Mountains and the associated foothills. Pine forests tend to be dominant throughout the coastal plain (Fig. 3.1).

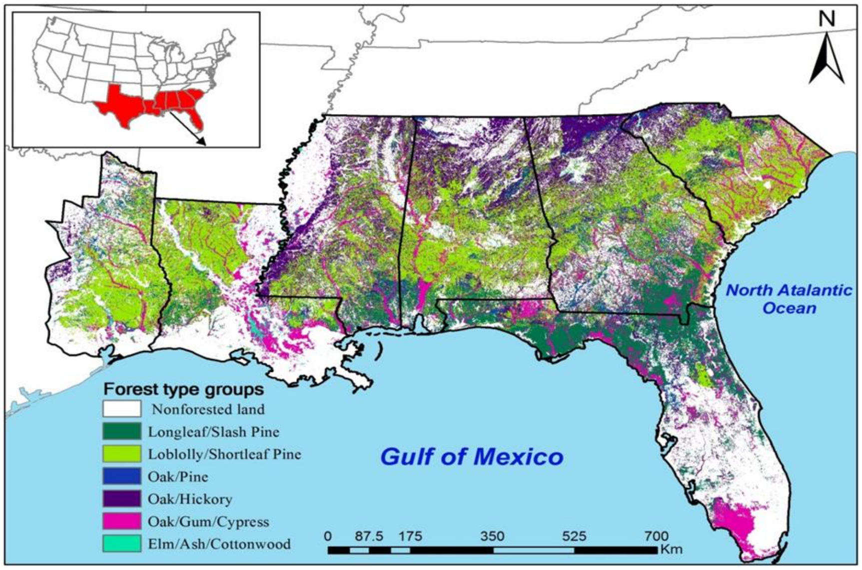

Figure 3.1 Map of major forest type groups invaded by tallow in the seven southeastern coastal states, United States. Ref-(Ruefenacht et al. 2008).

3.2.2 FIA Data:

The USDA Forest Service continually monitors US forests via the FIA program. FIA data are recognized as the most comprehensive forest dataset in the USA (Tinkham et al. 2018). FIA collects data at an intensity of one plot for every 2,428 hectares (6,000 acres) (McRoberts et al.

2005). Each FIA plot is divided into four subplots: one in the center and three each located 36.6 meters (120 feet) away at azimuths of 360, 120, and 240 degrees, respectively (O’Connell et al. 2015a). FIA monitors selected NNIPS at the subplot level. Tallow presence/absence and cover percentage data were collected from each subplot. Forest type groups and location (coordinates) of each plot were derived from the FIA data by using subplot/ plot condition and other required data base variables downloaded from the FIA DataMart (FIA DataMart 2019).

Previously, state forest inventories were measured on a periodic basis with complete state inventories conducted at intervals of approximately 7–15 years. Based on the legislation contained in the Farm Bill of 1998, the inventories began the transition from a periodic inventory to an annual system (USDA Forest Service 2016). Depending on the state, each FIA annual inventory cycle is designed to be completed in approximately 5–7 years (O’Connell et al. 2015b). A new cycle starts only after all FIA plots in a state in the current cycle have been sampled. About 10–20% of FIA plots were measured annually in the Southern states. Because of the staggered start to the annual inventory system across states, the cycle length, number of completely measured cycles, and starting year of annual FIA plots differed among states. For instance, the annual inventory program began in Georgia in 1998, South Carolina in 1999, Texas, Alabama, and Louisiana in 2001, Florida in 2002, and Mississippi in 2005. By 2019, South Carolina had uploaded four completely measured annual FIA inventory cycles of data in the FIA DataMart; Texas and Alabama had uploaded three cycles, and Florida, Mississippi, and Louisiana had uploaded two completely measured inventory cycles into the FIA DataMart.

As a result, there is asynchrony in cycle length and cycle starting point among states, which makes it challenging to compare tallow invasion trends at the regional level. Although many states in the region had completed more than two annual inventory cycles, only two completely measured annual plots can be found at the regional level. Hence, we considered only the two most recent and completely measured FIA cycles from each state in this analysis. The most recently completed cycle (2009–2019) is designated T2 throughout the article and the second most recently completed cycle (2002–2012) in each state is T1. We assume population expansion or change can be detected using repeatedly measured cycles (i.e., T1 and T2) across states.

3.2.3 Aggregating FIA data at the county level

We examined 528 counties from seven Southern coastal states. We found that 211 counties in T1 and 250 counties in T2 had at least one subplot infested with tallow. We found 128 out of 211 counties with five or fewer tallow-infested plots in T1 and 118 out of 250 in T2. In contrast, we found 83 counties in T1 and 132 in T2 with more than five infested plots. In our analysis, we used both presence probability and cover percentage to describe the invasion risk and severity. Presence probability accounts for the number of infested plots and total number of plots in a county. Cover percentage was calculated as the mean tallow cover percentage of all FIA plots at a county level. Not all counties had the same number of FIA plots; some had sufficient numbers whereas others had limited data. To overcome this problem, we used a neighborhood smoothing function to calculate the smoothed presence probability and cover percentage of each county. This approach is similar to that used by Fan et al. (2018).

$$P_i = \frac{\sum_{i}^{nb_i} S_i}{\sum_{i}^{nb_i} N_i} \tag{1}$$

Here, P_i is the presence probability of tallow for a county i using a neighbor smoothing function. S_i is the number of tallow presence plots in a county i, N_i is the total number of plots in a county i, and nb_i is the set of neighboring counties that share a boundary with a county i.

A similar approach was used to calculate tallow cover percentage at the county level.

$$C_i = \frac{\sum_{i}^{nb_i} T_i}{\sum_{i}^{nb_i} N_i} \tag{2}$$

Here, C_i is the cover percentage of tallow for a county i using a neighbor smoothing function. T_i is sum of tallow cover in a county i, N_i is the total number of infested plots in a county i, and nb_i is the set of neighboring counties that share a boundary with a county i.

3.2.4 Mapping invasion severity

In this study, the product of the estimated presence probability and mean cover rate (percent) that measure both the abundance and dominance of an invasive species was used as a proxy to quantify the invasion severity of tallow in a county. The empirical cumulative distribution function (ECDF, a sigmoid or exponential or spherical curve) of the product was calculated to quantify the distribution of the county-level invasion severity. A classification and regression tree (CART) model was used to partition the ECDF into different segments or categories by using the product as the only predictor, so that all counties were classified into different severity classes (Fan et al. 2018, Figure 3.2). Those counties with varying invasion severities were then mapped spatially to reflect the geospatial invasion patterns of tallow across the southeastern coastal states. The initial CART model was then pruned using an optimal complexity parameter value derived through the cross-validation approach. The terminal nodes of the pruned CART model explicitly specify the severity classes of infested counties. The T1 data were used to construct the CART model to delineate the severity classes of infested counties. This model (same variables and cutoff values) was then applied to T2 data to delineate potential changes in severity during the most recent inventory cycle (2009–2019).

3.2.5 Quantifying invasibility of forest type groups to tallow invasion

Invasibility of the recipient forest type groups was examined by using the mean presence probability and cover percentage for all severity classes. Using mean presence probability as the x-axis and mean cover percentage as the y-axis, a triangle (half of the rectangle) was plotted for each forest type group in each severity class for both inventory cycles T1 and T2. The area of the triangle and its change between inventory cycles were used as a proxy for measuring invasibility, with a larger area representing greater invasibility and vice versa, with a smaller area representing that a forest type group is more resistant to tallow invasion.

The presence probability and cover percentage were bootstrapped 1,000 times and the area of the corresponding triangle was computed for each forest type group by severity class. The Student's t-test was conducted to compare the statistical significance in the change of the area inside a triangle for each forest type over time. All analyses were done using the R language (R Development Core Team, 2014) and an a priori significance level of $\alpha = 0.05$ was set for statistical tests. The CART model of invasion severity was constructed using the rpart package, and other analyses, including Student's ttest, were conducted using the R base package.

3.3 Results

3.3.1 Mapping invasion severity

More than 72% of the counties in the region contained at least one FIA plot infested with tallow in two periods. The CART model of invasion severity (the product of presence probability and cover percentage at the county level) classified the infested counties into four (high, moderate, low, and minimal) severity classes using three cutoff values of the product (Fig. 3.2 A and B). In T1, 90, 89, 90, and 89 counties and in T2, 123, 119, 64, and 78 counties were classified as high, moderate, low, and minimal severity, respectively (Fig. 3.2 C and D). The number of counties with high and moderate invasion severity have increased over time (Fig. 3.3). We found nearly 31% of all counties moved into a higher severity class between T1 and T2, suggesting further focal expansion of tallow populations. We also observed that nearly 39% of the counties remained in the same severity class over time and 5% of the counties showed a decrease in severity class. Negative changes were found mostly on the edges of the invasion range, where a lower level of risk already existed. In contrast, no change or increase was observed in areas with higher levels of invasion (Fig.3.3).

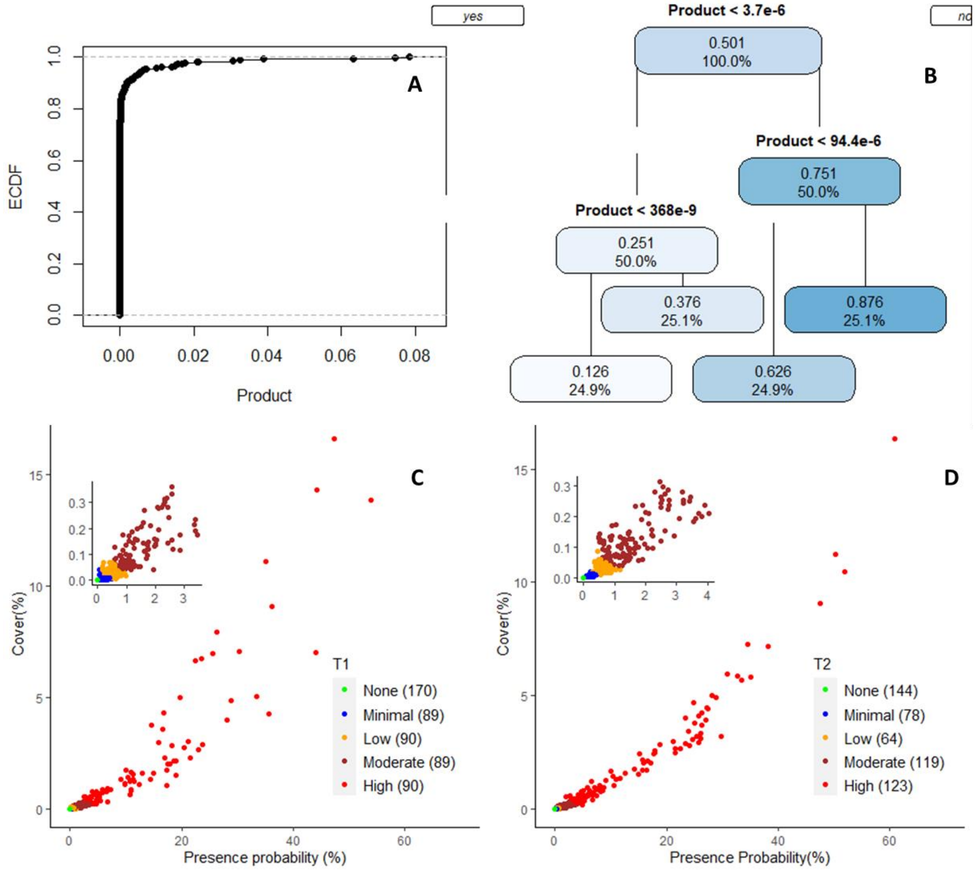

Figure 3.2**.** The empirical cumulative density function (ECDF) curve of the product of presence probability and cover percent of tallow.

In Fig. 3.2 a proxy of invasion severity (A), four invasion severity classes identified by the classification and regression tree (CART) model (B), and the relationship between presence probability and cover percent by severity class in cycle T1 (C) and T2 (D). Number of counties in each severity class is shown in the legend.

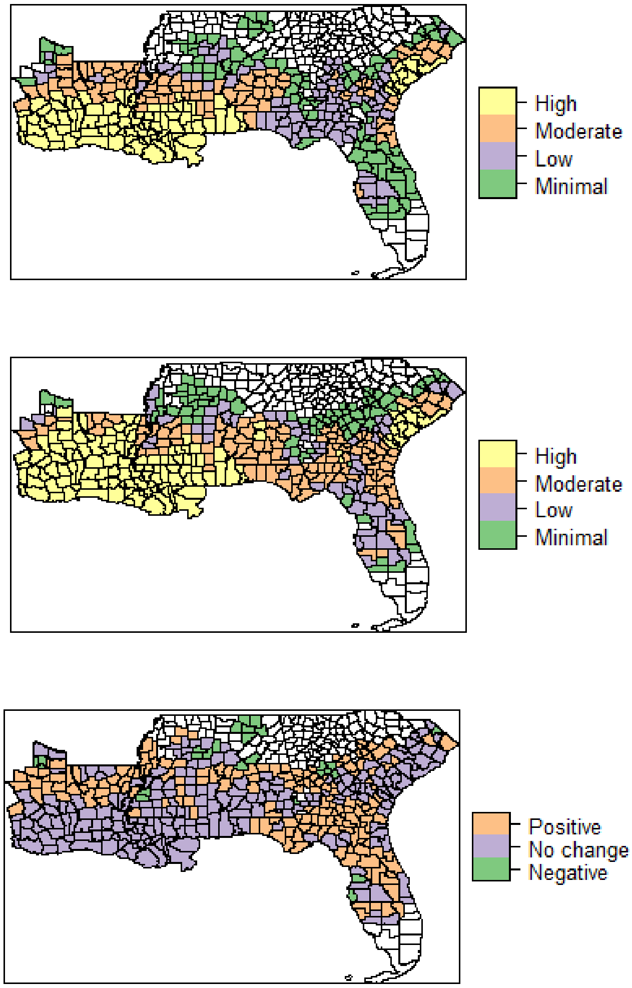

Figure 3.3 Spatial representation of invasion severity of Chinese tallow showing the distribution of different severity classes.

In Fig. 2.3, Cycle T1 (A), in cycle T2 (B), and changes between T1 and T2 (C). Of the 528 counties, 166 counties showed a positive change, 205 showed no change, 26 showed a negative change, and 131 remained non-infested.

3.3.2 Invasibility of forest type groups

The highest invasibility (the largest area of the triangle) was observed in the elm/ash/cottonwood forest type group (Fig. 3.4 and Tables 3.1 and 3.2). In contrast, the longleaf/slash pine forest type group had the lowest invasibility. This trend existed in both cycles across all severity classes (Fig. 3.4). High invasibility was also observed in the oak/gum/cypress forest type group in both cycles in all severity classes. We observed increases in the presence probability over time for almost all forest type groups, although the cover percentage did not follow the same pattern. For instance, cover percentage in the high-severity class increased in the longleaf/slash pine, loblolly/shortleaf pine, and oak/gum/cypress forest type groups and decreased in the oak/hickory, oak/pine, and elm/ash/ cottonwood forest type groups (Table 3.1). The Student’s t-test showed that in the high-severity class, the area of the triangle significantly increased between T1 and T2 for all forest type groups except the oak/hickory forest type group (Table 3.2). Detailed comparisons for other forest type groups by severity class are shown in Table 3.2 and Figure 3.4.

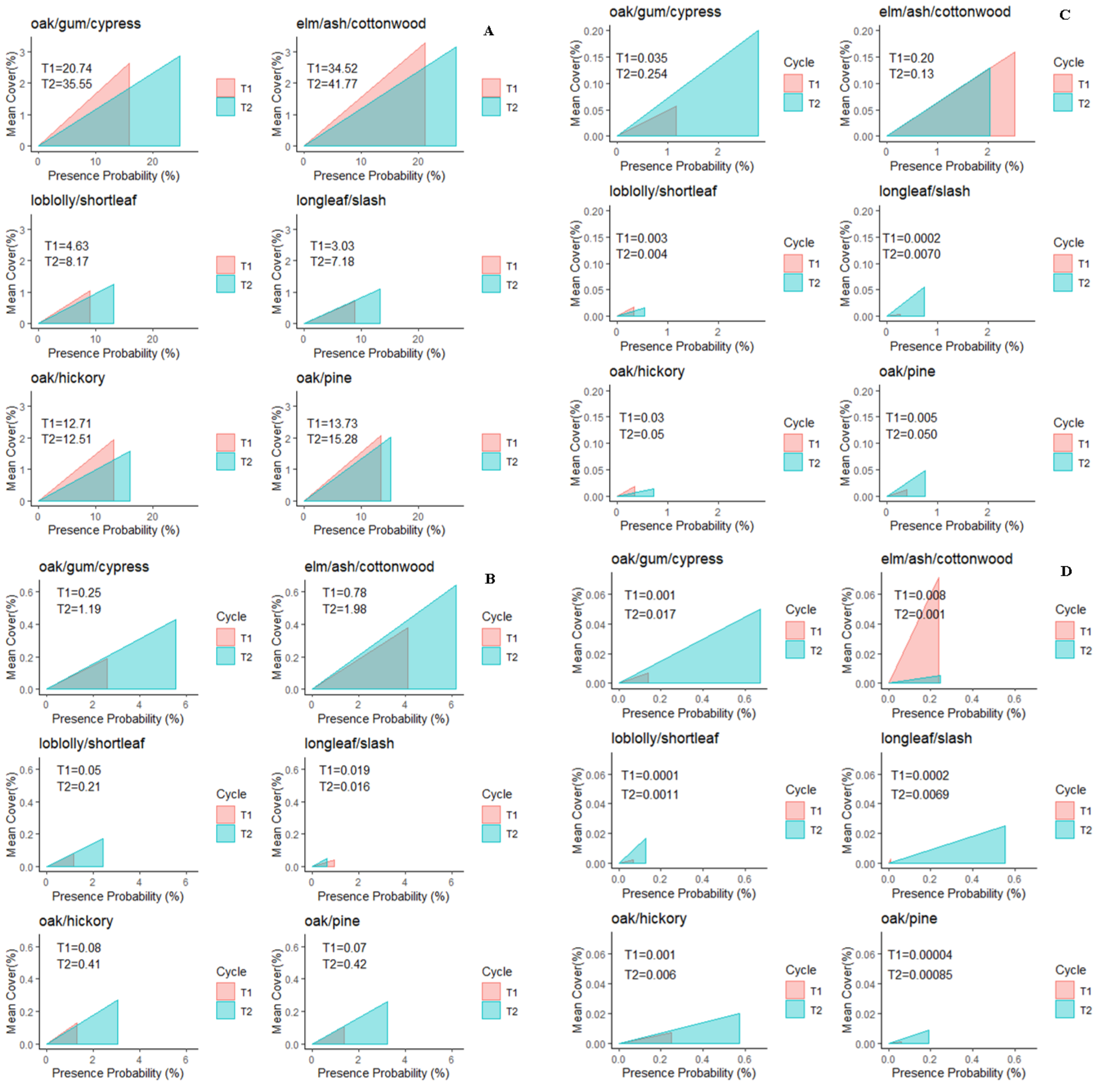

Figure 3.4 Area inside triangles for different forest type groups over two cycles. High (A), moderate (B), low (C), and minimal (D) severity classes. The numbers reported for T1 and T2 are the area of each triangle.

Table 3.1 The presence probability and cover percentage of Chinese tallow by identified severity classes in the Southern US

Severity class	Forest type groups	T1		T2		Change	
		Presence %	Cover %	Presence %	Cover %	Presence %	Cover %
High	EAC	21.15	3.29	26.50	3.15	5.35	-0.14
	LSL	8.98	1.03	13.06	1.25	4.08	0.22
	LLS	8.79	0.70	13.28	1.07	4.49	0.37
	OGC	15.78	2.62	24.64	2.87	8.86	0.25
	OH	13.11	1.94	15.95	1.57	2.84	-0.37
	OP	13.34	2.05	15.05	2.04	1.71	-0.01
Moderate	EAC	4.08	0.37	6.16	0.64	2.08	0.27
	LSL	1.16	0.08	2.42	0.17	1.26	0.09
	LLS	0.94	0.04	0.63	0.05	-0.31	0.01
	OGC	2.61	0.19	5.52	0.43	2.91	0.24
	OH	1.30	0.14	3.05	0.27	1.75	0.13
	OP	1.35	0.11	3.25	0.26	1.90	0.15
Low	EAC	2.55	0.17	2.03	0.13	-0.52	-0.04
	LSL	0.33	0.02	0.53	0.02	0.20	0.00
	LLS	0.27	0.00	0.75	0.06	0.48	0.06
	OGC	1.18	0.06	2.79	0.20	1.61	0.14
	OH	0.35	0.02	0.72	0.03	0.37	0.01
	OP	0.40	0.01	0.76	0.05	0.36	0.04
Minimal	EAC	0.24	0.07	0.25	0.00	0.01	-0.07
	LSL	0.07	0.00	0.13	0.02	0.06	0.02
	LLS	0.10	0.00	0.54	0.02	0.44	0.02
	OGC	0.14	0.01	0.67	0.05	0.53	0.04
	OH	0.26	0.01	0.57	0.02	0.31	0.01
	OP	0.06	0.00	0.18	0.01	0.12	0.01

Note: EAC, elm/ash/cottonwood; LSL, loblolly/shortleaf pine; LLS, longleaf/slash pine; OGC, oak/gum/cypress; OH, oak/hickory; OP oak/pine

Table 3.2 Results of Student's t-test performed to compare area inside triangles between two cycles (T1 and T2) in each severity class for all forest type groups using bootstrapping (n = 1,000)

Severity class	Forest type groups	Mean (SD); n=1000		t-stat (df=1998)	Relative change
		T1	T2		
High	EAC	34.68(6.64)	41.9(5.95)	<0.001	0.21
	LSL	4.65(0.5)	8.21(0.64)	<0.001	0.77
	LLS	3.07(0.69)	7.19(1.03)	<0.001	1.34
	OGC	20.76(2.26)	35.59(2.78)	<0.001	0.71
	OH	12.77(1.89)	12.52(1.57)	<0.001	-0.02
	OP	13.78(2.29)	15.34(2.1)	<0.001	0.11
Moderate	EAC	0.8(0.42)	2.01(0.67)	<0.001	1.52
	LSL	0.05(0.01)	0.2(0.03)	<0.001	3.33
	LLS	0.02(0.01)	0.02(0.01)	<0.001	-0.22
	OGC	0.25(0.07)	1.19(0.23)	<0.001	3.75
	OH	0.09(0.03)	0.41(0.09)	<0.001	3.66
	OP	0.07(0.04)	0.42(0.12)	<0.001	4.64
Low	EAC	0.23(0.16)	0.15(0.13)	<0.001	-0.32
	LSL	<0.01(<0.01)	<0.01(<0.01)	<0.001	NA
	LLS	<0.01(<0.01)	0.02(0.01)	<0.001	NA
	OGC	0.04(0.02)	0.28(0.08)	<0.001	7.01
	OH	<0.01(<0.01)	0.01(0.01)	<0.001	NA
	OP	<0.01(<0.01)	0.02(0.01)	<0.001	NA
Minimal	EAC	0.02(0.03)	<0.01(<0.01)	<0.001	NA
	LSL	<0.01(<0.01)	<0.01(<0.01)	<0.001	NA
	LLS	<0.01(<0.01)	0.01(0.01)	<0.001	NA
	OGC	<0.01(<0.01)	0.02(0.01)	<0.001	NA
	OH	<0.01(<0.01)	0.01(0)	<0.001	NA
	OP	<0.01(<0.01)	<0.01(<0.01)	<0.001	NA

3.4 Discussion:

3.4.1 Invasion severity of Chinese tallow over time

In the Southern US, tallow invasion has increased over time. Infestation intensity was higher in two sub-regions (Miller et al. 2013): the Gulf of Mexico coastal plain, stretching from Texas to Florida (western epicenter) and the Atlantic coastal plain of South Carolina and Georgia (eastern epicenter). The increase in infested area and infestation intensity extended northeast from the western epicenter and northwest from the eastern epicenter. Infestation intensity was higher in the western epicenter than in the eastern epicenter. Other research identified those areas as locations where tallow was historically introduced for commercial planting (Howes 1949, Pile et al. 2017b, Miller et al. 2013). Our findings from the remeasured FIA data found that these epicenters of tallow infestation became more pronounced over time both in extent and intensity.

Tallow invasion severity classes in this study were quantitatively derived from the FIA plot data in the Southern coastal states. A similar approach was applied in Fan et al. (2018) using one-time measured FIA data to classify invasion severities of NNIPS in the midwestern US. Remeasured data provided an opportunity to compare invasion severity over time at the regional level. Counties in the high invasion severity class were concentrated around the two epicenters in both cycles. Examination of temporal changes in invasion severity showed that most of the counties with the highest level of invasion severity stayed at that level. Some counties with a lower number of infested plots, especially at the outer edge of the epicenter, decreased in invasion severity. Increases in severity occurred outside of but adjacent to the two epicenters.

Using remeasured FIA plot-level data, Yang (2019) used a CART model to classify four subregions based on tallow presence in the Southern US. Yang (2019) classified the low elevation coastal plain near the western epicenter (i.e., near Texas) as the area with the most severe tallow invasion. Our classification approach differed in that we considered both presence probability and cover percentage of FIA plots and aggregated them to the county level before classification. As such, we identified high-severity zones in both the eastern and western epicenters.

3.4.2 Invasibility of forest type groups

Invasibility of forest type groups was examined using presence probability, cover percentage, and the product of both in different severity classes. In prior work, Fan et al. (2012) explored tallow invasion solely in eastern Texas, where they found the highest presence probability in the oak/gum/cypress forest type group, followed by longleaf/slash pine, elm/ash/cottonwood, loblolly/shortleaf, and oak/hickory groups. Most of the counties in eastern Texas and Louisiana were classified as high severity in our analysis. Furthermore, this study was regional and covered a wider range, with overall presence probabilities that were relatively lower than Fan et al. (2012). However, our results for the high-severity class were similar to those of Fan et al. (2012). The most severely infested forest type groups in both cycles were elm/ash/cottonwood and oak/gum/cypress. In general, tallow can invade both wet and dry sites (Miller 2003), but the severe infestation sites were wet forest sites and coastal prairies (Miller 1998). Our findings also support this conclusion, as we found that elm/ ash/cottonwood and oak/gum/cypress were severely infested by tallow invasion in both measurement cycles.

At the other end of the invasion spectrum, we found longleaf/slash pine and loblolly/shortleaf pine forest type groups were the least infested in both cycles at the regional level. The greatest changes in presence probability over time were observed in oak/gum/cypress, elm/ash/cottonwood, longleaf/ slash pine, and loblolly/shortleaf pine. Presence probabilities in these forest type groups increased by 8.86, 5.35, 4.49, and 4.08%, respectively. Fan et al. (2012) also found the highest spread rate in oak/gum/cypress forests. In our analysis, we found invasibility was already higher in oak/gum/cypress in T1, and the rate of change in presence probability was higher than for other forest type groups. Thus, the oak/gum/ cypress forest type group is highly susceptible to tallow invasion compared with other forest type groups across severity classes.

The graphical tool (Figure 3.4) provides a useful approach to compare the invasibility of different communities by simultaneously considering the presence probability (abundance) and cover percentage (dominance) of NNIPS involved (Catford et al. 2012, Fan 2015, Fan et al. 2018). The largest areas inside a triangle were observed in the elm/ash/cottonwood forest type group in both cycles whereas the smallest was in the longleaf/slash pine forest type group. Invaded area increased over time in all but the oak/hickory forest type group at the regional level. Although the longleaf/slash pine forest type group had the lowest severity of invasion, it has the highest relative increment between T2 and T1 (> 134%). We also found higher rates of increment in the area under the triangle in the loblolly/shortleaf pine (>77%), oak/gum/ cypress (>72%), and elm/ash/cottonwood (>21%) forest type groups between T2 and T1. In the moderate-severity class, areas inside a triangle were significantly higher in T2 than in T1 for all forest type groups except longleaf/slash pine. The highest relative increment in area inside a triangle between the two cycles was seen in the oak/pine forest type group (>464%), followed by the oak/gum/cypress forest type group (>375%).

In the high-severity class, tallow presence (abundance) increased across all forest type groups but cover percentage (dominance) increased, primarily in pines (lowland flatwoods) and oak/gum/cypress (bottomland). In elm/ash/cottonwood, oak/pine, and oak/hickory forest type groups, the mean dominance declined, although tallow presence increased. These results suggest that monitoring of tallow spread should focus on longleaf/slash pine, loblolly/shortleaf pine, and oak/gum/ cypress forest type groups. Further, the longleaf/slash pine forest type group demonstrated a higher level of invasibility at the regional level, whereas elm/ash/cottonwood forest type groups displayed a lower level of invasibility by tallow. Further, Fan (2012) compared the invasibility of loblolly/ shortleaf pine and oak/gum/cypress forest type groups in eastern Texas and suggested that oak/gum/cypress was more susceptible to tallow invasion than was loblolly/shortleaf pine. Our results are consistent with those findings. In both cycles, we found a significantly higher level of tallow invasion in the oak/gum/cypress forest type group as compared with the loblolly/shortleaf pine forest type group.

There are multiple potential confounding factors influencing community invasibility, including disturbance, climate, and geographic features. That comparisons of the invasibility of different forest type groups were made by invasion severity class helped to a certain degree to filter out the potential impact of these confounding factors, because each subregion identified in this study is more homogeneous in invasion severity (propagule pressure), geographical features, climate, and disturbance regimes (Fan et al. 2012). Specifically, geographical features (e.g., slope and elevation), propagule pressure levels, and disturbances (e.g., hurricanes, flooding, wildfires, and management activities) tend to increase the presence probability of tallow between inventory cycles, but climate and community conditions will have greater effects on the change in dominance of the invasive tallow (Yang et al. 2021). The western epicenter seemed to expand more rapidly than the eastern epicenter between the two inventory cycles, as more counties around the western epicenter had been converted to the high-severity class in the most recent inventory cycle. This might be related to more favorable climate (e.g., high temperatures and rainfall), flat terrain, and more frequent and intensive disturbances such as hurricanes and tropical storms along the western Gulf coastal plain. Moser et al. (2016) stated that the oak/hickory and elm/ash/cottonwood forest type groups were highly susceptible to invasive shrubs in the midwestern US. These Midwestern-forest types were heavily exposed to disturbance because of their proximity to human development and the history of forest management and exploitation of those types; such disturbances were likely facilitators of the invasive species spread (Moser et al. 2009a). Pile et al. (2017a) compared the invasibility of natural and disturbed forests to tallow infestation and found that natural forests were more resistant to the invasion because of greater woody species richness and biomass. Other factors, including physiographic zone, hurricanes, storms, distance to nearest road and water, stand age, treatments, and land ownership play important roles in the extent and severity of tallow infestation (Yang 2019, Gan et al. 2009). Gan et al. (2009) also showed that slope was the key factor for tallow invasibility in flat areas, as flooding and higher soil moisture levels facilitate its invasion. We examined only forest type groups in relation to tallow invasion severity classes. In the future, these other variables should be included in analyses of temporal changes in invasion severity.

3.5 Conclusions:

Based on the USDA Forest Service's FIA data, tallow has become one of the most aggressive invaders in diverse Southern forestlands. Using the product of presence probability and cover percentage as a proxy, we classified tallow invasion severity into four classes in the South coastal US states from eastern Texas to South Carolina. The highest severity class occurred in counties in coastal South Carolina and Georgia and at the opposite end of the region in western Louisiana and eastern Texas. Among all severity classes, elm/ash/cottonwood and oak/gum/cypress forest type groups were most susceptible to tallow invasion, followed in a descending order by oak/ pine, oak/hickory, loblolly/shortleaf pine, and longleaf/slash pine forest type groups, based on the invasibility measure: the area of the triangle that reflects both the abundance (presence probability) and dominance (cover percentage) of invading tallow populations. The invasion severity and invasibility measures provide useful tools to assess the relative threats of different forest communities across the Southern forestland, allowing forest managers to prioritize risk areas and forest types to control and mitigate the invasion and post invasion spread of tallow. Management of tallow invasion should be focused on the highly susceptible oak/gum/cypress and elm/ ash/cottonwood forests. In addition, considering the high value and rapid increase of tallow’s cover percent in longleaf/ slash pine forests in the high-, low-, and minimal-severity regions, mechanical and chemical treatments to remove tallow should be used to protect and conserve this endangered ecosystem. Because tallow is largely disturbance-dependent and may completely replace the invaded forest and form a monoculture stand, intensive monitoring is critical in frequently disturbed coastal areas (e.g., high- and moderate-severity counties) so that proactive measures can be implemented in a timely manner. For instance, prescribed fire in combination with stand management has been recommended to mitigate tallow’s growth to prevent stand-replacement incidence.

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4 Evaluating the effect of biotic and abiotic stresses to develop mitigation methods for oak decline

4.1 Introduction

There is increasing concern over rising tree mortality. Increased heat stress, water deficits, and frequent and prolonged drought periods, among others, are causing tree mortality across the globe (Castro et al. 2009; Van Mantgem et al. 2009). Recently, in the Southern USA, Oak decline has become increasingly severe as numerous fully stocked and over-stocked oak forests across the Ozarks of Arkansas and Missouri are approaching physiological maturity. It has become of great concern for the future of forests across the region and the eastern US. The etiology of oak decline is imperfectly understood, but oak crown dieback and tree mortality are usually attributed to an interrelated series of predisposing, inciting, and contributing factors (Manion 1991).

Numerous oak decline events have been recorded in the Ozarks (Voelker et al. 2008). Among them, mortality after a late 1990's drought accounted for 15-28 percent of red oak basal area loss in the highlands (Starkey et al. 2004). The intensity and frequency of these events are expected to increase (Dai 2011). Drought and physiologically mature red oak group species (*Quercus* spp. L.; subgenus *Erythrobalanus*) are associated with the periodic widespread decline episodes that impacted over 1.38 million hectares across the Ozarks by 2010 (Spetich et al. 2017).

Current forests differ from historical ecosystems in the Missouri Ozarks (Hanebrry et al. 2012). Historically, Ozark forests consisted mainly of white oaks, red oaks, and shortleaf pine (*Pinus echinata*). Due to the excessive timber harvest (mainly shortleaf pine) during the 1800s and early 1900s wildfires, forests in the region shifted to the oak-hickory forest (Guyette and Kabrick 2002; Cunningham 2006). Pine forests were concentrated in the past on poor soil quality, but now red oaks exist on those sites. Many of those red oaks are reaching or surpassing their longevity. They, thus are expected to be on the verge of decline as recent oak decline events were mainly associated with physiologic maturity and drought. Therefore, it is imperative that management methods are developed to help mitigate these effects. With that in mind, we planned to examine disturbance, silvicultural treatments, and historical forest conditions to understand and develop oak decline mitigation methods.

As the threat of more frequent and intense drought increases (Vose et al. 2016), the oak decline is likely to become a greater problem across the fourteen state central hardwood forest (CHF) region of the eastern U.S.(Oak et al. 2016). Oak decline analysis based on ten years of data (1999-2010) stated that the red oak decline reached its peak from 2008 to 2009, two years after the end of the 2006 drought (Spetich et al. 2017). Researchers noticed a time lag between actual mortality events and the occurrence of inducing factors (Sohar et al. 2014). Fan et al. (2012) suggested that the lag effect of drought events on oak mortality may last up to ten years. Prior studies of oak decline have examined risk factors, regeneration dynamics, spatial and temporal patterns, modeled alternate harvesting techniques, and developed hazard indices (Fan et al. 2006, 2011, 2012; Spetich and He 2008; Spetich et al. 2011, 2017; Wang et al. 2013; Yang et al. 2021). However, recommendations must be refined to help managers design treatments that meet site-specific needs. Additionally, stands are typically carried well beyond decline susceptible ages prior to regenerating them.

Comparing and contrasting any shift in vegetation across the region will help understand the future of the highlands. The USDA Forest Service continuously monitors highlands forests via Forest Inventory and Analysis (FIA). FIA data for the highlands have been available since 1989. Furthermore, historical individual tree species records from earlier French/Spanish land surveys, the U.S. General Land Office's original Public Land Survey (PLS), are available. These tree records have been widely used to reconstruct historical land-use and species composition (Hanberry et al.2014; Grubh 2010). Some researchers pointed out some species preferences bias while recording PLS trees, but the data were expected to provide a good representation of historical species composition and land-use cover, specifically over broad spatial scales (Schulte and Mladenoff 2001; Grubh 2010). Comparing species composition will illustrate the potential consequences of oak decline in the Ozarks. In this study, we also examine the mortality and establishment rate of major species groups in the Highlands.

Oak mortality in the Ozark is likely influenced by tree characteristics, stand dynamics, and climatic factors such as extreme drought and wildfires (Shifley et al. 2006; Yang et al. 2021). Crown dieback is an indicator of tree stress recorded in FIA data. Further, Crown dieback is one of trees' earliest signs of drought-induced stress. Healthy crowns are essential for growth and development. Diameter growth would be slower on trees with crown dieback and the extreme dieback ultimately leads to tree mortality. We assumed that trees grown in severe and continuous drought should have lower diameter growth than trees grown under more favorable growing conditions. We also assumed red oaks group might have a higher reduction in growth rate due to crown dieback trees compared to other species groups.

Previously, several oak decline risk groups had been identified to construct a stand hazard model for a small area in the Ozarks to quantify the severity of oak decline based on the data from a long-term ecological monitoring project (the Missouri Ozark Forest Ecosystem Project (MOFEP)) (Fan et al. 2006, 2011). Even though the model showed high predictive power for the studied landscape over 16 years (1990-2006), it is unknown how the model will perform for the entire Ozarks. Another study (Fan et al. 2012) based on FIA data (1999-2010) showed that oak decline varied spatially and temporally in the Ozarks and was highly correlated with recurrent droughts. Further, Moser et al. (2003) found that the spatial pattern of precipitation across the highlands was correlated to the landscape pattern of oak mortality. In addition, the previous model did not include site, landscape, and regional variables other than tree and stand variables. This may greatly limit its applicability to other areas with different extreme climatic regimes, such as droughts. Further, since historical vegetation cover changed in the Ozarks (Hanberry et al. 2014), we suspect that such a shift in vegetation cover might impact oak mortality.

A modeling system that includes all known factors contributing to the oak decline is critical to developing applicable oak decline mitigation methods in the Ozarks. Comparing current and historical species composition provides a better understanding of recent oak mortality. Further, comparing recent oak mortality in ownership classes and different disturbance regimes, including natural disturbance and silvicultural treatments, helps us better understand how management and disturbances can impact oak mortality. Furthermore, comparing recent oak mortality with historical drought at different spatial domains provides a better understanding of recent oak mortality and its association with historical drought across spatial and temporal scales.

The primary objective of this chapter is to examine recent oak mortality and its association with biotic and abiotic factors such as present and historical forest conditions, climatic zone, ownership, disturbances, and silvicultural treatments in the Ozarks to develop or update the existing stand hazard model. The goal here is to identify silvicultural treatments that mimic conditions in which stands exhibited mortality. This study also compares and contrasts present species composition to the historical tree species and examines the changes in vegetation over time. Furthermore, it compares and contrasts ingrowth and mortality within major species groups in the Missouri Ozark using remeasured trees from FIA to understand the future of tree species in the Highlands. Finally, it examines annual growth and morality in stressed oak vs. not stressed oak to understand how drought-induced stress impacts oak growth and facilitates mortality over time. With that in mind, management recommendations developed in this study will likely have broad relevance throughout the CHF region and across the eastern U.S. as the Ozarks act as a sentinel for the future of oak-dominated forests across the eastern U.S.

4.2 Methodology

4.2.1 Study area

The Ozarks ecoregion is in the central USA across parts of Missouri, Arkansas, Oklahoma, and Kansas. Most of the Ozarks are from Southern Missouri to northern Arkansas (Fig. 4.1). Four levels of hierarchical ecosystems shapefiles of the United States were downloaded from the USDA Forest Service website (https://www.fs.fed.us/rm/ecoregions/products/map-ecoregions-united-states/). The four levels include polygons of ecological subsections within sections within provinces within the conterminous United States. The Ozarks section and associated subsections were selected. The Ozarks is a section under the central interior broadleaf forest province, containing 17 subsections (Cleland et al. 2007). In FIA data, those subsections were coded as Ecological Subsection Code (ECOSUBCD). The highlands include a high plateau of steep and low rolling hills and are mainly dominated by oak-hickory and oak-pine forest types (Mcnab et al. 2007).

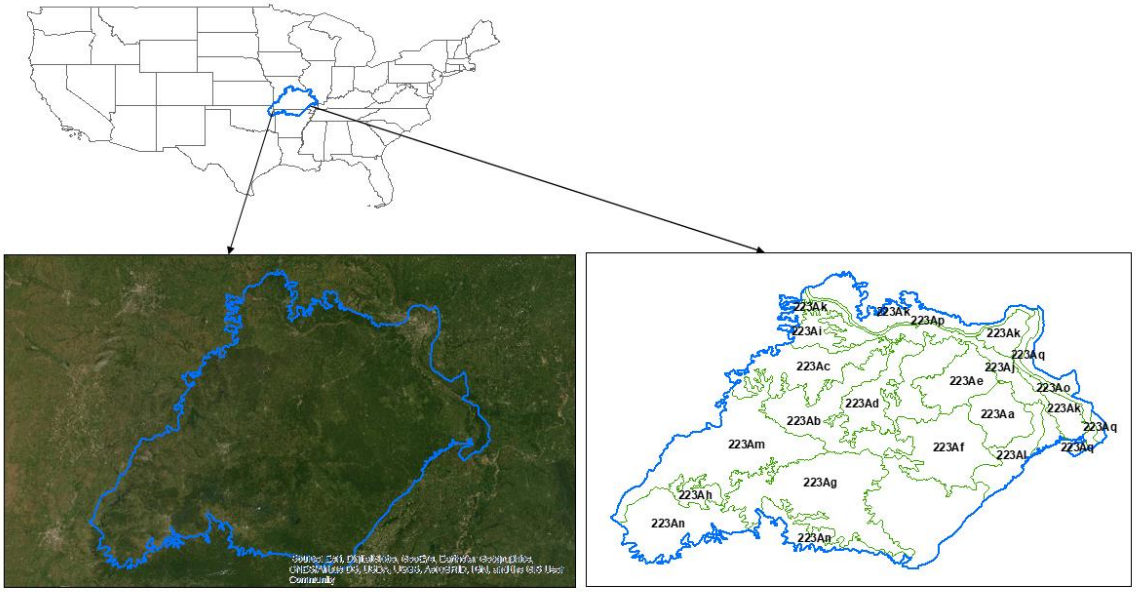

Figure 4.1 The Ozarks boundary. Most of the highlands is located in Missouri and Arkansas. The highlands contents 17 different ecological subsections (i.e. ecoregions).

4.2.2 FIA data

Publicly available FIA subplot level data were downloaded from the FIA DataMart (https://apps.fs.usda.gov/fia/datamart/CSV/datamart_csv.html, data access date: 5/30/2020). FIA collects stand-level information from the US forest land. It has permanent plots established in approximately 4.83X4.83 kilometers throughout the Southern states. There is one plot for roughly every 2428.11 hectares. FIA data inside the Ozark boundary were selected. In Missouri, data measured from 1989 to 2019 were available in the DataMart. Two different datasets were made; one to compare with historical vegetation and the other to analyze the mortality rate. All available trees greater than five-inch DBHs were utilized to compare current species composition with historical tree composition. As such, we used five cycles of data: cycle 4(1989), cycle 5 (1999-2003), cycle 6 (2004-2008), cycle 7 (2009-2013), and cycle 8 (2014-2019). For the mortality analysis, we used FIA data that had been remeasured annually four times.

In Missouri, FIA started its annual measurement scheme beginning in 1999. The Annual inventory program was designed to complete a cycle in five years; however, the target was not always met due to funding and other limitations. There were four annual inventory cycle data available for the state of Missouri: cycle 5 (1999-2003), cycle 6 (2004-2008), cycle 7(2009-2013), and cycle 8(2014-2019). Data in Cycle 8 were incomplete. Each annual plot is nested with four subplots. All large trees (>5" DBH) were selected from subplots remeasured four times. As such, 39,490 trees (Table.4.1) were utilized in this analysis. Individual tree-level data and associated stand-level information were gathered from the FIA database. Data include individual tree level information such as diameter, height, crown condition, and status, stands level information such as stocking, basal area, ecoregion, and location (see the data wrangling scheme in Fig 4.2.).

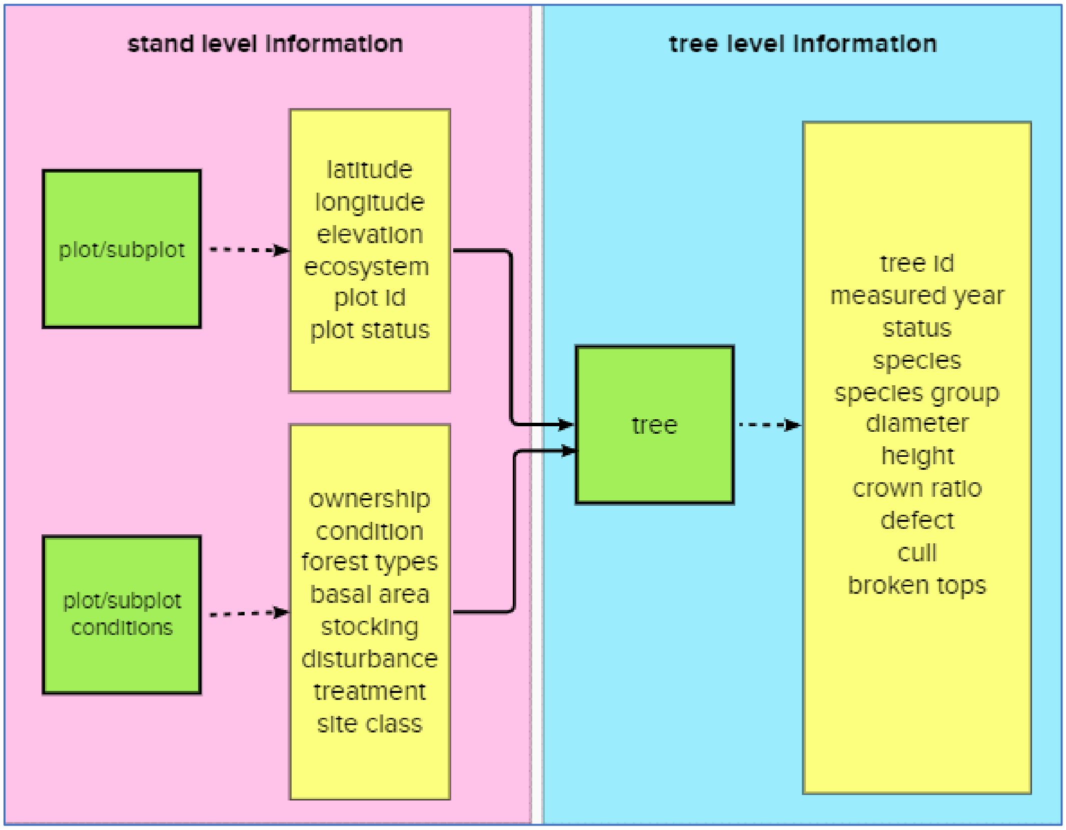

Figure 4.2 FIA data wrangling scheme. Individual tree level information was gathered using both tree and stand level conditions.

As a baseline tree record, we chose all live trees greater than 5 inches dbh from cycle 5. Harvested and missing trees in the dataset were ignored. There were 31,037 live trees in cycle 5 (Table 4.1). If baseline trees' status changed to dead in the following cycle, i.e., cycle 6, those trees were considered dead trees. Similarly, new tree records in the same subplot were classified as ingrowth. As such, we analyzed 39,940 individual tree records that had been remeasured four times in the Missouri Ozarks. In Table 4.1, NAs mean null records on the respective measurement cycles; those were ingrowth records from past cycles.

Table 4.1 Tree counts, status, and measurement cycles recorded by FIA from the Missouri Ozarks.

Cycle	Dead	Live	NAs	PrevDead**	Ingrowth	Total
5 (1999-2003)	0	31,037	8,903	0	0	39,940
6 (2004-2008)	1,881	29,572	6,340	0	2,147	39,940
7 (2009-2013)	2,312	29,850	3,364	1,881	2,533	39,940
8 (2014-2019)*	2,819	29,604	0	4,193	3324	39,940

*cycle 8 is not completely measured, **PrevDead= Number of trees already dead in previous cycle.

4.2.2.1 Major species grouping classes

In FIA data, each tree record has a species code (SPCD) and species group code (SPGRPCD). This study mainly focuses on oaks mortality; the FIA data classify oaks into four major species groups; selected white oaks (SPGRPCD=25), selected red oaks (SPGRPCD=26), other white oaks (SPGRPCD=27), and other red oaks (SPGRPCD=28). We combined these four groups into two broad groups: white oaks and red oaks. The white oaks group includes white Oak (*Quercus alba),* swamp white oak (*Quercus bicolor),* overcup oak (*Quercus lyrata),* bur oak (*Quercus macrocarpa),* swamp chestnut oak (*Quercus michauxii),* chinkapin oak (*Quercus muehlenbergii),* chestnut oak (*Quercus prinus),* and post oak (*Quercus stellate).* Similarly, the red oaks group includes scarlet oak (*Quercus coccinea*), Southern red oak (*Quercus falcata*), blackjack oak (*Quercus marilandica*), water oak (*Quercus nigra*), pin oak (*Quercus palustris*), willow oak (*Quercus phellos*), northern red oak (*Quercus rubra*), and black oak (*Quercus velutina*). Additional species groups were made using the species dominance threshold. Species groups higher than 6% of total sample trees were classified into individual groups. All others included less than 6% of the total sample size were classified into the "Others" grouping. Seven major groups were made: White oaks, Red oaks, hickory, pine, Other Eastern Softwood (OES), Other Eastern Soft Hardwood (OESH), and others. Hickory group includes hickory species (*Carya spp.*) and pecan (*Carya illinoinensis*), pine is shortleaf pine (*Pinus echinata)*, OES includes eastern redcedar (*Juniperus virginiana*), and OESH includes elm species (*Ulmus spp.*), sycamore species (*Platanus spp.*), hackberry species (*Celtis spp.*), etc.

4.2.2.2 Crown dieback and annual growth rate

FIA measures crown dieback for live trees greater than 5" DBH. Percent of dead material on a tree due to the dieback is quantified into 21 crown dieback codes (CDIEBKCD). The CDIEBKCD is optional in phase 2 and required in phase 3 (i.e., Forest Health Monitoring (FHM)) (O'Connell et al. 2015). This means not all 39,940 trees had crown dieback info. Thus, we have chosen remeasured trees from FHM plots for crown dieback analysis. We reclassified CDIEBKCD into two broad classes: dieback (>5%) and no dieback (0-5%).

4.2.3 Historical data

We used historical individual tree species derived from original General Land Office (GLO) field notes and earlier French/Spanish land survey documents by the geographic resources center, Department of Geography, University of Missouri. We obtained a shapefile loaded with individual tree species; the Geographic Resource Center interpreted these individual records. They have interpreted 69 historical individual species in the Ozarks with 247,377 individual trees. They were grouped into seven major species groups, similar to FIA-based species groups (Table 4.2).

Table 4.2 Historical witness tree count and associated major species groups in the Missouri Ozarks.

Major species group	Count
White oaks	123,796
Red oaks	76,346
Pine	19,547
Hickory	11,931
Other eastern softwood (OES)	333
Other eastern soft hardwood (OESH)	8,575
Others	6,849
Total count	**247,377**

Hanberry et al. (2014) reconstructed historical forest conditions in the Missouri Ozarks using these witness trees. They have reconstructed seven major forest conditions: Oak Closed Woodland, Oak Forest, Oak Open Woodland, Oak Savanna, Oak/Pine Closed Woodland, Oak/Pine Forest, and Oak/Pine Open Woodland (Fig.4.3). We obtained geospatial data of these reconstructed forest conditions and processed to examine associations between recent oak mortality and historical forest conditions. Actual FIA plot locations were obtained through spatial data requests from the USDA Forest Service. Actual FIA plot’s locations were spatially joined with historical forest conditions.

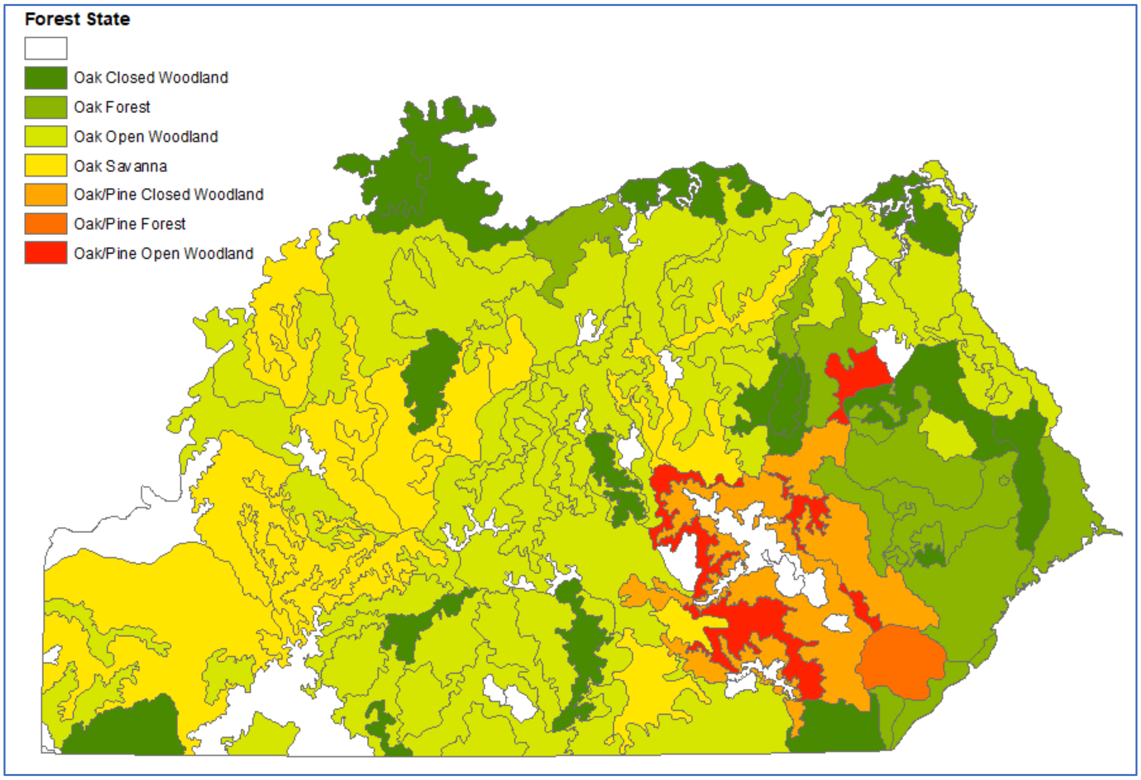

Figure 4.3 Historical forest conditions in Missouri Ozarks. Forest conditions were reconstructed by Hanberry et al. (2014) using historical public land survey (PLS) witness trees data. White space in this map was due to insufficient trees data during survey.

4.2.4 Drought data

The monthly Palmer Drought Severity Index (PDSI) for each climatic division in Missouri was downloaded from the National Climatic Data Center (https://www7.ncdc.noaa.gov/CDO/CDODivisionalSelect.jsp). They reported monthly PDSI indicating the severity of a wet or dry spell. PDSI ranges mainly from -6 to +6; negative values indicate dry and positive values indicate a wet period. Monthly reported PDSI values were reclassified as growing (May to October) and nongrowing (November to April) season. Missouri has six climatic divisions (Fig. 4.4), each with monthly PDSI values in the database.

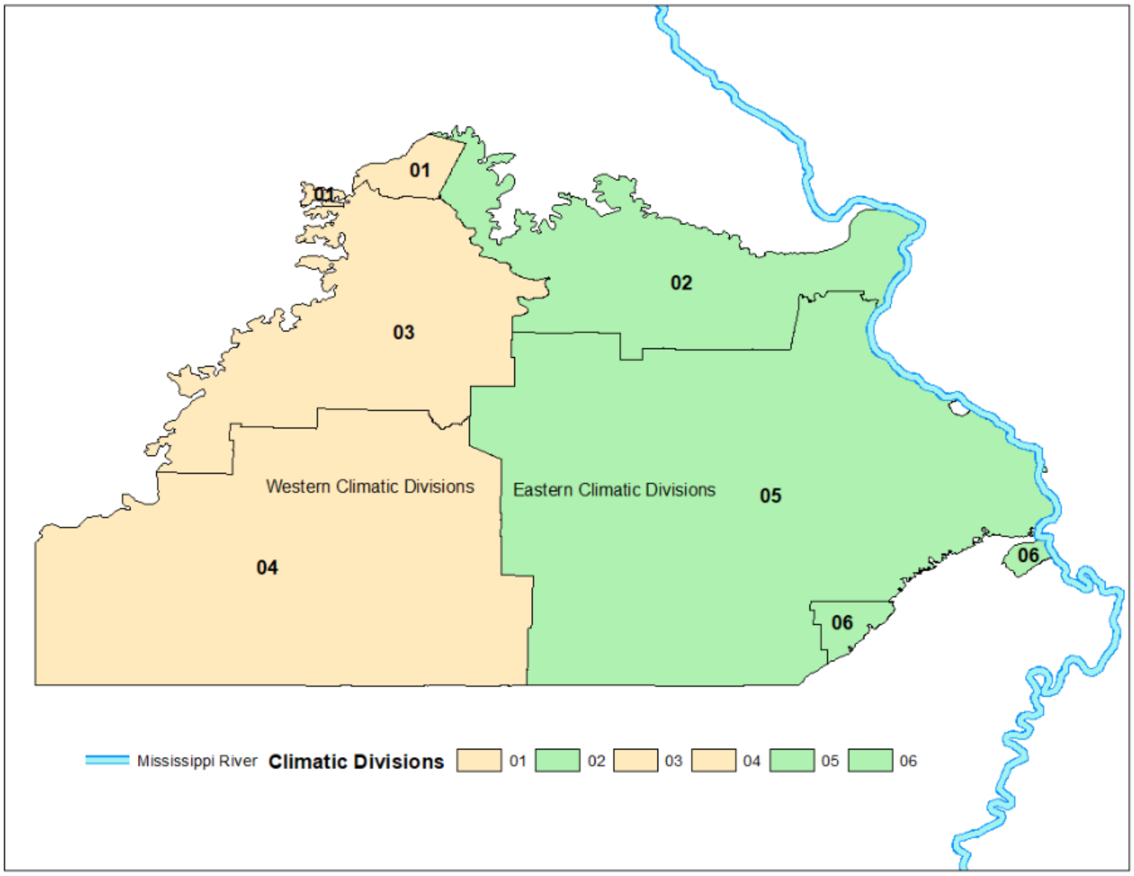

Figure 4.4 Climatic divisions in the Missouri Ozarks. Western climatic divisions include division 1, 3, and 4. The Eastern climatic division includes divisions 2, 5, and 6.

Missouri includes six climatic divisions: the Ozarks has all of them. However, more than half of the Ozark area is in divisions 4 and 5 (See Fig. 4.4). Region-wide drought was suspected as a potential cause of the oak decline in the Ozark (Fan et al.2012). Further, the frequencies of extreme drought events in eastern Missouri were lower than in western Missouri (Drew and Chen 1997). We suspect that the climatic divisions may relate to oak mortality. Thus, we classified six climatic divisions into two broad classes based on location; Eastern climatic divisions (divisions 2, 5, and 6) and Western climatic divisions (divisions 3, 4, and 1).

4.2.5 Methods

4.2.5.1 Historical and FIA species dominance proportion

The proportion of historical forest species groups was estimated using equation 1. Similarly, the proportion of FIA species groups was estimated using equation 2. A comparison between these two proportions was made to examine the change in dominance across the region.

$$H_i = \frac{\sum h_i}{\sum T} \tag{1}$$

Here H_i is the historical proportion of species group *i*. h_i is the number of historical tree counts of species group *i*, T is the total number of historical trees counted in the entire Missouri Ozark.

$$S_i = \frac{\sum s_i}{\sum T_i} \tag{2}$$

Here S_i is the proportion of species group *i* in FIA data. s_i is a number of tree counts of species group *i*, T_i is the total number of trees of all species groups.

4.2.5.2 Mortality and ingrowth proportion

Mortality and ingrowth proportions were calculated using equations 3 and 4, respectively.

$$p_i = \frac{\sum d_i}{\sum t_i} \tag{3}$$

Here p_i is the mortality proportion of the species group *i*. d_i is a number of dead trees of a species group *i*, t_i is the total number of trees recorded in a species group *i*.

$$\mathrm{q_i} = \frac{\sum \mathrm{I_i}}{\sum \mathrm{t_i}} \tag{4}$$

Here q_i is the ingrowth proportion of species group *i*. I_i is number of ingrowth trees of species group *i*, t_i is the total number trees recorded in species group *i*.

4.2.5.3 Estimating annual growth rate on crown dieback trees

Crown dieback data are only available on FHM plots. Annual growth rate was calculated using remeasured dbh of individual trees. We calculated relative annual growth for each tree using equation 5.

$$\text{annual diameter growth (\%)} = \frac{\left(\left(\text{dbh}_\text{j} - \text{dbh}_\text{i}\right) / \text{dbh}_\text{i}\right)}{\text{year}_\text{j} - \text{year}_\text{i}} * 100 \qquad (5)$$

Here, dbh_j is dbh measured at cycle j (j=cycle6, cycle7, cycle8), dbh_i is the dbh measured at cycle *i* (i=cycle 5, cycle 6, and cycle 7). year_j, and year_i are measurement year in cycle *i* and *j*.

4.3 Results

4.3.1 Historical vs recent species composition

We compared historical PLS witness tree records to the FIA data (measured between 1989 and 2019) in the Missouri Ozarks. From FIA data, only live trees greater than five inches dbh were considered in this analysis. Trees from all available FIA subplots and cycles were used (not limited to remeasured plots). The PLS data suggested that oaks covered more than two-thirds of all species: white oak group 48.68%, and the red oak group 29.96% (Fig. 4.5), followed by Shortleaf pine 6.76%, hickory species 5.52%, and other species 9.08%. Based on FIA data (1989-2019), the proportion of red and white oak groups decreased in recent decades (Fig. 4.5). The red oak group decreased at a higher rate than the white oak group. Red oaks decreased at the highest rate between FIA cycles 4 and 5 and continuously decreased over time at a higher rate than white oak and other species in the Ozark (Fig. 4.5).

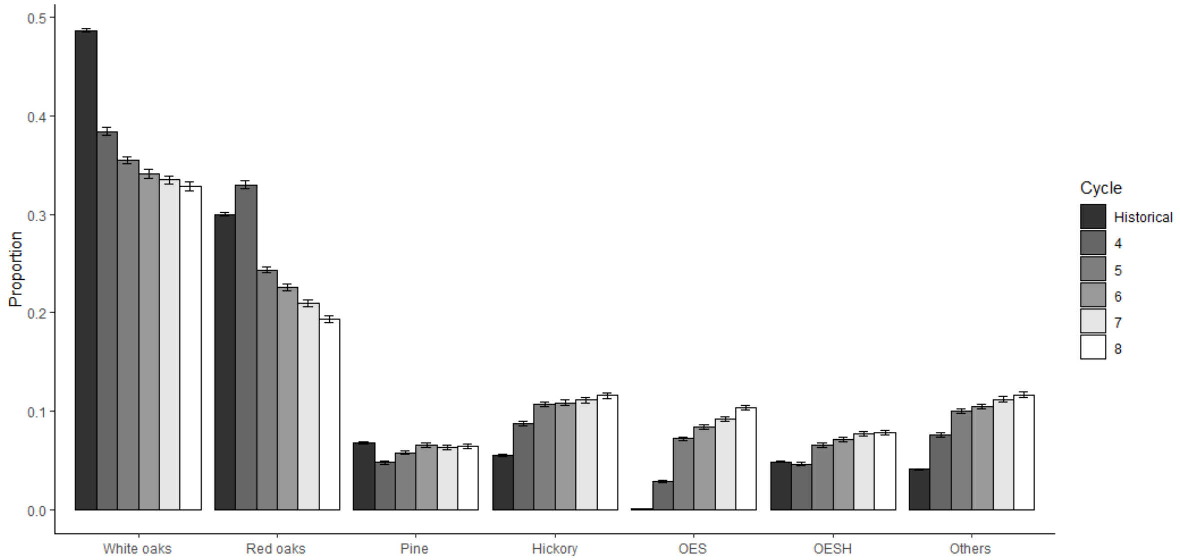

Figure 4.5 Proportion of historical vs current major species groups in the Missouri Ozark. Proportion represents a ratio between selected species group count to the total number of trees measured FIA cycle or PLS records. Error bar represents 95% confidence interval for the observed count of the major species group's proportion. Only trees with a greater than 5-inch dbh were used in this process. Here OES is other eastern softwood, OESH (Other eastern soft hardwood).

In the Missouri Ozark, the white oak group was the most dominant species group, followed by the red oak group. Historically, the white oak group proportion was 48.68% of all species recorded. However, the dominance of white oaks has decreased in recent decades. FIA inventory data showed that the white oak proportion decreased by more than 10% by 1989 in cycle 4 compared to the historical proportion. The white oak decreased in recent decades; by 38.41%, 35.5%, 34.12%, 33.49%, and 32.91% respectively in cycles 4, 5, 6, 7, and 8.

The highest proportion of red oaks measured in 1989 measured (i.e., cycle 4), 32.99% (which was 3.03% higher than the historical proportion). After cycle 4, the red oak proportion continuously decreased over time; 24.34%, 22.60%, 21.00%, and 19.57 % in cycles 5, 6, 7, and 8 respectively. The shortleaf pine proportion decreased between historical records and cycle 4, but the proportion increased over FIA inventory cycles. Besides these three species groups, the proportions of all other species groups increased compared to their respective historical proportions. The majority of these groups continuously increased over inventory cycles. Species groups: other eastern softwood (OES), hickory, and others increased over time. Historically, shortleaf pine was the third most dominant species group, but in recent decades, hickory, other eastern softwood, and other eastern hardwood overtook pine.

4.3.2 Mortality vs ingrowth

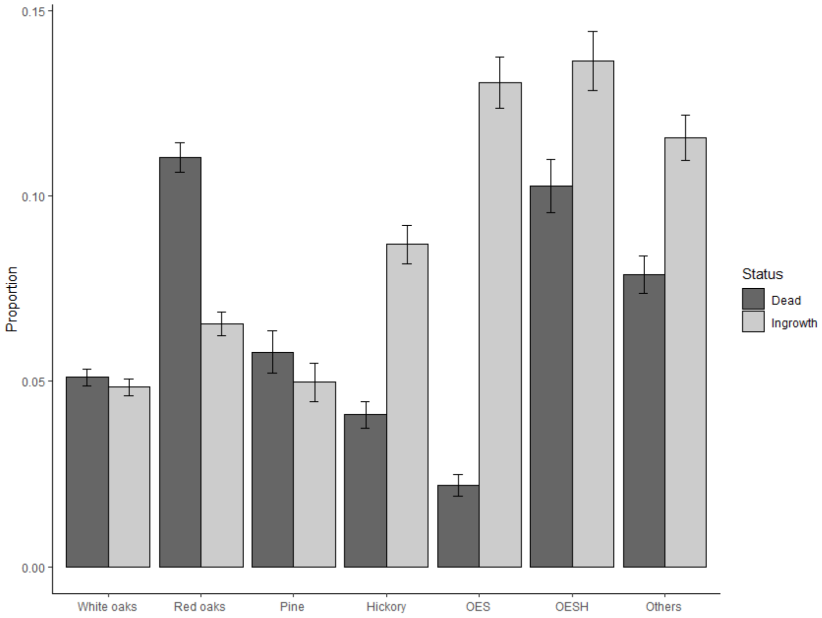

Figure 4.6 Dead and ingrowth proportion in major species groups in the Missouri Ozarks during 2004-2019. Here OES is other eastern softwood, OESH (Other eastern soft hardwood). Mortality and ingrowth proportion was calculated based on four-time remeasured subplots and used only larger trees (greater than 5-inch diameter). Only red oaks species group had a significant negative recruitment (mortality proportion was significantly higher than the ingrowth proportion). Error bar represents 95% confidence intervals for the observed dead and ingrowth proportion.

Between 2004 and 2019, species group performance relative to the mortality and ingrowth proportion differed among groups (Fig. 4.6). Ingrowth was greater than mortality for species groups OES, hickory, OESH, and others. Mortality and ingrowth were similar for white oaks and pine. The red oaks' morality proportion was nearly double than the ingrowth; in contrast, hickory and OES got more than double ingrowth than morality (Fig 4.6).

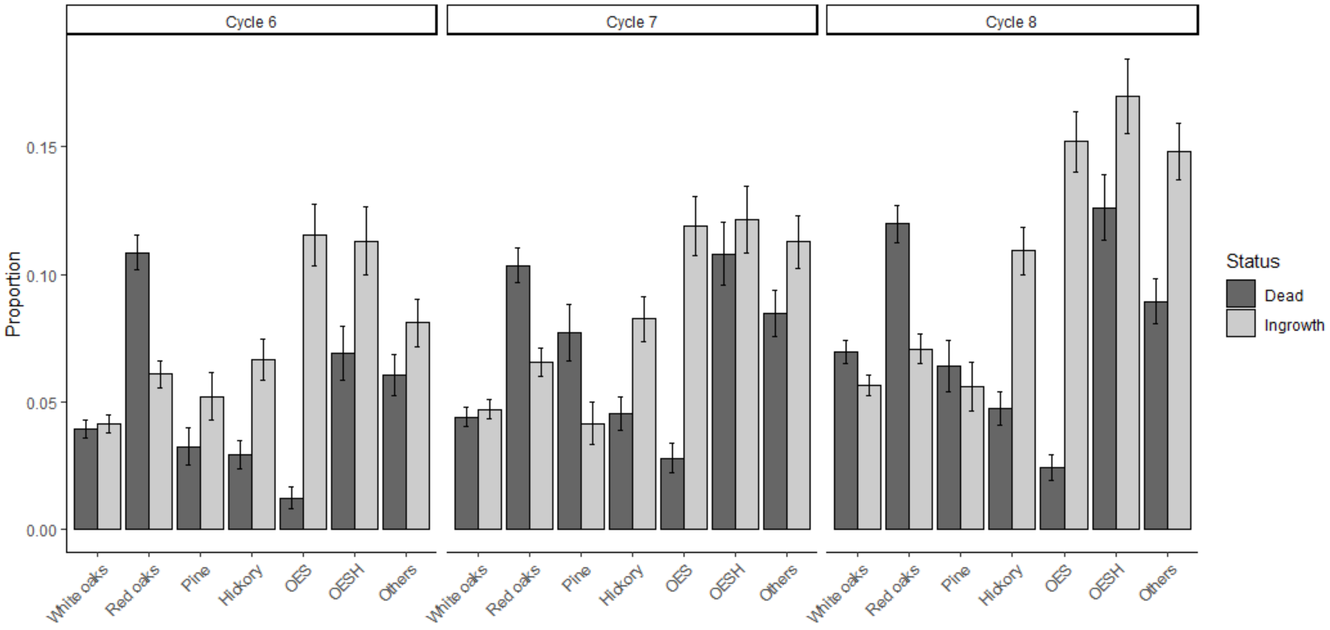

Figure 4.7 Dead and ingrowth proportion in Missouri Ozark for three recent FIA cycles. X-axis = species groups and y-axis = proportion of dead and ingrowth trees. Error bars indicate 95% confidence intervals for the observed dead and ingrowth proportion.

Higher mortality rates were observed in red oak groups in all cycles compared to other species groups except OESH in cycle 8 (Fig. 4.7). Moreover, ingrowth proportions were consistently lower than the mortality proportions over time which puts the red oak groups future dominance at risk. The mortality rate increased over time in the OESH group, but their ingrowth was higher than the mortality proportion. Species groups such as OES, hickory, and others had a lower mortality rate in all cycles and higher ingrowth proportions. As such, their dominance increased in recent decades.

4.3.3 Mortality and drought

NOAA records monthly PDSI for individual climatic divisions. Based on two broad climatic division classifications, eastern and western (Fig. 4.4), a time-series graph was constructed (Fig 4.8) to show the average PDSI overtime in the climatic division.

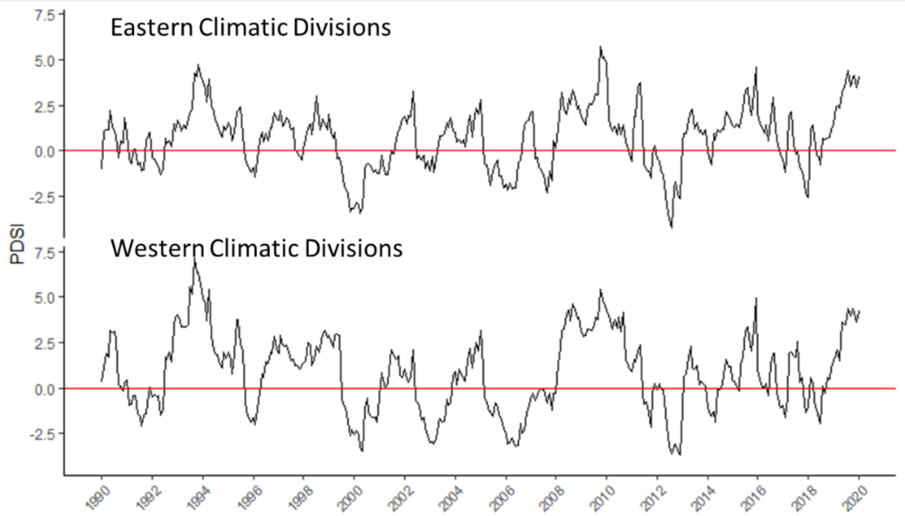

Figure 4.8 Average PDSI overtime in reclassified climatic divisions in Missouri Ozarks (1990-2020).

In 1999, both eastern and western climatic divisions were exposed to severe drought (Fig. 4.8). From 1999 to 2008, the eastern climatic division received two or more wet seasons than the western division.

4.3.3.1 Reconstructed drought indices

Monthly PDSI was reconstructed for the growing season (April to July). The impact of historical drought on tree mortality was examined using past growing seasons' average PDSI, which gives us a better understanding of how long the drought impact lasted. On average, the western climatic division was exposed to prolonged and continuous drought between 2000 and 2008, while the eastern division had some excellent wet seasons. Cumulative average PDSI, using the past five growing seasons average, rarely became negative in the eastern, but western divisions were continuously exposed to severe drought (Fig. 4.9). Another series of cumulative drought events were seen between 2012 and 2014, similar to the 2000 and 2008 events, the western division was exposed to a more extended drought than the eastern division.

Figure 4.9 Average PDSI using past five growing seasons in western and eastern climatic division of Missouri Ozarks

4.3.3.2 Correlation between annual mortality and drought

Mortality proportions for major species groups were estimated for each climatic division. The average annual mortality rate was calculated by dividing the average cycle length. Average growing seasons PDSI for each climatic division was estimated based on the average of the past ten growing seasons. Correlation analysis was run to see how the annual mortality rate correlated with historical drought events in the region. Correlation coefficients can be found in tables 4.3 and 4.4.

Table 4.3 Correlation coefficients between annual mortality rate and average growing seasons PDSI from measured year to past ten growing seasons in the eastern climatic division

AvgPDSI (growing seasons)	White oaks		Red oaks		Pine		Hickory		OES		OESH		Others	
	r	p	r	p	r	p	r	p	r	p	r	p	r	p
YRn	0.21	0.44	-0.17	0.53	0.07	0.79	-0.04	0.89	0.24	0.38	0.39	0.14	0.65	0.007**
YRn : YRn-1	0.04	0.89	-0.16	0.55	0.16	0.55	-0.05	0.86	-0.07	0.80	0.34	0.20	0.56	0.03*
YRn : YRn-2	0.24	0.38	0.11	0.68	0.36	0.17	0.26	0.32	0.04	0.87	0.54	0.03*	0.46	0.08.
YRn : YRn-3	0.31	0.24	0.28	0.30	0.62	0.01*	0.45	0.08.	0.12	0.66	0.54	0.03*	0.37	0.15
YRn : YRn-4	0.51	0.05.	0.33	0.21	0.62	0.01*	0.67	0.005**	0.37	0.15	0.66	0.01*	0.51	0.04*
YRn : YRn-5	0.66	0.01*	0.40	0.12	0.70	0.003**	0.69	0.003**	0.42	0.10	0.52	0.04*	0.54	0.03*
YRn : YRn-6	0.72	0.002**	0.36	0.17	0.52	0.04*	0.55	0.03*	0.25	0.35	0.52	0.04*	0.45	0.08.
YRn : YRn-7	0.68	0.004**	0.28	0.30	0.43	0.09.	0.35	0.19	0.00	0.99	0.39	0.13	0.33	0.21
YRn : YRn-8	0.71	0.002**	0.23	0.38	0.34	0.20	0.21	0.43	-0.08	0.76	0.50	0.05.	0.09	0.75
YRn : YRn-9	0.72	0.002**	0.23	0.39	0.36	0.17	0.26	0.34	-0.13	0.63	0.46	0.08.	0.17	0.52
YRn : YRn-10	0.59	0.02*	0.17	0.54	0.23	0.40	0.10	0.73	-0.07	0.79	0.39	0.13	0.03	0.92

Table 4.4 Correlation coefficients between annual mortality rate and average growing seasons PDSI from measured year to past ten growing seasons in the western climatic division

AvgPDSI (growing seasons)	White oaks		Red oaks		Pine		Hickory		OES		OESH		Others	
	r	p	r	p	r	p	r	p	r	p	r	p	r	p
YRn	-0.24	0.37	-0.53	0.04*	-0.08	0.77	-0.27	0.32	0.02	0.95	0.27	0.32	0.31	0.24
YRn : YRn-1	-0.52	0.04*	-0.64	0.01*	0.36	0.17	-0.16	0.55	0.51	0.05.	0.36	0.17	0.42	0.10
YRn : YRn-2	-0.27	0.32	-0.54	0.03*	0.53	0.04*	0.13	0.63	0.62	0.01*	0.53	0.04*	0.49	0.06*
YRn : YRn-3	-0.15	0.58	-0.47	0.07.	0.49	0.05.	0.13	0.64	0.56	0.02*	0.64	0.01*	0.49	0.06*
YRn : YRn-4	0.00	0.99	-0.37	0.16	0.53	0.04*	0.27	0.30	0.61	0.01*	0.71	0.002**	0.56	0.02*
YRn : YRn-5	0.27	0.31	-0.17	0.52	0.40	0.13	0.31	0.24	0.70	0.003**	0.57	0.02*	0.48	0.06*
YRn : YRn-6	0.29	0.27	-0.09	0.74	0.29	0.28	0.17	0.53	0.70	0.003**	0.44	0.09.	0.31	0.24
YRn : YRn-7	0.32	0.22	-0.01	0.98	0.39	0.13	0.21	0.44	0.67	0.004**	0.34	0.20	0.16	0.57
YRn : YRn-8	0.33	0.22	0.10	0.72	0.43	0.10	0.29	0.28	0.37	0.16	0.17	0.53	0.18	0.51
YRn : YRn-9	0.42	0.10	0.14	0.60	0.44	0.09.	0.33	0.21	0.59	0.02*	0.33	0.21	0.09	0.74
YRn : YRn-10	0.20	0.45	0.20	0.45	0.34	0.19	0.33	0.21	0.09	0.75	0.18	0.52	0.26	0.33

We did not find any negative correlation between average PDSI and annual mortality rate in the eastern climatic division. However, we found a significant negative correlation between average PDSI of the past two growing seasons and white oaks group annual mortality rate in the western climatic division. Similarly, there was a significant negative correlation between the average PDSI of past growing seasons (up to four growing seasons) and the annual mortality rate for the red oaks group. The relation between red oak mortality and average PDSI can be visualized by comparing the three seasons' average PDSI in a linear regression plot (Fig. 4.10).

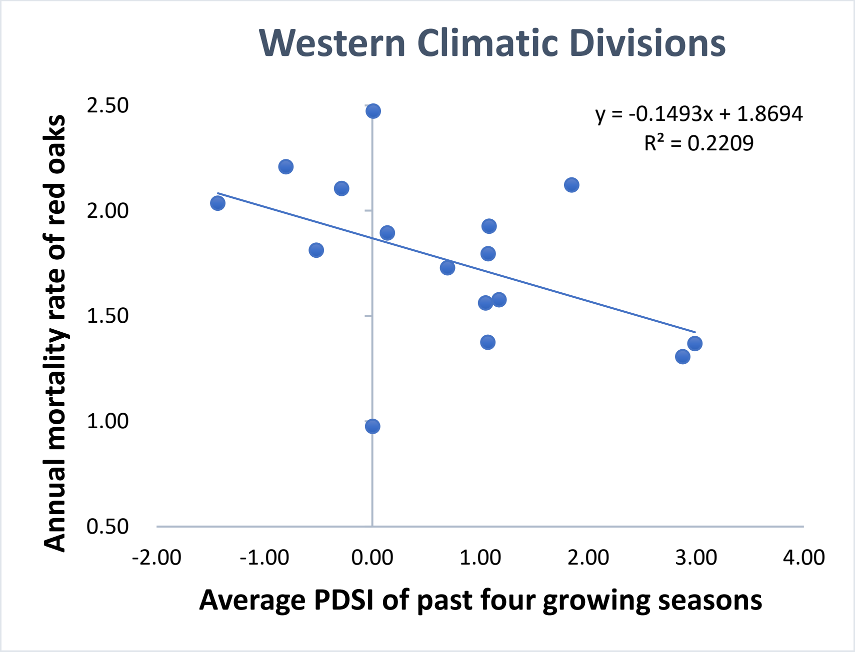

Figure 4.10: Linear relation between red oak annual mortality and the past four seasons average PDSI in the western climatic divisions.

4.3.4 Historical forest condition and recent oak motility

We examined how land-use impacts recent oak mortality. We examined red and white oak groups and compared their mortality proportions over historical forest conditions.

4.3.4.1 Comparing white and red oak mortality over time

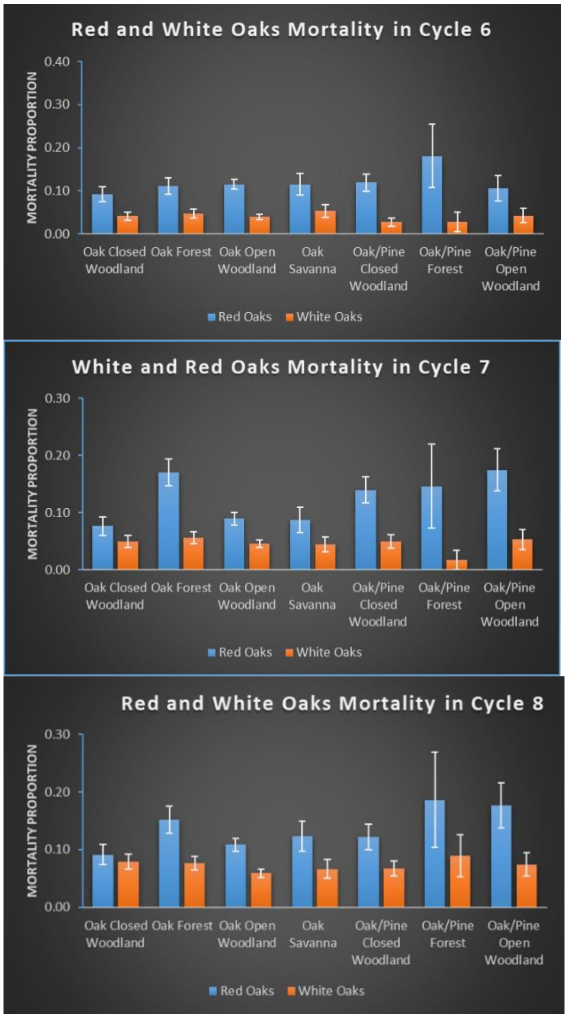

Figure 4-11: Comparing white and red oak mortality to historical forest conditions in the Missouri Ozarks. The X-axis in all three figures represents historical forest conditions and y-axis represents mortality proportion. Error bars indicate 95% confidence intervals for the observed red and white oak mortality proportion.

Figure 4.11 Red and white oak mortality in cycles 6-8 compared across historical forest conditions.

In cycle 6, red oak mortality proportions were significantly higher than white oaks across all historical forest conditions (Fig.4.11). Red and white oaks' mortality proportions were dissimilar across all historical forest conditions and cycles. In cycle 7, red oaks' mortality proportions were significantly higher than white oaks except in the oak closed woodland forest condition. We are 95% confident that there was sufficient evidence to conclude that red and white oak mortality proportions differ among these forest conditions. In cycle 8, red oaks mortality proportions were significantly higher in all forest conditions except the oak closed woodland and oak/pine forest.

Since the red oaks group had the highest mortality rate, we examined it separately to observe how red oaks mortality varied with historical forest conditions. Figure 4.12 shows how red oaks mortality in FIA cycles 6-8 differs among historical forest conditions. We performed a chi-square test of independence to compare red oak mortality proportions to the historical forest conditions.

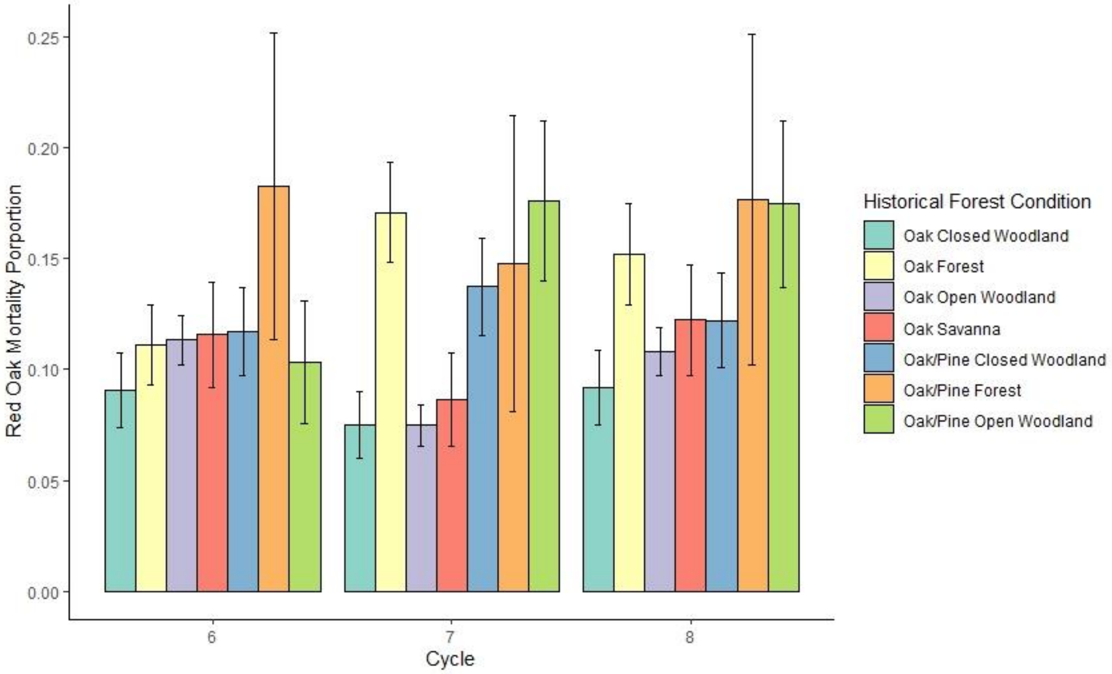

Figure 4.12 Bar plot of red oak group mortality proportion vs historical forest conditions in the Missouri Ozark for three recent FIA cycles. Error bars indicate 95% confidence intervals for the observed red oak mortality proportion. Sample size based on cycle 6; oak closed woodland (1048), oak forest (1108), oak open woodland (3044), oak savanna (640), oak/pine closed woodland (948), oak/pine forest (104), oak/pine woodland (417).

We found red oaks’ mortality proportions depended on the historical forest conditions. We observed the lowest red oaks morality proportion on oak closed woodland across all cycles (Fig. 4.12). Oak closed woodland had a significantly lower mortality rate than oak/pine forest, and oak/pin closed woodland across all cycles. Oak, and oak/pine forest conditions consistently experienced higher red oak mortality. Further, we performed a post-hoc pairwise chi-square test for multiple comparisons. A detail of the post-hos test can be found in Table 4.5.

Table 4.5 Post-hoc pairwise chi-square test for multiple comparison of red oaks mortality proportion to the historical forest conditions using recent FIA data.

Historical forest condition	P value of chi-square test		
	Cycle 6	Cycle 7	Cycle 8
Oak Closed Woodland : Oak Forest	0.14	0.00***	0.00***
Oak Closed Woodland : Oak Open Woodland	0.05**	1.00	0.15
Oak Closed Woodland : Oak Savanna	0.12	0.47	0.06**
Oak Closed Woodland : Oak/Pine Closed Woodland	0.06*	0.00***	0.04**
Oak Closed Woodland : Oak/Pine Forest	0.00***	0.03**	0.02**
Oak Closed Woodland : Oak/Pine Open Woodland	0.52	0.00***	0.00***
Oak Forest : Oak Open Woodland	0.88	0.00***	0.00***
Oak Forest : Oak Savanna	0.83	0.00***	0.12
Oak Forest : Oak/Pine Closed Woodland	0.72	0.05**	0.08*
Oak Forest : Oak/Pine Forest	0.04**	0.69	0.66
Oak Forest : Oak/Pine Open Woodland	0.73	0.88	0.36
Oak Open Woodland : Oak Savanna	0.92	0.37	0.34
Oak Open Woodland : Oak/Pine Closed Woodland	0.80	0.00***	0.28
Oak Open Woodland : Oak/Pine Forest	0.04**	0.02**	0.07*
Oak Open Woodland : Oak/Pine Open Woodland	0.59	0.00***	0.00***
Oak Savanna : Oak/Pine Closed Woodland	0.99	0.00***	1.00
Oak Savanna : Oak/Pine Forest	0.08*	0.10	0.22
Oak Savanna : Oak/Pine Open Woodland	0.59	0.00***	0.03**
Oak/Pine Closed Woodland : Oak/Pine Forest	0.08*	0.91	0.21
Oak/Pine Closed Woodland : Oak/Pine Open Woodland	0.51	0.09*	0.02**
Oak/Pine Forest : Oak/Pine Open Woodland	0.04	0.63	1.00

*** significant different at 99% significant level, ** significant different at 95% significant level, * significant different at 90% significant level

4.3.5 Oak mortality and ownership

We classified ownership into two groups: public and private. Public ownership includes forestland owned by the USDA Forest Service, other federal agencies, and state, and private ownership includes forest owned by private and Native American people. Most forestlands were privately owned. The red oak proportion did not differ statistically between the private and public ownerships. However, the red oak mortality rate was higher when comparing public forestland to privately-owned forests. The Chi-square test shows that the red oak mortality proportion was higher in public-owned forestland than in private land at each cycle (Table 4.6 and Fig. 4.13). We also investigated red oak presence percent in these different ownerships.

Table 4.6 Red oak mortality in Missouri Ozark over three different FIA measurement cycles and forest ownership

	Cycle 6			Cycle 7			Cycle 8		
Ownership	Dead	Live	Mortality proportion (%)	Dead	Live	Mortality proportion (%)	Dead	Live	Mortality proportion (%)
Public	308	2,235	12.11	303	2,146	12.37	310	1,977	13.55
Private	554	4,850	10.25	484	4,749	9.25	578	4,561	11.25

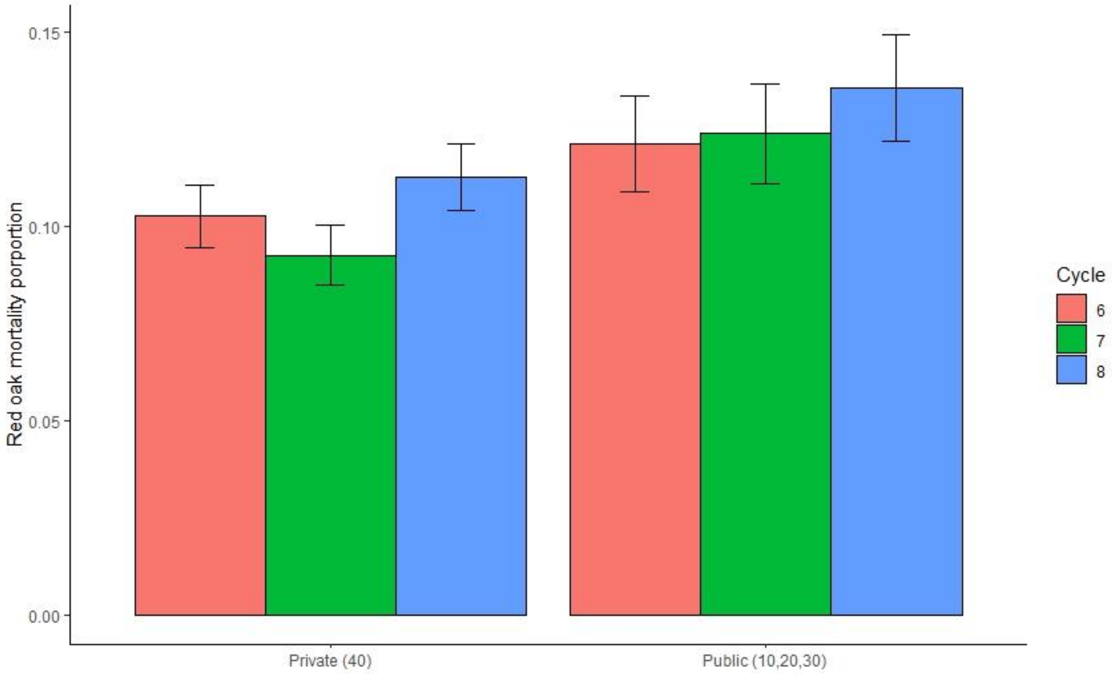

Figure 4.13 Bar plot of red oak group mortality proportion vs Ownership in the Missouri Ozark. Error bars indicate 95% confidence intervals for the observed red oak mortality proportion.

4.3.6 Red oak mortality and treatments

Approximately 10% of FIA subplots received stand treatments in each cycle. Treatments include removal of trees, site preparation, regeneration, and other silvicultural treatments. Subplots in which red oak occurred were further classified into two groups; treated and not treated. Red oak presence probability in both classes was similar. The mortality rate between these two groups was analyzed using the chi-square test. We found a higher red oak mortality rate in treated subplots than in untreated subplots in all cycles, but only cycles 7 and 8 were statistically significant (Fig. 4.14).

Table 4.7: Red oak mortality in Missouri Ozark over three different FIA measurement cycles and silvicultural treatments

	Cycle 6			Cycle 7			Cycle 8		
Treatment	Dead	Live	Mortality proportion (%)	Dead	Live	Mortality proportion (%)	Dead	Live	Mortality proportion (%)
Treated	61	453	11.87	98	421	18.88	116	486	19.27
Not treated	802	6,636	10.78	697	6,478	9.71	774	6,059	11.33

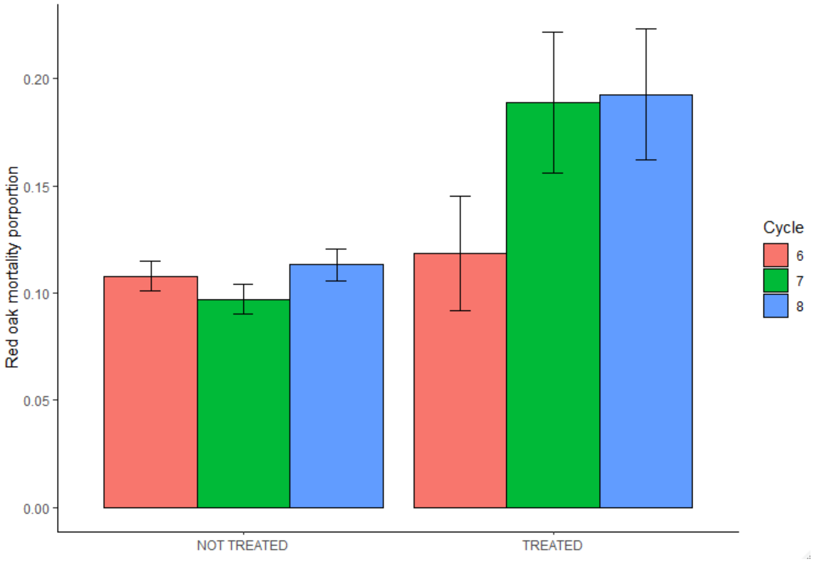

Figure 4.14: Bar plot of red oak group mortality proportion vs treatments in the Missouri Ozark. Error bars indicate 95% confidence intervals for the observed red oak mortality proportion.

4.3.7 Red oak morality and disturbances

Disturbed stands had a significantly higher red oak mortality proportion than the non-disturbed stands in cycles 7 and 8. Particularly stands impacted by climate related disturbances, including drought, wind, ice, and flooding.

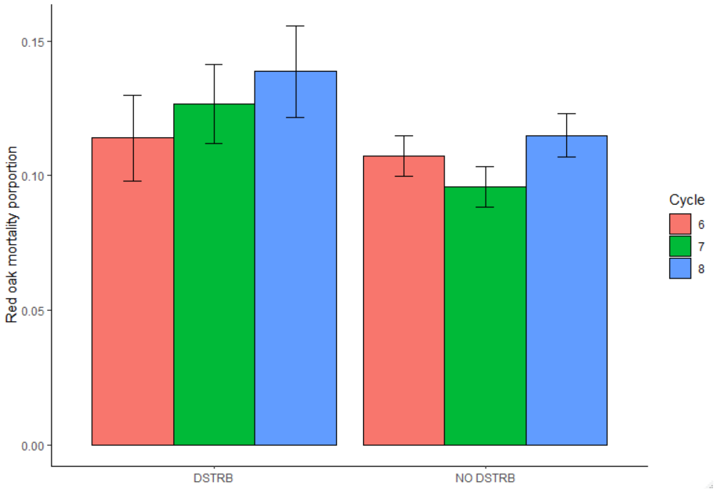

Figure 4.15 Bar plot of red oak group mortality proportion vs disturbance in the Missouri Ozarks. Error bars indicate 95% confidence intervals for the observed red oak mortality proportion.

Table 4.8 Red oak mortality in Missouri Ozark over three different FIA measurement cycles and disturbance

	Cycle 6			Cycle 7			Cycle 8		
Disturbance	Dead	Live	Mortality proportion (%)	Dead	Live	Mortality proportion (%)	Dead	Live	Mortality proportion (%)
Disturbed	167	1299	11.39	236	1628	12.66	209	1298	13.86
Not disturbed	696	5790	10.73	559	5271	9.58	681	5247	11.48

4.3.8 Crown dieback and relative diameter growth

Crown dieback data can be seen in table 4.9. It shows a higher dieback proportion in red oaks compared to white oaks. We also calculated these trees' annual growth and compared the growth rate (Fig 4.16). The annual diameter growth in both white and red oak species groups was lower in crown dieback trees. The annual growth rate in red oaks without crown dieback was similar in both climatic divisions across all cycles (Fig. 4.16). The annual growth rate was > 1.5% across all cycles and climatic divisions. However, the annual growth of red oak with dieback was consistently lower in the western climatic division compared to the eastern climatic across all cycles.

Table 4.9 Dieback records in Oak groups in Ozark

Cycle	Group	Eastern			Western		
		Dieback	No dieback	Dieback proportion (%)	Dieback	No dieback	Dieback proportion (%)
6	Red oaks	25	229	9.84	7	105	6.25
6	White oaks	19	465	3.93	11	152	6.75
7	Red oaks	55	356	13.38	19	143	11.73
7	White oaks	43	676	5.98	18	259	6.50
8	Red oaks	76	303	20.05	27	257	9.51
8	White oaks	82	739	9.99	25	403	5.84

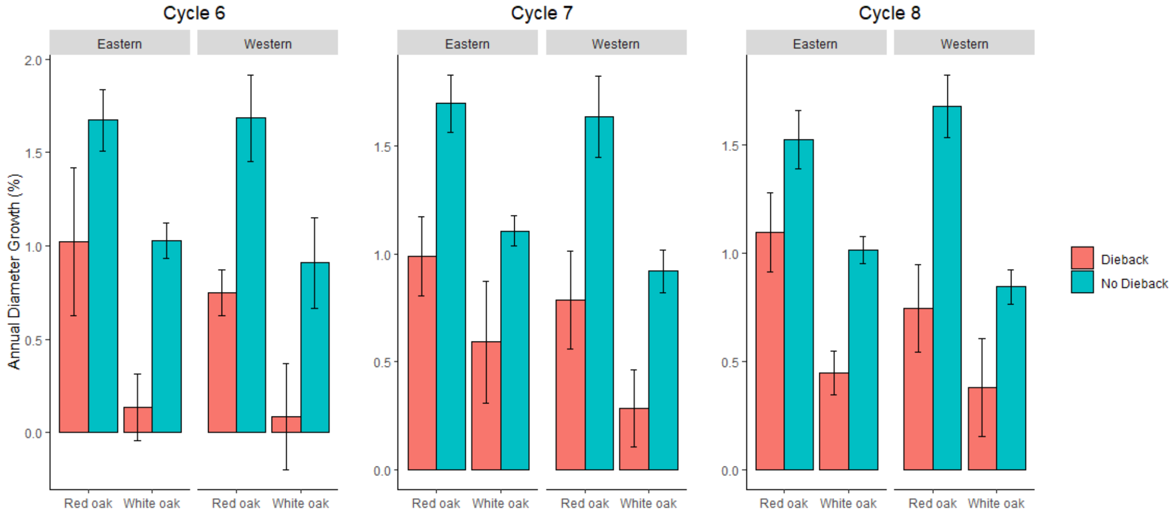

Figure 4.16 Annual diameter growth percent for red and white oak trees with and without crown dieback. Bar graph represents mean diameter growth and error bars indicate 95% confidence intervals for the observed annual diameter growth.

4.4 Discussions

Species composition in the Ozarks has changed at the landscape level over recent decades. Historically, the Highlands was dominated by oaks and pine. However, species' dominance changed over time. Oaks and pine dominance has been reduced compared to their historical level of dominance. Less commercially valuable species such as hickory, other eastern hardwoods, and maple have increased as a greater proportion of all trees in the region. Comparisons were made using historical PLS data and FIA data. However, PLS data may have a species preference bias. Nevertheless, PLS is the best available data of historical species composition in the Ozarks.

The red oaks group is at serious risk in the Ozarks not only because they have the highest rate of mortality but also because they have the lowest recruitment. On the flip side, some economically less desirable species such as hickory, maple, and other eastern hardwood have the lowest rate of mortality and highest rate of recruitment. This phenomenon shows up in all recent FIA measurement cycles. Thus, we recommend that managers should focus on minimizing oaks morality and maximizing its recruitment because the recruitment defines the future stand composition. The low proportion of red oaks ingrowth in recent FIA data suggest that red oaks will be less dominant in the future unless oaks regeneration can be increased through forest management. Hanberry et al. (2014) recommended that historical low-density forest ecosystems such as woodlands and savannas (Fig. 4.3) are potentially good sites for restoring moderate drought-tolerant species. Our results also suggest that lower density ecosystems such as oak closed woodlands had significantly lower red oaks mortality rates compared to higher stocked ecosystems such as oak forests and oak/pine forests (Fig. 4.12). Thus, we suggest adopting silvicultural treatments, which can create more open space and make a favorable condition for red oaks (both in terms of maximizing ingrowth and minimizing mortality).

Juvenile oaks often grow slower than their competitors; they are primarily intolerant or intermediate to the shade. Thus, Smith (1993) recommended silvicultural intervention to obtain consistent & successful oaks' natural regeneration. This implies that if forests are left undisturbed, oak forests are likely to shift toward more shade tolerance species. Our results also show the same pattern, the Ozark is experiencing decreasing oak proportion over time, and other species are increasing. Thus, we recommend adequate disturbance on the stands, which will provide more sunlight to the floor and create favorable conditions for oaks regeneration. As such, oaks can compete with more shade tolerance species. Particularly, Scarlet oak can grow faster than its competitor during the juvenile stage, but since they are shade-intolerant, openings are needed on the stand to guarantee its survival; otherwise, more shade tolerance species will replace it.

Severe droughts have led to widespread tree mortality across many forests globally and regionally (Choat et al. 2018). Soil water availability decreased with consecutive seasons of drought. Severe droughts and elongated periods of drought reduce soil water availability, which produces stress on the trees and ultimately leads to tree mortality. Indeed, drought caused a significant decline in tree growth (Camarero et al. 2018). Previous literature suggested that red oak groups are the most susceptible to oak decline, and the decline was reportedly more severe on droughty, nutrient-poor sites (Kabrick et al. 2008). Our analysis also suggests that red oaks are sensitive to extreme drought events but vary by spatial domain. Since FIA data do not provide comprehensive information about tree death, we used generic live or dead from tree status, which may not be sufficient to explain oak mortality due to the drought. Thus, we also examined the annual growth rate of trees with and without crown dieback. The crown dieback is one of the early indicators of stress on the tree. Extreme drought causes dieback in trees; thus, comparing the growth rate between dieback and non- dieback trees gives us a better understanding of drought's impact on trees. Red oaks relative growth rates were sensitive to dieback, as we found higher relative annual diameter growth on no crown dieback red oaks compared to those with dieback crown trees of red oaks. The difference between trees with and without crown dieback was large in the red oak groups.

This study examined how the average PDSI from previous growing seasons impacted recent mortality. Instead of using the lag of individual growing seasons or yearly PDSI, we used average PDSI of up to 11 past growing seasons PDSI. In doing so, we highlight the impact of continuous drought on tree mortality. From Fan et al. (2012), we know that red oak mortality was significantly correlated to the historic drought. We assume that the average PDSI values reflect the overall drought trends in the region, and it should be a meaningful variable in explaining red oak mortality. We found a significant negative correlation between red oak mortality with the average PDSI up to the past four seasons in the western climatic division (Fig. 4.10 & Table 4.4). Fan et al. (2012) used cross-correlation analyses to examine the correlation between red oak mortality and drought index. They used single growing season PDSI and one-year cumulative PDSI and found that cumulative PDSIs were correlated with red oak mortality for up to 10 years. However, our results suggest that drought impacts on oaks are also spatially varied in Ozarks. The western climatic division was more prone to drought-induced mortality than the eastern climatic division. Our analysis differs from Fan et al. (2012) because we used cumulative PDSI of up to ten previous growing seasons.

We also compared red oak mortality between treated and not treated and disturbed and not disturbed stands. Treated and disturbed stands had higher mortality compared to not disturbed or not treated (tables 4.7 & 4.8). This may be due to the short-term impact of the disturbance or treatment in the stands. Trees might die due to the machine or other instruments used during treatment. However, disturbance and treatment lead to a healthier forest in the long term. Thus, we found private forests had less red oak mortality than public forest lands. We think private forests are more resilient than public forests because more active forest management, including silvicultural treatments, is expected in the majority of private forests compared to public forests (O'Laughlin and Cook 2003). Furthermore, confounded with physiological maturity as Black and Scarlet oaks are substantially shorter-lived trees than most/all of the other oaks. Public land is likely to have a longer rotation than private. These are likely making private forests relatively healthier than public forests.

4.5 Conclusions

In this chapter, we explored species composition change in the Ozarks by comparing FIA data and pre-European settlement tree records from the original PLS. We have further evaluated the effect of biotic and abiotic stresses on oak mortality. Results show that the tree species composition in the Ozarks has changed; oak and pine's dominance has reduced, and less commercially valuable species such as hickory, maple, and other eastern hardwood proportions have increased. Results also show a higher mortality rate and lower ingrowth rate in red oaks group, which puts red oaks group at serious risk. The red oaks mortality rate in the private forests and the historically open or lower density woodland type ecosystem were lower. Lower density stands, actively treated, not only provide an opportunity for ingrowth but are also likely to reduce red oak's mortality. Further, results show the red oak groups were more sensitive to severe drought events. The impact of severe drought depends on the spatial location. Red oaks on the western side of Ozark were more prone to drought than on the eastern.

White oak recruitment nearly offset the mortality proportion, while red oak mortality was significantly higher than the recruitment. The future of red oak can be secure if we successfully promote red oak recruitment in the Ozarks on suitable, less drought-prone sites. Further, Management should promote more heterogenous stand structures and species composition. Uniform distribution of basal area among species that are suitable for local site conditions may provide increased resilience to drought and related stressors (Pile et al. 2019). Some management options for reducing the impact of oak decline include managing the physiological age of susceptible trees, favoring decline-resistant species such as white oak on drought-prone sites, and prescribed fire. We recommend promoting red oaks in the woodland-type ecosystems as we see significantly lower red oaks mortality in that ecosystem than in higher-density ecosystems.

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5 Resistance, recovery and resilience of different tree species related to severe drought in the Ozarks

(Under revision in Forest Science)

5.1 Introduction

A tree's ability to adjust to a changing environment determines its growth and survival (Young and Giese 1990). Tree growth is species specific, but it can be impacted directly by rainfall (Stoddard and Stoddard 1987), and growing season drought can be particularly impactful because it can significantly reduce tree growth (Au and Maxwell 2022). Severe droughts impact tree growth and stand productivity (Rötzer et al. 2017) by slowing growth and initiating tree death (DeSoto et al. 2020). Drought is a worldwide phenomenon, and its effect on tree dieback and mortality has been documented worldwide (Lloret et al. 2011); for instance, several countries have experienced an extensive decline in oak species (Gentilesca et al. 2017).

In the Ozarks of Arkansas and Missouri in the United States, drought has been identified as the inciting factors of oak decline over the past several decades (Kabrick et al. 2008, Spetich et al. 2016). Since 1998, three widespread and severe drought events have been recorded in the Ozarks: 1998 -2001, 2005-2007, and 2012-2013. Red oak mortality in the region has been high following drought (Fan et al. 2012). Jenkins & Pallardy (1995) found that severe drought influenced red oak growth. They also noticed that the historical growth of the red oak species was highly correlated with the historical Palmer Drought Severity Index (PDSI) in the Ozarks. Red oaks are the most susceptible oak species to oak decline, and a higher proportion of decline symptoms, such as crown dieback, occurs in red oaks compared to other coexisting species or species groups (Kabrick et al. 2008).

The Ozarks has already been exposed to multiple severe drought events, with more droughts expected in the future (Wehner et al. 2011, Cook et al. 2020). Past studies have established that white oaks typically have a much higher survival rate than red oaks during drought-induced oak decline events (Fan et al. 2012). A better, quantifiable understanding of the response of red oaks to this environmental stress is needed.

A tree's response to drought can be quantified by employing three indices: resistance, recovery, and resilience (Lloret et al. 2011). We will refer resistance, recovery, and resilience as RRR for the remainder of this chapter. Resistance can be understood as a trees' ability to withstand stresses induced by drought, recovery is the ability to overcome those stresses, and resilience is the ability a tree to sustain itself in a new environment. Radial growth, such as basal area increment (BAI), is often used to quantify these indices (Schwarz et al. 2020). As droughts are responsible for slow growth, severe drought stress on a tree can be observed in its radial expansion rate. Thus, we assume drought-induced stress on Ozark's trees or tree species' responses to drought can be explained by examining basal area growth.

We did not note any published research addressing the RRR of major species groups in relation to drought in the Ozarks. Such studies have been done elsewhere, such as in other parts of the central hardwood region (Au and Maxwell 2022). They compared hickories and white oaks' response to drought and found hickories performed better in resilience than white oaks. In the Appalachian hardwood forest, Keyser and Brown (2016) found white oaks outperformed red oaks in terms of resistance, recovery, and resilience. Slow-growing species are less sensitive to drought because they have a conservative resource-use strategy (Ouédraogo et al. 2013). In contrast, fast-growing species are less resistant and resilient even though they might have better recovery rates (Martínez-Vilalta et al. 2012, Serra-Maluquer et al. 2018).

Oak decline often involves the interaction of drought with predisposing factors such as site quality and tree age (Hoffard et al. 1995), which influence future forest health, productivity, and resilience of forests (Dinon et al. 2013). Information regarding the effects of drought on species RRR is vital to forest managers because the severity and frequency of drought are expected to increase across the globe (Cook et al. 2014). Understanding factors influencing a tree's response to droughts, such as RRR, are necessary to understand future stand dynamics and can be helpful for adaptive forest management (Serra-Maluquer et al. 2018, Zhang et al. 2022). These are crucial for developing long-term tree survival strategies in the Ozarks. Thus, this study focused on quantifying major tree species' RRR and identifying the influencing factors.

5.2 Materials and methods

5.2.1 Study areas

This study focused on the Ozarks in Southern Missouri and northern Arkansas, USA (Fig. 5.1). The highlands include a plateau of steep and low rolling hills with forest that is mainly dominated by oak-hickory and oak-pine forest types (McNab et al. 2007). The Ozarks is located in the central interior broadleaf forest province, containing 17 subsections (Cleland et al. 2007). Those 17 subsections were coded as ECOSUBCD (ecological subsection code) that were used as an explanatory variable in the statistical models.

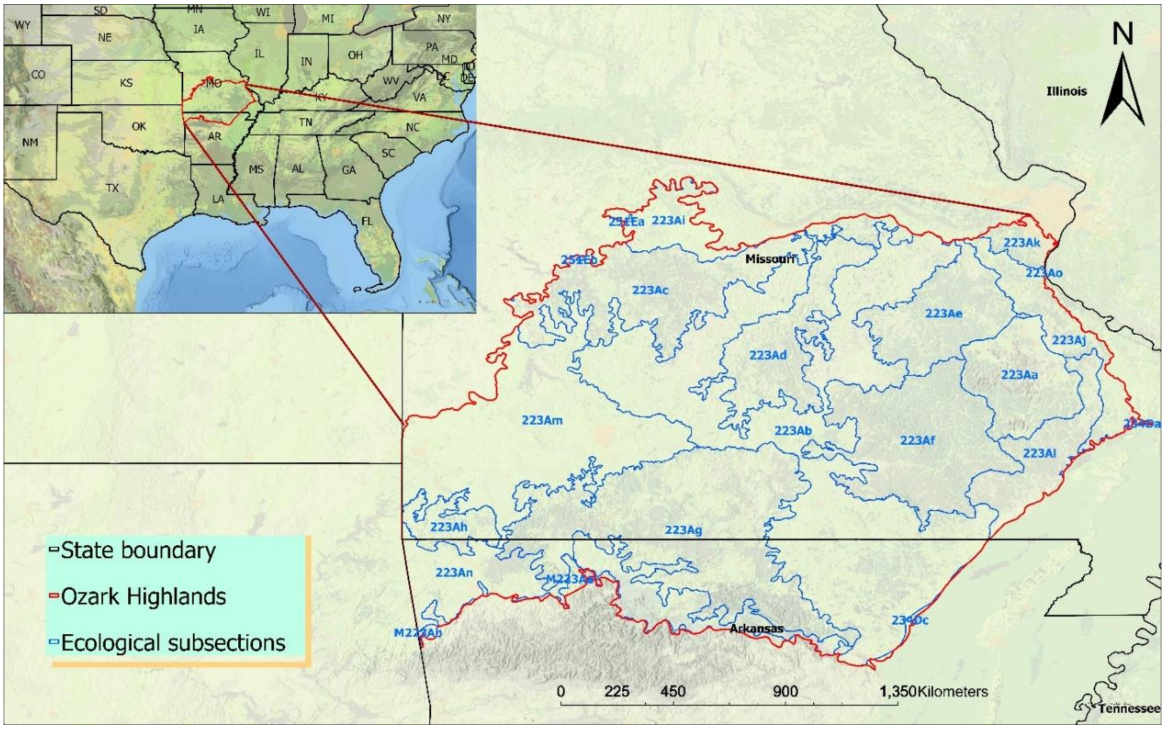

Figure 5.1 Study area covers Ozarks in Missouri and Arkansas.

5.2.2 FIA data

We used FIA data within the study area (Fig. 5.1), wherein each tree was remeasured four times during the study period (1999-2020). We limited data selection to re-measured plots. Annual measurement in Missouri began in 1999 and in Arkansas in 2000. We removed defective trees, such as those that are dead and have broken tops and culls. Thus, we selected live trees ≥ 5” dbh with no cull, no broken tops, and minimal defect (< 2%).

We selected 12,331 measured individual trees and classified them into six major species groups: eastern redcedar (*Juniperus virginiana* L.), hickory species (pignut hickory (*Carya glabra* Mill.), black hickory (*Carya texana* Buckley) and mockernut hickory (*Carya tomentosa* Nutt.)), red oaks (black oak (*Quercus velutina* Lamarck), scarlet oak (*Quercus coccinea* Muenchausen), and northern red oak (*Quercus rubra* L.)), shortleaf pine (*Pinus echinata* Mill.), white oaks (white oak (*Quercus alba* L.) and post oak (*Quercus stellata* Wangenheim)), and others (Table 5.1). The white oaks group was the most abundant, representing nearly 38% of the trees, followed by the red oaks group at 15.70%, shortleaf pine at 15.05%, and the other species group at 12.72%, hickories at 11.56%, and eastern redcedar at 6.97% respectively. Table 5.2 lists tree count by species group and average diameter by year.

Table 5.1 Tree counts by species groups* per measurement time.

Species/Species group*	Tree counts	Percent	Average DBH (in Inches)**			
			T1	T2	T3	T4
Eastern redcedar	860	6.97	6.84	7.30	7.75	8.19
Hickory	1,426	11.56	7.73	8.11	8.49	8.96
Others	1,568	12.72	7.98	8.62	9.20	9.82
Red oaks	1,936	15.70	9.14	9.93	10.73	11.60
Shortleaf pine	1,856	15.05	9.39	10.00	10.52	11.07
White oaks	4,685	37.99	8.75	9.23	9.75	10.33
total	12,331	100	8.56	9.11	9.67	10.27

* Major species were divided into six species groups. Here, red oak includes scarlet oak, northern red oak and black oak, white oak includes post oak and white oak, and hickory includes pignut hickory, black hickory and mockernut hickory. ** Average DBH of remeasured trees: first measurement (T1) trees measured between 1999 and 2005, second measurement (T2) measured between 2004 and 2010, third measurement (T3) measured between 2009 and 2015 and fourth measurement (T4) measured between 2014 and 2020.

Table 5.2 Average diameter of the Ozark trees by species and inventory year

	Average DBH in inch (tree counts)					
Year	Eastern redcedar	Hickory	Others	Red oaks	Shortleaf pine	White oaks
1999	5.98 (36)	7.62 (204)	8.5 (137)	9.14 (314)	9.7 (236)	8.69 (695)
2000	6.84 (135)	7.65 (343)	7.92 (449)	9.32 (475)	9.21 (538)	8.65 (1,153)
2001	6.58 (91)	8.02 (233)	7.9 (154)	9.01 (351)	9.44 (361)	9.04 (811)
2002	6.72 (210)	7.6 (335)	7.67 (324)	9.44 (476)	9.95 (308)	8.79 (1,202)
2003	6.99 (141)	7.88 (132)	8.13 (163)	8.9 (158)	9.2 (217)	8.67 (423)
2004	6.91 (133)	8.05 (315)	8.69 (308)	9.9 (372)	10.16 (314)	9.04 (891)
2005	7.02 (201)	7.89 (285)	8.15 (390)	9.66 (482)	9.37 (564)	8.99 (1,151)
2006	7.11 (258)	8.16 (365)	8.54 (336)	9.87 (437)	10.14 (392)	9.39 (1,091)
2007	7.17 (183)	8.02 (298)	8.13 (280)	9.92 (463)	10.17 (379)	9.2 (1,111)
2008	7.4 (98)	8.39 (162)	8.72 (212)	9.66 (150)	10.05 (196)	8.98 (447)
2009	7.54 (148)	8.36 (312)	9.46 (338)	10.8 (404)	10.79 (312)	9.73 (883)
2010	7.62 (173)	8.21 (289)	8.66 (402)	10.44 (482)	9.91 (577)	9.5 (1,153)
2011	7.58 (258)	8.57 (365)	9.08 (336)	10.65 (437)	10.66 (392)	9.94 (1,091)
2012	7.61 (183)	8.4 (298)	8.81 (280)	10.75 (463)	10.73 (379)	9.74 (1,111)
2013	7.89 (98)	8.78 (162)	9.34 (212)	10.54 (150)	10.53 (196)	9.53 (447)
2014	8.15 (110)	8.76 (262)	9.81 (308)	11.59 (333)	11.4 (258)	10.22 (680)
2015	8.01 (188)	8.64 (208)	9.61 (277)	10.96 (343)	10.22 (405)	9.81 (804)
2016	8.16 (190)	8.8 (301)	9.65 (353)	11.88 (343)	11.06 (291)	10.41 (939)
2017	8.01 (222)	9.1 (251)	9.72 (234)	11.34 (326)	11.13 (377)	10.6 (711)
2018	8.16 (134)	9.07 (258)	9.79 (294)	11.41 (319)	11.11 (266)	10.16 (858)
2019	8.57 (99)	9.06 (258)	9.94 (315)	12.05 (362)	11.58 (348)	10.61 (883)
2020	8.6 (151)	8.64 (68)	10.07 (170)	10.55 (104)	10.09 (118)	10.42 (205)

5.2.3 Drought event in 2012

We explored historical drought data for 72 Ozarks counties from Arkansas and Missouri. Based on data (available at https://www.drought.gov/), The Ozarks began undergoing severe drought from May 2012 through February 2013. We downloaded historical PDSI data from the National Oceanic and Atmospheric Administration (NOAA)/National Centers for Environmental Information (NCEI) (https://www.ncei.noaa.gov/pub/data/cirs/climdiv/climdiv-pdsidv-v1.0.0-20220108). Based on the historical PDSI data, we derived the growing season (April, May, June, and July) average PDSI for the Ozarks area (Fig. 5.2). Historical PDSI data revealed a severe drought across the Ozarks during the 2012 growing season. There were no severe drought events during the five years before 2012 and no severe events after 2012 across the Highlands area.

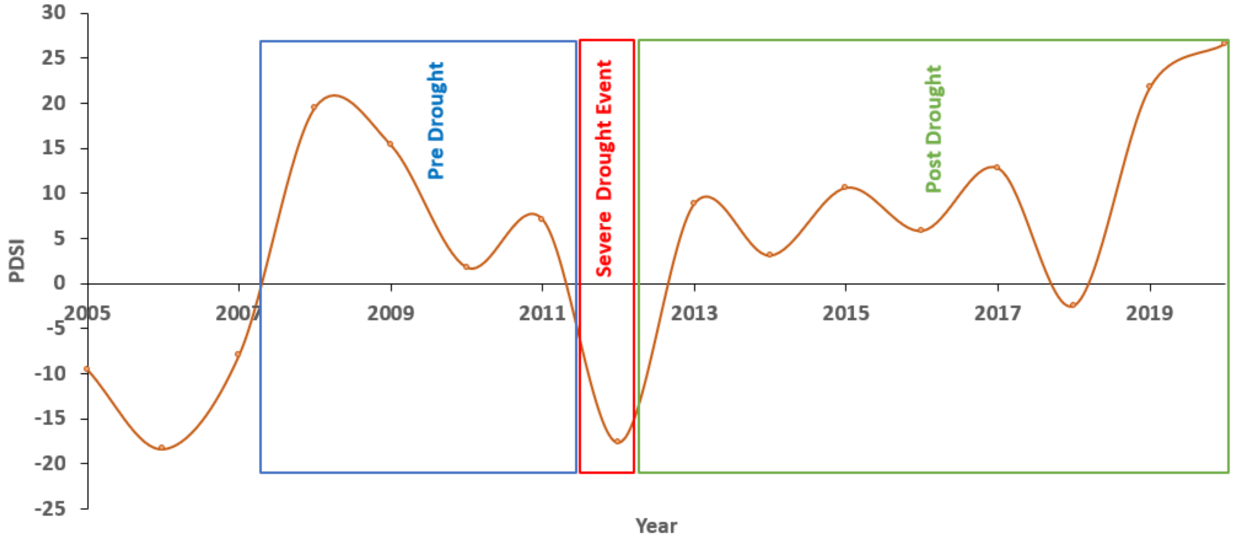

Figure 5.2 Historical average PDSI values on growing season across Ozarks' of Missouri and Arkansas. (source: www.ncei.noaa.gov/pub)

5.2.4 Estimating annual basal area increment (BAI)

The growing season in the Ozarks occurs between May and October, as noted by Fan et al. (2012). Research indicates potential prolonged growing seasons due to global warming over the past decade (Grossiord et al. 2022). Frisner (1942) estimated the potential growth of tree species at the end of each month. He estimated that most trees grow in April, May, June and July at the cumulative rate of 3, 22, 75, and 100 percent, respectively. We considered Frisner (1942) monthly growth potential while calculating growth by year. We used measured year and month from FIA data and calculated the actual growth interval based on Frisner (1942). Annual BAI per tree was calculated based on equation 1, which considers compound interest while calculating the growth. Equation 1 was adapted from Husch et al. (2002).

$$BAI_{T(x)} = \left(\frac{BA_{T(x)}}{BA_{T(x-1)}} \right)^{(1/n)} \text{.......................................(1)}$$

Here,

$BAI_{T(x)}$ = annual basal area increment
$BA_{T(x)}$ = basal area measured in time $T_{(x)}$ with x equal to 1, 2, 3, or 4
$BA_{T(x-1)}$ = basal area measured in previous time $T_{(x-1)}$.
n = growth interval in years (adjusted with monthly growth potential)

Growth interval (n) was calculated as follows;

n = remaining growth in year $T_{(x-1)}$ + complete growth years between $T_{(x-1)}$ and $T_{(x)}$ + growth in a year $T_{(x)}$. For example; if a tree was measured in 2014 May (i.e., $T_{(x-1)}$) and 2020 June (i.e., $T_{(x)}$), in this case remaining growth in year $T_{(x-1)}$ = (1-(22/100)) = 0.88, complete growth years between $T_{(x-1)}$ and $T_{(x)}$ = 5 (i.e., 2015, 2016, 2017, 2018, and 2019), and growth in year $T_{(x)}$ = 75/100 = 0.75. Thus n = 0.88+5+0.75 = 6.63 years.

5.2.5 Quantifying resistance, recovery, and resilience

Since the FIA program measures trees in approximately five-year intervals in the Ozarks, we assumed the severe drought in 2012 would impact trees measured two or three years later due to the documented lag effect (Fan et al. 2012). To capture the impact of drought on growth using FIA data, we calculated BAI during the drought period (Dr) from trees measured in 2014 and 2015. Pre- and post-BAI is needed to estimate RRR. Thus, we selected individual trees measured in 2014-2015 that were also measured before drought (i.e., before 2012) and after the drought period. We found 1,593 individual trees- which met this requirement. Those trees were measured three times; first in 2009-2010, second in 2014-2015, and third in 2019-2020. As such, the BAI pre-drought (PreDr), during drought (Dr), and post-drought (PostDr) were based on 2009-2010, 2014-2015, and 2019-2020 measurements, respectively.

Resistance is the capacity of trees to withstand the impact of drought. It can be quantified with a ratio of growth during the drought (Dr) and growth before the drought event (PreDr), i.e., Resistance = Dr/PreDr (Lloret et al. 2011). Recovery is the ability of trees to retain a growth rate comparable to that before the drought. It can be quantified using the ratio of post-drought growth (i.e., PostDr) and during drought growth (Dr.), i.e., recovery = PostDr/Dr. Furthermore, resilience is the capacity of trees to reach a growth rate similar to the pre-disturbance level. It compares pre- and post-drought growth, which can be calculated with post-drought growth (PostDr) and pre-drought growth (PreDr), i.e., resilience = PostDr/PreDr, a product of resistance and recovery.

5.2.6 Statistical analysis

Of the 1,593 trees that were measured both prior to and after the drought 77 trees had negative growth. Negative DBH growth is often the result of manual error; however, negative growth during drought is most likely due to water deficit during the growing seasons (Pastur et al. 2007). In addition, trees can also lose bark between measurements resulting in seemingly negative growth in the data. There were also other outliers, with some trees' annual basal area increment of over 25%. Thus, we removed the outliers. Responses that were less than one interquartile range (IQR) below the first quartile and more than one IQR above the third quartile were removed. As such, in the statistical analysis, we used more than 1,200 tree records. Analysis of variance (ANOVA) was performed to compare RRR of major species groups. Tukey's HSD postdoc test was used to identify the significant difference among major species groups.

Further, a mixed-effect model was run to evaluate the fixed and random effects on resistance, resilience, and recovery. Table 3 shows all variables used in the model and its descriptions. The FIA subplot was used as a random effect. The best model was identified based on the lowest AIC. All statistical analysis was performed in R (R Core Team 2014), and the linear mixed effect model was run using the “lme4” package (Bates et al. 2015) in R with the following equation 2.

$$\mathrm{Y} = \mathrm{X}\beta + \mathrm{Zu} + \varepsilon \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots..(2)$$

Here, y is the response variable (resistance, recovery, or resilience) with N (total number of trees) x 1 column vector, $\boldsymbol{X}$ is a N x p (explanatory variables-listed in Table 3) matrix, $\boldsymbol{Z}$ is the N x q^j design matrix for the q random effects and j (FIA subplots) groups, u is a q^j x 1 vector of random effects and ε is residual with N x 1 vector.

Table 5.3: Variables used in linear mixed effect model

Variable	Variable definition	Data types	Explanation
species groups	major tree species groups in the Ozarks	factors	six species groups; red oaks (RO), white oaks (WO), hickory (HI), shortleaf pine (SP), eastern red cedar (ERC), and others (OT)
DBH	diameter at breast height in a tree (inch)	numeric	
BA	basal area of a tree (ft2)	numeric	
HT	total height of the tree (ft)	numeric	
BAL	basal area of a larger tree in a plot (ft2)	numeric	
CR	crown ratio of a tree (%)	numeric	
SLOPE	slope of the plot (%)	numeric	
ASPECT	aspect	numeric	
PHYSCLCD	physiographic class code	factors	five levels: 21(flatwood),22(rolling uplands),23(moist slopes and coves),24(narrow floodplains/bottomlands), and 32 (small drains)
SITECLCD	site class code		three site classes: 4 (85-119 cubic feet/acre/year; 5(50-84 cubic feet/acre/year); 6(20-49 cubic feet/acre/year)
PreDr_AvgPDSI	pre-drought average PDSI	numeric	Five years average growing seasons PDSI prior to the drought period
Dr_AvgPDSI	during drought avg PDSI	numeric	Five years average growing seasons PDSI during the drought period
PostDr_AvgPDSI	Post-drought average PDSI	numeric	Five years average growing seasons PDSI post-drought period
ECOSUBCD	ecological subsection code	factors	seven subsections: 223Ab, 223Ag, 223Ah, 223An, M223Aa, M223Ab

5.3 Results

5.3.1 Basal area increment (BAI)

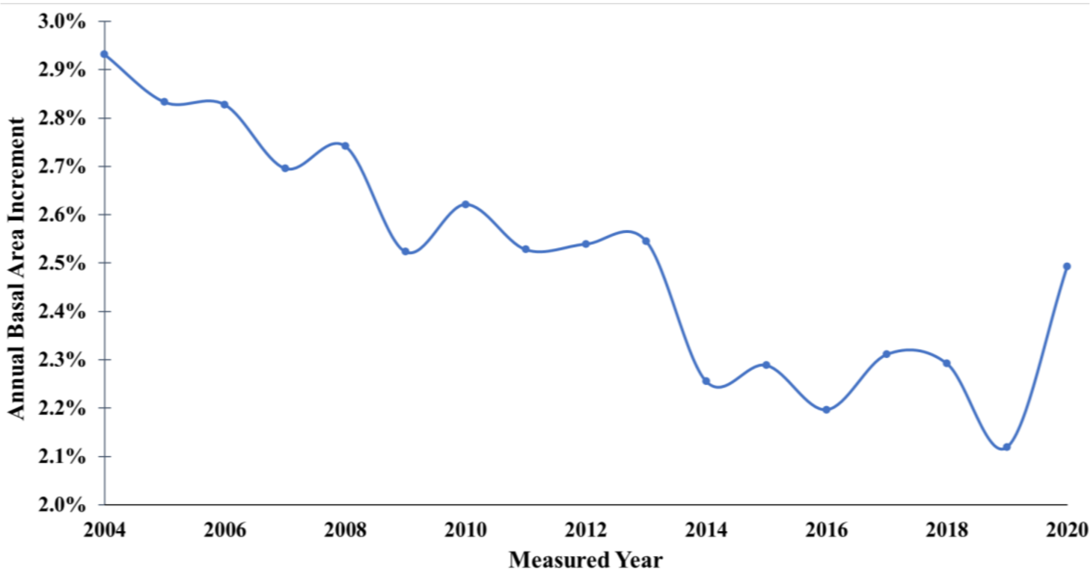

Figure 5.3 Annual basal area increment per tree based on all species.

Basal area increments` of all species (Fig. 5.3) show an overall decreasing trend of annual growth rate over time; however, an abrupt drop can be seen after 2014 (i.e., after the severe drought events 2012-2013). For all species combined, the highest annual basal area growth rate was in 2004 (2.95%), and the lowest growth rate was in 2019 (2.09%). There were also relatively low growth rates in 2014 and 2016; trees measured in 2014 had a 2.19% annual growth rate, and 2016 had a 2.18% annual growth rate. Additionally, the average annual BAI rate (%) was calculated by species group based on equation 1 (Fig. 5.4). Red oaks had the highest annual growth rate, and hickories had the lowest.

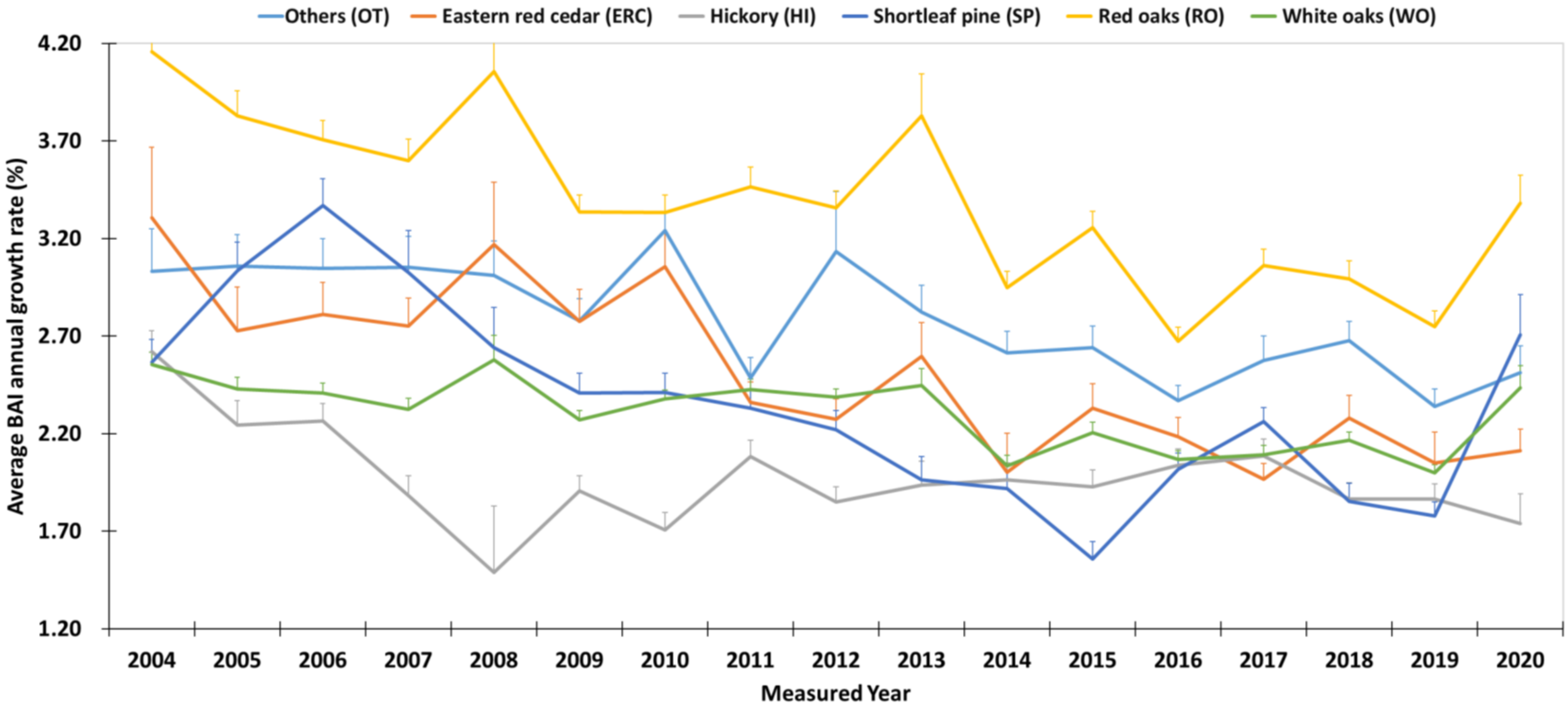

Figure 5.4 Average annual growth rate for each forest types of groups. Note: Error bars represents SE of mean. Annual basal area growth rate (%) in x-axis and year of measurement in y-axis.

5.3.2 Resistance, recovery, and resilience

We examined potential species groups' RRR differences. Hickories (HI) had the highest rate of resistance, followed by White Oaks (WO), others (OT), Red Oaks (RO), Eastern Red Cedar (ERS), and Shortleaf Pine (SP) (Fig. 5.5). The highest recovery was found in SP, followed by WO. Even though shortleaf showed the poorest resistance, their recovery rate was the highest compared to other species groups. Furthermore, we see the lowest level of resilience in RO, followed by ERC, OT, and SP. HI had the highest level of resilience, followed closely by WO.

Further, we performed one-way ANOVA, which shows the significant difference in RRR among the species groups. Pairwise Tukey's HSD comparison can be found in Figure 5. HI had a significantly higher resistance compared to all species groups except WO. The WO had significantly higher resistance than SP and ERC. We did not find any significant differences between RO, SP, and ERC (Fig. 5.5). SP had significantly higher recovery compared to all other species groups; however, we did not find any significant difference among others. RO had the lowest rate of resilience compared to others; however, only two species groups, HI and WO, had significantly higher resilience than RO (Fig. 5.5). WO and HI RRR responses were similar with no significant difference between the two species groups.

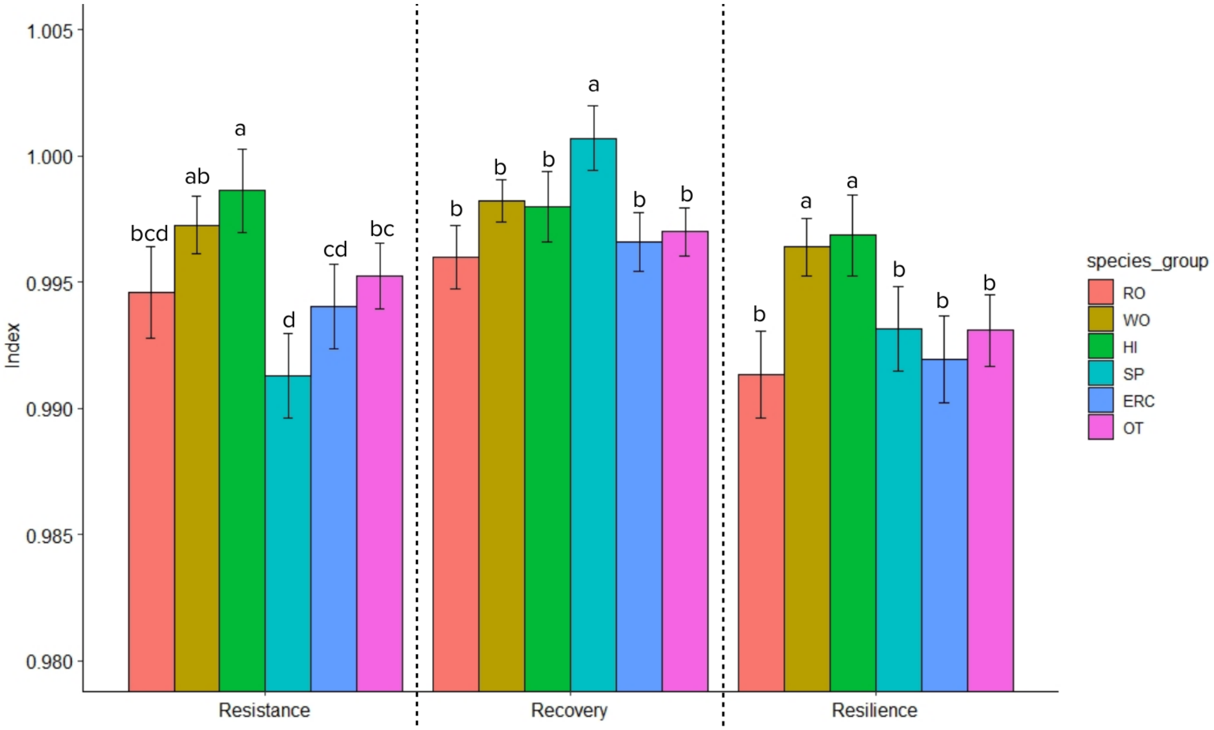

Figure 5.5 Resistance, recovery, and resilience by major species groups. Note: Error bars represent the 95% confidence interval of the mean. Species codes; ERC (Eastern red cedar), Hickory (HI), Red Oaks (RO), Shortleaf Pine (SP), White Oak s(WO), and Others (OT).

5.3.3 Factors affecting resistance, recovery, and resilience

The outcomes of the best-fitted linear mixed models are listed in table 5.4. Fixed effects of species groups significantly affect all RRR. The larger variance of random effects (subplots) on resistance and resilience indicates the greater impact of the random effect on these models. The random effect has a lower impact on the recovery model (Table 5.4). Furthermore, the resistance of species was positively impacted by the BAL, the total height of the tree (HT), slope, and drought, and negatively impacted by the crown ratio (CR).

The recovery of species was significantly influenced by dbh and ecological subsection (ECOSUBCD). Results show trees with larger dbh had a higher recovery rate. Trees located in the ECOSUBCD 223An (Springfield Plateau subsection)-northwest Arkansas and M223Ab (Boston Hills subsection) had a higher recovery rate than those in the 223Ab (central plateau subsection)-northeast Arkansas. During drought average PDSI (Dr_AvgPDSI) had a negative impact on the recovery but was not statistically significant.

BAL, HT, CR, SLOPE, and site class significantly impacted resilience. We found that the productive site classes had low resilience. Resilience increased with increasing BAL, HT, and SLOPE and decreased with increasing CR. Also, resilience significantly increased in less productive sites (site class codes 5 and 6) compared to the productive site (site class code 4).

Table 5.4 : Tree and stand-level characteristics for predicting resistance, recovery, and resilience obtained from the best-fitted linear mixed-effect model

	Resistance				Recovery				Resilience			
	Predictors	Estimate	SE	p	Predictors	Estimate	SE	p	Predictors	Estimate	SE	p
Fixed effects	intercept	98.780	0.257	<0.001***	intercept	98.860	0.238	<0.001***	intercept	97.730	0.363	<0.001***
	RO	ref			RO	ref			RO	ref		
	WO+	0.429	0.113	<0.001***	WO+	0.200	0.083	0.016*	WO+	0.618	0.115	<0.001***
	HI+	0.558	0.134	<0.001***	HI+	0.244	0.100	0.015*	HI+	0.716	0.136	<0.001***
	SP+	-0.179	0.136	0.188	SP+	0.448	0.097	<0.001***	SP+	0.351	0.144	0.015*
	ERC+	0.162	0.115	0.161	ERC+	0.135	0.085	0.113	ERC+	0.377	0.119	0.001**
	OT+	0.303	0.151	0.045*	OT+	0.127	0.097	0.193	OT+	0.450	0.155	0.004**
	BAL	0.004	0.001	<0.001***	BA	-0.605	0.345	0.079.	BAL	0.003	0.001	<0.001***
	HT	0.011	0.003	<0.001***	DBH	0.098	0.042	0.019*	HT	0.013	0.003	<0.001***
	CR	-0.009	0.004	0.013*	ASPECT	0.000	0.000	0.093.	CR	-0.009	0.004	0.012*
	SLOPE	0.009	0.003	0.006**	223Ab	Ref			SLOPE	0.011	0.003	0.001**
	Dr_AvgPDSI	0.213	0.102	0.039*	223Ag++	0.087	0.063	0.169	SITECLCD4	ref		
					223Ah++	0.186	0.224	0.407	SITECLCD 5+++	0.566	0.216	0.009**
					223An++	0.186	0.075	0.014*	SITECLCD 6+++	0.640	0.219	0.004**
					234Dc++	-0.174	0.315	0.582				
					M223Aa++	-0.112	0.187	0.552				
					M223Ab++	0.824	0.267	0.002**				
					PreDr_AvgPDSI	0.140	0.096	0.147				
					Dr_AvgPDSI	-0.116	0.073	0.116				
Random effects	variance of residual	0.9607			variance of residual	0.5851			variance of residual	0.9441		
	variance by random effect (plot)	0.209			variance by random effect (plot)	0.0578			variance by random effect (plot)	0.2719		
	number of plots	429			number of plots	430			number of plots	431		
	observation	1275			observation	1258			observation	1287		

Note: '+' is species group ref is RO; '++' ECOSUBCD ref is 223Ab; '+++' SITECLCD ref is 4. Significant levels; '.' for 10%,'*' for 5%, '**' for 1%, '***' for 0.1%

5.4 Discussion

RRR of individual species and species groups are logical indicators to explore the impact of severe drought or other disturbances on tree growth and development. In this study, we estimated the RRR of major species (groups) found in Ozarks using publicly available and continuously measured FIA data. The impact of severe drought on tree growth has been studied in the past (Rötzer et al. 2017, Zhang et al. 2022). Most of these studies have used annual tree growth analysis and tree ring data or dendrometers to collect tree growth data. Ring data are a valuable asset that can provide lifelong biological trends of an individual tree (Helama 2015). Radial growth pre-, during, and post-drought is used in quantifying the response of trees to severe drought events.

In this study, we used the same concept to describe the impact of severe drought on Ozark Highland trees. We calculated the radial growth (i.e., BAI) on each tree pre-, during, and post-reference drought period. Instead of growth ring analysis, we used repeatedly measured trees from the FIA data across the region. Each tree was remeasured three times; pre-, during, and post-drought in approximately five years intervals. Since FIA does not measure the same tree annually or core the tree to obtain an annual growth rate, we selected trees that appropriately represent pre-, during, and post-drought BAI (see FIA data selection process in the methods section). Quantification of RRR was based on basal area growth of individual trees. Explanatory variables such as stand conditions and individual characteristics such as diameter and height represent the initial measurement conditions. We also calculated five-year average growing seasons PDSI and assigned them to each tree based on measured years.

We used one-way ANOVA and a mixed effect model to investigate the species groups' response to RRR. Both models showed significant differences among species groups within each of the RRR categories. ANOVA used continuous response variables and species groups as a categorical covariate, but the linear-mixed model considered both random and fixed effects. A linear mixed effect model is recommended if experiments have both random and fixed effects (Yang 2010). We used the FIA subplot as a random effect and other stand-level and tree characters as fixed effects. We observed larger random effects variances in the linear mixed models (Table 5.4); thus, we believe the linear mixed model better describes the species groups' RRR in the Ozarks. We found some differences in these two models; for instance, ANOVA showed RO's resistance was significantly lower than only HI, but mixed effect models showed it was significantly lower than HI, WO, and OT. Similarly, ANOVA showed recovery of RO was significantly lower than SP, but the mixed-effect model showed RO was significantly lower than SP, WO, and HI. Further, ANOVA showed the resilience of RO was significantly lower than WO and HI, but the mixed effect model showed the RO resilience was significantly lower than all groups (Table 5.4).

We found significantly higher RRR in the white oak group compared to the red oaks in the Ozark (Table 5.4). Our results are consistent with Keyser and Brown (2016), who found that white oaks in the Appalachian hardwood forests displayed better RRR than red oaks. Shortleaf pine had the best recovery rate among others, but the resistance is significantly lower than red oaks (Fig. 5.5), suggesting that it had a lower growth rate during the 2012 drought period and excellent growth in post-drought. Our results indicate that the hickories performed the best followed closely by white oaks in terms of RRR on average when compared to other species groups. In particular, hickories have a significantly higher RRR rate than red oaks (Table 5.4). Our results align with Au and Maxwell (2022); hickory species showed better resilience than white oaks in the central hardwood regions (Au and Maxwell 2022).

Among six species groups in the Ozarks, RO is the fastest growing species group, and HI is the slowest (Fig. 5.4). Furthermore, RO has a wide variation in the growth rate while HI and WO have shown consistence growth. Between 2009-2020, the average annual basal area growth on HI was 1.91% (min:1.85% and max: 2.09%), WO 2.24% (min: 2% and max:2.43%), and RO 3.2% (min: 2.67% and max: 3.83%) (Fig. 5.4). These temporal changes in the growth rate suggest that RO can grow extremely well in favorable conditions, but it will be the first victim of adverse drought conditions. Additionally, the fast-growing species are less resistant and resilient even though they might have a better recovery rate (Serra-Maluquer et al. 2018); in contrast, slow-growing trees are less sensitive to drought (Ouédraogo et al. 2013). Slow-growing species are adapted to use limited available resources, making them less likely to be impacted by drought. Further, our resilience model shows that species located on less productive sites had higher resilience than productive sites. Trees grow slower in the less productive sites. Lower growth on the less productive site made those trees more resilient. Less productive sites often have a higher percentage of slower growing HI and WO than RO; for instance, we observed nearly 70% of HI and WO (selected for linear mixed model) fall under the less productive site class (i.e., SITECLCD 6; with productivity 20-49 cubic feet/acre/year). This phenomenon can be seen among major species groups as well. For instance, red oaks have the highest growth rate, and hickories have the lowest growth rate (Fig. 5.4); if we compare these two species' responses to the drought, we can see RO's weakest performance to RRR than HI.

A higher rate of red oaks mortality has been observed in past studies: for instance, red oaks mortality rate was significantly higher than for white oaks, hickories, shortleaf pine, and other non-oak species (Fan et al. 2006, 2012). Our analysis showed RO's poor performance concerning RRR relative to the 2012 drought. BAI on RO continually fell mainly during and after the drought events (Fig. 5.4). In contrast, HI has the most robust performance in response to the severe drought. Thus, HI species are more likely to increase their dominance in the Ozarks in the future. The dominance of hickory species increased over time as there was greater ingrowth of hickories relative to mortality (Nepal 2022). This indicates that hickory has been flourishing in the Ozarks. Compared to hickories and white oaks, red oaks performance was poor in terms of resistance and resilience. This indicates that red oaks will likely continue to be impacted by future severe drought in the Ozarks.

5.5 Conclusions

Severe drought slows tree growth and increases the probability of tree mortality. A positive species' response to drought is critical for long-term survival. Understanding trees RRR response is essential to a better understanding of how a tree responds to severe drought. In the Ozarks, we quantified major species' responses to severe drought in terms of RRR. We used the 2012 severe drought as a reference. RRR were quantified based on radial tree growth (i.e., annual basal area increment) before, during, and after the severe drought. The results suggest that fast-growing red oaks are less resilient to severe drought and likely to be threatened by future severe drought events. The resilience of slow-growing species such as white oaks and hickories performed better during and after the 2012 severe drought in the Ozarks. Further, hickories followed closely by white oaks demonstrated better resistance compared to its peers. Thus, we suspect the dominance of hickories will increase in the future if severe droughts continue to impact the region. Similarly, the dominance of white oaks is likely to increase or remain the same as it was the second most resilient and resistant species group relative to the 2012 severe drought.

5.6 References

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6 Summary and Conclusions

Spatial and temporal aspects of forest health risks in the Eastern U.S. were examined. This dissertation research focused on two major forest health risks in the regions: NNIPS and oak decline/mortality. We developed a multiscale modeling framework for quantifying regional forest health risks. Influencing risk factors were examined, and mitigation approaches were recommended. We used secondary data sources such as the FIA and Census data, LandFire, and historical land use and re-quantified them into multiple spatial domains, including stand, landscape, and region. We set up four major research efforts to achieve the overall objective, which is developing a multiscale modeling framework of forest health risks: 1) a multiscale modeling framework was developed, and influencing factors for the invasion were identified, 2) a quantitative and analytical approach was developed to understand the invasive stages and invasiveness of the nonnative invasive tree species, 3) effect of biotic and abiotic stressors on oak mortality were evaluated, and mitigation methods were recommended and 4) resistance, recovery, and resilience of trees in response to the extreme climatic stress was examined.

6.1 A multiscale modeling framework and influencing factors for the invasion

Invasive species cause significant damage to the native forest ecosystems. The Southern state of Alabama is also experiencing negative consequences such as habitat degradation, ecological instability, and biodiversity loss due to NNIPS. More than 50% of Alabama's forested lands are invaded with at least one NNIPS. Seven major NNIPS were identified based on FIA data for the state of Alabama: Japanese honeysuckle (Lonicera japonica Thunb), Privet (Ligustrum L.), Japanese climbing fern (Lygodium japonicum (Thunb.)), Chinese lespedeza (Lespedeza cuneata (Dum. Cours.)), Silk tree (Albizia julibrissin Durazz.), Chinese tallow tree (Triadica sebifera (L.) Small), and Rose (Rosa L.). We used more than 5,000 permanent FIA plots in Alabama measured between 2001 to 2019 over three measurement cycles to test the suitable modeling unit for quantifying large-scale invasion patterns and associated factors. NNIPS heavily infest Alabama forestlands, and Alabama's forested lands infested with at least one invasive species have been increasing over time; 41.1%, 50.8%, and 54.8% during the past three FIA measurement cycles. Lonicera japonica (Thunb.) was the most abundant NNIPS in Alabama. In the last three measurement cycles, the percentages of infestation in Alabama forestlands by Lonicera japonica (Thunb.) were 26.36%, 32.05%, and 33.83%, respectively. FIA data were aggregated with multiple spatial units; five levels of hydrological units, three levels of ecological units, and a county level. Invasion indices -based on the presence/absence and average cover of invasive species were calculated for all spatial units. The best modeling unit was identified based on Moran's test. Counties level modeling unit provides the best Moran's I over all measurement periods. Influencing factors of invasion were modeled based on spatial lag models. Our models show a positive and statistically significant lag coefficient in all measurement cycles. Our models also show that the invasion index decrease with increases in public forest areas and the area of the modeling unit itself. In contrast, road length, the population density of neighbors, and households positively correlated with the invasion index.

6.2 A quantitative and analytical approach to understanding the invasive stages and invasiveness of the nonnative invasive tree species

Quantifying the invasion severity of NNIPS is vital for developing appropriate mitigation and control measures. We examined more than 23 thousand FIA plots from the Southern U.S. to develop an alternative method to classify and map the invasion severity of the Chinese tallow (Triadica sebifera). Remeasured FIA plot-level data were used to examine the spatiotemporal changes in the presence probability and cover percentage. Four invasion severity classes were identified using the product of presence probability and cover percentage. Chinese tallow invasion severity increased over time, with 90 and 123 counties being classified into the highest severity classes for the first and second measurements. Further, the invasibility of major forest-type groups by severity class was examined using the product of the county-level mean presence probability and mean cover percentage of Chinese tallow as a proxy of invasibility. Longleaf/slash pine (Pinus palustris/P. elliottii) forests were highly resilient to the Chinese tallow invasion. In contrast, elm/ash/cottonwood (Ulmus spp./Fraxinus spp./Populus deltoides) and oak/gum/cypress (Quercus spp./Nyssa spp./Taxodium spp.) forest-type groups were vulnerable to invasion.

6.3 Effect of biotic and abiotic stressors on oak mortality and mitigation methods

We examined 39,940 continuously monitored trees measured between 1999 and 2019 by FIA from the Missouri Ozark and evaluated the effect of biotic and abiotic stresses on oak mortality. We also explored tree species composition change by comparing FIA data and pre-European settlement tree records measured by the original Public Land Survey (PLS) from the U.S. General Land Office. We found that the species composition in recent decades has changed in the Missouri Ozark; oak and pine's dominance has been reduced, and less commercially valuable species such as hickory, maple, and other eastern hardwood proportions have increased. We also found a higher mortality rate and lower ingrowth rate in red oak groups, which puts red oak groups at serious risk. Further, our analysis showed that the red oak groups were more sensitive to extreme drought events. The impact of severe drought depends on the spatial location. Red oaks on the western side of Ozark were more prone to drought than on the eastern.

We found a lower red oaks mortality rate in the private forests and the historically open or lower-density woodland-type ecosystem. Lower density and actively treated stands provided a better opportunity for ingrowth and experienced lower red oak mortality. Thus, we recommend active forest management with frequent disturbance to create open space and remove older trees.

6.4 Resistance, recovery, and resilience of trees to the extreme climatic stress

Previous research has shown that severe drought can slow tree growth and that tree mortality (especially among red oaks) was associated with extreme drought. The Ozarks have been exposed to multiple severe drought events and are expected to experience more in the future. Species resilient to those extreme events are more likely to survive over the long term. This study examined the resistance, recovery, and resilience of major tree groups associated with extreme drought using pre-, during, and post-drought BAI. Even though individual BAI is an essential component of stand dynamics, we firmly believe that the altered tree growth can be used to determine important spatial or temporal patterns. We examined 12,331 greater than 5" DBH individual live tree records in Ozarks. These trees were measured by FIA between 1999 & 2020, with each tree measured four times and had no major defects (e.g., cull & broken tops). This analysis used the 2012 severe drought as a reference event. Pre-, during, and post-annual BAI was calculated based on the reference drought event, and species response to the drought was quantified in terms of resistance, recovery, and resilience. A linear mixed-effect model was used to examine the impact of tree's and stand's levels characteristics on the resistance, recovery, and resilience. The result showed that species groups significantly affected all resistance, recovery, and resilience. Notably, red oaks groups had lower resistance, recovery, and resilience while hickory had a higher. Results also showed that the BAL, total height, and slope positively impacted the resistance and resilience of individual trees. Similarly, Dbh and ecosection had a significant impact on recovery, and crown ration also had a negative effect on resistance and resilience. Our results also showed that the trees on the productive sites had low resilience.

6.5 Future direction

This research evaluated a multiscale modeling framework based on cumulative invasion severity, including all major invasive species measured. Individual invasive species characteristics are different. Thus, the invasiveness of individual species and the invasibility of forests to invasive species might differ. So, we recommend that future research adapt our approach to quantify and understand individual species so we can get a better understanding. Also, since FIA will collect more data in the future, our analytical approach can be improved with those additional data sets.

Regarding the oak response to severe drought, we used FIA data to calculate radial growth. FIA measures tree on average five years intervals. Annual radial growth based on FIA may not be perfect. Thus, we recommend future research using ring data and quantifying annual radial growth. Furthermore, species composition is changing in the Ozarks: the dominance of more resilience to severe drought is increasing. The dominance of more resilient species is outpacing the less resilient species. Further research should also focus on developing adaptive mechanisms by promoting more resilient species in such forest productivity can be intact.