

**EFFECTS OF MANAGEMENT FOLLOWING DIFFERENT TECHNIQUES TO
RESTORE NATIVE EARLY SUCCESSIONAL PLANT COMMUNITIES**

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DEDICATION

I dedicate this thesis first and foremost to my wife, Lexie Powell. The encouragement, patience, and support I have received from her throughout my time working on my M.S. degree, has been much more than I deserve. I also dedicate this thesis to my parents, Kerry and Jennifer Powell. They have supported my pursuit of a career in wildlife management monetarily, emotionally, and spiritually. I could never repay all I have been given, and I will never forget the encouragement and love I have been shown throughout my life.

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ABSTRACT

Early successional plant communities are being restored across the eastern USA in response to declines in native early successional plant communities and associated wildlife. Previous research has indicated that early successional communities can be restored via seedbank response and that planting native grasses and forbs is not necessary for many wildlife management objectives. I evaluated effects of burning and mowing following restoration of native plant communities via 1) planting native grasses and forbs and 2) seedbank response without planting across 11 replicated sites in Tennessee and Alabama, 2019-20. I compared vegetation composition and structure, openness at ground level, forage availability, and nutritional carrying capacity (NCC) in 2019 and 2020 following 4 treatments (Seedbank burned (SB), Seedbank mowed (SM), Planted burned (PB), Planted mowed (PM)), and tall fescue Control (CL)). The combination of planting and mowing increased grass coverage. Forb coverage declined in all treatments from 2018 to 2020 except SB, where it increased. Visual obstruction above 25 cm was greater in all treatments than CL, which enhanced nesting cover for wild turkey (*Meleagris gallopavo*) and bedding cover for white-tailed deer (*Odocoileus virginianus*). Openness at ground level, which is important for northern bobwhite (*Colinus virginianus*) and wild turkey broods, was 30% greater in SB than PM, which provided the least openness among treatments. Coverage of bobwhite food and deer forage plant species was greatest in SB. Selected available deer forage (kg/ha) was greater in all treatments compared to CL, and NCC (deer days/ha) was greatest in SB. Spring-, summer-, and fall-flowering forbs, which are a focus of conservation programs designed to enhance habitat for

pollinators, increased most in SB. My results provide insight into how disturbance management techniques may alter plant community composition and structure soon after restoration. I recommend managers consider using seedbank response without planting when restoring early successional plant communities previously dominated by nonnative grasses and to use prescribed fire instead of mowing to maintain these plant communities. Furthermore, my results highlight how planting native grasses and forbs is not necessary to restore native early successional plant communities on many sites in the eastern United States.

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INTRODUCTION

Native early successional plant communities are in decline across the eastern United States (Brennan 1991, Noss et al. 1995, Noss 2013). Historically, natural and anthropogenic disturbances, such as fire, wind events, and flooding, periodically set-back succession and maintained early successional plant communities (Pyne 1982, Williams 1989, Oliveri 1993, Vitousek et al. 1997, Burris and Haney 2005, Haney et al. 2008, Chandler et al. 2009). The suppression of fire, shifts in land-use practices, and urbanization have contributed to changes in the landscape across the southeastern United States (Brennan 1991, Kirkland and Hart 1999, Pruitt 2000, Harper 2007, Sauer et al. 2011, Mcchesney and Anderson 2015). Increasing intensity in agricultural practices and greater farm size have reduced the area represented by native early successional plant communities in the southern United States (Brennan 1991, Heard et al. 2000). Forest maturation and urbanization also have contributed to the decline in these communities (Williams 1989, Ramankutty and Foley 1999, Drummond and Loveland 2010). Approximately 5.7 million hectares of early successional plant communities were lost to land development in the United States from 1982–2015 (USDA 2018)

Restoration of early successional plant communities on both private and public lands has been prompted by population declines in many wildlife species commonly associated with early successional plant communities. The federal Farm Bill provided technical and financial assistance for wildlife management practices on private lands for the first time in 1985 (Weldon et al. 2010). Practices approved for private-land wildlife conservation broadened significantly with each reauthorization of the Farm Bill (Weldon et al. 2010), and the Farm Bill has been a significant contributor to the overall restoration and conservation of early successional plant communities with the goal of providing components of habitat for various wildlife species,

maintaining diverse native plant communities, and improving soil and water quality (Heard et al. 2000). Additionally, many state wildlife agencies have included early-succession management in their State Wildlife Actions Plans (Tennessee 2015, Kentucky 2013, North Carolina 2005, Georgia 2015), which determine wildlife management practices on state-owned lands.

Early successional communities can be restored or established using different methods. Numerous studies have evaluated restoration techniques, but most studies have focused on establishment and restoration through planting native warm-season grasses and forbs (Washburn et al. 2000, Harper et al. 2003, Jones et al. 2004). Recommended native grass and forb planting establishment can cost between \$450/ha and \$900/ha (Monroe et al. 2017, GeFellers et al. 2020). However, planting is not the only option. Recent research suggests restoration using selective herbicide applications along with the seedbank response is just as or more effective as planting and with considerable cost savings (GeFellers et al. 2020). Restoration by allowing the seedbank to respond and using strategic herbicide applications can cost approximately \$125/ha (GeFellers et al. 2020). Restoration efforts most often focus on converting nonnative grasses to a native grass-forb community (Madison et al. 1995, Washburn et al. 2000, Harper and Gruchy 2009, Hall et al. 2012).

Following restoration, disturbance is necessary to maintain early successional plant communities. Burning, mowing, and disking are common practices used to maintain early succession. However, the type of disturbance can influence plant species composition and the resulting plant community (Gruchy and Harper 2014). Herbicides are often used in conjunction with disturbance to promote native early successional plant communities (GeFellers et al. 2020).

Prescribed fire is an effective and efficient way to maintain early successional plant communities (Waldrop and Goodrick 2012). The effects of burning vary with fire frequency, fire

intensity, and fire timing (Stanturf et al. 2003, Harper et al. 2016). Using frequent prescribed fire (1–2-year frequency) will reduce tree and shrub cover and maintain an herbaceous plant community (Robertson and Hmielowski 2013). Plant communities shaped by frequent fire can provide greater amounts of forage for white-tailed deer (Nanney et al. 2018, Glow et al. 2019). Less frequent fire (3–5-year frequency) allows tree seedlings and scattered shrubs to develop (Briggs et al. 2002). Areas managed with fire on a 5–7-year interval can develop dense, young tree and shrub cover. Areas managed on a 3–7-year fire-return interval can provide important habitat components for shrubland birds (Briggs et al. 2002, Schlossberg et al. 2010). Prescribed fire consumes debris, creates openness at ground level, and stimulates the seedbank (Buckner and Landers 1979, Gruchy and Harper 2014).

Fire timing and intensity also can affect plant composition after disturbance. Low-intensity fire can remove litter and maintain shrub height and coverage (Zuckerberg and Vickery 2006). Greater fire intensity can reduce shrub or small tree coverage and increase fuel consumption (Drewa 2003). Fire timing (or season of burning) may influence plant composition. Burning in the early portion of the growing season may increase coverage of native warm-season grasses (Whitehead and McConnell 1980, Manley 1994, Gruchy and Harper 2014). Burning during the dormant season may increase grass and forb coverage while setting back growth of sprouting trees and shrubs, but resprouting continues (Waldrop et al. 1992, Brockway et al. 2002, Robertson and Hmielowski 2013). Burning during the latter portion of the growing season may reduce dominance of woody species and promote increased forb coverage (Lewis et al. 1964, Gruchy et al. 2009, Harper et al. 2016). Fire frequency, intensity, and timing are interconnected, and each must be considered when making burning decisions as related to effects on the plant community (Slocum et al. 2003, Harper et al. 2016).

The most common method for maintaining early successional plant communities is mowing, and it is often chosen as an alternative to prescribed fire (Dykes 2005, Golden et al. 2012, Gudlin et al. 2019). However, mowing can be expensive because of labor, fuel, and depreciation of equipment (Wigginton and Meyerson 2018). Early successional plant communities managed by mowing often are dominated by grasses and lack openness at ground level (McCoy et al. 2001, Dykes 2005, Gruchy and Harper 2014). Litter build-up is encouraged by mowing and contributes to the lack of openness at ground level, which impedes movement by many small wildlife species (McCoy 2001, Harper et al. 2007). The accelerated build-up of litter can suppress forbs and reduce species richness. Mowing is used by most landowners to maintain fields and reduce woody coverage, but studies suggest mowing once annually does not reduce woody coverage (Dykes 2005, Welch et al. 2004, Gruchy et al. 2009, Gruchy and Harper 2014).

Many wildlife species require early successional plant communities to provide food and cover, including northern bobwhite (*Colinus virginianus*), grasshopper sparrow (*Ammodramus savannarum*), field sparrow (*Spizella pusilla*), eastern meadowlark (*Sturnella magna*), Henslow's sparrow (*Ammodramus henslowii*), dickcissel (*Spiza americana*), hispid cotton rat (*Sigmodon hispidus*), least shrew (*Cryptotis parva*), and many species of pollinators (Whitmore 1981, Ginsberg 1983, Teer 1996, Althoff et al. 1997, Hunter et al. 2001). Other species, such as song sparrow (*Melospiza melodia*), wild turkey (*Meleagris gallopavo*), long-tailed weasel (*Mustela frenata*), eastern cottontail (*Sylvilagus floridanus*), red fox (*Vulpes vulpes*), and white-tailed deer (*Odocoileus virginianus*), do not require early successional plant communities, but their habitat may be enhanced with more diverse sources of food and cover if early successional communities are present.

We developed a field experiment across a large geographical area to assess plant community structure and composition in fields that previously were dominated by tall fescue and subsequently restored by planting native grasses and forbs or by seedbank response without planting but instead using strategic herbicide applications to control undesirable plant species (GeFellers et al. 2020). Our research objectives were to compare the effects of burning and mowing on vegetation composition and structure in early successional communities that had been established using the two methods. Previous research has evaluated the effects of common management techniques in planted communities, but because seedbank response has only recently been described and evaluated as a restoration technique, no research has yet compared disturbance practices between the two restoration types. We wanted to evaluate how vegetation composition changed following two years of burning or mowing and how the resulting food and cover values changed for the two most popular game species in the region, white-tailed deer and wild turkey, as well as a conservation-priority gamebird species, northern bobwhite.

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**CHAPTER I. VEGETATION STRUCTURE AND FOOD AVAILABILITY
FOLLOWING DISTURBANCE IN EARLY SUCCESSIONAL PLANT COMMUNITIES**

ABSTRACT

Fields dominated by nonnative grasses, such as tall fescue (*Schedonorus arundinaceus*), are being restored to native plant communities across the eastern U.S. However, disturbance is necessary to maintain early successional communities, and plant response to different management practices is of interest to managers to guide habitat enhancement for various wildlife species. We evaluated effects of burning and mowing following restoration of native plant communities via 2 methods (1. planting native grasses and forbs and 2. seedbank response without planting), across 11 replicated sites in Tennessee and Alabama, 2019-20. We compared vegetation composition and structure, openness at ground level, forage availability, and nutritional carrying capacity (NCC) following 4 treatments (Seedbank burned [SB], Seedbank mowed [SM], Planted burned [PB], Planted mowed [PM]), and tall fescue Control (CL). The combination of planting and mowing increased grass coverage, whereas units that were established via seedbank response and managed by burning had greater forb coverage. Visual obstruction above 25 cm was greater in all treatments than CL, which enhanced nesting cover for wild turkey (*Meleagris gallopavo*) and bedding cover for white-tailed deer (*Odocoileus virginianus*). Openness at ground level, which is especially important for northern bobwhite (*Colinus virginianus*) and wild turkey broods, was 30% greater in SB than PM, which provided the least openness among treatments, by the second year of treatment. Similarly, coverage of bobwhite food and deer forage plant species was greatest in SB. Selected deer forage biomass (kg/ha) was greater in all treatments than in CL, and NCC (deer days/ha) was greatest in SB by year 2. Our results highlight differences in plant composition and structure following management that can help managers decide which management technique best fits their

objectives. Furthermore, our results illustrate planting is not necessary to restore and maintain native plant communities on most sites.

KEY WORDS burning, early succession, mowing, native plant restoration, planting, seedbank response, white-tailed deer

Federal and state agencies have worked to reverse the decline of native early successional communities in recent years. Shifts in land-use practices, urbanization, fire suppression, and forest maturation all have contributed to significant landscape changes across the eastern United States and a decline of early successional communities (Brennan 1991, Noss et al. 1995, Askins 2001, Noss 2013). The loss of native early successional plant communities has corresponded with a decline in wildlife populations that require those communities (Brady et al. 1998, Kirkland and Hart 1999, Pruitt 2000, Askins 2001, Mcchesney and Anderson 2015). In response, USDA-Natural Resources Conservation Service (hereafter NRCS) offers cost-share programs that provide technical and financial assistance to landowners to restore these communities on private lands (USDA 2016). Many state wildlife agencies also have included early-succession management in their State Wildlife Actions Plans (Tennessee 2015, Kentucky 2013, North Carolina 2005, Georgia 2015), which guide wildlife management practices on state-owned lands. All of these programs commonly promote use of herbicides to kill nonnative grasses and subsequent planting of native grasses and forbs to restore native plant communities. However, research has highlighted problems commonly associated with planting, especially planting failure, cost, and the relative inability to control nonnative invasive plants without harming planted species (Barnes 2004, Rowe 2010). A more practical and efficient approach to restoring

native plant communities on sites previously dominated by nonnative species involves managing the seedbank response instead of planting (GeFellers et al. 2020). Selective spot-spray applications of herbicides in areas with only a seedbank response (i.e., no planting) restored native plant communities and improved cover and food resources for various wildlife species equal to or better than planting and was 3.7X less expensive (Harper et al. 2021).

Regardless of restoration technique, disturbance is necessary to maintain early successional plant communities, but the type of disturbance can influence the resulting plant community (MacDougall and Turkington 2007, Gruchy and Harper 2014, Harper 2017). Burning and mowing are common practices used to maintain early successional plant communities, and herbicides may be used in conjunction with disturbance to promote these communities (Nanney et al. 2018, GeFellers et al. 2020). The most common method for maintaining early successional plant communities is mowing, and it is often chosen as an alternative to prescribed fire (Dykes 2005). However, mowing can be expensive because of labor, fuel, and depreciation of equipment (Wigginton and Meyerson 2018). Early successional plant communities managed by mowing often are dominated by grasses and lack openness at ground level (McCoy et al. 2001, Dykes 2005, Gruchy and Harper 2014). Litter build-up is encouraged by mowing in planted fields, which suppresses forb coverage and contributes to a lack of openness at ground level, which impedes movement for many small wildlife species (McCoy 2001, Harper et al. 2007, Doxon and Carroll 2010).

Contrary to mowing, prescribed fire is an effective and efficient practice to maintain high quality early successional plant communities (Brockway et al. 2001, Waldrop and Goodrick 2012). The effects of burning vary with fire frequency, fire intensity, and fire timing (Stanturf et al. 2002, Harper et al. 2017). Prescribed fire consumes debris, creates openness at ground level,

and stimulates the seedbank (Buckner and Landers 1979, Gruchy and Harper 2014). Moreover, research has indicated frequent burning can increase deer forage availability in forested or woodland communities, but there is little information available describing how fire influences deer forage availability in early successional communities, such as old-fields (Lashley et al. 2011, Nanney et al. 2018, Glow et al. 2019).

We developed a field experiment to assess plant community structure and composition in fields that previously were dominated by tall fescue and subsequently restored by planting native grasses and forbs or by seedbank response without planting but instead using strategic herbicide applications to control undesirable plant species (GeFellers et al. 2020). We used mowing and burning to maintain the restored communities, as these methods commonly are used to manage native early successional plant communities in the eastern U.S. Our research objective was to compare the effects of burning and mowing in combination with strategic herbicide applications as approaches for maintaining early successional communities that had been established using the two different methods (i.e., planting and seedbank response). Research evaluating management techniques exists for planted communities, but no research has compared disturbance practices between planted communities and communities established using seedbank response only. Specifically, we wanted to evaluate plant communities following mowing or burning and assess the resulting food and cover values for the two most popular game species in the region, white-tailed deer (hereafter deer; *Odocoileus virginianus*) and wild turkey (*Meleagris gallopavo*), as well as a conservation-priority gamebird species, northern bobwhite (*Colinus virginianus*), which has experienced a significant population decline over the past 5 decades. We hypothesized burning would stimulate increased deer forage, increased coverage of bobwhite food plants, greater openness at ground level, and greater visual obstruction above 1 m compared

to mowing because the thatch resulting from mowing suppresses seedbank response and plant growth. We hypothesized seedbank response treatment units would have greater deer forage availability, greater coverage of bobwhite food plants, and increased openness at ground level than planted treatment units, regardless of disturbance technique, because planted units would have greater grass coverage and fewer forbs. Lastly, we hypothesized tall fescue-dominated controls would have the least deer forage availability, coverage of bobwhite food species, openness at ground level, and visual obstruction above 1 m because sod-forming grasses suppress germination of the seedbank and reduce relative abundance of forbs.

STUDY AREA

We conducted our study at 11 sites in Tennessee and Alabama. Each site was represented by a 0.8 to 1.6-ha field dominated by tall fescue. Six of the study sites were on Tennessee Valley Authority (TVA) property in Bedford, Hamblen, Jefferson, Monroe, and Sevier Counties, TN, and Franklin County, AL. One study site was in Cades Cove within the Great Smoky Mountains National Park in Blount County, TN. Three study sites were on Tennessee Wildlife Resources Agency (TWRA) property in Cocke, Cumberland, and White Counties, TN. Another study site was on Alabama Department of Conservation and Natural Resources (ADCNR) property in Jackson, AL. This study was initiated in 2015 by partitioning each field into 3 equally-sized treatment units (Planted, Seedbank, and tall fescue Control). Treatment units varied in size from 0.8 to 2.0 ha. Tall fescue was controlled in all study areas in November 2015 prior to initiating 2 establishment treatments (Planted and Seedbank) in spring 2016 as described by GeFellers (2019). Evaluation of the establishment techniques was described by GeFellers et al. (2020) and Harper et al. (2021). Elevation at study sites ranged from 180 m at the Franklin County, AL site

to 658 m above mean sea level at the Cumberland County, TN site. Soils at 10 of the 11 sites were loam or silt/loam, whereas one site (Jackson County, Alabama) had silt clay (Soil Survey Staff 2021).

METHODS

In spring 2019, we divided each of the establishment treatment units (Planted and Seedbank) at each site into 2 units and randomly assigned a management treatment (Mow and Burn) to each. Thus, we created four treatment units that varied in size from 0.36 to 0.95 ha at each of the 11 sites: Planted burn (PB), Planted mow (PM), Seedbank burn (SB), and Seedbank mow (SM). We also maintained the tall fescue-dominated control (CL) at each site.

Burning Treatments

Prescribed fires were conducted by location managers at each site in the appropriate units. Location managers installed firebreaks around units that were selected to burn using either disking, tilling, or application of water. Backing fires were used at each site to establish a blackened area on the downwind side of the burn unit adjacent to the firebreak, and all units were burned with backing fire for complete fine fuel consumption unless conditions demanded heading fires to consume fuels. Average flame lengths were 0.8 m, and all fires were low to moderate intensity. We conducted prescribed fire at each site in 2019 and 2020 during the late dormant season (February-early April), which is consistent with when most prescribed fire is implemented to maintain fields enrolled in conservation programs (Dryden 2001, Harper et al. 2007). All burns were conducted within the following prescription parameters: relative humidity 20–50%, wind speed 0–16 km/h, temperature -1–24 °C, and cloud cover <50%.

Mowing Treatments

Mowing was conducted in 2019 and 2020 by location managers or contractors using a tractor-mounted rotary mower during the dormant season (February-March). All units were mowed to a height of approximately 25.4 cm (10 in). Control units were maintained by annual late-winter mowing, consistent with how fallow fields dominated by tall fescue are commonly managed in the region (Dykes 2005).

Herbicide Treatments

We used selective herbicide applications in 2019 and 2020 to control undesirable species throughout the study in all treatment units. We used spot-spray applications with 15-L (4-gal) backpack sprayers (Solo USA, Newport News, Virginia) or a 95-L (25-gal) ATV sprayer (Cabelas, Sydney, Nebraska) equipped with a spray gun (Green Garde®, H.D. Hudson Manufacturing Company, Chicago, Illinois). We used selective herbicide applications once during spring and once during summer to control undesirable warm-season species, such as johnsongrass (*Sorghum halepense*), sericea lespedeza (*Lespedeza cuneata*), or bermudagrass (*Cynodon dactylon*). We made one application during fall/winter to control undesirable cool-season species, such as common henbit (*Lamium amplexicaule*) or purple deadnettle (*Lamium purpureum*), if needed. We used selective applications in Planted units only in accordance with recommendations of Private Lands Biologists with ADCNR and TWRA to remain in compliance with what is permitted and recommended to landowners enrolled in state conservation programs. We controlled nonnative invasive species when possible with herbicides that would not damage planted species in Planted units; however, there are no options to control some invasive species, such as bermudagrass or sericea lespedeza, without harming planted native grasses and forbs. Undesirable vegetation was categorized as nonnative invasive plants, but we also controlled 2

native species, blackberry (*Rubus spp.*) and black locust (*Robinia pseudoacacia*), when coverage exceeded 30%. Nonnative invasive species were controlled with selective herbicide applications in Seedbank units when they occurred, regardless of coverage. Control units were spot-sprayed with a mixture of triclopyr (0.10 kg ai/ha) and fluroxypyr (0.31 kg ai/ha) using an ATV sprayer in May 2019 to control woody encroachment. No additional herbicide treatment was used in Control units.

Data Collection

We recorded vegetation composition, measured vegetation structure and litter depth, and collected deer forages in each treatment and control mid-June-early August 2019 and 2020 (once at each site each year).

Vegetation Composition

We used line-point intercept sampling along 4 50-m transects placed systematically in all treatment units. We recorded vegetation to species at 2-m intervals. We calculated percent coverage of various plants or plant groups by dividing the number of detections by the total number of sampling points (n=25) on that transect. We split plants into 6 plant groups including grass, forbs, native grasses, nonnative grasses, semi-woody, and woody. Grass included all grass species detected. We further divided grass into native and nonnative grasses which included all native grasses detected and nonnative grasses detected. Forb included all broadleaf herbaceous species that were detected. Woody included tree and shrub species. Semi-woody included brambles and vines. We then calculated percent coverage of each plant or plant group for each unit by averaging percent cover from all 4 transects in that unit (Table A.1). Vegetation coverage was used to calculate and observe changes in percent coverage of bobwhite food and deer forage plant species following management. We considered plants producing either seed or soft mast

readily consumed by bobwhite as bobwhite food plants, and we included plants that have been identified as selected by deer in our region of study as deer forage plant species (Rosene and Freeman 1988, Johnson et al. 2018, Lashley et al. 2011, Nanney et al. 2018, Harper et al. 2021).

Forage Samples

We collected 2 forage samples along each transect in a 0.5-m² PVC frame (total of 8 per treatment unit). We positioned the frame adjacent to the transect at random locations between 0 and 50 m. All live portions of plants at and below 2 m were collected in each frame. We placed samples in forage collection bags and sorted them in the lab. We removed all stems and lignified portions of the plants during sorting. Leaves and tender growing portions of herbaceous plants and brambles, as well as twig ends and leaves of woody species and vines, were retained for weighing and nutrition analysis. Retained plant parts represented what is normally selected by deer (Lashley et al. 2014). We collected and identified plant species selected by deer and we separated young and old portions of leaves and stems for nutritional analysis. Selectivity of plant species by deer at each site was determined by Harper et al. (2021). Young, tender growing portions of vegetation are more palatable and nutritious than older, more lignified plant parts; therefore, we weighed and analyzed young and old portions of the plants we collected separately.

We dried forage samples in walk-in drying ovens at 50°C for 72 hours and shipped dried samples to the Agriculture Service Laboratory at Clemson University for wet chemistry nutrient analysis (Ondarza and Ward 2013). We used forage biomass calculations and nutrient analysis results to estimate deer days/ha using an explicit nutritional constraints model (Hobbs and Swift 1985). Constraints for the model were set at crude protein (CP) levels of 14%. Crude protein levels of 14% represents nutritional requirements needed to support adult females during late pregnancy and peak lactation and exceed nutritional requirements for adult body maintenance in

adult females during summer and requirements for adult males that are growing antlers and gaining body mass during summer (Asleson et al. 1996, NRC 2007, Lashley et al. 2011, Nanney et al. 2018, Harper et al. 2021).

Vegetation Structure

We measured vegetation structure by estimating visual obstruction along a modified vegetation profile board (Nudds 1977). The vegetation profile board was 2 m in height and was separated into 5 alternating orange and white sections. The bottom 0.25-m stratum represented visual obstruction at the level where bobwhite and other small wildlife species occur. Visual obstruction up to 0.5 m represented cover important to brooding wild turkey (Metzler and Speake 1985, Peoples et al. 1995). The upper 1.5 m was separated into 3 0.5-m strata which were used to evaluate structure important for deer fawns and adult deer (Huegel et al. 1986, DePerno et al. 2003). We recorded 2 visual obstruction measurements along each transect at the 14- and 34-m mark. One member of the research crew knelt at plot center (24 m) and estimated visual obstruction by placing each of the 5 profile board sections into 1 of 6 categories (e.g., 0 = no vegetation, 1 = 1-20% obstruction, 2 = 21-40% obstruction, etc.). We calculated average visual obstruction for each of the 5 profile board sections and compared across treatment units and years.

We recorded ground-sighting distance measurements at the 14-m mark and at the 34-m mark on the transect tape to provide an index of openness at ground level using a sight-tube made from PVC pipe to observe from 10.2-cm above ground (Gruchy and Harper 2014). The observer looked horizontally through the sight-tube while another team member placed a colored 5.1-cm diameter PVC target in front of the sight-tube. The target was moved horizontally away

from the sight-tube until the bottom 15 cm of the target was completely obscured by vegetation. The distance between the tube and target was recorded and averaged for each treatment unit.

DATA ANALYSIS

We used a randomized block, repeated measures study design with treatment unit serving as the experimental unit. We averaged measurements from the 4 transects in each treatment unit to calculate the mean for each treatment unit. Results were obtained using analysis of variance (ANOVA), with linear models fit using package “nlme” (Pinheiro et al. 2021) in program R (R Core Team 2020). Autocorrelation between remeasurements was checked and considered negligible for all models. Analysis determined the effects of treatment, year, and treatment $\times$ year interactions on plant composition, coverage of bobwhite and deer food plant species, visual obstruction, openness at ground level, deer forage estimates, and NCC estimates. If we documented treatment $\times$ year effects, we created interaction plots to assist with interpretation. If only treatment or year effects were significant, we used package “emmeans” (Lenth 2018) to compare means within year among treatments and between years within treatments using Tukey’s Honest Significant Difference (HSD) p-value adjustment. We used a significance level of $\alpha=0.05$ for all contrasts.

RESULTS

We detected treatment, year, and treatment $\times$ year effects for grass coverage. By the second year of treatment, the Planted treatments were similar to CL in total grass coverage (Table A.1). Grass coverage increased 17% from 2019 to 2020 in all treatments except PM, which increased by 34%. Coverage of native grass was greater in all treatment units than CL, and increased 26 –

29% in all treatments except PM, which increased 73% (Table A.1). Coverage of nonnative grass was greatest in CL, and remained similar across years by treatment, but was approximately 60% greater in the Planted treatments than Seedbank treatments. We detected treatment and treatment $\times$ year effects for forb coverage. Forb coverage was greater in all treatment units than CL, and ranged from 12% in CL to 82% in SB in 2020 (Table A.1). Forb coverage decreased 24% in PM, 19% in PB, and 3% in SM, but increased 8% in SB from 2019 to 2020. All treatment units had greater semi-woody coverage than CL. Semi-woody and woody coverage remained similar among treatment units; we did not detect differences across years.

We detected treatment and year effects for coverage of bobwhite food plants, and we detected treatment effects for coverage of deer forage plants (Table A.1). Coverage of bobwhite food plants and deer forage plants were greater in all treatment units than CL, and coverage of both tended to be greater in Seedbank units than Planted units and greater in burned units than mowed units. Bobwhite food plants and deer forage plants were most abundant in SB and represented a majority of the plant species in SB.

We also detected treatment effects for availability of selected deer forage biomass as well as NCC (Tables A.2 and A.3). Availability of selected deer forage and NCC were greater in all treatments than CL. Ironically, almost all of the deer forage collected in Planted units was from plants germinating from the seedbank as $\leq 6\%$ of the deer forage collected in PM and PB was represented by planted species. Deer days/ha were approximately 8X greater in SB than CL (Table A.3).

Visual obstruction at the <25 cm stratum was greater than 97% and similar among all treatments and CL in 2019 and 2020 ($F = 0.64$, $P = 0.636$). However, visual obstruction at the 26–50 cm strata ($F = 11.62$, $P < 0.0001$) and 51–100 cm strata ($F = 10.54$, $P < 0.0001$) was

greater in all treatment units than CL. In general, burning tended to maintain taller structure than mowing at the 51–100 cm stratum and above following the second management treatment (Figure A.1). Openness at ground level was >20% greater in SB than CL (Table A.4).

DISCUSSION

Our study compared effects of the 2 most common management techniques to maintain early successional plant communities in the eastern U.S., and we related those effects to habitat conditions for the 2 most common game species and an iconic gamebird undergoing precipitous population decline. Our data indicated burning leads to increased forb coverage and taller structure when compared to mowing, which supported several initial hypotheses. Burning also resulted in greater cover of bobwhite food plants and deer forage plants, and greater NCC for deer than mowing. Planting native grasses, in contrast, increased grass coverage, limited forb coverage, and reduced food availability for focal species. We did not detect a treatment effect for openness at ground level, which was counter to our hypothesis. Following GeFellers et al. (2020) and Harper et al. (2021), our data continue to demonstrate that planting native grasses and forbs, whether managed by mowing or burning, is not necessary to enhance the food or structural requirements of deer, wild turkey, or bobwhite when converting fields dominated by nonnative grasses to native early successional plant communities.

Coverage of grass tended to increase in all treatments from 2019 to 2020, but especially in PM. By the second year of treatment, grass coverage in the Planted treatments averaged 85%, far in excess of what bobwhite require or use. Unger et al. (2015) reported planted native grass coverage increased from 49% to 77% following dormant-season burning, and bobwhite avoided those areas. In our Seedbank treatment units where native grasses were never planted, total grass

coverage increased from approximately 60% in 2019 to nearly 70% in 2020, and native grass coverage averaged 52% by the second year. Bobwhite nest and brood locations in Kentucky averaged 15% and 5% coverage of native grasses, respectively (Brooke et al. 2016).

Additionally, deer do not select grass to eat during the growing season (Korschgen et al. 1980, Lashley et al. 2011, Nanney et al. 2018, Harper et al. 2021), and wild turkeys select forb-dominated plant communities for brooding similar to bobwhite (Healy 1985, Campo et al. 1989, Spears et al. 2007), further supporting the belief that early successional plant communities managed for these species should not have >20 – 30% grass coverage. Gruchy and Harper (2014) and Brooke and Harper (2016) evaluated techniques to reduce density of planted native grasses and recommended heavy disking following burning or applications of glyphosate or imazapyr to achieve 10 – 30% grass coverage.

As grass coverage increased, forb coverage tended to decrease in PB and PM, remained similar in SM, but increased 8% in SB from 2019 to 2020. Grman et al. (2020) reported planted native grasses suppressed forb coverage and reduced plant diversity. We observed similar effects on forb coverage after management in units that were restored by planting. Before we applied management treatments, the planted units averaged 64% coverage of forbs (Harper et al. 2021). By 2020, PB averaged 52% and PM averaged 45% forb coverage. Mowing also led to reduced forb coverage. Fields restored via seedbank response without planting averaged 72% forb coverage before any management occurred (Harper et al. 2021). By 2020, forb coverage averaged 82% in SB and 61% in SM.

Increased forb coverage can improve components of habitat for many wildlife species. Coverage of bobwhite food plants obviously is important for bobwhite, especially during winter months, when energy requirements are high and food limitations during this time can result in

decreased survival (Robel 1963, Roseberry 1964, Madison et al. 2002). We considered 113 plants as bobwhite food plants, the majority of which were forbs. Grasses only comprised 19% of quail food plants. The combination of seedbank response and burning resulted in greater coverage of bobwhite food plants than any other restoration/management combination. Additionally, cover provided by forb-dominated communities typically facilitate movement of bobwhite and wild turkey broods while supporting more insects than grass-dominated communities (Hill 1985, Burger et al. 1993, Harper et al. 2001, Fettinger et al. 2002, Jamison et al. 2002).

Deer forage biomass estimates were similar among all treatment units. However, >95% of deer forages collected in planted units in 2020 were not planted species, but instead were species that responded from the seedbank. Previous studies conducted in the same region as our study reported deer selected forbs during spring and summer when individuals are lactating and growing (Lashley et al. 2011, Nanney et al. 2018, Harper et al. 2021). We stress the importance of forbs for deer because more private landowners and state agencies manage property for deer in the eastern U. S. than any other species (McShea 2012), and if proper management for deer benefits a wide array of wildlife, then attention to deer management is warranted even if deer is not a species of conservation concern (Harper et al. 2021). Simply eliminating coverage of nonnative grass and allowing the seedbank to respond allowed greater forb coverage and increased the NCC for deer by approximately 800%.

Coverage of semi-woody and woody plants were similar among treatments, and semi-woody coverage was greater in treatment units than CL. Coverage of semi-woody and woody plants is considered desirable for deer, wild turkey, and bobwhite. Blackberry and raspberry (*Rubus occidentalis*) were included in semi-woody, and several woody species we detected were

selected deer forages, and the structure these woody species provide is important for nesting wild turkey (Badyaev 1995, Moore et al. 2010). Coverage of semi-woody/woody species is requisite for bobwhite, which has been characterized as a shrubland-obligate species (Crosby et al. 2015). Brooke et al. (2015) reported bobwhite locations averaged <40 m from woody cover during both the breeding and nonbreeding seasons, and Klimstra and Roseberry (1975) identified preferred nesting cover as idle fields in the perennial weed/bramble/early shrub phase of succession, which is consistent with the vegetation present in our Seedbank treatment units.

Visual obstruction as influenced by the structure of vegetation may determine occupancy and use of an area by various wildlife species (Swanson et al. 1999, West et al. 2016). Visual obstruction and habitat use by certain wildlife species may vary as vegetation type, density, and height changes (Winter et al. 2005). Deer bedsites generally are located where taller vegetation provides greater visual obstruction (Huegel et al. 1986, Uresk et al. 1999, DePerno et al. 2003, Chitwood et al. 2017). Wild turkeys typically select nest locations with greater visual obstruction from 0.5–1.5 m, and nest success may be greater with increased visual obstruction from 0.5–1 m (Badyaev 1995, Kilburg et al. 2014, Johnson et al. *In press*). All of our treatments provided greater visual obstruction at the 0.5–1 m level than CL. Visual obstruction at the 1.25–1.5 m level was most influential on microsite selection by bobwhite during the breeding season in Kentucky (Brooke et al. 2015), and our burned treatments had the same structure, more so than mowed treatments.

Openness at ground level is important for bobwhite and wild turkey, especially for broods. Insect and seed availability are inconsequential when adult bobwhites and broods cannot effectively forage because of thick, impeding vegetation (Barnes et al. 1995, Taylor et al. 1999, Collins et al. 2009, Moorman et al. 2013). Although we did not detect differences among

treatments, SB tended to have greater openness at ground level than planted units by the second year of treatment. Gruchy and Harper (2014) and McCoy et al. (2001) reported mowing increased litter depth and reduced openness at ground level when compared to disking or burning. Our data also indicate an increase in grass coverage to the extent present in our treatments, and especially the Planted treatments, reduces openness at ground level and may limit movement of young bobwhites and wild turkey broods.

MANAGEMENT IMPLICATIONS

Disturbance is required to maintain early successional communities in the eastern US, regardless of establishment technique. Following 2 years of evaluating the effects of burning and mowing on vegetation composition and structure, we recommend managers use fire over mowing to set-back succession and maintain early seral-stage communities, especially if increased food for bobwhite and deer and enhanced structure at multiple levels for deer, wild turkey, and bobwhite are management objectives. Fire frequency should be determined by plant community response. We recommend selective spot-spray herbicide applications to reduce undesirable plant species and encourage colonization of additional desirable species, and we encourage managers to consider using seedbank response and natural colonization of plants instead of planting native grasses and forbs to more effectively and efficiently promote and enhance habitat for bobwhite, wild turkey, and deer. Although planting is increasingly justified as a means to ensure the availability of nectar resources for pollinators, fostering and maintaining the seedbank response with prescribed fire encourages a range of desirable forbs that benefit pollinators along with the species addressed in this study (GeFellers et al. 2020, Harper et al. 2021).

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APPENDIX A: CHAPTER I TABLES AND FIGURES

Table A.1. Mean coverage (%) and standard error (SE) of plant groups detected in 5 early successional plant community treatments across all study sites (n = 11) in Tennessee and Alabama, USA, June–August 2019 and 2020.

life form	Treatment ^A														
	CL			PB			PM			SB			SM		
	x ^A	SE		x	SE		x	SE		x	SE		x	SE	
2019 ^B															
Grass ^C	98.8	0.4	Aa	69.2	5.21	Ba	66.4	6.95	Ba	57.6	6.42	Ba	61.9	6.26	Ba
NWSG _D	20.2	6.16	Aa	44.5	6.63	Ba	39.9	7.32	Ba	40.2	6.88	Ba	42.3	7.97	Ba
NNG ^E	87.2	5.1	Aa	26.8	7.04	Ba	26.0	6.42	Ba	17.3	5.76	Ba	16.8	6.29	Ba
Forb ^F	8.8	4.71	Aa	64.4	5.48	Ba	58.7	7.0	Ba	75.8	4.73	Ba	63.1	6.79	Ba
Semi-woody _G	1.9	0.98	Aa	20.6	5.23	Ba	15.8	4.64	Ba	18.1	4.24	Ba	21.3	5.37	Ba
Woody _H	0.3	0.14	Aa	2.4	0.78	Aa	3.8	1.39	Aa	5.3	3.08	Aa	5.2	2.3	Aa
BWFP ^I	2.36	0.96	Aa	32.0	6.93	Ba	24.4	5.55	Ba	40.0	8.17	Ba	35.1	8.4	Ba
DFP ^J	3.4	1.78	Aa	52.8	7.77	Ba	44.4	7.61	Ba	67.2	7.15	Ca	60.9	6.81	BC _a
2020															
Grass	98.0	1.8	Aa	80.7	4.16	AB _{Ca}	89.0	3.51	AB _b	67.2	2.84	Ca	72.3	5.80	BC _a
NWSG	9.7	3.27	Aa	56.6	6.64	Ba	69.2	5.52	Bb	50.7	5.27	Ba	54.6	6.99	Ba
NNG	96.5	2.28	Aa	30.0	7.14	Ba	26.1	5.35	Ba	15.5	3.53	Ba	17.7	5.27	Ba
Forb	12.2	4.64	Aa	52.3	7.25	Ba	44.6	6.59	Ba	81.9	4.78	Ca	60.9	6.41	BC _a
Semi-woody	2.5	1.22	Aa	15.6	3.89	Ba	16.5	3.77	Ba	22.6	4.76	Ba	21.5	4.04	Ba
Woody	2.3	1.19	Aa	2.3	0.81	Aa	3.8	1.31	Aa	5.4	1.78	Aa	5.8	1.73	Aa
BWFP	9.6	2.56	Aa	35.6	5.79	Ba	26.6	3.73	Ba	57.4	5.86	Ca	41.1	4.18	BC _a
DFP	8.7	3.53	Aa	48.3	6.79	Ba	43.5	7.47	Ba	78.7	3.91	Ca	57.3	6.37	BC _a

^A Within year, treatments with the same uppercase letter did not differ (P > 0.05)

^B Between years, treatments with the same lowercase letter did not differ (P > 0.05)

^C Percent coverage of all grasses present in each treatment ($F^{4,40} = 3.21$, P < 0.02)

^D Percent coverage of native warm-season grasses (>90%) including broomsedge bluestem (*Andropogon virginicus*), little bluestem (*Schizachyrium scoparium*), purpletop (*Tridens flavus*), and beaked panicgrass (*Panicum anceps*) ($F^{4,50} = 7.92$, P < 0.0001)

^E Percent coverage of nonnative grasses (>90%), including tall fescue (*Schedonorus arundinaceus*), johnsongrass (*Sorghum halepense*), bermudagrass (*Cynodon dactylon*), and Japanese stiltgrass (*Microstegium vimineum*) ($F^{4,40} = 75.2$, $P < 0.0001$)

^F Percent coverage of forbs ($F^{4,50} = 2.79$, $P < 0.035$)

^G Percent coverage of semi-woody includes brambles and vines ($F^{4,40} = 7.57$, $P < 0.0001$)

^H Percent coverage of trees and shrubs ($F^{4,40} = 2.38$, $P < 0.067$)

^I Percent coverage of bobwhite food plants ($F^{4,40} = 19.17$, $P < 0.0001$)

^J Percent coverage of deer forage plants ($F^{4,40} = 32.03$, $P < 0.0001$)

Table A.2. Mean $\pm$ SE deer forage biomass (kg/ha) by treatment for all study sites (n = 11) in Tennessee and Alabama, USA, June–August 2019 and 2020.

Treatment ^A	2019 ^B	2020
CL	104 $\pm$ 73 A	102 $\pm$ 50 A
PB	707 $\pm$ 161 B	523 $\pm$ 118 B
PM	623 $\pm$ 211 B	548 $\pm$ 127 B
SB	629 $\pm$ 168 B	745 $\pm$ 154 B
SM	647 $\pm$ 120 B	604 $\pm$ 108 B
<i>F</i> ^{4,40}	9.58	
<i>P</i>	<0.0001	

^A Treatments with the same uppercase letter did not differ ($P > 0.05$)

^B No effect of year ($P = 0.211$)

Table A.3. Mean $\pm$ SE nutritional carrying capacity (deer days/ha) by treatment for all study sites (n = 11) in Tennessee and Alabama, USA, June–August 2019 and 2020.

Treatment ^A	2019 ^B	2020
CL	37 $\pm$ 16 A	19 $\pm$ 8 A
PB	192 $\pm$ 42 B	206 $\pm$ 48 B
PM	164 $\pm$ 39 B	185 $\pm$ 41 B
SB	212 $\pm$ 50 B	279 $\pm$ 57 B
SM	209 $\pm$ 51 B	171 $\pm$ 34 B
<i>F</i> ^{4,40}	7.04	
<i>P</i>	0.0002	

^A Treatments with the same uppercase letter do not differ ($P > 0.05$)

^B No effect of year ($P = 0.42$)

Table A.4. Mean $\pm$ SE openness at ground level (cm) by treatment for all study sites (n = 11) in Tennessee and Alabama, USA, June–August 2019 and 2020.

Treatment ^A	2019 ^B	2020
CL	77.6 $\pm$ 4.9 Aa	71.5 $\pm$ 8.4 Aa
PB	75.3 $\pm$ 5.5 Aa	68.2 $\pm$ 5.5 Aa
PM	78.7 $\pm$ 8.5 Aa	66.4 $\pm$ 5.3 Aa
SB	79.7 $\pm$ 5.0 Aa	86.2 $\pm$ 6.5 Aa
SM	74.7 $\pm$ 7.4 Aa	76.1 $\pm$ 8.4 Aa
<i>F</i> ^{4,40}	1.69	
<i>P</i>	0.17	

^A Treatments with the same uppercase letter do not differ ($P > 0.05$)

^B Between years, treatments with the same lowercase letter do not differ ($P > 0.05$)

Table A.5. Species that were included in bobwhite food plants category and species that were used to calculate deer forage availability and NCC at all sites (n=11) across Tennessee and Alabama, USA, June–August 2019 and 2020.

common name	scientific name	life form	QF ^a	DF ^b
black raspberry	<i>Rubus occidentalis</i>	Bramble	X	X
blackberry	<i>Rubus spp.</i>	Bramble	X	X
cat greenbrier	<i>Smilax glauca</i>	Bramble	X	X
common greenbrier	<i>Smilax rotundifolia</i>	Bramble	X	X
sensitive briar	<i>Mimosa microphylla</i>	Bramble	X	
multiflora rose	<i>Rosa multiflora</i>	Bramble	X	X
northern dewberry	<i>Rubus flagellaris</i>	Bramble	X	X
rose spp.	<i>Rosa spp.</i>	Bramble	X	X
saw greenbrier	<i>Smilax bonanox</i>	Bramble	X	X
Amaranthus spp.	<i>Amaranthus spp.</i>	Forb	X	
American black nightshade	<i>Solanum americanum</i>	Forb	X	
American burnweed	<i>Erechtites hieraciifolius</i>	Forb		X
American hog peanut	<i>Amphicarpaea bracteata</i>	Forb	X	
American pokeweed	<i>Phytolacca americana</i>	Forb	X	X
Asiatic dayflower	<i>Commelina communis</i>	Forb	X	
bigpod sesbania	<i>Sesbania herbacea</i>	Forb	X	
bigroot morningglory	<i>Ipomoea pandurata</i>	Forb	X	
black-eyed Susan	<i>Rudbeckia hirta</i>	Forb		X
broadleaf dock	<i>Rumex obtusifolius</i>	Forb	X	
bushy aster	<i>Symphyotrichum dumosum</i>	Forb		X
butterfly pea	<i>Clitoria mariana</i>	Forb	X	
Canada goldenrod	<i>Solidago canadensis</i>	Forb		X
Carolina geranium	<i>Geranium carolinianum</i>	Forb	X	X
Carolina horsenettle	<i>Solanum carolinense</i>	Forb	X	

Table A.5. Continued

common name	scientific name	life form	QF ^a	DF ^b
clammy groundcherry	<i>Physalis heterophylla</i>	Forb	X	
common evening primrose	<i>Oenothera biennis</i>	Forb	X	
common lespedeza	<i>Lespedeza striata</i>	Forb	X	
common milkweed	<i>Asclepias syriaca</i>	Forb	X	
common ragweed	<i>Ambrosia artemisiifolia</i>	Forb	X	X
common St. Johnswort	<i>Hypericum perforatum</i>	Forb	X	
curly dock	<i>Rumex crispus</i>	Forb	X	
cutleaf eveningprimrose	<i>Oenothera laciniata</i>	Forb	X	
daisy fleabane	<i>Erigeron annuus</i>	Forb		X
everlasting peavine	<i>Lathyrus latifolius</i>	Forb	X	
giant ragweed	<i>Ambrosia trifida</i>	Forb	X	X
gray-headed coneflower	<i>Ratibida pinnata</i>	Forb		X
horseweed	<i>Conyza canadensis</i>	Forb		X
Illinois bundleflower	<i>Desmanthus illinoensis</i>	Forb	X	X
ivyleaf morningglory	<i>Ipomoea hederacea</i>	Forb	X	
late boneset	<i>Eupatorium serotinum</i>	Forb		X
naked-flowered ticktrefoil	<i>Desmodium nudiflorum</i>	Forb	X	X
Palmer pigweed	<i>Amaranthus palmeri</i>	Forb	X	
panicled-leaf ticktrefoil	<i>Desmodium paniculatum</i>	Forb	X	X
Pennsylvania smartweed	<i>Polygonum pensylvanicum</i>	Forb	X	
prairie tea	<i>Croton monanthogynus</i>	Forb	X	
purple coneflower	<i>Echinacea purpurea</i>	Forb		X
red clover	<i>Trifolium pretense</i>	Forb	X	X
roundhead lespedeza	<i>Lespedeza capitata</i>	Forb	X	
sensitive partridge pea	<i>Chamaecrista nictitans</i>	Forb	X	X
showy partridge pea	<i>Chamaecrista fasciculata</i>	Forb	X	X
sicklepod	<i>Senna obtusifolia</i>	Forb	X	

Table A.5. Continued

common name	scientific name	life form	QF ^a	DF ^b
slender lespedeza	<i>Lespedeza virginica</i>	Forb	X	
smooth ticktrefoil	<i>Desmodium laevigatum</i>	Forb	X	X
spanishneedles	<i>Bidens alba</i>	Forb	X	X
stiff-haired sunflower	<i>Helianthus hirsutus</i>	Forb	X	
stiff ticktrefoil	<i>Desmodium obtusum</i>	Forb	X	X
swamp sunflower	<i>Helianthus angustifolius</i>	Forb	X	
three-seeded mercury	<i>Acalypha rhomboidea</i>	Forb	X	X
trailing lespedeza	<i>Lespedeza repens</i>	Forb	X	
violet spp.	<i>Viola spp.</i>	Forb		X
white clover	<i>Trifolium repens</i>	Forb		X
white old-field aster	<i>Symphyotrichum pilosum</i>	Forb		X
white vervain	<i>Verbena urticifolia</i>	Forb		X
wild bean	<i>Strophostyles helvola</i>	Forb	X	
wild lettuce	<i>Lactuca virosa</i>	Forb		X
wooly croton	<i>Croton capitatus</i>	Forb	X	
yellow rocket	<i>Barbarea vulgaris</i>	Forb	X	
yellow woodsorrel	<i>Oxalis stricta</i>	Forb		X
annual foxtail	<i>Setaria viridis</i>	Grass	X	
bahiagrass	<i>Paspalum notatum</i>	Grass	X	
barnyardgrass	<i>Echinochloa crus-galli</i>	Grass	X	
beaked panicgrass	<i>Panicum anceps</i>	Grass	X	
dallisgrass	<i>Paspalum dilatatum</i>	Grass	X	
deertongue	<i>Dichanthelium clandestinum</i>	Grass	X	
Dichanthelium spp.	<i>Dichanthelium spp.</i>	Grass	X	
fall panicum	<i>Panicum dichotomiflorum</i>	Grass	X	
frank sedge	<i>Carex atherodes</i>	Grass	X	
giant foxtail	<i>Setaria faberi</i>	Grass	X	

Table A.5. Continued

common name	scientific name	life form	QF ^a	DF ^b
goosegrass	<i>Eleusine indica</i>	Grass	X	
johnsongrass	<i>Sorghum halepense</i>	Grass	X	
knotroot foxtail	<i>Setaria parviflora</i>	Grass	X	
Panicum spp.	<i>Panicum spp.</i>	Grass	X	
Paspalum spp.	<i>Paspalum spp.</i>	Grass	X	
rice cutgrass	<i>Leersia oryzoides</i>	Grass	X	
Setaria spp.	<i>Setaria spp.</i>	Grass	X	
thin paspalum	<i>Paspalum setaceum</i>	Grass	X	
vaseygrass	<i>Paspalum urvillei</i>	Grass	X	
American beautyberry	<i>Callicarpa americana</i>	Shrub	X	X
American plum	<i>Prunus americana</i>	Shrub	X	
bicolor lespedeza	<i>Lespedeza bicolor</i>	Shrub	X	
coralberry	<i>Symphoricarpos orbiculatu</i>	Shrub	X	
elderberry	<i>Sambucus canadensis</i>	Shrub	X	X
Chinese privet	<i>Ligustrum sinense</i>	Shrub	X	
winged sumac	<i>Rhus copallinum</i>	Shrub	X	X
black cherry	<i>Prunus serotina</i>	Tree	X	
black locust	<i>Robinia pseudoacacia</i>	Tree	X	
black oak	<i>Quercus velutina</i>	Tree	X	
boxelder	<i>Acer negundo</i>	Tree	X	X
common persimmon	<i>Diospyros virginiana</i>	Tree	X	X
eastern redcedar	<i>Juniperus virginiana</i>	Tree	X	
flowering dogwood	<i>Cornus florida</i>	Tree	X	X
green ash	<i>Fraxinus pennsylvanica</i>	Tree	X	X
common hackberry	<i>Celtis occidentalis</i>	Tree	X	X
honeylocust	<i>Gleditsia triacanthos</i>	Tree	X	
loblolly pine	<i>Pinus taeda</i>	Tree	X	

Table A.5. Continued

common name	scientific name	life form	QF ^a	DF ^b
red maple	<i>Acer rubrum</i>	Tree	X	X
sassafras	<i>Sassafras albidum</i>	Tree	X	
sweetgum	<i>Liquidambar styraciflua</i>	Tree	X	
Japanese honeysuckle	<i>Lonicera japonica</i>	Vine	X	X
muscadine	<i>Vitis rotundifolia</i>	Vine	X	X
poison ivy	<i>Toxicodendron radicans</i>	Vine	X	X
trumpet creeper	<i>Campsis radicans</i>	Vine		X
Virginia creeper	<i>Parthenocissus quinquefolia</i>	Vine	X	X
grape	<i>Vitis spp.</i>	Vine	X	X

^a X signifies species that were considered bobwhite food plants

^b X signifies species that were considered deer forages.

Average Percent Visual Obstruction by Nudds Board Stratum 2019

Stratum 5 151-200 cm	3.4%, A	38.3%, B	39.6%, B	29.3%, B	25.9%, AB
Stratum 4 101-150 cm	13.5%, A	54.6%, B	51.4%, B	43.2%, B	39.6%, B
Stratum 3 51-100 cm	38.4%, A	78.5%, B	73.3%, B	67.7%, B	68.4%, B
Stratum 2 26-50 cm	78.0%, A	97.0%, B	94.9%, B	91.1%, B	93.6%, B
Stratum 1 0-25 cm	97.5%, A	100.0%, A	99.2%, A	99.5%, A	99.8%, A
	CL	PB	PM	SB	SM

Average Percent Visual Obstruction by Nudds Board Stratum 2020

Stratum 5 151-200 cm	1.0%, A	24.4%, B	16.4%, B	27.5%, B	9.5%, AB
Stratum 4 101-150 cm	10.9%, A	42.3%, B	28.7%, B	41.1%, B	21.5%, AB
Stratum 3 51-100 cm	32.5%, A	71.4%, B	60.3%, B	63.8%, B	50.4%, B
Stratum 2 26-50 cm	72.6%, A	94.0%, B	94.7%, B	86.8%, B	87.4%, B
Stratum 1 0-25 cm	98.9%, A	99.8%, A	100.0%, A	98.9%, A	100.0%, A
	CL	PB	PM	SB	SM

Figure A.1. Average percent visual obstruction during June–August 2019 and 2020 at all study sites (n=11) in Tennessee and Alabama following burn and mow treatments. Different letters between treatments within a stratum represent significant differences in visual obstruction.

APPENDIX B: INCLUSION OF SOIL DISTURBANCE DATA FOR CHAPTER I

Disking is often used to set-back succession to the first seral stage and encourage increased coverage of annual plants. Previous research has indicated that intensity of disking can influence plant composition. Light disking commonly is used to disturb areas, but the results of light disking on vegetation are shorter-lived when compared to more intense disking because light disking does not disturb perennial root systems like deep or heavy disking (Greenfield et al. 2003). Native warm-season grass coverage may be reduced by disking and coverage of bare ground may be increased (Gruchy and Harper 2014). Disking can incorporate thatch into the soil, reducing litter cover and depth, and provide openness at ground level (McCoy et al. 2001, Harper et al. 2007, Gruchy and Harper 2014). Timing of disking also can influence plant composition (Jones et al. 1993, Olinde 2000, Carver et al. 2001). Plant composition following disking can be difficult to predict because it can be influenced by composition of the seedbank, site history, soil pH, soil moisture, and nutrient availability (Olinde 2000).

A firebreak was installed (February-March) at ten of the eleven study sites to divide mowed units from burned units. Firebreaks were approximately 4.5 meters (15 ft) wide and served as a disked treatment. Plant composition data and deer forage plants were collected in firebreaks in the same manner as the mow and burn treatment units except the size of the firebreaks did not allow the same number of sampling points as in the mow and burn treatment units. We used line-point intercept sampling along two 50-m transects placed systematically in all treatment units. The data collected from the firebreak plots are summarized and provided below. These data were not included or considered in chapters 1 and 2 because of the small size of the firebreaks and the reduced number of sampling plots.

Considering the firebreaks as a disked treatment, 10 sites contained 6 management treatment units: Planted burned (PB), Planted disked (PD), Planted mowed (PM), Seedbank burned (SB), Seedbank disked (SD), Seedbank mowed (SM), and tall fescue Control (CL).

Tillage (disking or tilling) reduced coverage of total grasses and native warm-season grasses (Table B.1) when compared to burn and mow treatments, which is similar to Gruchy and Harper (2014). Burning and mowing promoted greater forb coverage than disking, likely because disked areas had less total plant coverage and more bare ground. Bobwhite food plant coverage was similar in all management units. Similarly, deer forage plant coverage tended to be similar in SD, SM, PB, PD, and PM, but deer forage plant coverage tended to be greater in SB than SD and PD (Table B.2). Additionally, diskling reduced coverage of semi-woody plants (brambles and vines) when compared to mowing and burning. Woody coverage was similar in all treatments. In 2019, deer forage biomass (kg/ha) was similar in all management units, but greater than CL. In 2020, deer forage biomass in disked units was less than in 2019 and less than burned or mowed management units.

APPENDIX B: LITERATURE CITED

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Table B.1. Mean coverage (%) and standard error (SE) of plant groups detected in 7 early successional plant community treatments across study sites (n = 10) in Tennessee and Alabama, USA, June–August 2019 and 2020.

	Treatment ^A													
	CL		PB		PD		PM		SB		SD		SM	
life form	x	SE	x	SE	x	SE	x	SE	x	SE	x	SE	x	SE
2019														
Grass ^A	99.0	0.4	69.1	5.5	20.2	2.6	66.2	7.3	58.2	6.7	16.8	4.5	60.8	6.5
NWSG ^B	19.1	6.3	43.9	6.9	9.1	2.8	40.8	7.6	39.3	7.2	7.0	2.9	40.1	8.0
NNG ^C	87.9	5.3	29.2	7.0	10.9	3.5	27.2	6.6	18.9	5.8	9.3	3.8	18.1	6.4
Forb ^D	8.3	4.9	61.8	5.0	36.5	3.0	56.4	6.9	73.9	4.5	41.1	2.0	60.8	6.7
Semi-woody ^E	1.2	0.7	19.9	5.4	4.0	1.1	16.0	4.9	17.8	4.4	4.9	1.7	21.5	5.6
Woody ^F	0.3	0.1	2.6	0.8	1.1	1.0	4.2	1.4	5.8	3.2	0.1	0.1	5.7	2.3
BWFP ^G	1.7	0.7	31.9	7.3	21.1	4.8	24.1	5.8	41.2	8.5	26.4	5.4	36.6	8.7
DFP ^H	1.9	0.9	49.1	6.9	29.8	4.3	41.1	7.2	64.7	7.0	33.9	4.2	58.3	6.6
2020														
Grass	99.8	0.1	81.9	4.2	27.2	3.9	89.5	3.6	66.0	2.7	18.5	4.3	71.5	6.0
NWSG	9.6	3.4	70.1	7.3	12.4	3.9	78.7	5.3	47.7	4.6	7.9	3.4	51.5	7.1
NNG	98.6	0.8	31.0	7.4	18.6	4.5	27.0	5.5	16.2	3.6	10.2	4.3	19.0	5.4
Forb	8.0	2.1	48.8	6.7	32.5	3.6	41.3	5.9	80.2	4.7	41.0	3.6	57.6	5.8
Semi-woody	1.6	0.8	15.4	4.0	5.2	1.5	16.8	3.9	22.7	5.0	6.5	2.2	21.5	4.2
Woody	2.1	1.2	2.5	0.8	1.8	0.9	4.8	1.3	5.9	1.8	0.1	0.1	6.3	1.7
BWFP	8.6	2.4	34.7	6.0	25.8	4.4	27.3	3.8	55.3	5.7	32.2	5.3	41.1	4.4
DFP	5.8	2.0	44.3	5.8	30.4	4.4	40.4	7.1	76.9	3.6	37.2	4.2	53.5	5.4

^A Percent coverage of all grasses present in each treatment

^B Percent coverage of native warm-season grasses

^C Percent coverage of nonnative grasses

^D Percent coverage of forbs

^E Percent coverage of semi-woody includes brambles and vines

^F Percent coverage of trees and shrubs

^G Percent coverage of bobwhite food plants

^H Percent coverage of deer forage plants

Table B.2. Mean $\pm$ SE deer forage biomass (kg/ha) by treatment for study sites (n = 10) in Tennessee and Alabama, USA, June–August 2019 and 2020.

Treatment	2019	2020
CL	33 $\pm$ 15	61 $\pm$ 22
PB	602 $\pm$ 129	451 $\pm$ 98
PD	558 $\pm$ 123	158 $\pm$ 37
PM	595 $\pm$ 220	536 $\pm$ 132
SB	536 $\pm$ 147	629 $\pm$ 108
SD	427 $\pm$ 100	159 $\pm$ 41
SM	558 $\pm$ 87	546 $\pm$ 96

**CHAPTER II. CHANGES IN PLANT COMPOSITION FOLLOWING DISTURBANCE
IN EARLY SUCCESSIONAL COMMUNITIES**

ABSTRACT

Restoration of native early successional plant communities in the eastern United States has been a conservation focus for more than two decades. Early successional communities provide components of habitat for many wildlife species and perform ecosystem services that enhance soil and water quality. Following restoration, disturbance is necessary to prevent these communities from progressing into woody-dominated mid-successional communities, and such management often changes community composition. We evaluated the effects of prescribed fire and mechanical disturbance following restoration of native plant communities via 2 methods (1. planting native grasses and forbs and 2. seedbank response without planting), across 11 replicated sites in Tennessee and Alabama, 2018-20. Specifically, we evaluated how disturbance altered vegetation composition following 4 treatment combinations (Planted mowed [PM], Planted burned [PB], Seedbank mowed [SM], Seedbank burned [SB]), and tall fescue (*Schedonorus arundinaceus*) Control (CL) from pre-disturbance conditions. Grass coverage increased in all treatment units, but tended to increase more in mowed treatments than burned treatments. Forb coverage declined in all treatments except SB where it increased. Similarly, spring-, summer-, and fall-flowering forbs, which are a focus of conservation programs designed to enhance habitat for pollinators, increased most in SB. Semi-woody and woody plant species increased more in treatment units than CL, but coverage of woody species remained low. Our results provide insight into how disturbance management techniques may alter plant community composition soon after restoration, and we recommend managers use fire over mowing if increased forb coverage is important in helping meet their management objectives. Furthermore, our results highlight how planting native grasses and forbs is not necessary to restore native early

successional plant communities on many sites dominated by nonnative grass in the eastern United States.

KEY WORDS disturbance management, early successional plant communities, mowing, prescribed fire, seedbank response, succession, vegetation composition

Native early successional plant communities have been in decline across the eastern United States for many years (Brennan 1991, Noss et al. 1995, Noss 2013, Keyser et al. 2019). Approximately 5.7 million hectares of early successional plant communities were lost to land development in the United States from 1982–2015 (USDA 2018). As a result, many wildlife species associated with early successional plant communities have experienced population declines (Brennan 1991, Knopf 1994, Hunter et al. 2001, Brennan and Kuvlesky Jr. 2005, USDA 2009). Federal and state initiatives have increased restoration of these plant communities on private and public land across the eastern United States with the goal of providing components of habitat for various wildlife species, maintaining a diverse native plant community, and improving soil and water quality. More specifically, the USDA-Natural Resource Conservation Service (NRCS) offers cost-share programs that provide technical and financial assistance for conservation of early successional communities on private lands (Heard et al. 2000, USDA 2016). Additionally, early-succession management has been included in many State Wildlife Action Plans (Tennessee 2015, Kentucky 2013, North Carolina 2005, Georgia 2015), which guide wildlife management practices on state-owned lands.

Early successional communities can be restored or established using different methods, and numerous studies have evaluated restoration techniques. Restoration of these communities

typically involves converting a plant community dominated by nonnative grasses and forbs to native plants, and control of nonnative grasses is common (Madison et al. 1995, Washburn et al. 2000, Harper and Gruchy 2009, Hall et al. 2012, GeFellers et al. 2020). Planting native grasses and forbs following control of nonnative species is ubiquitously recommended and practiced (Barnes 2004, Burger 2005, Mittelhauser et al. 2005, Wortley et al. 2013). Restoration via planting commonly costs \$450 to \$900/ha, and seeding mixtures designed specifically for pollinators can cost \$2500/ha or more (Monroe et al. 2017, Williams and Lonsdorf 2017, GeFellers et al. 2020). However, recent research suggests restoration using selective herbicide applications along with the seedbank response is just as or more effective as planting and with considerable cost savings (Harper et al. 2021, GeFellers et al. 2020).

Following restoration, disturbance is necessary to maintain early successional plant communities. The type of disturbance, as well as the frequency, intensity, and timing of disturbance, may affect vegetation composition (Fynn et al. 2004, MacDougall and Turkington 2007, Gruchy and Harper 2014, Harper 2017). Different disturbance regimes are used to benefit various wildlife species. Prescribed fire is commonly promoted to maintain early successional communities, consume thatch, and stimulate the seedbank (Buckner and Landers 1979, Gruchy and Harper 2014). Mowing is a common method for maintaining openings, but mowing can lead to accumulated thatch and reduced plant species richness, which may have negative effects on habitat quality for some wildlife species (Dykes 2005, Golden et al. 2012, Harper 2017, Gudlin et al. 2019).

Previous research has evaluated the effects of management on early successional communities restored via planting, but no research has compared vegetation change following disturbance on sites that were planted to those restored using the seedbank only (McCoy et al.

2001, Greenfield et al. 2003, Gruchy and Harper 2014). Composition of planted communities may be different from those comprised of seedbank response only (GeFellers et al. 2020), and disturbance thus may further lead to differential plant community composition. Additionally, the type of disturbance may influence plant composition (Gruchy and Harper 2014), and certain restoration/disturbance combinations may promote different plant communities and alter plant species richness, evenness, and diversity, which can affect resource availability for wildlife, including availability of pollen and nectar resources (McCoy et al. 2001, Van Nuland et al. 2013, Halbritter et al. 2015). The evaluation of plant composition change following different disturbances in plant communities restored using planting or seedbank response can provide managers with valuable insight that could help them reach their management objectives.

We implemented a field experiment across Tennessee and north Alabama to assess the effects of common field management practices on plant composition in fields previously dominated by tall fescue (*Schedonorus arundinaceus*) and restored by planting a mixture of native forbs and grasses or restored by allowing the seedbank to respond without planting and instead using strategic herbicide applications to control undesirable plant species (GeFellers et al. 2020). Our research objective was to compare the effects of burning and mowing on vegetation composition in early successional communities that had been established using the 2 methods. We hypothesized burning would stimulate increased forb coverage, including spring-, summer-, and fall-flowering forbs important to pollinators, because burning consumes the litter layer and stimulates germination of the seedbank (Buckner and Landers 1979, Harper 2007, Gruchy and Harper 2014). We hypothesized mowing would lead to increased grass coverage because many perennial grasses spread by tillering or with increased bunch size and do not rely necessarily on germination from the seedbank. We hypothesized that regardless of disturbance type, seedbank

response units would have greater coverage of native forbs and grasses and less coverage of nonnative forbs and grasses than planted units because herbicide applications are limited in planted units because of potential harm to planted species. Lastly, we hypothesized that burning and mowing would result in similar amounts of woody and semi-woody (including brambles and vines) species because those plants typically resprout following fire and mowing unless set-back more than once during the growing season.

STUDY AREA

We conducted our study at 11 sites in Tennessee and Alabama (Figure C.1). Each site was represented by a 0.8 to 2.0-ha field dominated by tall fescue. Six of the study sites were on Tennessee Valley Authority (TVA) property in Bedford, Hamblen, Jefferson, Monroe, and Sevier Counties, TN, and Franklin County, AL. Three study sites were on Tennessee Wildlife Resources Agency (TWRA) property in Cocke, Cumberland, and White Counties, TN. One study site was in Cades Cove within the Great Smoky Mountains National Park in Blount County, TN. Another study site was on Alabama Department of Conservation and Natural Resources (ADCNR) property in Jackson, AL. Our study was initiated in 2015 by partitioning each field into 3 equally-sized treatment units (Planted, Seedbank, and tall fescue Control). Treatment units varied in size from 0.8 to 2.0 ha. Tall fescue was controlled with an application of glyphosate at all study areas in November 2015 prior to initiating 2 establishment treatments (Planted and Seedbank) in spring 2016 as described by GeFellers (2019). Evaluation of the establishment techniques was described by GeFellers et al. (2020) and Harper et al. (2021). Elevation at study sites ranged from 180 m at the Franklin County, AL site to 658 m above mean sea level at the Cumberland County, TN site. Soils at 10 of the 11 sites were loam or silt/loam,

whereas one site (Jackson County, Alabama) had silt clay (Soil Survey Staff 2021). All fields had been maintained in an open condition by mowing or haying for at least 15 years prior to the study.

METHODS

We divided the Planted and Seedbank units at each site into two equal-sized units and randomly assigned mowing or burning to each in the spring of 2019. This approach created four treatment units that varied from 0.36 to 0.95 ha at each of the 11 sites: Planted mow (PM), Planted burn (PB), Seedbank mow (SM), and Seedbank burn (SB). We maintained the tall fescue-dominated control (CL) at each site with an annual late-winter mowing, which is consistent with how idle fields dominated by tall fescue are maintained in the region (Dykes 2005).

Burning treatments

Location managers installed firebreaks around the appropriate units and implemented prescribed fire in those units at each site. We used backing fires at each site to establish a blackened area on the downwind side of the burn unit adjacent to the firebreak, and if conditions allowed, backing fires were used to burn the entire unit. Flanking and heading fires were used when conditions did not allow backing fires to consume fuels. All fires were considered relatively low-intensity with flame lengths averaging 0.8 m. We conducted all burns at each site in 2019 and 2020 during the late dormant season (February-early April), which is consistent with when most prescribed fire is implemented to maintain fields enrolled in conservation programs (Dryden 2001, Harper et al. 2007). All burns were conducted within the following prescription parameters: relative humidity 20–50%, wind speed 0–16 km/h, temperature -1–24 °C, and cloud cover <50%.

Mowing treatments

Location managers or contractors mowed the appropriate units at each site in 2019 and 2020 using tractor-mounted rotary mower during the dormant season. All units were mowed to a height of approximately 25 cm.

Herbicide treatments

Throughout the study, we used selective herbicide applications to control nonnative species in all treatment units. We used 15-L backpack sprayers with wands (Solo USA, Newport News, Virginia) and a 95-L ATV sprayer (Cabelas, Sydney, Nebraska equipped with a spray gun (Green Garde®, H.D. Hudson Manufacturing Company, Chicago, Illinois) to make spot-spray applications in units to reduce coverage of undesirable species (i.e., nonnative invasive plant species and 2 native species, blackberry [*Rubus* spp.] and black locust [*Robinia pseudoacacia*], if coverage exceeded 30% to remain in compliance with recommendations from state agency Private Lands biologists with the Alabama Department of Conservation and Natural Resources (ACDNR) and the Tennessee Wildlife Resources Agency (TWRA) and NRCS biologists for conservation programs). We conducted spot-spray applications once during spring and once during summer to control undesirable warm-season species, such as johnsongrass (*Sorghum halepense*), sericea lespedeza (*Lespedeza cuneata*), or bermudagrass (*Cynodon dactylon*). We conducted one fall/winter application to control undesirable cool-season species, such as common henbit (*Lamium amplexicaule*) or purple deadnettle (*Lamium purpureum*), if needed. We used spot-spray applications with selective herbicides in PL units only in accordance with recommendations of Private Lands Biologists with ADCNR and TWRA to remain in compliance with what is permitted and recommended to landowners enrolled in state conservation programs. We used selective herbicides when possible to control nonnative invasive species and protect

species that were planted. Such applications limited control of some species, such as bermudagrass or sericea lespedeza, that cannot be killed without damaging planted species. We controlled nonnative invasive species with spot-spray applications in Seedbank units whenever they occurred, regardless of percent coverage. We used triclopyr (0.10 kg ai/ha) and fluroxypyr (0.31 kg ai/ha) and an ATV sprayer in May 2019 to spot-spray CL units to control woody encroachment and maintain a tall fescue control. No additional herbicide treatment was used in CL units.

Data Collection

We recorded vegetation composition in each treatment unit and CL mid-June-early August 2020. We used line-point intercept sampling along 4 50-m transects in all treatment units to obtain vegetation coverage. Each transect was systematically placed equal distance from one another and each transect was at least 10 m from the edge of each unit. Each transect had 25 sampling points, and we recorded vegetation to species at 2-m intervals. We calculated percent coverage of various plants and plant groups by dividing the total number of detections on each transect by the number of sampling points on the transect. We then calculated the average species or plant group coverage across each unit by averaging the percent coverage estimates of all the transects in that unit. We used the USDA plants website (plants.usda.gov) to divide plant species into 4 plant groups and to further divide plant groups into other categories, such as native or nonnative. Our 4 main plant groups were grass, forb, semi-woody, and woody. Grass included all grass species detected. Forb included all broadleaf herbaceous species detected. Semi-woody included all bramble and vine species detected. Woody included all tree and shrub species detected. We further divided grass into native and nonnative grasses, and cool-season (CSG) or warm-season grasses (WSG). We further divided forb into native and nonnative forbs and spring-, summer-,

and fall-flowering native forbs. Flowering season was determined for each species using the USDA plants website. We also divided plants into a native species group and nonnative species group, which included grasses, forbs, semi-woody, and woody species.

We calculated Simpson's *E* index and Shannon-Weiner index values for each treatment at each site to determine average plant species evenness and diversity. Simpson's *E* index indicates how evenly abundance is distributed among species, whereas Shannon-Weiner index evaluates species richness and evenness to calculate a diversity score. The maximum value for Simpson's *E* index is one and values nearer one represents greater evenness in the plant community. Values for Shannon-Weiner index range from zero to four and greater values represent greater plant diversity.

DATA ANALYSIS

We used coverage data collected in 2018 in the same fields prior to disturbance (prescribed fire or mowing) to calculate percent difference in plant group coverage and assess how each disturbance practice affected the plant community. We subtracted percent coverage of plant groups in 2018 from the percent coverage of plant groups in 2020 to calculate percent difference in each treatment following disturbance. We used percent change for species richness, Simpson's evenness, and Shannon's Diversity indices because these metrics were not already in percentages. We calculated percent change by subtracting means from 2018 by means from 2020 and then dividing that calculation by the original (2018) mean and multiplying by 100. The data from 2018 were collected during the same time period (mid-June-early August) in the same fields using the same sampling protocol as used by GeFellers et al. (2020).

We used a randomized block study design with treatment unit serving as the experimental unit. The response for plant groups was the difference between averaged measurements from 2018 and 2020 in each treatment unit. The response for species richness, Simpson's evenness, and Shannon's diversity was the percent change between averaged measurements from 2018 and 2020 in each treatment unit. We conducted analysis of variance (ANOVA) in program R (R Core Team 2020) to determine the effects of treatment on vegetation composition and diversity indices; if treatment effects were significant, we used package "emmeans" (Lenth 2018) to compare means using Tukey's Honest Significant Difference (HSD) p-value adjustment. We used a significance level of $\alpha=0.05$ for all contrasts.

RESULTS

Grass coverage did not differ by treatment following disturbance from 2018 to 2020 ($F = 0.7$, $P = 0.55$). Total grass coverage increased in all treatments, though mowing tended to increase total grass coverage more than burning (Table C.1). Total grass coverage increased 18.9% and 12.3% in SM and PM and 10.5% and 2.7% in SB and PB, respectively.

Change in coverage of total native grasses differed by treatment ($F = 8.96$, $P < 0.0001$). The change in coverage of total native grasses was greater in all management treatments than CL ($P < 0.02$), but all management treatments were similar ($P > 0.3$). Change in coverage of total nonnative grasses did not differ by treatment ($F = 0.3$, $P = 0.86$); however, coverage of total nonnative grasses averaged nearly 70% greater in Planted treatments than Seedbank treatments.

Change of total warm-season grass coverage differed by treatment ($F = 14.69$, $P < 0.0001$). Total warm-season grasses increased in all management treatments similarly ($P > 0.78$),

but decreased in CL. We detected treatment effects for difference in coverage of native warm-season grass ($F = 12.7$, $P < 0.0001$) and all management treatments increased the coverage of native warm-season grass at a greater rate than CL. All management treatments influenced the change in native warm-season grass coverage similarly ($P > 0.65$), but mowing tended to increase coverage of native warm-season grass more than burning. Native warm-season grass increased 18.1% and 26.1% in SM and PM, and 10.7% and 13.8% in SB and PB, respectively.

Change in nonnative cool-season grass coverage differed by treatment ($F = 5.07$, $P < 0.001$) and SB ($P < 0.002$) and PB ($P < 0.007$) reduced nonnative cool-season grass coverage at a greater rate than CL, but change in nonnative cool-season grasses was similar between CL, SM ($P > 0.06$), and PM ($P > 0.06$). All management treatments reduced coverage of nonnative cool-season grasses similarly ($P > 0.65$), but burning tended to reduce them more than mowing.

Change in total forb coverage differed by treatment ($F = 8.31$, $P < 0.0001$) following disturbance from 2018 to 2020. Forb coverage decreased in SM, PB, and PM (-14%, -18%, and -30%, respectively), but forb coverage increased 15% in SB. Change in coverage of native forbs differed by treatment ($F = 13.35$, $P < 0.0001$), which decreased similarly ($P > 0.15$) among SM, PB, and PM (-4.6%, -16.7%, and -28.7%, respectively), but increased 45.7% in SB. Change in nonnative forb coverage did not differ by treatment ($F = 0.74$, $P = 0.567$), and coverage of nonnative forbs was similar ($P > 0.65$).

Change in coverage of spring- ($F = 8.72$, $P < 0.0001$), summer- ($F = 18.08$, $P < 0.0001$), and fall-flowering forbs ($F = 11.14$, $P < 0.0001$) differed by treatment. Coverage of spring-flowering forbs increased more in SB than PB, PM, and CL ($P < 0.002$) (Figure C.2). Coverage of summer-flowering forbs increased more in SB, SM, and PB than CL ($P < 0.0022$). The increase of summer-flowering forbs in PM did not differ from CL ($P = 0.14$). Coverage of

summer-flowering forbs increased most in SB, and the increase was greater ($P < 0.03$) in SB than SM, PB, and PM. Coverage of fall-flowering forbs increased more in SB, SM, and PB than CL ($P < 0.05$). Coverage of fall-flowering forbs increased more ($P < 0.006$) in SB than SM, PM, and CL.

Change in coverage of semi-woody species differed by treatment ($F = 4.29$, $P < 0.001$). Coverage of semi-woody plants increased more in SB and SM than CL ($P < 0.012$), but change of coverage of semi-woody plants was similar among all management treatments ($P > 0.36$). Coverage of woody species did not vary among treatments ($F = 1.86$, $P = 0.13$) following disturbance.

Change in species richness differed by treatment ($F = 13.53$, $P < 0.0001$) following disturbance from 2018 to 2020. Species richness was reduced similarly ($P > 0.7$) in SM, PB, PM, and CL by -11.57%, -23.85%, -18.29%, and -63.43%, respectively, but increased 8.39% in SB. All treatments had less negative effects on species richness than CL ($P < 0.003$). We detected treatment effects for Shannon-Weiner index ($F = 17.84$, $P < 0.0001$). Diversity decreased similarly ($P > 0.32$) in all treatments (SB -0.7%, SM -3.14%, PB -15.37%, PM -7.93%), but the decrease was greater ($P < 0.0001$) in CL (-56.53%). Simpson's evenness index did not differ by treatment ($F = 0.63$, $P < 0.63$), but mowing tended to increase evenness more than burning.

DISCUSSION

Our study compared the effects of mowing and burning on the trajectory of composition in restored early successional plant communities. Burning led to increased native forb coverage in communities responding from the seedbank, but not in areas that were restored by planting native forbs and grasses, which partially supported our hypothesis that burning would increase

forb coverage. Burning also resulted in increased coverage of spring-, summer-, and fall-flowering forbs in both seedbank response and planted communities, but the increase in coverage was much greater in seedbank response units. Notably, neither burning nor mowing increased coverage of planted forbs in planted units. Grass coverage increased in all treatment units, but mowing tended to increase grass coverage at a greater rate than burning. Plant communities restored using planting and seedbank restoration often respond differently to disturbance; hence, restoration method and disturbance type should be considered in efforts to meet management objectives. Additionally, our study clearly demonstrated that planting is not necessary to establish and maintain native early successional plant communities on a majority of sites, regardless of the type of disturbance implemented.

Total grass coverage increased at a similar rate in all treatment units, consistent with other studies that used dormant- and early growing-season fire to maintain early successional plant communities (Whitehead and McConnell 1980, Manley 1994, Brockway et al. 2001, Gruchy and Harper 2014). In 2018, grass coverage was greater in Planted units than in Seedbank units, and grass coverage remained greater in Planted units following disturbance. In fact, grass coverage in all treatments was in excess of what is selected by most wildlife that use early successional communities (Herkert 1994, Granfors et al. 1996, Warren and Anderson 2005, Unger et al. 2015, Brooke et al. 2016). The coverage of native grass that was maintained in the Seedbank units (approximately 50%) substantiated that planting native grasses is not necessary when restoring native plant communities on most sites in the eastern US where tall fescue occurs. In fact, following planting, native grasses commonly become too dense for management objectives within 2 – 3 years, necessitating management to decrease grass coverage and increase forb coverage and overall plant diversity (Gruchy and Harper 2014, Brooke and Harper 2016).

The occurrence of nonnative plant species, especially nonnative grasses, is a common management concern when restoring and managing early successional plant communities, and periodic disturbance can favor establishment of nonnative species (Kuebbing et al. 2014). In summer 2020, following the initial application of glyphosate in autumn of 2015 to kill tall fescue in the treatment units, nonnative cool-season grass coverage was only 1 – 6.5%, indicating a single fall herbicide application is very effective at controlling this nonnative perennial cool-season grass (GeFellers et al. 2020). Although the reduction in total nonnative grasses was similar among treatments, coverage of nonnative grasses remained greater in Planted units than in Seedbank units, which is attributed to the inability to spot-treat various warm-season grasses (such as bermudagrass) in Planted treatments because herbicide treatment would also kill planted grass species.

Mowing should be used only when prescribed fire is not available as a management tool. A desirable forb component usually is an objective when managing early successional plant communities (Harper 2017, Meissen et al. 2019). Forbs provide forage, seed, cover, and nectar for a wide variety of wildlife species (Robel 1963, Healy 1985, Steffan-Dewenter and Tschardt 2001, Lashley et al. 2011, Nanney et al. 2018). In Seedbank and Planted units, mowing reduced forb coverage whereas burning increased forb coverage in Seedbank units. Forb coverage increased only in the SB treatment, including a 45.7% increase in coverage of native forbs. Forb coverage did not increase in PB, likely because there was nearly 80% grass coverage in PB by 2020. Grman et al. (2020) also reported that planted native grasses suppressed forb coverage. Mowing in PM led to nearly 90% grass coverage by 2020. Mowing typically promotes thatch build-up (Gruchy 2007) and could further suppress forb response in both SM and PM.

The combination of seedbank restoration and burning increased coverage of spring-, summer-, and fall-flowering forbs at a greater rate than any other restoration/management combination, providing pollinators with more food and nest structure resources. Many species of pollinators are dependent on early successional plant communities, and forb coverage is critical for pollinator food resources (Ginsberg 1983, Teer 1996, Althoff et al. 1997, Hunter et al. 2001, Wilkerson et al. 2014). Greater coverage of forbs provides more pollen and nectar resources to a wide range of insect pollinators, but the availability of pollen and nectar resources throughout the growing season should be considered when restoration focuses on pollinators (Steffan-Dewenter and Tscharntke 2001). Insect pollinators also require nest sites, and many species nest in forb stems (Black et al. 2011).

A variety of plant species with relatively even distribution can be an important consideration when managing early successional plant communities as increased species diversity and evenness may provide more food or cover resources with better distribution for wildlife through the year with different timing of plant phenology (Levine and D'Antonio 1999, Wilsey and Potvin 2000, Steffan-Dewenter and Tscharntke 2001, Tracy and Sanderson 2004, Fontaine et al. 2005). Plant species richness has been linked to more diverse insect populations as well as increased nutritional carrying capacity for species such as white-tailed deer (Harper et al. 2021, Knops et al. 1999). Although relatively few private landowners may be interested in plant species richness or insect populations, per se, we stress how management of early successional communities for greater plant diversity is important for deer because more private landowners manage their land for deer than any other species (McShea 2012). As private landowners learn the value in promoting increased diversity of native plants, enhanced habitat for pollinators and their populations may result by default. Species evenness increased following

all treatments, meaning the plant communities following burning and mowing had a more equal abundance of each species present in the plant community. Evenness may be especially important for pollinators that rely on floral resources in Spring, Summer, and Fall. However, we documented an increase in species richness only in SB where litter was consumed, allowing the seedbank to germinate, and reduce grass dominance. There was a negative trend in species diversity following all treatments, but SB had the least negative change with <1% change. The combination of seedbank restoration and burning may help with plant community persistence, productivity, and overall function by increasing species richness and evenness.

There were no trends in change of coverage of semi-woody or woody plants. Coverage of both semi-woody and woody species were relatively low prior to treatment, but after two years of management, semi-woody coverage increased to approximately 20% in Seedbank treatments. Presence of various semi-woody plants, such as blackberry and northern dewberry (*Rubus flagellaris*), provides additional food and cover resources for many wildlife species (Badyaev 1995, Moore et al. 2010, Nanney et al. 2018). Although we primarily used low-intensity backing fire in the burn treatments, fire intensity was sufficient to at least top-kill semi-woody or woody plant species. In both fire and mow treatments, semi-woody and woody species continued to resprout and persist. These results are consistent with other studies that report dormant-season fire or annual mowing may control but not eradicate woody species (Drewa et al. 2002, Dykes 2005, Gruchy et al. 2009, Robertson and Hmielowski 2014)

MANAGEMENT IMPLICATIONS

Disturbance is required to maintain early successional communities in the eastern US, regardless of establishment technique. After documenting the change in vegetation composition following

the two most common disturbance practices used to manage fields and other open areas in the eastern US, we recommend managers consider using prescribed fire over mowing if increased forb coverage is important in helping meet their management objectives. Forb coverage is important with regard to forage for white-tailed deer (*Odocoileus virginianus*), nectar for pollinators, seed production for birds, and cover for gamebird broods and many songbird species. Where increased grass coverage is desirable, planting and mowing can be used to achieve management goals. However, few if any management objectives would warrant more grass coverage than what we documented following establishment from the seedbank and prescribed fire. Although we used fire and mowing 2 years in succession, the frequency and timing (season of burning) of disturbance may be altered depending on management objectives and plant community response. Managers should consider using the seedbank response instead of planting when restoring native plant communities following eradication of nonnative grasses where they dominate the site because flexibility in herbicide use is an important consideration when many herbicides required to control undesirable plants also would kill planted species.

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APPENDIX C: CHAPTER II TABLES AND FIGURES

Table C.1. Mean percent (%) coverage and standard error of all species groups detected in 5 early successional plant community treatments across all study sites (n = 11) in Tennessee and Alabama, USA.

	Treatment																			
	PM				PB				SM				SB				CL			
life form	2018	SE ^A	2020	SE	2018	SE	2020	SE	2018	SE	2020	SE	2018	SE	2020	SE	2018	SE	2020	SE
Grass ^B	78.6	5.3	88.2	3.2	78.6	5.3	80.7	4.2	60.8	9.2	72.2	5.8	60.8	9.2	67.2	2.8	95.4	2.2	98.0	1.8
NG ^C	61.3	6.7	69.1	5.5	61.3	6.7	56.7	6.6	46.9	9.4	54.6	7.0	46.9	9.4	50.7	5.0	35.5	8.4	9.7	3.2
NNG ^D	21.1	6.9	26.0	5.4	21.1	6.9	29.9	7.1	12.0	2.8	17.7	5.3	12.0	2.8	15.4	3.5	86.5	4.0	96.4	2.3
NNCSG ^E	5.3	2.2	5.0	2.8	5.3	2.2	1.4	0.4	6.7	2.0	6.5	3.1	6.7	2.0	1.0	0.4	85.0	3.7	95.8	2.3
NWSG ^F	61.0	6.6	77.0	5.2	61.0	6.6	69.4	6.9	45.1	9.2	53.4	7.0	45.1	9.2	50.0	5.0	33.8	8.5	9.8	3.2
CSG ^G	5.3	2.2	17.6	2.8	5.3	2.2	18.1	0.6	6.8	2.5	9.2	3.7	6.8	2.5	2.4	1.4	85.0	2.9	95.7	2.3
WSG ^H	75.9	5.2	85.0	3.4	75.9	5.2	79.5	4.0	55.3	8.6	66.9	5.6	55.3	8.6	65.6	3.1	48.2	7.2	15.1	3.8
Forb ^I	64.3	6.7	44.6	6.6	64.3	6.7	52.3	7.2	71.1	3.5	60.9	6.4	71.1	3.5	81.9	4.8	40.6	7.3	12.1	4.6
NF ^J	48.7	7.8	34.7	6.1	48.7	7.8	40.5	6.6	51.6	5.7	49.2	6.4	51.6	5.7	75.2	5.2	26.5	7.6	9.4	4.2
NNF ^K	25.8	6.0	14.8	4.2	25.8	6.0	20.5	7.2	30.1	5.1	25.3	4.5	30.1	5.1	25.6	6.7	18.2	2.9	3.3	1.2
SPFF ^L	11.1	4.7	12.6	3.1	11.1	4.7	19.3	5.2	9.7	2.7	29.8	5.9	9.7	2.7	42.4	5.2	1.7	0.8	5.5	1.8
SUFF ^M	15.7	2.7	30.9	4.3	15.7	2.7	43.7	6.5	12.8	2.5	48.5	6.2	12.8	2.5	72.2	5.0	11.6	3.8	8.5	2.5
FAFF ^N	28.0	7.0	37.5	5.8	28.0	7.0	28.1	7.3	35.7	7.2	48.7	6.5	35.7	7.2	74.6	5.0	16.8	6.7	9.4	3.1
Semi-woody ^O	13.1	3.9	16.5	3.7	13.1	3.9	15.6	3.9	11.5	3.7	21.5	4.0	11.5	3.7	22.6	4.8	8.7	3.4	2.6	1.2
Woody ^P	0.5	0.1	4.5	1.3	0.5	0.1	2.3	0.8	0.2	0.1	5.8	1.7	0.2	0.1	5.4	1.8	0.4	0.2	2.3	1.2
NS ^Q	85.9	4.9	87.4	3.8	85.9	4.9	81.2	6.0	82.0	4.3	85.6	3.4	82.0	4.3	95.6	1.2	56.4	5.9	22.0	6.0
NNS ^R	41.9	7.0	38.6	6.2	41.9	7.0	45.6	8.0	39.7	5.8	38.4	6.4	39.7	5.8	39.5	6.3	90.9	2.9	96.8	1.8

^A Standard Error (SE) for percent coverage of plant groups

^B All grasses present in each treatment

- ^C NG consists of all native grasses present in each treatment
- ^D NNG consists of all non-native grasses present in each treatment
- ^E NNCSG consists of all non-native cool-season grasses present in each treatment
- ^F NWSG consists of all native warm-season grasses present in each treatment
- ^G CSG consists of all cool-season grasses in each treatment
- ^H WSG consists of all warm-season grasses in each treatment
- ^I All forbs present in each treatment
- ^J NF consists of all native forbs present in each treatment
- ^K NNF consists of all non-native forbs present in each treatment
- ^L SPFF (Spring-flowering forbs) consists of all forb species detected that flower in spring
- ^M SUFF (Summer-flowering forbs) consists of all forb species detected that flower in summer
- ^N FAFF (Fall-flowering forbs) consists of all forb species detected that flower in fall
- ^O Semi-woody consists of bramble and vine species
- ^P Woody consists of tree and shrub species
- ^Q NS consists of all native plant species present in each treatment
- ^R NNS consists of all nonnative plant species present in each treatment

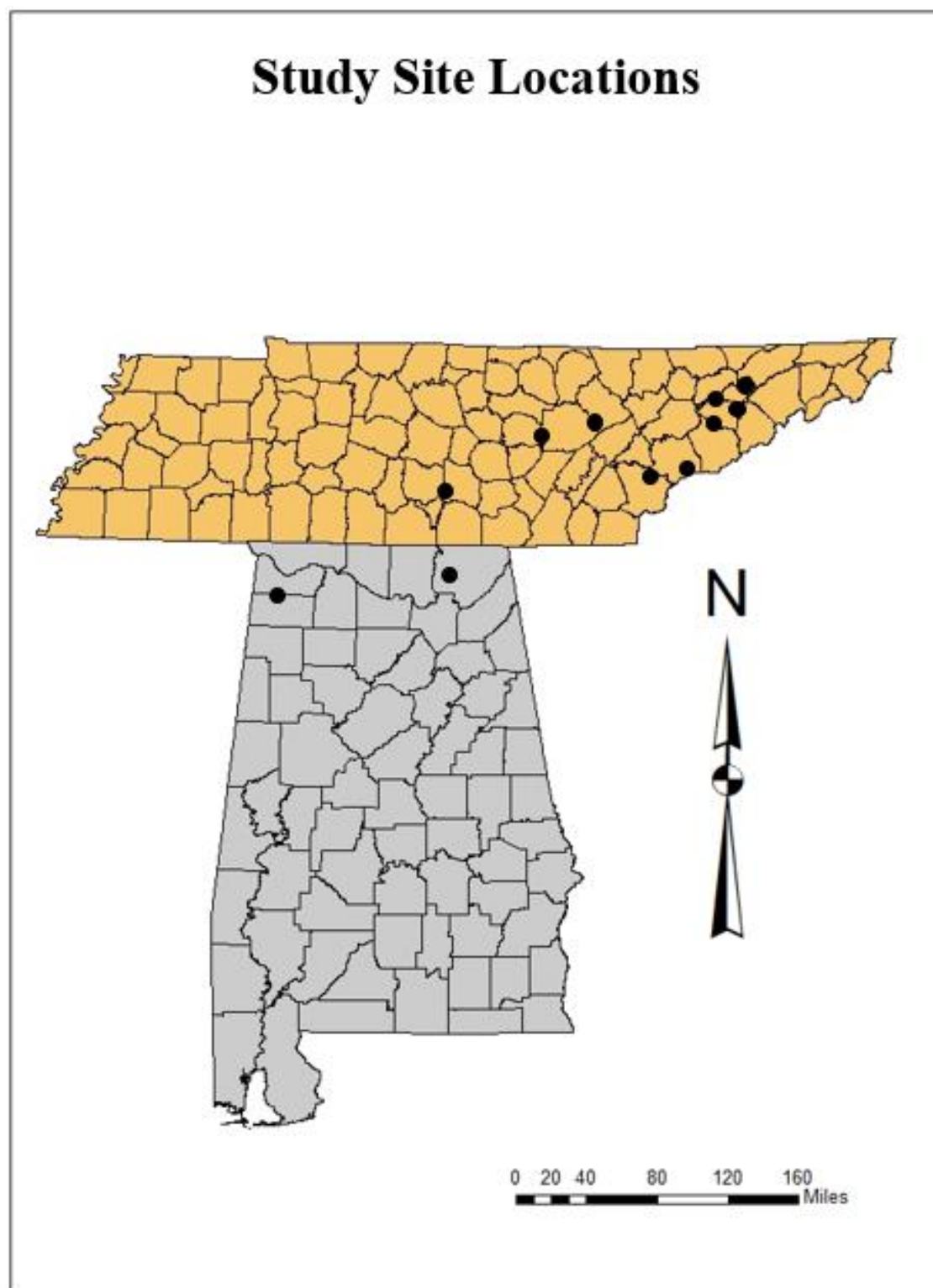


Figure C.1. Map of 11 study site locations in Tennessee and Alabama, USA (2018-2020)

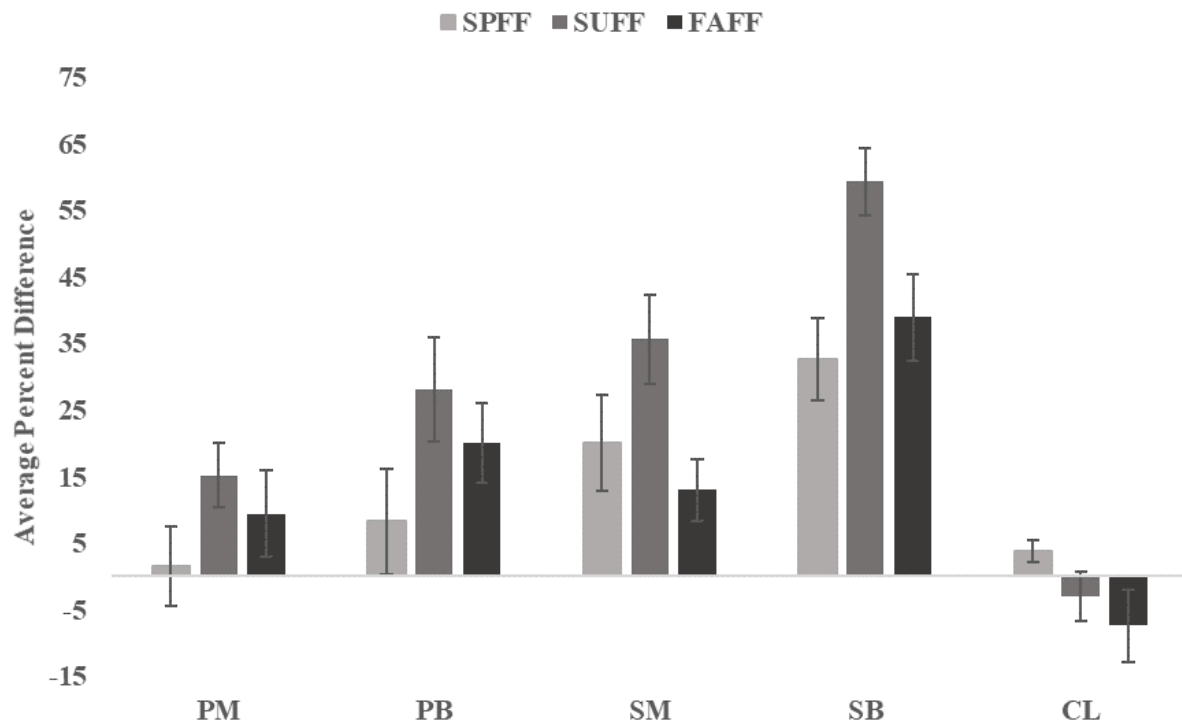


Figure C.2. Mean percent difference in spring- (SPFF), summer- (SUFF), and fall-flowering forb (FAFF) coverage during June–August 2018 and 2020 at all study sites (n=11) in Tennessee and Alabama following burn and mow treatments.

CONCLUSION

Restoration of early successional plant communities on both private and public lands has been prompted by population declines in many wildlife species commonly associated with early successional plant communities and these plant communities are being restored across the South. Disturbance is required to prevent early successional plant communities from entering a mid-successional stage dominated by woody sprouts and the type of disturbance has effects on vegetation composition. The type of disturbance managers and landowners use should be determined by the predicted response and their management objectives. However, few management objectives would warrant more grass coverage than what we documented following SB. Fire should be used to manage early successional communities when forb coverage is an objective and if managers are considering restoring early successional plant communities with wildlife species such as white-tailed deer, wild turkey, and northern bobwhite in mind the combination seedbank response/burning should be used. Strategic herbicide applications are essential to maintaining native early successional plant communities following disturbance. Flexibility in herbicide use is an important consideration when many herbicides required to control undesirable plants also would kill planted species. Although herbicide use is limited in areas planted to native warm-season grasses and forbs, nonnative plants that can be controlled should not be ignored. For example, johnsongrass can be killed with imazapic in planted areas without damaging planted species. However, there are no herbicides that will kill bermudagrass or sericea lespedeza without also killing planted species. Although we used fire and mowing two years in succession, the frequency and timing of disturbance may be altered depending on management objectives and plant community response. Finally, restoration and management of early successional plant communities, regardless of restoration/disturbance combination, takes

patience and commitment but these communities when managed correctly will be ecologically functional and provide components of habitat for many wildlife species.

VITA

Bonner L. Powell was born in Oxford, Mississippi. His life in Mississippi allowed him many opportunities to hunt and fish while also allowing him to explore and learn outdoors. Bonner attended Mississippi State University and received his Bachelor of Science degree in Wildlife, Fisheries, and Aquaculture Science. After completing his degree at Mississippi State University, he decided to pursue his Master of Science degree at the University of Tennessee.