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Effects of prescribed fire and selective herbicide (Imazapyr) on biodiversity in intensively managed pine stands of Mississippi

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EFFECTS OF PRESCRIBED FIRE AND SELECTIVE HERBICIDE (IMAZAPYR) ON BIODIVERSITY IN INTENSIVELY MANAGED PINE STANDS OF MISSISSIPPI

By

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Prescribed fire and imazapyr are two silviculture tools used to control hardwood midstory competition in intensively managed, mid-rotation pine (*Pinus* spp.) stands but also may support conservation of biodiversity in the southeastern United States. Therefore, I investigated select measures of biodiversity response, small mammals, reptiles, amphibians, carabid beetles, songbirds, and vegetation communities, to fire and imazapyr treatments in intensively managed, mid-rotation pine stands of east-central Mississippi. I used a randomized complete block design of 6 stands (blocks) with 4, 10-ha treatment plots assigned randomly a treatment of burn only, herbicide only, burn + herbicide, or control. I applied dormant season prescribed fires every 3 years beginning in January 2000 and a one-time application of imazapyr in September 1999 using 877 ml/ha (12.0 liquid oz./ac; Arsenal®, BASF 2006). I sampled avifauna, herpetofauna, small mammal, and carabid beetle communities using appropriate sampling techniques for attaining species-specific relative abundance. I also measured vegetation structure and biomass. Vegetation and bird communities exhibited significant responses to treatments. Imazapyr had the greatest initial impact on communities followed by a long-term effect of repeated prescribed fires on a 3 year fire-return interval. Combining fire and imazapyr perpetuated high-quality browse for white-tailed deer (*Odocoileus virginianus*), plant species richness, high-priority bird species relative abundances, and diversity of landscape-level vegetation structure and biomass by creating a two-tier vegetation structure (pine canopy and herbaceous understory). Independent treatments also were more effective management approaches to sustain biodiversity than controls by maintaining or increasing overall species richness specifically soon after treatment application. Most responses of other wildlife communities were time-limited suggesting the possibility of greater effects of factors other than treatments such as long-term disturbance regimes (e.g., forest management practices, climate trends), proximity of treatment plots to wetlands, and landscape-level population dynamics including characteristics within and among stands. Combined and independent applications of these treatments will support biodiversity conservation, sustainable forestry objectives, and concomitant timber management goals. Long-term conservation of biodiversity within an intensive timber management matrix also may benefit from future investigations of multiple-herbicide tank mixtures, population dynamics of indicator species, and landscape-level biodiversity responses across multiple strata.

Key Words: birds, carabid beetles, community response, fire, forest management, habitat management, herpetofauna, imazapyr, Mississippi, pine, *Pinus*, rodents, small mammals, vegetation.

DEDICATION

I dedicate this research to Katie and Luke, my wife and son.

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CHAPTER 1

VEGETATION RESPONSE TO FIRE AND IMAZAPYR APPLICATION IN INTENSIVELY MANAGED, MID-ROTATION PINE STANDS OF MISSISSIPPI

Historically, nearly 30 million ha of longleaf pine (*Pinus palustris*) forests covered the southeastern United States from the Gulf Coastal Plain to the eastern shore (Frost 1993). Other pine species, including slash (*P. elliotii*), loblolly (*P. taeda*), and short-leaf (*P. echinata*), followed a gradient of fire tolerance inland from the Lower to Upper Coastal Plains and into the Piedmont. Although lightning-ignited fires occurred, anthropogenic-caused fire was a main influence on southern landscapes (Komerak 1974, Hudson 1976, Keeley 1981, Gill 1981, Noss 1989, Landers et al. 1995, Masters et al. 1995). Early European settlers learned the value of fire from Native Americans for clearing areas of brush and managing wildlife habitat (Hudson 1976, Williams 1989, Pyne et al. 1996, Bonnicksen 2000, Carroll et al. 2002), reducing fuel loads (Williams 1989, Johnson and Hale 2002) and reducing human parasites (Bonnichsen et al. 1987). However, alterations to fire frequency regimes and extensive human settlement changed pine-dominated forests over time (Carroll et al. 2002, Stanturf et al. 2002), allowing hardwood encroachment into former pine savannahs. During the 1900s, fire management became fire suppression with anti-fire messages delivered by the American Forestry Association, U.S. Forest Service, and state forestry agencies (Forman and Godron 1986, Frost 1993, Pyne et al. 1996, Pyne 2001, Van Lear et al. 2005). With less fire on the landscape, fuel loads increased causing greater fire intensities when lightning or arson ignition occurred (Wade and Lunsford 1989, Johnson and Hale 2002). Pyric-adapted species faced increased competition as their competitiveness edge under frequent fire conditions was limited (Ware et al. 1993, Bond and Van Wilgen 1996, Engstrom et al. 2001, Masters et al. 2003). Some thin-barked hardwoods, with the advantage of shade tolerance, out-competed photo-phylic herbaceous plants and pine (Platt and Schwartz 1990, Ware et al. 1993, Engstrom et al. 2001). Abundant woody shrubs and vines inhibited herbaceous understory plants restricting many pyric species to patches of sunlight amid blankets of debris (Sousa 1984, Masters 1991, Masters et al. 1993). Eventually, extensive fire research established guidelines regarding fuel load limits, ignition patterns, and ideal climatic variables enabling managers to use fire as a management tool (Komerak 1967, Wade and Lunsford 1989, Bessie and Johnson 1995, Schimmel and Granstrom 1997). Fire soon became an important silvicultural tool in short-rotation intensive forestry, used to prepare seedbeds, control understory and hardwood competition, and enhance wildlife habitat (Chen et al. 1975, Landes 1975, Buckner 1981, Mobley and Balmer 1981, Waldrop et al. 1987, Van Lear 2000).

Prescribed fire, which is the controlled application of fire, occurs either during the dormant or growing seasons (Wade and Lunsford 1989). Dormant season prescribed fires (ambient temperature $\leq 16^{o}$ C) are preferred in intensive forest management with less risk of crown scorch, restricted pine growth, and climatic variability than growing season prescribed fires (Wade and Lunsford 1989, Robbins and Myers 1992). Consistency of climate conditions 1 to 3 days after a winter cold front producing 1.3-2.5 cm of rain usually coincide with prescribed fire guidelines (Wade and Lunsford 1998). Dormant season prescribed fires reduce fuel loads, alter above ground vegetation structure, and scarify seeds but may not kill roots (Wade and Lunsford 1989, Waldrop et al. 1992). Prolific re-sprouting from undamaged root stocks can be common post-burn, although past research debates if season of fire differentiates root kill efficacy (Komerak 1965, Waldrop et al. 1992, Sparks et al. 2002). Early growing season fires may be more effective for hardwood reduction if timing of fire application occurs when hardwoods are releasing stored nutrients into stems. However, greater flame height and length associated with growing season fires increases risk of crown scorch (Waldrop et al. 1992). Plant survival differs between seasons of fire by reducing cool- or warm-season plants with dormant or growing season burns, respectively (Towne and Owensby 1984, Hulbert 1988, Biondini et al. 1989, Howe 1994). Dormant and growing season burns promote post-burn herbaceous plant cover including pyric-adapted species of forbs and legumes (Hodgkins 1958, White et al. 1991, Robbins and Myers 1992, Masters et al. 1993, Sparks et al. 1998).

Fuel characteristics and climatic variables may be the ultimate influential factors of fire intensity despite season of burn (Bessie and Johnson 1995, Schimmel and Granstrom 1997, Sparks et al. 1999). Following prescriptions with recommended conditions (i.e., ambient temperature, wind speed, fuel moisture) helps reduce fire's stochastic nature (Wade and Lunsford 1989) but limit number of burning degree days. In addition, smoke management issues and liability concerns support the need for alternative silviculture tools when burning is infeasible, such as selective herbicides (Brennan et al. 1998, Burger et al. 1998, Haines et al. 2001, Wigley et al. 2002).

Use of selective herbicides is common in intensive forestry (Shepard et al. 2004, Wagner et al. 2004). Through competition control, selective herbicides benefit intensively managed pine much the same as burning without many of the issues associated with burning (Brennan et al. 1998, Wigley et al. 2002). Site preparation and mid-rotation applications have been shown to increase pine yield (Elwell 1967, Grano 1970, Smith and Schimdtling 1970, Glover et al. 1989, Miller et al. 1991, Borders and Bailey 2001, Siry 2002, McInnis et al. 2004, Wagner et al. 2004) and overall stand quality (White 1975, Blake et al. 1987, McComb and Hurst 1987, Stewart 1987, Freedman 1991, Miller and Zutter 1989, Miller and Miller 2004, McInnis et al. 2004) by reducing competition using herbaceous weed control or release (Shepard et al. 2004), direct injection of unwanted trees (Hawley 1929), and overall woody stem control (Klingman 1961). Selective herbicides can effectively control hardwoods, target-specific suites of plants including exotics and invasive plants (Williams 1997, Miller 2003, Shepard et al. 2004), and benefit a wide array of wildlife species (Wagner et al. 2004). However, herbicide effectiveness differs across soil conditions and chemicals used (Morrison and Meslow 1983, Miller and Witt 1990, Miller and Miller 1999, Miller and Miller 2004).

Selectivity of modern forest herbicides enables managers to reduce unwanted vegetation and avoid damage to preferred species (Guynn et al. 2004, Miller and Miller 2004). Herbicides enter plants through roots, stems, or leaves depending on herbicide type and application method (Kidd 1987, Bovey 2001). Active ingredients of each herbicide can dictate their effective range of controlled plant species. Considering all possible forest herbicide active ingredients, imazapyr (Arsenal; BASF Corporation, Research Triangle Park, North Carolina), sulfometuron (Oust), hexazinone (Velpar), glyphosate (Accord® and generic products; Dow AgroSciences, Indianapolis, Indiana), metsulfuron (Escort®; E. I. du Pont de Nemours and Company), and triclopy (Garlon, Dow AgroSciences, Indianapolis, Indiana) are primarily (90%) used in the southeastern United States (Shepard et al. 2004). However, hexazinone and metsulfuron do not control some exotic invasive plants such as Johnson (*Sorghum halepense*) and Bermuda (*Cynodon dactylon*) grasses. Tank mixtures can include $\geq$ 1 herbicide at different concentrations, broadening vegetation control (Shepard et al. 2004). However in mid-rotation pine, woody competition control is the primary purpose of herbicide application with the most common herbicides of Arsenal AC (imazapyr) or Arsenal AC + Escort ® (Shepard et al. 2004).

Imazapyr's selectivity is driven primarily by herbicide absorption, translocation, rate of metabolism, and site sensitivity to the herbicide (Shaner and Moorthy Mallipudi 1991). As with most herbicides, imazapyr can control a wide variety of plants if adequate amounts reach sites of action. It is absorbed through foliage and roots and translocated ultimately via phloem to meristemic tissues and organs where it inhibits acetohydroxyacid synthase (AHAS) enzyme, an enzyme responsible for the synthesis of branched amino acids such as leucine, isoleucine, and valine (Little and Shaner 1991). Of the imidazolinone herbicides, imazapyr is least absorbed by roots and generallyhas greater absorption through foliage (Little and Shaner 1991). Imazapyr, similar to other imidazolinone herbicides, can inhibit its own translocation by reducing the sink strength of meristematic regions when it inhibits AHAS (Little and Shaner 1991). Intolerant plants may avoid fatality by metabolizing imazapyr to a less toxic compound (Shaner and Moorthy Mallipudi 1991). Loblolly pine, commonly cultivated in intensive forestry within the southeastern U.S., constricts translocation from xylem to phloem with most absorbed imazapyr remaining in needles (Esau 1977, Minogue 1990).

Imazapyr's effectiveness and preference in intensively managed pine is due to its selective control of unwanted plants with minimal effects on crop trees especially at low-application rates typical of intensive forest management practices (e.g., *Pinus* spp.; BASF 2006, Shepard et al. 2004). For wildlife management purposes, effective removal of an established hardwood midstory without affecting overstory pine results in a two-tier vegetative structure similar to pine savannahs within the southeastern U.S. Coastal Plain. Grasses, forbs, semi-woody vines, and some legumes are favored ground coverage for many wildlife species that proliferate after herbicide application in mid-rotation, intensively managed pine (Thompson 2002, Iglay et al. 2010*a*). However, herbicide application may not be a replacement for fire as it lacks fuel reduction and seed scarification capabilities (Brennan et al. 1998). However, although herbicide application may cover an area more homogeneously than a prescribed fire creeping across the land, but because herbicide activity can vary by soil conditions and chemicals used (Morrison and Meslow 1983, Miller and Witt 1990, Miller and Miller 1999, Miller and Miller 2004), effects of vegetation communities will most likely also be heterogeneous across the land.

Prescribed fire and imazapyr have some similar effects on understory vegetation structure and diversity in intensively managed pine with resultant impacts to wildlife communities. Dormant season prescribed fire benefits many vertebrate and invertebrate wildlife species by altering vegetation structure (Howard et al. 1959, Bendell 1974, Grelen 1975, Lyon et al. 1978, Taylor 1981, Means and Campbell 1981, Landers 1987). Direct mortality tends to be greater for organisms with limited mobility or dispersal capabilities, such as invertebrates (Lamotte 1975, Lyon et al. 1978, Pippin and Nichols 1996). However, unburned patches and burrows across an area provide refuge, and direct mortality from fire is rare for most species assemblages (Bendall 1974, Speake et al. 1979, Means and Campbell 1981, Jackson and Milstrey 1989, Lips 1991). Similar to fire, imazapyr indirectly affects wildlife species (Myllymaki 1975, Borrecco et al. 1979, Sullivan and Sullivan 1982, Morrison and Meslow 1983, Anthony and Morrison 1985, McComb and Hurst 1987, Lautenschlager 1993, Clark et al. 1996, BASF 2006). Direct impacts are relatively absent as toxicity to wildlife is minimal or absent (Miller et al. 1991, Tatum 2004), and application rates are low with only one or 2 applications per stand rotation (~30 years; Wagner et al. 2004). However, surfactants used in tank mixtures can range from nontoxic to very toxic (Geisy et al. 2000, Tatum 2004).

Direct effects to vegetation by prescribed fire and imazapyr generally favors early-successional vegetation associations used by northern bobwhite (*Colinus virginianus*; hereafter bobwhite, Brennan et al. 1998, Guynn et al. 2004, Jones and Chamberlain 2004, Miller and Miller 2004, Welch et al. 2004), eastern wild turkey (*Melleagris gallapavo silvestris*; Dickson and Wigley 2001, Miller and Conner 2007) and white-tailed deer (*Odocoileus virginiaus*; hereafter deer; Edwards et al. 2004, Mixon et al. 2009, Iglay et al. 2010 *b*). Using a single application of imazapyr followed by repeated prescribed fire, now known as Quality Vegetation Management (QVM), has been shown to greatly improve wildlife habitat in pine stands (Brennan et al. 1998, Thompson 2002, Edwards et al. 2004, Woodall 2005). These favorable conditions are achieved as combining prescribed fire and imazapyr reduces hardwood midstory competition (Thompson 2002, Woodall 2005, Iglay et al. 2010*ab*), thus promoting herbaceous ground cover and plant diversity (Sousa 1984, Masters et al. 1993, Sparks et al. 1998, Thompson 2002, Wigley et al. 2002, Guynn et al. 2004, Miller and Miller 2004). Pietz et al. (1999) and Thompson et al. (1991) attribute greater amounts of sunlight reaching the forest floor as a significant stimulant of increased understory herbaceous cover. Although mid-rotation thinning can have similar results, increasing thinning volume may hinder forest management goals and general thinning effects are typically short-lived (Guo and Shelton 1998, Miller et al. 1999, Pietz et al. 1999). However, an initial imazapyr application targeting undesirable hardwood species followed by a frequent fire-return interval regime (~3-years) should provide adequate hardwood fatality and prolonged hardwood stem control, respectively. Greater effectiveness of woody stem control by imazapyr may support greater competition control, fuel reduction, and seed scarification by dormant season burns if prescribed burns follow imazapyr application. Additionally, snags and concomitantly down woody debris also can increase post prescribed fire and/or imazapyr application benefitting many wildlife species (McComb and Rumsey 1983, Lautenschlager et al. 1995, Wigley et al. 2002).

Justification

Planted pine covers an estimated 18 million ha in the southern United States, nearly 20% of southern forests, and 2.2 million ha in Mississippi (USDA Forest Service 2007). Short-rotation (27-32 years) loblolly pine management typically includes clear-cutting followed by site preparation, 1-2 commercial thins, and fertilizing. Although habitat management specifically for wildlife is generally limited on industrial forest landscapes, active management promotes a diversity of stand conditions from early (stand initiation) to late succession (late rotation; Habin et al. 1993, O'Hara 1998) interspersed with non-managed stands.

Disturbances can perpetuate biodiversity (Turner et al. 2001, White and Jentsch 2001, Rundel et al. 1998), and although frequent disturbances favor disturbance-dependent or –tolerant species, timber management incorporating biocomplexity objectives could compensate for disturbance-intolerant species by creating a mosaic of stand treatments, vegetative structures, species, and successional stages across a landscape (Hunter 1990, Franklin 1993, Heljden et al. 1998, Carey et al. 1999*ab*, Tilman 1999) . Managed forests can provide ecosystem benefits such as wildlife habitat, protection of water quality, and carbon sequestration while offering a mosaic of successional stages across the landscape (Petraitis et al. 1989, Greenberg et al. 1994, McLeod and Gates 1998, Vogt et al. 1999). Therefore, it is essential to determine optimal management approaches of intensively managed, mid-rotation pine (*Pinus spp.*) stands to meet forestry and biodiversity objectives (Wear and Greis 2002, Stein et al. 2005). Prescribed burning and imazapyr application during mid-rotation also can help meet habitat and forest management goals by increasing habitat quality and reducing competition from hardwood midstory (Pienaar et al. 1983, Shiver 1994, Quicke 2002, McInnis et al. 2004, Sladek et al. 2008, Iglay et al. 2010*ab*).

Considering the acreage in intensively managed pine in the southeastern United States alone, pairing forestry and wildlife management goals could serve to conserve local and regional biodiversity (Miller et al. 2009). Forest managers are increasingly expected to incorporate management efforts contributing to the conservation of biodiversity (Sustainable Forestry Initiative 2005), but the most commonly applied management practices at mid-rotation, such as commercial thinning and fertilizing, may only provide short-term (< 4 years) benefits for conservation (Wood 1986, Peitz et al. 1999, Iglay et al. 2010*a*). Because species diversity is a criterion for sustainability (Grumbine 1990, Hunter 1990, Bourgeron and Jensen 1994, Kaufmann et al. 1994, Sustainable Forestry Initiative 2005), management tools that enable conservation of biological diversity within intensively managed forests would benefit conservation of biodiversity, sustainable forestry objectives, and social expectations and needs (Hunter 1990, Burton et al. 1992, Hartley 2002, Carnus et al. 2006, Sustainable Forestry Initiative 2005). Without proper management for biodiversity, managed forests may become increasingly biologically simplified causing declines in habitat quality for wildlife (Perry 1994, Carey et al. 1999), ecosystem function (Canham et al. 1990, Franklin 1993, Tilman 1999), site productivity, watershed quality, and carbon sequestration (Harmon et al. 1996, Ponge et al. 1998, Vogt et al. 1999). Addition of mid-rotation treatments, such as prescribed fire and imazapyr, may enhance wildlife habitat within post-thin stands supporting concepts of sustainable forestry by supporting conservation of biodiversity (Hartley 2002, Carnus et al. 2003, Sustainable Forestry Initiative 2005). Such treatments also may improve value of the landbase for fee-based hunting (e.g., Miller et al. 2010), providing economic incentive for such activities. However, information on combined influence of fire and imazapyr, especially across multiple years, is lacking (Chen et al. 1977, Bernardo et al. 1992, Harrington 1998, Brockway and Outcalt 2000).

Overall, prescribed burning and imazapyr can reduced hardwood competition and improve habitat quality for manay wildlife speces in intensively managed pine (*Pinus* spp.) stands (Van Lear 2000, Miller and Miller 2004, McInnis et al. 2004). These activities have variable direct (vegetation) and indirect (fauna) effects (Taylor 1981, Means and Campbell 1981, Landers 1987, Lautenschlager 1993, Clark et al. 1996, BASF 2006). Imazapyr applications have become favored over prescribed fire due to liability concerns, smoke management issues, and burning degree day restrictions (Brennan et al. 1998, Burger et al. 1998, Haines et al. 2001, Wigley et al. 2002). However, fire's influence on ecosystems of southeastern United States, including pine forests, support its use for effective wildlife management through the 21st century (Brennan et al. 1998, Van Lear et al. 2005), specifically in stands of intensively managed pine.

Reintroducing fire to seed banks harboring pyric-adapted species may be the single most beneficial wildlife management practice within pine-dominated systems of the southeastern United States. A major concern regarding such habitat restoration involves ability of management practices to restore areas to pre-treatment composition. Within pyric environments, factors of seed availability, adequate fire disturbance, and pre-fire composition may determine restoration outcomes (Armour et al. 1984, Stickney 1986, Rego et al. 1991). Plant frequency also may depend on pre-treatment distribution (Mueller-Dombois and Ellenberg 1974). Without recent fire history (> 100 years) in many intensively managed pine stands of the southeast, response of pyric-adapted plants in the seed bank may be limited and require multiple burn treatments. The possible greater impact of burning combined with imazapyr on vegetation structure may achieve significantly better alterations of plant succession otherwise unattainable with independent treatments and provide a greater amount of seed catchment opportunities and enhanced plant establishment through reduced competition (Sparks et al. 1998), if seeds are available. However, information regarding effects of reintroducing fire in intensively managed pine of the southeastern United States across a full fire-return rotation is lacking.

From a habitat management perspective, imazapyr may or may not be a viable replacement for fire in pine forests of the southeastern United States (Brennan et al. 1998). Conversely, herbicides may be used with fire in a complimentary manner to achieve management objectives. Although past research suggests community responses similar between use of fire and imazapyr, concurrent comparisons of independent and combined treatments are lacking (Lyon et al. 1978, McInnis et al. 2004). Replicated field experiments with adequate controls are the best approach for understanding effects of imazapyr and prescribed fire on vegetation and wildlife communities. Past field research concerning imazapyr has focused most on site preparation (Shepard et al. 2004) and non-industrial timberlands (Howell et al. 1996, Edwards et al. 2004, McInnis et al. 2004, Mixon et al. 2009), not mid-rotation release on industry land.

Understanding the concomitant effects of prescribed fire and imazapyr on the vegetative community is critical to understanding biodiversity response to treatments in mid-rotation, intensively managed pine stands of Mississippi. Therefore, I examined effects of prescribed fire and imazapyr on the vegetation structure, coverage, and biomass of mid-rotation, intensively managed pine stands of east-central Mississippi to characterize treatments by vegetation variables.

Study Area and Design

My study area was in the Interior Flatwoods Area (Pettry 1977) and Upper Coastal Plain Regions of east-central Mississippi in Kemper County between Scooba and DeKalb, MS, USA. The 9,600 ha landscape containing my research plots was composed of intensively managed pine (*Pinus* spp.) stands (70%) of various ages, mature pine-hardwood (17%), hardwood (10%), and non-forested areas (3%). Early accounts of species composition described a mixture of loblolly pine (*Pinus taeda*) and hardwoods in open, frequently burned forests (Adair 1977, Bourne 1904, Lowe 1913, Perkins 1973). Original harvest of virgin forests occurred between 1912 and 1941 by the Sumter Lumber Company (Perkins 1973). From 1941-1967, Flintkote Company practiced very conservative selective cutting (McKee 1972). Weyerhaeuser NR Company has been intensively managing these forests since 1962 consisting of clear-cuts followed by site preparation and planting (McKee 1972).

Within my study area, I used 6 mid-rotation (16-19 years) pine stands, managed on short-rotation (25-32 years) by Weyerhaeuser NR Company, selected for this project in 1998. Each stand was commercially thinned 2-5 years prior to project initiation, 59-120 ha in size, and fertilized immediately post-thin and again in winter 2001 following Weyerhaeuser NR Company protocols with diammonium phosphate (127-283.5 kg/ha, $\bar{x}$ =153.4 kg/ha) and/or urea (381–448 kg/ha, $\bar{x}$ =222.8 kg/ha). Sites indexes ranged from

65-79 ($\bar{x}$ = 74.8). Within each stand, I created 4, 10-ha treatment plots (286 m X 350 m) with $\geq$ 50 m buffers and randomly assigned each plot to a treatment (burn only, herbicide only, burn + herbicide, control). Imazapyr was applied to all herbicide treated plots in September 1999 via skidder with a tank mixture of 877 ml/ha (12.0 liquid oz./ac) of Arsenal® (imazapyr; BASF 2006), 0.5% Timbursurf90 (Timberland Enterprises, Inc. Monticello, AR) as a surfactant, and water for dilution at a rate of 150-187 L/ha (16-20 gals/ac). Prescribed burns were applied using drip torches in January 2000 and 2003 and February and March 2006 within recommended environmental conditions of fine-fuel moisture (10-20%), ambient temperature ($\leq 16^{o}$C), wind speed (in stand: 1.61-3.22 kph, open locations: 4.83-32.19 kph), and relative humidity (30-55%, Wade and Lunsford 1989).

Methods

Prescribed Fire

I reduced variability in fire intensity by following burn prescriptions (Wade and Lunsford 1989). Within one month prior and post burn, I measured fuel composition using a 7.6 m line-intercept and recording the first-encountered fuel type crossing the transect every 3 cm from 1.4 m (breast height) to the ground. I calculated relative coverage of exposed debris (pine needles and leaves), living herbaceous vegetation, living woody vegetation, dead vegetation such as dead grasses, forbs, vines, and shrubs, and exposed bare ground at 4 random points in 2000 and 3 random points in 2003 and 2006 per treatment plot. I used rebar to mark intercept locations pre-burn for repeated measures post-burn. Immediately prior to each burn, I measured in-stand temperature (^{o}C) and relative humidity (%) using a Digital max/min Thermohygrometer (Forestry Suppliers, Jackson, Mississippi, USA), wind speed using a Skywatch Xplorer 1 Anemometer (JDC Electronics SA, Switzerland), and fine-fuel moisture using fine-fuel moisture sticks at plot center. I placed fine-fuel moisture sticks in each plot $\geq$ 24 hours prior to burning. I weighed sticks at plot center immediately prior to burning, sealed them in individual Ziploc® bags, dried them at 80^{o}C until constant weight was achieved, and weighed them again for dry weight. I used the difference between wet and dry weight for fine-fuel moisture. During burns, I used 4, 3, and 5 pairs of randomly placed wooden stakes in 2000, 2003, and 2006, respectively, with each pair spaced 5 m (15.24 m spacing in 2003). I marked each stake in 0.5 m increments to measure flame height. I also used them to measure mean residence time (s) and mean rate of spread (m/s) with stop watches.

Vegetation Response

I measured vegetation structure and coverage and plant biomass production in summer (May-July) 1999-2008 and winter (January-March) 1999-2007. I used 1999 vegetation structure, coverage, and biomass as pre-treatment, baseline data. During summer, I measured vegetation structure $\geq$ 50 m from plot edge to avoid edge effects at 9 randomly placed points as follows: one at each avian survey point (n = 4/plot; see Chapter 2), one at each center insect pitfall trap (n = 2/plot; see Chapter 5), and one at each drift fence array (n = 3/plot; see Chapters 3 and 4). During 2002 and 2003, I replaced center insect pitfall traps with 2 cover board points. In winter, I used 2 points on a small mammal trapping grid instead of the insect pitfall traps (see Chapter 3). In both seasons, I measured overstory and midstory trees by recording species and diameter at breast height (dbh; cm) of all trees ≥ 10 cm dbh within a 0.04 ha plot (overstory), and height (m) and dbh of all trees 1.3 cm dbh $\leq x <$ 10 cm dbh in a 0.004 ha plot (midstory, Wegner 1984). For snags ≥ 2 m in height and ≥ 10 cm dbh, I recorded dbh, height, one of 5 decay classes (Thomas et al. 1979), woodpecker activity presence/absence, and cavity presence/absence within a 0.04 ha plot. I used a Nudds board divided into 6, 0.3 m sections (1.8 m height) with each section alternating orange and white to estimate foliage density (Nudds 1977). I recorded foliage density per section at 10% interval estimates 1999-2003 and 5% interval estimates 2004-2008 from a 1 m viewing height at plot center, 11.3 m from the board in each of the 4 cardinal directions. I used a convex spherical densiometer to estimate percentage canopy cover (Lemmon 1957). During winter only, I sampled downed woody material using a 30.3 m line intercept from plot center oriented in a random direction. For each occurrence of downed woody material ≥ 8 cm diameter at the widest point and ≥ 100 cm in length, I recorded the point at which it crossed the line, length (cm), diameter (cm), and assigned one of 5 decay classes (McCarthy and Bailey 1994). For wood piles, I also measured height (m).

From May-June, I recorded plant species < 1.4 m in height, exposed bare ground, and exposed debris along a randomly placed 7.6 m line intercept from plot center with direction determined by spinning a pencil (30.3 m line in 1999). At every 3 cm, I measured any plant crossing the line. After entering data, I recorded ‘space’ for all gaps in layers occupied but not covered by a plant or other category to standardize all lines to 7.6 m intervals.

I measured plant productivity using 1-m^2 clip hoops placed randomly diagonally across each treatment plot. In summer, I used 20 hoops/plot (10 hoops/plot 1999 and 2000) and in winter 10 hoops/plot. I increased number of subsamples/plot based on initial estimates of variability and desired precision. I clipped all plants < 1.3 cm diameter, separated leaves from stems, and placed them in brown paper lunch bags marked with hoop number, treatment plot, collection date, plant species acronym, and designated leaves and growing stems as consumable plant parts for white-tailed deer. I dried all samples at 80^{o}C until constant weight was obtained and extrapolated biomass estimates (kg/ha).

Statistical Analysis

I used 2 approaches for analyses. First, I assessed vegetation response to treatments and then used ordination to help visualize segregation of treatment plots by vegetation structure, cover, and biomass variables. I used basal area (m^2/ha) of overstory and midstory trees (pine or hardwood), canopy cover (%), Nudds board cover (%), and downed woody debris (cm^3/ha) for vegetation structure and forage class coverage (%), biomass (kg/ha), and species richness for understory plant diversity. I classified species into forage class groups of forbs (herbaceous annual and perennial plants), grasses (Family: Poaceae), herbaceous vines, legumes (Family: Fabaceae), sedges and rushes (non-grass), semi-woody vines, woody plants, and woody vines.

I tested the hypothesis of no difference in mean basal area, canopy cover, Nudds board cover, down woody debris, understory plant coverage, biomass, and species richness among treatments within years using mixed models, repeated measures analysis of covariance in SAS Proc Mixed (SAS Institute Inc., Cary, North Carolina, USA). I used main effects of treatment, year, and treatment × year on vegetation structure and understory plant diversity. I used 4 levels of treatment main effects (burn, herbicide, burn + herbicide, control), random effect of stand (n = 6), repeated measures of year (n = 9; 2000-2008), and subject of stand × treatment (Littell et al. 2006). I used pre-treatment year (1999) measurements as baseline covariates because pre-treatment vegetation characteristics could affect post-treatment response (Milliken and Johnson 2002). I selected an appropriate covariance structure for each variable from among those following a time series: 8-banded Toeplitz (7-banded for winter), heterogeneous compound symmetry, heterogeneous auto-regressive, and auto-regressive. I selected the covariance structure that minimized Akaike's Information Criterion corrected for small sample size (Littell et al. 2006, Gutzwiller and Riffell 2007). I used Kenward-Roger correction for denominator degrees of freedom for repeated measures to avoid inflated Type I error (Littell et al. 2006, Gutzwiller and Riffell 2007). I used the LSMEANS SLICE option to identify a treatment effect within years following a significant interaction (Littell et al. 2006), and LSMEANS PDIFF to conduct pair-wise comparisons (Littell et al. 2006). All year references in results refer to years post-treatment, with 2000 being year 1. My *a priori* significance level was $\alpha=0.05$.

To help visualize segregation of treatment plots by treatment using vegetation structure, coverage, and biomass variables, I used Principal Components Analysis (PCA) with correlation matrices in PC-ORD (version 5.10, MJM Software, Gleneden Beach, Oregon, USA; Taylor et al. 1993, Morrison et al. 1998, ter Braak and Smilauer 1998). I conducted separate analyses for vegetation structure (e.g., basal area, canopy and Nudds board cover, and downed woody debris), forage class coverage, and forage class biomass. Although forage class coverage and biomass each assess understory plant diversity, they represent different information and thus required separate analyses. I used randomization tests of eigenvalues with 999 runs and resulting test statistic, Rnd-Lambda, as a stopping rule to determine total number of non-trivial components (Peres-Neto et al. 2005). I used Rnd-Lambda because it restricts introducing additional noise into analysis results and is robust with non-normal data and uncorrelated variables (Peres-Neto et al. 2005). I deemed eigenvector loadings $\geq |0.3|$ significant for interpretation and examined variables by year to minimize sampling units to variable ratios as suggested by Pillar (1999). Whenever > one component was considered non-trivial, I graphed components for a visual representation of treatment plot segregation. I did not use PCA to reduce dimensionality of my data or to determine treatment effects. Whenever summary and individual statistics were deemed important (e.g., Nudds board section one, Nudds summary for sections 1-3), I used summary variables.

Results

I collected 888 samples between vegetation structure and coverage (n=456) and biomass (n=432) samples with 10,344 subsamples across 10 years. I observed 387 plant species (163 forbs, 10 ferns, 45 grasses, 16 herbaceous vines, 25 legumes, 28 sedges and rushes, 10 semi-woody vines, 76 woody plants, and 14 woody vines) . Six species [sawtooth blackberry (*Rubus argutus* Link), Japanese honeysuckle (*Lonicera japonica* Thunb.), poison ivy (*Toxicodendron radicans* L. Kuntze), blackberry/dewberry (*Rubus* spp.), slender woodoats (*Chasmanthium laxum* L. Yates), and muscadine grape (*Vitis rotundifolia* Michx.)] were predominate (55%) in biomass samples.

Vegetation Structure

All treatments achieved some level of midstory hardwood control and increase in understory plant species richness compared to controls. Fire was less effective than imazapyr for reducing woody plants, particularly midstory hardwoods, but always stimulated changes in low-level visual obstruction (e.g., Nudds board sections 1-3) every 3 years. Burn + herbicide could be considered the greatest intensity treatment as evidenced by significant initial reductions in low-level visual obstruction and basal area of midstory hardwoods and increased volume of downed woody debris of class 2 and basal area of hardwood snags. Understory forage class coverage of burn + herbicide plots typically consisted of herbaceous plants such as forbs, grasses, legumes, and semi-woody vines.

I used 21 vegetation structure variables for principal components analysis (Table 1.1). Burn + herbicide and control plots appeared as treatment extremes at either end of PC1 for all years post-treatment based on line intercept data. Vegetation structure variables did not differentiate treatment plots as well as line intercept variables. Common trends of treatment plot differentiation by PCA included less midstory basal area in treated sites than controls. Consequently, upper level visual obstruction (e.g., Nudds board sections 4-6) was greater in controls than treated plots. Unlike ANCOVA results (see below), burn treatments (burn only and burn + herbicide) were generally more similar to each other as were herbicide only to control plots.

Summer Vegetation Structure ANCOVA

Thirteen of 19 summer vegetation structure variables differed among treatments within years (Table 1.2). Overstory and midstory pine and hardwood basal areas and canopy coverage had significant pre-treatment covariates. Post-treatment, hardwood overstory basal area was less in herbicide treatments (herbicide only and burn + herbicide) than controls in years 6 and 8 but similar between burn only and all treatments in year 6 and burn only and control in year 8 (Table 1.3). Hardwood midstory basal area was greatest in control plots 6 out of 9 years and least in burn + herbicide. Hardwood midstory tree height was greater in herbicide treatments ($\bar{x}$ = 5.80 m) than burn only and controls year 1 ($\bar{x}$ = 3.51 m), greater in burn treatments ($\bar{x}$ = 5.43 m) than herbicide only and controls year 7 ($\bar{x}$ = 4.32 m), and least in burn + herbicide in year 9 [$\bar{x}$ = 3.73 m (burn + herbicide) vs. $\bar{x}$ = 4.92 m (other treatments and control)]. Pine midstory tree height was greater in herbicide only than burn + herbicide year 8 with burn only and control intermediate. Hardwood snag height was greater in herbicide treatments than burn only and control in year 1 ($\bar{x}$ = 8.67 m vs. $\bar{x}$ = 1.24 m).

All sections and summary variables of Nudds' board differed among treatments within years (Table 1.2). Treatment intensity (i.e., burn + herbicide > herbicide only > burn only > control) created a gradient of Nudds board cover from greater coverage in controls and burn only followed by herbicide only and burn + herbicide in year 1 (Table 1.3). After year 1, cover for Nudds board section one (0.0 – 0.3 m) was generally greater in treated plots versus controls with herbicide treatments exhibiting greater coverage than burn only. Nudds board section one cover in burn treatments seemed to vary relative to fire frequency with reduced coverage following 2nd and 3rd burns (years 4 and 7) with fast recovery of visual obstruction in subsequent years, especially burn + herbicide. Nudds board section two (0.3 - 0.6 m) cover was generally least in burn only and controls 5 of 9 years but similar to burn + herbicide for 2 years and similar between independent treatments in years 6 and 8. Nudds board section three cover (0.6 – 0.9 m) was greater in herbicide only and control ($\bar{x}$ = 84%) than burn treatments ($\bar{x}$ = 66.5%) year 4, greatest in herbicide only year 7 ($\bar{x}$ = 85% vs. $\bar{x}$ = 58.7%), and greater in herbicide treatments than controls year 8 ($\bar{x}$ = 68.5% vs. $\bar{x}$ = 52%) with burn only intermediate to all treatments. Nudds board section four cover (0.9 – 1.2 m) was least in herbicide treatments year 2 ($\bar{x}$ = 55.5% vs. $\bar{x}$ = 77.5%) but greater in herbicide only and control than burn treatments in years 4 and 7. Nudds board section five and six cover (1.2 – 1.8 m) were generally greater in control and herbicide only than burn treatments. Nudds board sections 1 -3 cover was greatest in herbicide only during 2nd and 3rd burn years but greater in herbicide treatments in years 5 and 6 (i.e., except for herbicide only, being intermediate to all treatments in year 6) and least in controls years 8 and 9. Nudds board sections 4-6 cover shifted from greater coverage in burn only and controls to herbicide only and controls up until year 7.

Overstory pine basal area and canopy coverage differed among treatments across all years (Table 1.2). Overstory pine basal area was least in controls ($\bar{x}$ = 18.89 m^2/ha vs. $\bar{x}$ = 21.2 m^2/ha). Canopy coverage was greater in control ($\bar{x}$ =93.1%) than herbicide treatments ($\bar{x}$ = 89.9%) with burn only intermediate to control and herbicide only.

Line Intercept ANCOVA

Six of 9 forage classes and debris coverage had significant treatment × year interactions with woody plants and woody vines having significant pretreatment covariates (Table 1.4). Other than years 1 and 2 when debris coverage was greatest in burn + herbicide and herbicide only, respectively, debris coverage tended to be twice as great in controls compared to burn + herbicide with independent treatments intermediate (Table 1.5). Forb coverage was up to 6 times greater in burn + herbicide than herbicide only and controls 3 of 7 years. Following 2nd and 3rd burns, burn treatments supported forb coverage 6 to 13 times greater than herbicide only and controls, respectively. Grass coverage was nearly 3 times less in herbicide treatments in year 1 but was 3-4 times greater in burn + herbicide than controls in years 6-9 with herbicide only intermediate to burn only and controls. Sedges and rushes coverage also was least in herbicide treatments in year 1 but greatest in burn + herbicide in years 3 ($\bar{x}$ = 9.7% vs. $\bar{x}$ = 1.9%) and 4 ($\bar{x}$ = 4.7% vs. $\bar{x}$ = 0.2%). Semi-woody coverage was greatest in herbicide treatments 4 of 9 years with twice the coverage as controls. Coverage in herbicide only was 2 times greater than burn only and controls in year 4, but coverage in burn only was similar to herbicide only in years 8 and 9. Greater woody plant cover was found generally in independent treatments and controls, not burn + herbicide. During years 1-3, burn only and controls had greater woody plant coverage than herbicide treatments except for similarities between burn treatments in year 2. Woody vine coverage was greatest in controls every year and up to 5 times less in burn + herbicide. Woody vine coverage in herbicide only was intermediate among burn treatments for 3 years, and coverage in burn only was intermediate among herbicide treatments in year 9.

Species richness of understory plant coverage differed among treatments within years (Table 1.4). Species richness of herbicide treatments was initially significantly less than burn only and control ($\bar{x}$ = 8.4 vs. $\bar{x}$ = 15.7; Table 1.6). However by year 6, species richness was greater in burn treatments than herbicide only and controls.

Winter Vegetation Structure ANCOVA

Eight of 25 winter vegetation structure variables differed among treatments within years (Table 1.7). Overstory pine and hardwood snag basal areas, canopy coverage, and Nudds board summary sections 4-6 had significant pretreatment covariates. Midstory hardwood basal area was greater in controls than burn + herbicide 4 of 9 years with independent treatments intermediate (Table 1.8). However, in years 2 and 3, herbicide only was intermediate among burn + herbicide and all other treatments and burn treatments, respectively. Midstory hardwood tree height was greatest in controls in year 3 ($\bar{x}$ = 5.88 m vs. $\bar{x}$ = 4.21 m). Nudds board sections 1-4 had less coverage in burn treatments than herbicide only and control in year 1 and during years of all subsequent burns. Coverage in burn only tended to be less than most other treatments except for coverage in burn + herbicide during burn years. Down woody debris cover of class 2 was greatest in burn + herbicide in year 1 ($\bar{x}$ = 4.24 m^3/ ha vs. $\bar{x}$ = 1.06 m^3/ ha).

Seven winter vegetation structure variables differed among treatments across all years (Table 1.7). Midstory pine height was greater in control ($\bar{x}$ = 0.11 m, SE = 0.44) than burn + herbicide ($\bar{x}$ = 2.0 m, SE = 0.45) with independent treatments intermediate ($\bar{x}$ = 1.36 m, SE = 0.44). Hardwood snag basal area was greatest in herbicide treatments ($\bar{x}$ = 4.74 m^2/ ha vs. $\bar{x}$ = 1.84 m^2/ ha, SE = 0.75). Nudds board sections 5 ($\bar{x}$ = 39.74 %, SE = 2.34) and 6 ($\bar{x}$ = 38.94 %, SE = 2.06) and summary variable for sections 4-6 ($\bar{x}$ = 40.94 %, SE = 2.13) had greatest coverage in controls. Least coverage for section 5 ($\bar{x}$ = 22.27 %) and summary variable for sections 4-6 ($\bar{x}$ = 23.19 %) was in burn treatments. For Nudds board section 6, least coverage was in burn + herbicide ($\bar{x}$ = 16.13 %) followed by burn only ($\bar{x}$ = 22.49 %) and herbicide only ($\bar{x}$ = 30.11 %).

Summer Vegetation Structure PCA

Two non-trivial PCA axes of 1999 explained 46.6% of total variance (28.1% and 18.5% for axis 1 and 2, respectively). They displayed relative homogeneity among treatment plots except for 2 outlying burn only plots (Figure 1.1). Important vegetation structure variables consistently contributing to axes were midstory hardwood height, hardwood snag height and basal area, and both Nudds board summary variables. One PCA axis of 2000 explained 31.6% of total variance and segregated treatment plots as herbicide treated sites versus burn only and control with tightly grouped burn only plots surrounded by control plots. Herbicide treated plots generally had greater snag and overstory pine basal areas and greater midstory hardwood and hardwood snag height with less mid-story hardwood basal area and overall visual obstruction compared to burn only and control plots. Two PCA axes of 2001 explained 44.6% of total variance (24.4% and 20.2% for axes 1 and 2, respectively). Treatment plots were segregated similar to their distribution in 2000 but with less-defined segregation. Some additional important variables contributing to components were hardwood overstory and midstory pine basal areas, pine snag height, and canopy coverage (Figure 1.2). In 2002, PCA axis 1 explained 25.8% of total variance with 11 of 13 variables contributing significantly to the component. Treatment plots were not segregated by treatments in ordinal space, but instead, plots treated alike were dissimilar.

One PCA axis of 2003 explained 27.2% of total variance with burn + herbicide plots correlated positively. Burn + herbicide plots tended to have less midstory basal area (pine and hardwood) and midstory pine height, hardwood snag basal area and height, and visual obstruction of the top 3 sections of a Nudds board than other plots. In 2004, PCA axes 1 and 2 explained 24.7% and 18.3% of total variance. Most treated plots were tightly grouped at the center of the graph with control plots positively and neutrally correlated to PC1 and PC2, respectively (Figure 1.3). According to PC1, treated plots generally had less midstory basal area, midstory pine height, canopy coverage, and high Nudds board cover than control plots. Three PCA axes in 2005 explained 60.9% of total variance (27.3%, 17.7%, and 15.9% for axes 1, 2, and 3, respectively). Only PC1 segregated treatment plots by treatment separating burn only and burn + herbicide plots (Figures 1.4 and 1.5). Pine overstory and snag basal areas, pine snag height, and low Nudds board coverage were correlated positively with burn + herbicide plots, and midstory pine height, overall hardwood component (live and dead), and canopy coverage were correlated positively with burn only plots. However, half of the burn only plots were centered on PC1.

Two PCA axes in 2006 explained 43% of total variance (22.9% and 20.1% for axes 1 and 2, respectively). Both axes tightly clustered burn treated sites except for one outlying burn only plot on PC1 (Figure 1.6). For PC1, burn treated plots typically had less midstory pine basal area and height, midstsory and overstory hardwood basal area, canopy cover, and high Nudds board coverage than herbicide only and control plots.

However, on PC2, burned plots were relatively neutral among axis correlations with most clustered around zero. One PCA axis in 2007 explained 26.4% of total variance but did not segregate treatment plots by treatment in ordinal space. Important variables of PC1 with negative correlations were midstory hardwood basal area and height, hardwood overstory basal area, and canopy coverage. Pine and hardwood snag basal area and height and both Nudds board summary variables were correlated positively with PC1. Two PCA axes in 2008 explained 46.5% of total variance (26.8% and 19.7% for axes 1 and 2, respectively). Treated sites were correlated neutrally or positively with PC1 and relatively centered on PC2 (Figure 1.7). Treated sites positively correlated with PC1 trended towards less midstory and overstory hardwood basal areas, midstory hardwood height, pine and hardwood snag basal areas and heights, and canopy coverage than control plots. Control plots had greater low Nudds board coverage.

Line Intercept PCA

Only 1999 and 2005 PCA of line intercept data had > 1 important axis. Two axes of 1999 explained 44.5% of total variance (24.2% and 20.3% for axes 1 and 2, respectively) but did not segregate treatment plots by treatment in ordinal space. However, 2 axes of 2005 explained 51.3% of total variance (31.2% and 20.1% for axes 1 and 2, respectively) clearly segregating burn + herbicide and control plots with PC1. Burn + herbicide plots had greater coverage of exposed bare ground, forbs, grasses, herbaceous vines, sedges and rushes, and semi-woody vines than control plots in addition to greater understory plant species richness. Control plots had greater exposed debris and woody vine coverage. Although burn only and herbicide only plots appeared intermediate to treatment extremes, they were more similar to burn + herbicide and control plots, respectively (Figure 1.8). All treatment plots were centered on PC2.

Various patterns appeared in years with only one important axis. From 2000-2002, herbicide treated plots were more similar to each other as were burn only and control plots. Across all of these years, exposed debris, forbs, sedges and rushes, woody vines, and wood plants remained important variables. In 2004, independent applications of burning and herbicide moved towards an intermediate position among treatment extremes. They stayed in this intermediate position throughout the remainder of the sampling years (2006-2008) becoming less similar to treatment extremes than each other by the last 2 years of the study. From 2006-2008, coverage of exposed debris, forbs, grasses, and all vines and understory species richness were important variables. Woody vines and exposed debris coverage had opposite axis correlations.

Winter Vegetation Structure PCA

One PCA axis in 1999 explained 28.5% of total variance but did not segregate treatment plots by treatment. Two PCA axes in 2000 explaining 41.6% of variance (25.6% and 15% for axes 1 and 2, respectively) also did not segregate treatment plots by treatments. In 2001, 3 PCA axes explained 53.9% of total variance (25.8%, 14.8%, and 13.3% for axes 1, 2, and 3, respectively). Visual inspection of PC1 and PC2 showed segregation among treatment plots by treatment over PC1, specifically overall separation of treatment extremes and a slight grouping of burn + herbicide treatments (Figure 1.9). However, independent treatment plots were scattered intermediate among extremes, and one third of control plots (2 plots) were located among burn + herbicide plots. Treatment plots could not be segregated visually when inspecting plots of PC1 by PC3 and PC2 by PC3 other than along PC1 (Figures 1.10 and 1.11). According to important variables of PC1, burn + herbicide plots tended to have less midstory and overstory hardwood basal areas, pine snag basal area, overall Nudds board coverage, and volume of downed woody debris of class 1 than control plots. Three PCA axes in 2002 explained 54% of total variance (22.9%, 17.8%, and 13.3% for axes 1, 2, and 3, respectively) but did not segregate treatment plots by treatment in ordinal space. Graphing of PC2 by PC3 did reveal an outlying burn only plot and also tightly grouped treatment plots suggesting relative homogeneity among treatment plots regarding important variables (Figure 1.12). Midstory pine basal area and height, mid-story hardwood height, pine snag basal area, hardwood snag basal area and height, volume of downed woody debris in decay classes 1 and 2 were important variables among both axes.

Two PCA axes in 2003 explained 42.8% of total variance (26.6% and 16.2% for axes 1 and 2, respectively). Burned plots were neutrally to negatively associated with PC1 whereas herbicide only and control plots had neutral to negatively associations (Figure 1.13). According to important variables of PC1, burned plots had less midstory pine and hardwood basal areas, midstory pine height, overall Nudds board cover, and volume of downed woody debris of decay classes 2-5 than herbicide only and control plots. No differentiation of treatment plots by treatment was apparent by PC2. In 2004, burned plots were more integrated with other treatment plots along the only PCA axis that explained 26% of total variance. Three PCA axes in 2005 explained 53.4% of total variance (24.7%, 15.1%, and 13.6% for axes 1, 2, and 3, respectively). Burn treatments remained clustered in an overall neutral to positive association with PC1 (Figure 1.14).

Other principal components did not appear to differentiate treatment plots by treatments. Burn treatments had less midstory pine basal area and height, hardwood basal area, hardwood snag basal area and height, overall Nudds board cover, and volume of down woody debris of classes 2 and 4. Three PCA axes in 2006 explained 54.8% of total variance (24.5%, 17.7%, and 12.6% for axes 1, 2, and 3, respectively). Although burned plots still remained clustered, all treatment plots were close together on PC1 (Figure 1.15). Other axes did not segregate treatment plots by treatment (Figure 1.16). According to important variables of PC1, burn treatments had less midstory basal area (pine and hardwood), midstory pine height, canopy coverage, and all Nudds board coverage. Two PCA axes in 2007 explained 40.1% of total variance (24.3% and 15.8% for axes 1 and 2, respectively). Burned plots were sill separated from other plots on PC1 (Figure 1.17). Burned plots had less midstory basal area (pine and hardwood), midstory pine height, canopy coverage, overall Nudds board coverage, and volume of down woody debris of class 2.

Plant Biomass

Summer biomass of herbaceous plants (e.g., forbs, legumes, grasses, herbaceous vines) were generally least in controls and greatest in burn + herbicide. Burn treatments harbored greater species richness than herbicide only and controls for most years. Relatinoships among treatments for winter biomass was near opposite as herbicide only trended towards greater biomass of herbaceous plants and greater species richness than burn treatments and control.

Few important principal components segregated treatment plots by treatment according to visual inspections of graphed components for summer and winter biomass data. Summer biomass PCA revealed identical treatment extremes as did line intercept PCA (burn + herbicide and control). Burn + herbicide plots generally had greater herbaceous understory plant biomass than control plots. Only one year (2000) of winter biomass PCA had any apparent clustering of plots by treatment.

Summer Plant Biomass ANCOVA

Summer biomasses of herbaceous vines, sedges and rushes, and woody plants differed ($P < 0.05$) among treatments within years (Table 1.9). Herbaceous vine biomass was greatest in burn + herbicide in year 8 ($\bar{x}$ = 11 kg/ ha vs. $\bar{x}$ = 2 kg/ ha; Table 1.10). Sedges and rushes biomass was greater in burn + herbicide than control with independent applications intermediate in year 3 and greater in all treatments than control plots in year 9 ($\bar{x}$ = 19 kg/ ha vs. $\bar{x}$ = 4.3 kg/ ha). Woody plant biomass was least in treated sites versus controls in year 1 and less in burn + herbicide than burn only and controls in year 2 with herbicide only intermediate. Woody plant biomass was greatest in herbicide only in years 7 and 8 except with burn only intermediate in year 8.

Summer biomasses of forbs, legumes, and semi-woody vines differed among treatment across all years (Table 1.9). Forb biomass was greatest in burn + herbicide ($\bar{x}$ = 207 kg/ ha vs. $\bar{x}$ = 59 kg/ ha, SE = 33). Legume biomass was greatest in burn treatments ($\bar{x}$ = 66 kg/ ha vs. $\bar{x}$ = 13 kg/ ha, SE = 5). Semi-woody vine biomass was greater in herbicide treatments than control ($\bar{x}$ = 378.5 kg/ ha vs. $\bar{x}$ = 206 kg/ ha, SE = 48) with burn only intermediate to burn + herbicide and control.

Summer understory plant species richness constructed from biomass collections differed among treatments within years (Table 1.9). It was greatest in burn only in years 1 and 2 and least in herbicide only in year 1 (Table 1.10). Species richness alternated between the top and intermediate levels in burn + herbicide and burn only in years 7 and 8, respectively. Species richness in burn treatments was greatest in year 9 ($\bar{x}$ = 56.5 vs. $\bar{x}$ = 40).

Winter Plant Biomass ANCOVA

Winter biomass of grasses, sedges and rushes, semi-woody vines, woody plants, and woody vines differed among treatments within years (Table 1.9). Grass biomass was greatest in controls in year 1 (Table 1.11). In year 3, herbicide treatments had greater grass biomass than control with burn only intermediate. In year 6, burn + herbicide had the greatest grass biomass, and in year 8, burn treatments had the greatest grass biomass. Sedges and rushes biomass was greatest in burn treatments in year 3 and least in control. Semi-woody vine biomass was greatest in herbicide only and control in year 1. In year 5, it was greatest in herbicide treatments. Woody plant biomass was greatest in herbicide only and control in year 1 and control in year 2. In year 2, it was least in burn + herbicide with herbicide only intermediate among burn treatments. In year 3, it was greater in burn only and control than burn + herbicide with herbicide only intermediate. Woody vine biomass was greatest in herbicide only and control in year 1 but greatest in control in years 2, 4, and 6.

Winter understory species richness differed among treatments within years (Table 1.9). Species richness was greatest in herbicide only and control in year 1, and burn only and control in year 2 (Table 1.11). In year 3, it was greater in independent treatments than burn + herbicide with control intermediate. In year 7, it was greatest in herbicide only and control, and in year 8, it was greatest in burn treatments.

Summer Plant Biomass PCA

Most years only had one principal component except for 2000 with 2 PCA axes (27% and 20.6% for axes 1 and 2, respectively) and no principal components in 1999 and 2001. Most treatment plots were centered on PC1 in 2000, but PC2 had control plots more associated positively than treated plots (Figure 1.18). Treated plots tended to have greater forb, fern, and herbaceous vine biomass in 2000. From 2002 to 2008, burn + herbicide and control plots began to move to opposite ends of PC1. Burn + herbicide plots tended to have greater forb, fern, herbaceous vine, legume, sedges and rushes, grass, and semi-woody vine biomass than controls. Species richness also was greater in burn + herbicide plots.

Winter Plant Biomass PCA

Five of 9 years had ≥ 1 principal component. One PCA axis in 1999 explained 33.3% of total variance but did not segregate treatment plots by treatment in ordinal space. Most plots had a negative association with PC1 suggesting greater woody plant and woody vine biomass. Two PCA axes in 2000 explained 72.2% of total variance (49.1% and 23.1% for axes 1, and 2, respectively). Burned plots were associated negatively with PC1 with less biomass of sedges and rushes, grasses, herbaceous and semi-woody vines, and woody plants and vines (Figure 1.19). For 2001, 2004, and 2005, plots remained clustered over each year's PC1.

Discussion

Prescribed burning and imazapyr significantly altered midstory and understory vegetation structure, coverage, and biomass in mid-rotation, intensively managed pine stands in my study. Each treatment (burn only, herbicide only, burn + herbicide) reduced vegetative competition by differentially affecting plant forage classes, increased species richness, and manipulated vegetative structure. However, distinct differences between prescribed burning and imazapyr were evident as were differences among independent and combined treatments.

My results do not support one treatment type over the other or offer a panacea for vegetation management in an intensive timber management matrix. Each treatment has its own set of benefits and concerns when dealing with application, vegetative response, and long-term vegetative community maintenance for the manager to consider. Each creates a unique vegetation community based on differences in hardwood competition control, visual obstruction, and understory plant diversity response. Across landscapes of multiple stands, applying treatments at random to mid-rotation stands of similar past management histories could help perpetuate conservation of biodiversity.

Combining burning and imazapyr was the most effective treatment for reducing hardwood midstory competition, which is desirable for many wildlife species via creation of a stand condition comprised of a pine overstory and herbaceous understory. Past studies have found similar effectiveness of the combined treatment but did not compare to independent applications (Edwards et al. 2004, Jones and Chamberlain 2004). A one time, low-rate application of imazapyr was more effective at controlling hardwood competition than a single dormant season prescribed burn. Multiple prescribed burns (3 in this study) can reduce midstory hardwood competition. Had this study continued with additional burns on a 3-year fire return interval, dormant season prescribed burning may have achieved hardwood midstory control equivalent to low-rate imazapyr application. Responses ranging from reduced to increased hardwood stems seem to be a function of burning characteristics such as rate of spread, fuel moisture, and ambient temperature and ability of some hardwoods to re-sprout quickly after top-kills (e.g., *Liquidambar*; Allen 1960, Komerak 1965, Waldrop et al. 1992, Sparks et al. 2002). Prescribed burning and imazapyr, independently applied or combined, always reduced hardwood midstory competition compared to no treatment (control).

All midstory hardwood control treatments impacted understory vegetative structure consequently influencing visual obstruction. Treatments reduced significantly pre-treatment standing structures (i.e., hardwood stems) causing expanses of open space under a pine overstory by 1 year post treatment. By year 2, vegetative response to treatments was evident by greater ground-level to mid-Nudds board visual obstruction beyond that of the control treatment. Because imazapyr was only applied once, vegetation was able to continue growing after initial application whereas prescribed burns repeatedly impacted vegetation maintaining reduced vegetative structure throughout the study especially upper level Nudds board (> 0.9 – 1.8 m). However, imazapyr, with or without prescribed fire, continually maintained greater visual obstruction than prescribed burn only.

Differences in upper level Nudds board visual obstruction among treatments revealed a long-term effect of prescribed burning unattained by imazapyr. Herbicide only plots initially had less upper level visual obstruction than burn only plots but eventually, the lack of repeated disturbance (i.e., repeated burns) allowed plants uncontrolled by imazapyr to occupy space indexed with upper Nudds board sections (0.9 – 1.8 m) and obscure visibility. Repeated burning eventually reduced hardwood midstory competition and maintained less upper visual obstruction than herbicide only and controls. Without repeated perturbations, vegetative communities in herbicide only plots were based on initial species response to herbicide application. However, any species uncontrolled by imazapyr herbicide had a distinct competitive advantage (Iglay et al. 2010*a*).

Sites treated with imazapyr with or without fire favored imazapyr-tolerant plant species (Iglay et al. 2010*a*). These species, potentially dominating sites, significantly influenced visual obstruction. Hardwood midstory trees surviving imazapyr treatment also contributed to upper level visual obstruction in herbicide only and control plots. Upper level visual obstruction may have been reduced in herbicide only plots if greater rates of imazapyr were used (Wigley et al. 2002, Miller and Miller 2004, Guynn e t al. 2004). However, such increases also could be detrimental to pine tree growth and understory plant species diversity (Miller and Miller 2004, Guynn et al. 2004).

Forage class response varied by treatment type (prescribed fire and imazapyr herbicide), post-treatment vegetation structure, and competition within and among vegetative communities. Treatment extremes (burn + herbicide and controls) tended to segregate forage classes most whereas responses to independent applications remained intermediate to treatment extremes throughout most of the study. However, forage class responses sometimes favored a group of similar treatments (e.g., burn treatments or herbicide treatments) revealing direct effects of specific treatments.

Forbs and legumes were favored by dormant season prescribed burning with and without imazapyr. During summers following 2^{nd} and 3^{rd} burns, forb coverage increased in fire-treated plots. Similar increases in forbs immediately post-burn have been observed in past studies (Stransky and Harlow 1981, Sparks et al. 1998). Imazapyr application alone increased forb coverage by 2 years post-treatment but did not maintain greater forb coverage beyond initial release. Combining prescribed burning and imazapyr maintained greatest overall forb biomass across all years of study. Many forbs are early-successional species intolerant of shade but able to opportunistically establish themselves in disturbed areas. Frequent disturbances, as caused by shorter ($\leq$ 3 years) fire-return intervals, inevitably favor this type of life history strategy (Masters et al. 1993, Sparks et al. 1998). Legumes also favor early-successional environments but their fire-dependency, specifically seed scarification, would favor dormant season prescribed burning over imazapyr only (Grelen and Lewis 1981, Landers 1981, White et al. 1991, Masters et al. 1993, Sparks et al. 1998, Brennan et al. 1999). However, if improved seed catchment opportunities provided by treatments was the main driving force behind greater coverage and biomass of these forage classes then greater intensity treatments such as burn + herbicide would be optimal (Sousa 1984, Masters 1991*ab*, Masters et al. 1993, Sparks et al. 1998).

Grasses, sedges, and rushes had sporadic responses to treatments. Summer coverage of grasses supported combining fire and imazapyr. Because grass coverage in burn + herbicide was eventually greater than other treatments, fire and removal of hardwood midstory may interact to promote grass growth, possibly through reduction of canopy coverage (Peitz et al. 1999, Thompson et al. 1991). However, Waldrop et al. (1992) observed greater grass coverage due to greater treatment frequencies (annual burns versus 3 to 7-year fire return intervals). As with forbs and legumes, greater treatment intensities such as burn + herbicide instead of independent applications, may cause greater grass coverage. Sedges and rushes also favored burn + herbicide, most likely for many of the same reasons grasses respond well. However, subsample biomass collections were generally small (< 1.00 g) for most species of grasses, sedges, and rushes and bias from weighing light subsamples may have masked treatment effects. Large samples (> 10.00 g) generally occurred among a few dominant species such as shade-tolerant slender woodoats (*Chasmanthium laxum*; Iglay et al. 2010*a*). This species' ability to thrive under a closed canopy could have enabled its response to remain independent of treatments.

Semi-woody vines benefitted from low-rate application of imazapyr. Label recommendations for control of top contributing species to the semi-woody vine forage class (*Rubus argutus* and *Lonicera japonica*) suggested greater application rates than used in this study (BASF 2006, Iglay et al. 2010*a*). However, my low application rate controlled most competing forage classes such as woody plants and vines. Persistence of semi-woody vines after treatments were applied and their eventual domination of understory plant communities most likely influenced other vegetative responses such as species richness and proportional contributions of each species to forage classes (Iglay et al. 2010*a*).

Woody plant and vine forage classes were affected detrimentally by treatments. Low-rate imazapyr application seemed to adequately control woody species even though label recommendations for control of top-contributing species in woody forage classes were greater than those used (BASF 2006, Iglay et al. 2010*a*). However, according to winter biomass measurements, imazapyr did not appear to take full effect (complete kill) until the first growing season after application whereas prescribed burning was able to reduce woody biomass and overall cover immediately. Independent treatments were not as effective as burn + herbicide in controlling woody vegetation. The synergy of burn + herbicide may have been a function of initial, effective control by imazapyr prolonged by repeated dormant season burning (Edwards et al. 2004, Welch et al. 2004, Mixon et al. 2009, Iglay et al. 2010*a*).

Understory forage class response also was influenced by pronounced changes in canopy coverage and overall vegetative structure caused by treatments. Less canopy coverage may have supported greater biomass and coverage of forbs, legumes, and semi-woody vines (Thompson et al. 1991, Peitz et al. 1999, Sparks et al. 1998). Because burn + herbicide can maintain long-term reduced canopy coverage, herbaceous, shade-intolerant understory plant species inevitably have a greater window of time to establish. Repeated burns, whether with or without an initial imazapyr application, also can reduce stem height and litter depth and increase nutrient cycling. Because seed banks change over time due to seed viability and dispersal, management options promoting seed catchment opportunities (e.g., exposed soil, reduced competition for sunlight) such as repeated prescribed burns could support early-successional species. In forestry, thinnings can have a similar short-term effect by opening canopies and disturbing soil (Peitz et al.

1999, Thompson et al. 1991). However, understory plant competition is manipulated rarely beyond normal tree removal practices affecting canopy closure; whereas, hardwood midstory control methods such as those used in this study provide competition control at 2 vegetation levels, midstory and understory.

Changes in vegetation communities because of treatments also may have manipulated intra- and inter-specific competition. An underlining theme of vegetation management tools is competition control. Whether directly reducing competing species or creating ideal conditions for target species, vegetation management relies on a myriad of tools and site-specific prescriptions to manipulate stands to achieve management goals. Prescribed burning and imazapyr, independently and combined, manipulated vegetative communities in favor of specific groups of species. If treatments evenly impacted all understory plant species in addition to removing midstory hardwood trees, plant succession would most likely follow a typical relay floristics pattern (Egler 1954) or more commonly used climax concept (Clements 1936). However, differential control of well-established species enabled some to dominate sites and may have influenced vegetative community response (Collins et al. 1995, Iglay et al. 2010*a*). In this situation, an individualistic concept of plant association as proposed by Gleason (1926) is more appropriate and helps describe the long-term changes in vegetative communities of my sites.

Successional changes in vegetative communities on my sites were characteristic of each treatment. Control sites had relatively stable succession with a similar plant species composition throughout the study. Main fluctuations were caused by tree gaps and internal food plots created by weather events and hunters, respectively. Herbicide only sites received an initial successional set-back when biomass and coverage of species controlled by low-rate imazapyr application, such as hardwood midstory trees, were reduced. Well-established plants uncontrolled by imazapyr eventually outgrew and most likely outcompeted forbs and other shade-intolerant, early successional species (Iglay et al. 2010*a*). Burn only sites had continually changing forage class biomass and coverage with overall gradual changes in vegetative communities as the study progressed. Dynamic effects included sharp increases in forb coverage immediately post-burn, long-term reduction in midstory hardwood basal areas, and species introductions through expressions of the seed bank and dispersed seeds. Temporal changes of burn + herbicide sites were a reflection of burn only and herbicide only sites and treatment synergy. Woody plants detrimentally impacted by imazapyr application were removed quickly from sites by the initial dormant season burn. Species domination by uncontrolled well-established plant species occurred (Iglay et al. 2010*a*), but repeated burns created patchwork mosaics of understory vegetation communities according to differential fire intensities and plant tolerance to fire. Considering disturbance can perpetuate biodiversity (Turner et al. 2001, White and Jentsch 2001, Rundel et al. 1998), disturbance-tolerant species such as early successional species would benefit from repeated burns or multple herbicide applications.

Prescribed fire and imazapyr treatments created a mosaic of vegetation types in an intensively managed pine forest system and positively influenced species richness compared to controls. An initial decrease in species richness immediately post-treatment was expected because of effects of treatments on pre-treatment vegetative communities, specifically woody plant species (Jones and Chamberlain 2004, Miller and Miller 2004).

However, prescribed burning with or without imazapyr always promoted species richness especially after the 3rd burn, similar to past studies (Thanos et al. 1996, Sparks et al. 1998). The stochastic behavior of prescribed burns most likely supported greater species richness. Fluctuations in fuel load and moisture and in-stand abiotic conditions contribute to fire's unpredictable behavior (Wade and Lunsford 1989). Past studies have investigated effects of various burn condition characteristics on fire behavior, and standard prescribed burning recommendations are available (Wade and Lunsford 1989). However, heterogeneous burns across a landscape are inevitable, and vegetation communities respond accordingly.

Prescribed burning can have various effects on vegetative communities. In my study, prescribed burning was not as effective at removing midstory hardwood competition or woody plant species as imazapyr but was able to stimulate forbs and legume biomass. This combination of multiple vegetative responses to one treatment supports increased diversity of vegetative species and structure across the treated area. Applications of selective herbicides achieved by broadcast methods can cover the understory vegetation with chemical homogeneously compared to patchwork burnings of prescribed fire. However, herbicide activity can vary by soil conditions and chemicals used (Morrison and Meslow 1983, Miller and Witt 1990, Miller and Miller 1999, Miller and Miller 2004). When used independent of fire, imazapyr created vegetative communities with greater species richness than controls. However, combined with fire, imazapyr's ability to reduce woody plant biomass enabled fire to continually perpetuate a vegetative community of herbaceous, early-successional plant species (Masters et al. 1993).

Observed effects of prescribed burning and imazapyr on vegetative communities in this study indicate their potential to promote conservation of biodiversity within an intensively managed forest matrix. Bottom-up influenced systems greatly benefit from diverse vegetative communities such as those created by these treatments (Hunter 1990, Hunter and Price 1992, Power 1992). Variations in plant species and vegetation structure diversity across the landscape will inherently promote local and regional biodiversity indices if faunal community structure and diversity is related directly to vegetative response. Easily incorporated into standard intensive forestry management practices, prescribed burning and imazapyr applications at mid-rotation could help enhance current biodiversity levels within a working landscape.

Use of mid-rotation competition control methods benefits wildlife and forestry management. Progressive wildlife management by timber companies within the context of timber management goals may increase the appeal of timber lands for hunting leases and reduce management costs typically incurred by hunting leasers (e.g., food plot maintenance). Iglay et al. (2010*b*) and Mixon et al. (2009) found direct benefits of fire and herbicide application for deer forage. Summer biomass of moderate to high quality deer forages were greater in treated sites than traditionally managed stands (e.g., commercial thinning followed by fertilizing). Increased revenue from greater pine yield also can be a result of mid-rotation competition control (McInnis et al. 2004), but greater pine yield was not observed in this study (Nick Biasini, unpublished data, Mississippi State University; Jessica Smith 2004). However, reductions in woody vine biomass and midstory hardwood trees may assist pine tree harvesting by creating easier access to crop trees. Although many wildlife species depend on woody vines and hardwood trees, most species of concern in the southeastern United States prefer early seral stage vegetative communities resulting from repeated disturbances (Burger 2000). Forest managers could incorporate management for hardwood-dependent wildlife species within streamside management zones of intensively managed pine forest matrices.

Management Implications

Dormant season prescribed burning and imazapyr, combined and independently applied, can maintain or enhance biodiversity in mid-rotation, intensively managed pine forests of Mississippi by altering vegetative structure, coverage, and biomass. Combined, they compliment benefits of each treatment. As independent applications, they expand the manager's toolbox. Either way, delayed treatment effects (~ 2 yrs) should be expected and evaluations of management practices at ≥ 2 years post-treatment are recommended.

Managers are encouraged to use dormant season prescribed burning and imazapyr in similar situations but cautioned of the various effects possible. Prescribed burning is stochastic and managers are encouraged to adhere to site-specific prescriptions to ensure similar burn intensities as described in this study and reduce pine scorch (Wade and Lunsford 1989, Bessie and Johnson 1995, Schimmel and Granstrom 1997). Low-rate imazapyr applications can control midstory hardwood competition, but my study's results apply to only one application rate of one herbicide and one surfactant diluted with water. Selective herbicides can be applied in a variety of tank mixes at various application rates but need to be handled with care (Jones and Chamberlain 2004, Miller and Miller 2004). Most multiple herbicide tank mixtures have unknown effects and whenever used, vegetative response should be recorded to increase our knowledge of these management tools. Surfactants also come in a variety of forms and can not only affect herbicide uptake but also directly impact faunal communities (Tatum 2004). Although fire can be applied during dormant or growing seasons, I recommend using dormant season burns in similar conditions to avoid pine crown scorch and cause similar results observed here (Wade and Lunsford 1989, Sparks et al. 1998).

Prescribed fire and imazapyr impacts can vary by application conditions and site characteristics. Burn history and pre-treatment plant community composition can significantly impact vegetative response (Sparks et al. 1998, Iglay et al. 2010*a*). Managers should always assume treatment performance will be a function of these factors. Whenever possible, managers should seek information regarding treatment applications and consequential vegetative responses observed through scientific experimentation in similar management conditions.

Observed vegetative community responses to treatments in this study are most likely representative of general vegetative responses in intensively managed, mid-rotation pine stands of the southeastern United States. Six replicates per treatment reduced spatial heterogeneity and minimized intra-treatment variation. However, managers also should consider initial vegetative community structure and diversity when creating management prescriptions as these factors can affect vegetative response (Iglay et al. 2010*a*).

Even though none of the treatments described offer a best solution, prescribed burning seems to be a more effective management tool for conserving biodiversity. Its cost-effectiveness and multiple ecosystem benefits (e.g., fuel reduction, seed scarification, increased nutrient cycling) merit its use. However, selective herbicides play a vital role in providing effective competition control when prescribed burning is not feasible due to burn conditions (e.g., smoke management issues) or when dealing with invasive or exotic species (Wigley et al. 2002, Iglay et al. 2010*a*).

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Table 1.1 Vegetation structure variables and sampling unit labels used for principal components analysis to visualize differences among experimental units treated with factorial combinations of prescribed fire and imazapyr in intensively managed, mid-rotation pine stands of east-central Mississippi, 1999-2008.

Variable Code	Description
mid-pine BA	Mean midstory pine basal area
mid-pine HT	Mean height of midstory pine trees
pine BA	Mean overstory pine basal area
mid-hwd BA	Mean midstory hardwood basal area
mid-hwd HT	Mean height of midstory hardwood trees
hwd BA	Mean overstory hardwood basal area
ptsnag HT	Mean pine snag height
ptsnag BA	Mean pine snag basal area
hwdsnag HT	Mean hardwood snag height
hwdsnag BA	Mean hardwood snag basal area
Ccover	Percentage canopy cover
Nudds Low	Mean percentage cover of Nudds board sections 1-3 (0-90cm)
Nudds High	Mean percentage cover of Nudds board sections 4-6 (90-180cm)
Nudd4[1]	Mean percentage cover 4th Nudds board section (90-120cm)
Nudd5[1]	Mean percentage cover 5th Nudds board section (120-150cm)
Nudd6[1]	Mean percentage cover 6th Nudds board section (150-180cm)
DWD1[1]	Volume of downed woody debris in decay class 1
DWD2[1]	Volume of downed woody debris in decay class 2
DWD3[1]	Volume of downed woody debris in decay class 3
DWD4[1]	Volume of downed woody debris in decay class 4
DWD5[1]	Volume of downed woody debris in decay class 5
1BH	Burn + herbicide treatment plot
1B	Burn only treatment plot
1H	Herbicide only treatment plot
1C	Control treatment plot

[1] Vegetation structure variables for principal components analysis of winter data.

Table 1.2 Vegetation structure interactions and main effects with prescribed burning and imazapyr intensively managed, mid-rotation pine stands of Kemper County, Mississippi, summer 2000-2008, with pre-treatment data as a covariate.

	Pre-treatment		Treatment*Year		Treatment		Year	
Basal Area	*F*	*P*-Value	*F*	*P*-Value	*F*	*P*-Value	*F*	*P*-Value
Overstory Pine	13.24	≤ 0.001	0.89	0.610	6.80	≤ 0.001	49.09	≤ 0.001
Overstory Hardwood	158.64	≤ 0.001	1.80	0.025	2.80	0.070	24.59	≤ 0.001
Midstory Pine	41.08	≤ 0.001	1.44	0.117	0.78	0.524	1.00	0.448
Midstory Hardwood	29.33	≤ 0.001	4.47	≤ 0.001	10.79	≤ 0.001	10.73	≤ 0.001
Pine Snag	0.89	0.349	0.94	0.544	0.92	0.446	5.93	≤ 0.001
Hardwood Snag	0.73	0.409	0.87	0.637	1.06	0.406	4.72	≤ 0.001
Tree Height								
Midstory Pine	0.00	0.953	1.75	0.029	1.71	0.196	3.14	0.005
Midstory Hardwood	0.01	0.943	2.55	≤ 0.001	0.37	0.774	5.00	≤ 0.001
Pine Snag	2.37	0.131	1.28	0.195	0.70	0.557	8.56	≤ 0.001
Hardwood Snag	0.01	0.918	2.14	0.005	5.27	0.004	9.35	≤ 0.001
Nudds Board Section								
One	0.47	0.498	10.27	≤ 0.001	4.72	0.006	8.39	≤ 0.001
Two	0.06	0.815	7.53	≤ 0.001	2.70	0.057	18.51	≤ 0.001
Three	0.01	0.943	6.35	≤ 0.001	2.06	0.123	17.61	≤ 0.001
Four	0.60	0.441	5.23	≤ 0.001	9.97	≤ 0.001	11.80	≤ 0.001
Five	1.60	0.211	4.70	≤ 0.001	20.62	≤ 0.001	9.02	≤ 0.001
Six	2.24	0.147	4.21	≤ 0.001	25.90	≤ 0.001	8.86	≤ 0.001
Summary Nudds Board								
One-Three	0.01	0.926	8.13	≤ 0.001	1.38	0.263	15.78	≤ 0.001
Four-Six	1.09	0.309	4.83	≤ 0.001	19.07	≤ 0.001	10.37	≤ 0.001
Canopy Coverage	16.55	≤ 0.001	1.45	0.098	5.23	0.007	35.87	≤ 0.001

Table 1.3 Least square means (SE) of vegetation structure variables in intensively managed, mid-rotation pine stands treated with prescribed burning and imazapyr in Kemper County, Mississippi, summer 2000-2008.

Basal Area (m^2/ha)	Year	*P*-value	Burn $\bar{x}$	Herbicide $\bar{x}$	Burn + herbicide $\bar{x}$	Control $\bar{x}$	SE
Overstory Hardwood	1	0.870	0.71	0.67	0.61	0.73	0.12
	2	0.236	0.70	0.90	0.67	1.04	0.15
	3	0.589	0.76	0.30	0.47	0.98	0.38
	4	0.127	1.29	0.91	0.98	1.31	0.14
	5	0.442	1.27	1.09	0.88	1.32	0.21
	6[1]	0.039	1.27 AB	0.82 B	0.89 B	1.41 A	0.16
	7	0.085	1.55	1.25	1.17	1.99	0.24
	8	0.006	1.74 A	1.11 B	1.04 B	1.85 A	0.18
	9	0.051	1.21	0.55	0.83	2.14	0.41
Midstory Hardwood	1	0.003	1.05 AB	0.53 B	0.57 B	1.47 A	0.18
	2	≤ 0.001	1.04 B	0.80 B	0.57 B	1.60 A	0.17
	3	0.021	2.24 AB	0.66 C	0.94 BC	2.45 A	0.46
	4	≤ 0.001	0.91 B	0.57 BC	0.42 C	1.82 A	0.15
	5	0.301	1.10	1.19	1.59	3.11	0.82
	6	≤ 0.001	1.63 B	1.16 BC	0.59 C	3.07 A	0.25
	7	≤ 0.001	1.43 B	1.34 BC	0.54 C	3.44 A	0.28
	8	≤ 0.001	1.07 B	1.13 B	0.40 C	2.14 A	0.21
	9	≤ 0.001	0.96 BC	1.11 B	0.43 C	2.94 A	0.22
Tree Height (m)							
Midstory Pine	1	0.264	2.03	1.65	0.01	2.30	0.92
	2	0.445	4.29	1.33	0.01	4.88	2.46
	3	0.213	1.06	4.00	0.01	1.20	1.36
	4	0.279	-0.01	0.76	0.01	0.74	0.48
	5	0.171	-0.01	0.33	0.01	1.21	0.53
	6	0.225	3.83	1.46	0.01	1.35	1.30
	7	0.109	1.07	1.97	0.01	1.32	0.62
	8	0.044	1.13 AB	1.96 A	0.01 B	1.90 AB	0.61
	9	0.145	1.41	1.69	0.01	1.78	0.67

Table 1.3 (continued)

Tree Height (m)	Year	*P*-value	Burn $\bar{x}$	Herbicide $\bar{x}$	Burn + herbicide $\bar{x}$	Control $\bar{x}$	SE
Midstory Hardwood	1	≤ 0.001	3.97 B	5.52 A	6.07 A	3.64 B	0.42
	2	0.145	4.70	6.00	6.00	3.96	0.72
	3	0.954	6.28	6.78	5.65	6.67	1.54
	4	0.482	4.51	3.86	3.67	3.92	0.40
	5	0.876	5.56	4.83	3.87	4.81	1.46
	6	0.874	4.30	4.17	4.11	4.35	0.25
	7	≤ 0.001	5.49 A	4.11 B	5.37 A	4.53 B	0.24
	8	0.535	5.53	4.73	4.53	4.68	0.53
	9	0.036	5.01 A	4.71 A	3.73 B	5.04 A	0.34
Hardwood Snag	1	≤ 0.001	1.67 B	10.03 A	7.32 A	0.81 B	1.12
	2	0.171	3.31	9.51	7.42	0.01	3.14
	3	0.079	1.95	5.03	9.67	0.01	2.63
	4	0.997	2.81	2.75	3.26	2.84	1.89
	5	0.537	1.31	1.51	0.67	0.01	0.79
	6	0.389	1.65	1.67	2.82	0.01	1.14
	7	0.673	0.41	2.19	1.75	1.42	1.04
	8	0.223	1.73	1.86	2.72	0.01	0.92
	9	0.506	-0.02	0.48	0.65	0.01	0.38
Nudds Board Section (% covered)							
One	1	≤ 0.001	94 A	78 B	69 C	98 A	2
	2	0.584	86	88	93	95	5
	3	0.935	92	93	94	92	3
	4	0.006	89 B	97 A	91 B	86 B	2
	5	0.004	90 B	97 A	99 A	89 B	2
	6	0.020	84 B	89 AB	93 A	81 B	3
	7	0.002	83 B	95 A	90 AB	82 B	3
	8	≤ 0.001	87 B	89 B	96 A	71 C	2
	9	≤ 0.001	87 AB	82 B	91 A	71 C	3
Two	1	≤ 0.001	82 A	58 B	51 B	89 A	4
	2	0.588	85	76	84	89	7
	3	0.663	86	89	91	86	4
	4	0.005	80 B	95 A	85 B	84 B	3
	5	0.006	85 B	94 A	95 A	81 B	3
	6	0.006	75 BC	83 AB	89 A	73 C	4
	7	0.002	66 B	88 A	76 B	67 B	4
	8	≤ 0.001	73 B	77 AB	83 A	54 C	4
	9	0.007	72 A	70 A	74 A	57 B	4

Table 1.3 (continued)

Nudds Board	Year	P-value	Burn	Herbicide	Burn + herbicide	Control	SE
			$\bar{x}$	$\bar{x}$	$\bar{x}$	$\bar{x}$	
Three	1	$\leq$ 0.001	70 B	51 C	35 D	84 A	4
	2	0.445	81	66	70	80	8
	3	0.916	83	82	81	80	4
	4	$\leq$ 0.001	67 B	88 A	66 B	80 A	4
	5	0.052	73	88	85	77	4
	6	0.070	64	74	80	70	4
	7	$\leq$ 0.001	54 B	85 A	58 B	64 B	5
	8	0.035	62 AB	70 A	67 A	52 B	6
	9	0.479	61	61	61	54	4
Four	1	$\leq$ 0.001	60 B	41 C	27 D	77 A	5
	2	$\leq$ 0.001	80 A	54 B	57 B	75 A	5
	3	0.441	74	76	67	78	5
	4	$\leq$ 0.001	54 B	80 A	51 B	75 A	5
	5	0.065	66	83	71	72	5
	6	0.262	58	69	67	70	5
	7	$\leq$ 0.001	46 C	79 A	40 C	65 B	5
	8	0.253	54	64	55	52	5
	9	0.874	53	55	50	55	5
Five	1	$\leq$ 0.001	53 B	35 C	22 C	73 A	5
	2	$\leq$ 0.001	76 A	44 B	42 B	72 A	5
	3	0.032	69 AB	70 AB	57 B	78 A	5
	4	$\leq$ 0.001	45 B	72 A	40 B	72 A	5
	5	0.016	55 B	76 A	58 B	67 AB	5
	6	0.084	51	63	57	69	5
	7	$\leq$ 0.001	35 B	76 A	30 B	64 A	5
	8	0.162	44	58	44	49	5
	9	0.112	45	54	40	55	5
Six	1	$\leq$ 0.001	48 B	29 C	20 C	65 A	5
	2	$\leq$ 0.001	68 A	38 B	34 B	68 A	5
	3	0.002	63 A	62 A	46 B	75 A	5
	4	$\leq$ 0.001	39 B	65 A	33 B	68 A	5
	5	$\leq$ 0.001	46 B	73 A	45 B	62 A	5
	6	0.020	47 B	59 AB	51 B	69 A	5
	7	$\leq$ 0.001	32 B	74 A	25 B	63 A	5
	8	0.019	40 BC	56 A	36 C	53 AB	5
	9	0.011	42 AB	55 A	35 B	56 A	5

Table 1.3 (continued)

Nudds Board	Year	*P*-value	Burn $\bar{x}$	Herbicide $\bar{x}$	Burn + herbicide $\bar{x}$	Control $\bar{x}$	SE
Low (sections 1-3)	1	≤ 0.001	82 A	63 B	52 C	90 A	3
	2	0.641	84	77	82	88	6
	3	0.929	87	88	88	86	3
	4	0.004	79 B	93 A	81 B	84 B	3
	5	0.013	83 B	93 A	93 A	82 B	3
	6	0.017	74 B	82 AB	88 A	75 B	4
	7	≤ 0.001	68 B	89 A	74 B	71 B	4
	8	≤ 0.001	74 A	79 A	82 A	59 B	3
	9	0.012	73 A	71 A	75 A	61 B	3
High (sections 4-6)	1	≤ 0.001	53 B	35 C	23 C	72 A	5
	2	≤ 0.001	74 A	46 B	44 B	72 A	5
	3	0.032	69 AB	69 AB	57 B	77 A	5
	4	≤ 0.001	46 B	72 A	41 B	72 A	5
	5	0.007	55 B	77 A	58 B	67 B	5
	6	0.083	52	64	58	69	5
	7	≤ 0.001	38 B	76 A	32 B	64 A	5
	8	0.133	46	60	45	51	5
	9	0.145	47	55	42	55	5

[1] Treatments with the same letter do not differ significantly ($P > 0.05$).

Table 1.4 Interaction and main effects of year and treatment on relative coverage of understory plants in intensively managed, mid-rotation pine stands treated with prescribed burning, imazapyr, burning and imazapyr, or control in Kemper County, Mississippi, May-June 2000-2008.

	Pre-treatment		Treatment × Year		Treatment		Year	
Coverage variable	F	P-Value	F	P-Value	F	P-Value	F	P-Value
Bare Ground	100.70	≤ 0.001	0.72	0.827	1.50	0.246	3.99	≤ 0.001
Debris	0.11	0.737	12.95	≤ 0.001	3.12	0.037	52.70	≤ 0.001
Fern	30.84	≤ 0.001	1.29	0.186	1.20	0.321	0.31	0.960
Forbs	2.22	0.176	4.01	≤ 0.001	14.49	≤ 0.001	13.02	≤ 0.001
Grasses	0.13	0.725	3.45	≤ 0.001	10.42	≤ 0.001	16.03	≤ 0.001
Grass-like	3.80	0.055	5.73	≤ 0.001	9.25	≤ 0.001	21.66	≤ 0.001
Herbaceous Vine	2.88	0.127	1.39	0.159	1.69	0.205	25.97	≤ 0.001
Legumes	0.00	0.973	1.49	0.091	2.83	0.055	17.11	≤ 0.001
Semi-woody Vine	2.16	0.151	4.99	≤ 0.001	32.40	≤ 0.001	21.57	≤ 0.001
Woody Plants	34.40	≤ 0.001	5.67	≤ 0.001	16.41	≤ 0.001	16.40	≤ 0.001
Woody Vine	1.61	≤ 0.001	3.14	≤ 0.001	24.33	≤ 0.001	34.97	≤ 0.001
Species Richness	0.15	0.705	8.94	≤ 0.001	14.46	≤ 0.001	116.60	≤ 0.001

Table 1.5 Least square means (SE) of understory vegetation relative coverage (%) among plant forage classes in intensively managed, mid-rotation pine treated with prescribed burning and imazapyr in Kemper County, Mississippi, USA, May-June 2000-2008.

			Burn	Herbicide	Burn + herbicide	Control	
Coverage variable	Year	P-value	$\bar{x}$	$\bar{x}$	$\bar{x}$	$\bar{x}$	SE
Debris	1[1]	≤ 0.001	13.8 C	50.8 B	61.3 A	6.2 C	3.1
	2	0.002	4.2 B	16.3 A	8.4 B	4.0 B	2.2
	3	0.367	8.6	8.5	5.8	10.6	1.9
	4	0.016	18.9 AB	8.5 C	11.3 BC	24.3 A	3.6
	5	0.314	14.0	12.3	9.3	18.9	3.6
	6	≤ 0.001	16.2 BC	20.8 AB	8.2 C	25.9 A	2.2
	7	0.133	23.5	19.9	17.8	30.5	3.9
	8	≤ 0.001	21.7 BC	24.6 B	14.6 C	33.3 A	2.9
	9	0.003	25.3 BC	33.8 AB	21.2 C	43.1 A	3.9
Forbs	1	0.002	3.2 B	0.2 B	6.4 A	1.4 B	0.9
	2	0.004	11.1 BC	16.4 AB	24.3 A	7.0 C	3.1
	3	0.864	3.7	0.4	3.7	2.7	1.1
	4	≤ 0.001	23.2 A	5.3 B	28.3 A	3.3 B	4.3
	5	≤ 0.001	2.1 B	0.9 B	7.6 A	0.4 B	0.9
	6	≤ 0.001	3.1 B	0.9 B	7.7 A	1.3 B	1.2
	7	0.002	9.2 A	0.7 B	10.7 A	0.8 B	2.1
	8	0.007	2.1 AB	0.5 B	4.4 A	0.5 B	0.8
	9	0.064	2.3	1.0	2.0	0.5	0.5
Grasses	1	0.002	21.0 A	6.8 B	6.0 B	18.5 A	3.0
	2	0.366	20.1	13.4	17.9	14.4	3.0
	3	0.038	6.0 B	9.7 AB	15.4 A	3.4 B	2.9
	4	0.129	3.0	5.9	7.1	2.1	1.6
	5	0.067	4.4	1.0	6.2	0.6	1.6
	6	0.011	12.2 AB	7.3 BC	16.6 A	5.1 C	2.4
	7	≤ 0.001	10.7 B	7.5 BC	19.8 A	4.1 C	1.5
	8	≤ 0.001	10.0 B	5.6 BC	20.8 A	3.8 C	1.6
	9	≤ 0.001	10.5 B	4.1 C	16.9 A	2.9 C	1.4

Table 1.5 (continued)

Coverage variable	Year	P-value	Burn $\bar{x}$	Herbicide $\bar{x}$	Burn + herbicide $\bar{x}$	Control $\bar{x}$	SE
Grass-like	1	≤ 0.001	10.2 A	3.5 C	2.6 C	5.1 B	0.8
	2	0.314	2.9	1.5	0.8	1.8	0.8
	3	≤ 0.001	1.8 B	2.6 B	9.7 A	1.3 B	0.8
	4	≤ 0.001	0.5 B	0.1 B	4.7 A	0.0 B	0.8
	5	1.000	0.0	0.1	0.0	0.0	0.8
	6	0.618	0.1	0.1	1.3	0.0	0.8
	7	1.000	0.0	0.1	0.0	0.0	0.8
	8	0.654	0.9	0.3	1.6	0.3	0.8
	9	0.267	0.4	0.4	2.2	0.3	0.8
Semi-woody Vines	1	0.206	23.3	24.4	16.3	20.1	3.5
	2	0.841	30.6	31.6	29.4	28.1	3.5
	3	≤ 0.001	32.7 B	46.9 A	42.9 A	21.9 C	3.5
	4	≤ 0.001	26.7 C	57.8 A	38.0 B	24.5 C	3.5
	5	≤ 0.001	32.1 B	49.7 A	55.1 A	21.4 C	3.5
	6	≤ 0.001	23.6 B	35.4 A	41.1 A	16.0 B	3.5
	7	≤ 0.001	17.7 B	33.7 A	30.0 A	11.6 B	3.5
	8	≤ 0.001	19.3 B	26.2 AB	30.2 A	11.1 C	3.5
	9	0.011	18.0AB	22.4 A	24.2 A	11.3 B	3.5
Woody Plants	1	≤ 0.001	20.7 B	6.3 C	3.4 C	35.6 A	3.0
	2	≤ 0.001	22.5 B	10.9 C	17.0 BC	33.0 A	3.1
	3	≤ 0.001	17.7 A	8.6 B	5.9 B	16.7 A	2.1
	4	0.978	6.4	6.7	6.0	5.8	1.6
	5	0.013	9.5 A	8.5 A	5.7 AB	3.4 B	1.4
	6	0.488	14.1	13.5	10.8	10.5	2.1
	7	0.002	9.2 BC	13.7 A	6.6 C	10.0 B	1.2
	8	0.002	14.2 A	16.4 A	7.0 B	17.3 A	1.8
	9	0.149	12.1	9.7	9.4	8.0	1.3
Woody Vines	1	≤ 0.001	4.6 B	3.4 B	1.5 B	11.9 A	1.4
	2	≤ 0.001	6.5 B	7.4 B	1.7 C	11.6 A	1.1
	3	≤ 0.001	17.9 B	12.3 B	7.2 B	39.5 A	4.0
	4	≤ 0.001	16.9 B	13.5 BC	4.4 C	40.1 A	4.1
	5	≤ 0.001	27.6 B	23.2 BC	10.0 C	51.6 A	4.8
	6	≤ 0.001	20.4 B	16.7 BC	8.9 C	37.4 A	3.0
	7	≤ 0.001	17.9 B	18.5 B	8.7 C	38.9 A	3.0
	8	≤ 0.001	21.2 B	20.3 B	11.7 C	30.2 A	2.1
	9	≤ 0.001	21.8 BC	24.2 B	16.2 C	31.2 A	2.4

[1] Treatments with the same letter do not differ significantly ($P > 0.05$).

Table 1.6 Species richness of understory vegetation sampled by line intercepts in intensively managed, mid-rotation pine stands treated with prescribed burning and imazapyr in Kemper County, Mississippi, USA, May-June 2000-2008.

		Burn	Herbicide	Burn + herbicide	Control	
Year	P-value	$\bar{x}$	$\bar{x}$	$\bar{x}$	$\bar{x}$	SE
1[1]	≤ 0.001	15.2 A	8.8 B	8.0 B	16.2 A	0.8
2	0.005	17.2 A	12.1 B	12.6 B	14.3 B	1.0
3	0.564	9.8	8.6	8.6	9.1	0.7
4	≤ 0.001	8.3 A	5.9 C	7.0 B	5.3 C	0.4
5	0.010	7.4 A	5.6 B	5.7 B	5.2 B	0.5
6	0.002	12.3 A	9.6 B	11.7 A	8.9 B	0.7
7	≤ 0.001	13.2 A	10.5 B	13.6 A	9.6 B	0.6
8	≤ 0.001	12.8 A	10.8 B	12.9 A	8.8 C	0.6
9	≤ 0.001	15.5 A	11.8 B	16.2 A	8.9 C	0.8

[1] Treatments with the same letter do not differ significantly ($P > 0.05$).

Table 1.7 Vegetation structure interactions and main effects with prescribed burning and imazapyr in intensively managed, mid-rotation pine stands of Kemper County, Mississippi, winter 2000-2007, with pre-treatment data as a covariate.

	Pre-treatment		Treatment × year		Treatment		Year	
Basal Area	F	P-Value	F	P-Value	F	P-Value	F	P-Value
Overstory Pine	7.34	0.009	0.89	0.599	1.84	0.161	43.95	≤ 0.001
Overstory Hardwood	0.19	0.661	1.19	0.276	2.04	0.131	5.33	≤ 0.001
Midstory Pine	0.01	0.943	1.35	0.164	2.30	0.089	2.37	0.032
Midstory Hardwood	2.75	0.113	5.33	≤ 0.001	7.66	≤ 0.001	31.78	≤ 0.001
Pine Snag	0.21	0.649	0.68	0.847	0.05	0.984	4.73	≤ 0.001
Hardwood Snag	21.89	≤ 0.001	1.39	0.146	2.15	0.111	3.57	0.003
Tree Height								
Midstory Pine	0.03	0.856	1.06	0.406	4.22	0.011	2.69	0.016
Midstory Hardwood	5.07	0.036	1.81	0.028	0.95	0.434	14.60	≤ 0.001
Pine Snag	0.62	0.436	1.34	0.154	0.80	0.498	0.77	0.616
Hardwood Snag	3.58	0.066	1.05	0.409	5.00	0.004	6.27	≤ 0.001
Nudds Board Section								
One	1.58	0.231	8.65	≤ 0.001	11.49	≤ 0.001	61.53	≤ 0.001
Two	0.40	0.533	3.02	≤ 0.001	11.39	≤ 0.001	69.73	≤ 0.001
Three	1.53	0.228	2.53	≤ 0.001	10.98	≤ 0.001	70.17	≤ 0.001
Four	4.33	0.047	1.80	0.028	12.42	≤ 0.001	59.78	≤ 0.001
Five	0.09	0.767	1.41	0.132	19.53	≤ 0.001	40.23	≤ 0.001
Six	1.71	0.237	1.47	0.107	24.44	≤ 0.001	35.72	≤ 0.001
Summary Nudds Board								
One-Three	0.30	0.587	3.38	≤ 0.001	14.25	≤ 0.001	67.16	≤ 0.001
Four-Six	6.25	0.015	1.42	0.126	17.60	≤ 0.001	43.10	≤ 0.001
Canopy Cover	4.70	0.040	1.31	0.186	1.09	0.374	39.94	≤ 0.001
Downed Woody Debris								
Class 1	2.64	0.108	0.62	0.903	0.79	0.501	4.61	≤ 0.001
Class 2	2.24	0.139	1.73	0.039	0.32	0.812	2.80	0.013
Class 3	0.00	0.961	1.52	0.087	3.28	0.027	6.08	≤ 0.001
Class 4	0.42	0.517	1.16	0.290	1.53	0.213	0.82	0.570
Class 5	1.38	0.251	0.00	1.000	0.00	1.000	0.00	1.000
Total	2.59	0.111	1.01	0.455	2.70	0.054	4.00	≤ 0.001

Table 1.8 Least square means (SE) of vegetation structure variables in intensively managed, mid-rotation pine stands treated with prescribed burning and imazapyr in Kemper County, Mississippi, winter 2000-2007.

Basal Area (m^2/ha)	Year	P-value	Burn $\bar{x}$	Herbicide $\bar{x}$	Burn + herbicide $\bar{x}$	Control $\bar{x}$	SE
Midstory Hardwood	1	0.287	1.91	1.73	1.00	1.66	0.34
	2[1]	0.024	1.14 A	0.76 AB	0.30 B	1.20 A	0.21
	3	≤ 0.001	1.28 B	0.99 BC	0.51 C	2.11 A	0.24
	4	0.235	0.87	0.73	0.47	1.29	0.28
	5	0.085	0.10	0.02	0.14	0.28	0.07
	6	0.926	0.22	0.21	0.15	0.25	0.11
	7	≤ 0.001	1.29 B	1.34 B	0.47 C	2.78 A	0.25
	8	≤ 0.001	1.03 B	1.10 B	0.21 C	2.43 A	0.22
Tree Height							
Midstory Hardwood	1	0.200	3.82	3.55	3.57	3.35	0.15
	2	0.417	4.11	4.48	3.60	3.62	0.43
	3	0.003	4.04 B	4.61 B	5.88 A	3.98 B	0.36
	4	0.751	4.65	4.19	3.95	4.22	0.46
	5	0.352	2.74	2.64	2.17	3.69	0.60
	6	0.075	4.36	3.82	2.43	3.83	0.51
	7	0.641	5.70	5.27	4.82	5.55	0.52
	8	0.299	5.53	4.40	4.92	4.61	0.44
Nudds' Board Section (% covered)							
One	1	≤ 0.001	33 C	58 B	28 C	68 A	3
	2	≤ 0.001	43 B	53 A	26 C	50 AB	3
	3	0.745	73	81	81	79	6
	4	≤ 0.001	30 B	66 A	28 B	63 A	5
	5	0.023	59 B	76 A	78 A	67 AB	4
	6	≤ 0.001	54 B	77 A	76 A	68 A	4
	7	0.010	22 B	50 A	22 B	42 A	7
	8	0.020	44 B	65 A	58 A	56 AB	4
Two	1	≤ 0.001	26 B	40 A	22 B	49 A	5
	2	0.032	31 AB	36 A	20 B	37 A	5
	3	0.471	63	73	70	69	5
	4	≤ 0.001	29 B	59 A	24 B	57 A	5
	5	0.040	53 B	68 A	71 A	62 AB	5
	6	≤ 0.001	40 B	66 A	62 A	58 A	5
	7	0.003	18 B	38 A	17 B	33 A	5
	8	0.140	30	43	39	43	5

Table 1.8 (continued)

Nudds Board	Year	P-value	Burn $\bar{x}$	Herbicide $\bar{x}$	Burn + herbicide $\bar{x}$	Control $\bar{x}$	SE
Three	1	0.002	25 BC	33 AB	20 C	43 A	4
	2	0.035	27 A	30 A	15 B	31 A	4
	3	0.413	56	66	60	63	4
	4	$\leq$ 0.001	24 B	48 A	18 B	49 A	4
	5	0.040	47 B	63 A	62 A	61 A	4
	6	0.000	29 B	55 A	46 A	51 A	4
	7	0.016	16 BC	30 A	13 C	28 AB	4
	8	0.069	26	39	29	40	4
Four	1	0.009	26 B	28 AB	19 B	39 A	4
	2	0.031	24 A	25 A	12 B	29 A	4
	3	0.370	50	57	51	59	4
	4	$\leq$ 0.001	23 B	37 A	14 B	44 A	4
	5	0.008	43 B	58 A	51 AB	62 A	4
	6	$\leq$ 0.001	24 B	51 A	34 B	48 A	4
	7	0.032	16 AB	24 A	11 B	26 A	4
	8	0.001	24 B	40 A	26 B	44 A	4
Nudds' Board Summary (% covered)							
Low (sections 1-3)	1	$\leq$ 0.001	28 B	44 A	23 B	53 A	4
	2	0.006	34 A	40 A	20 B	39 A	4
	3	0.511	64	73	70	70	4
	4	$\leq$ 0.001	28 B	57 A	23 B	56 A	4
	5	0.032	53 B	68 A	70 A	63 AB	4
	6	$\leq$ 0.001	41 B	71 A	61 A	59 A	4
	7	$\leq$ 0.001	19 B	39 A	17 B	34 A	4
	8	0.072	34	49	42	46	4
Downed Woody Debris							
Class 2	1	0.048	0.75B	1.48 B	4.24 A	0.95 B	0.93
	2	0.260	1.70	1.08	4.06	1.77	1.10
	3	0.616	2.32	0.45	0.49	1.32	1.13
	4	0.178	0.34	0.10	0.26	1.36	0.44
	5	0.631	0.77	0.41	1.15	-0.02	0.66
	6	0.051	0.42	0.10	-0.07	1.35	0.38
	7	0.202	2.69	1.43	-0.10	2.39	0.98
	8	0.242	0.18	1.33	0.06	-0.02	0.52

[1] Treatments with the same letter do not differ significantly ($P > 0.05$).

Table 1.9 Interaction and main effects of year and treatment on plant biomass of understory plants in intensively managed, mid-rotation pine stands treated with prescribed burning, imazapyr, burning and imazapyr, or control in Kemper County, Mississippi 2000-2008.

	Pre-treatment		Treatment × year		Treatment		Year	
Summer[1]	*F*	*P*-Value	*F*	*P*-Value	*F*	*P*-Value	*F*	*P*-Value
Fern	0.54	0.466	1.11	0.347	0.97	0.416	1.95	0.065
Forbs	1.95	0.175	1.31	0.179	5.41	0.009	4.00	≤ 0.001
Grasses	0.95	0.332	1.08	0.373	1.31	0.277	1.63	0.119
Grass-like	0.00	0.971	2.39	≤ 0.001	4.06	0.013	12.98	≤ 0.001
Herbaceous Vine	0.00	0.973	1.69	0.037	1.87	0.160	3.36	0.003
Legumes	0.09	0.764	1.45	0.104	4.70	0.009	9.03	≤ 0.001
Semi-woody Vine	9.39	0.005	1.26	0.204	3.95	0.020	3.65	≤ 0.001
Woody Plants	0.34	0.564	4.87	≤ 0.001	2.49	0.092	16.72	≤ 0.001
Woody Vine	14.47	≤ 0.001	1.51	0.078	1.89	0.155	13.30	≤ 0.001
Species Richness	1.73	0.199	2.50	≤ 0.001	11.95	≤ 0.001	74.15	≤ 0.001
Winter[2]								
Fern	0.00	0.000	1.07	0.402	1.20	0.370	1.09	0.388
Grasses	18.19	≤ 0.001	3.77	≤ 0.001	5.10	0.004	10.65	≤ 0.001
Grass-like	0.05	0.824	9.42	≤ 0.001	0.93	0.435	53.50	≤ 0.001
Herbaceous Vine	0.00	1.000	1.12	0.333	0.52	0.672	1.47	0.183
Legumes	0.00	1.000	1.10	0.362	1.18	0.352	1.27	0.278
Semi-woody Vine	0.00	0.955	2.95	≤ 0.001	0.95	0.433	19.04	≤ 0.001
Woody Plants	5.63	0.021	6.13	≤ 0.001	9.07	≤ 0.001	33.20	≤ 0.001
Woody Vine	0.71	0.408	3.38	≤ 0.001	9.50	≤ 0.001	14.61	≤ 0.001
Species Richness	0.45	0.519	10.74	≤ 0.001	6.22	0.013	59.50	≤ 0.001

[1]Summer plant biomass was collected July 2000-2008.

[2]Winter plant biomass was collected January-February 2000-2007.

Table 1.10 Least square means (SE) of understory plant biomass in intensively managed, mid-rotation pine stands treated with prescribed burning and imazapyr in Kemper County, Mississippi, July 2000-2008.

			Burn	Herbicide	Burn + herbicide	Control	
Forage Class	Year	P-value	$\bar{x}$	$\bar{x}$	$\bar{x}$	$\bar{x}$	SE
Grass-like	1	0.016	17	10	6	18	3
	2	0.124	63	20	74	28	18
	3[1]	0.045	13 AB	19 AB	31 A	2 B	7
	4	0.107	4	5	14	5	3
	5	0.350	0	0	3	0	1
	6	0.245	18	7	9	3	5
	7	0.373	16	1	3	3	7
	8	0.367	10	6	11	3	3
	9	0.001	7 B	4 B	19 A	2 B	3
Herbaceous Vine	1	0.183	2	1	1	0	1
	2	0.508	2	3	2	5	2
	3	0.170	69	17	26	17	18
	4	0.225	2	2	9	4	3
	5	0.205	1	5	4	1	2
	6	0.046	5	3	13	2	3
	7	0.099	3	2	13	1	4
	8	0.032	3 B	3 B	11 A	0 B	3
	9	0.181	2	2	5	1	1
Woody Plants	1	≤ 0.001	201 B	68 BC	9 C	458 A	49
	2	0.005	377 A	213 AB	98 B	365 A	57
	3	0.369	195	102	185	205	45
	4	0.871	27	18	29	23	10
	5	0.635	98	50	68	66	26
	6	0.052	140	100	77	49	22
	7	0.035	73 B	164 A	45 B	42 B	31
	8	0.023	171 AB	217 A	75 B	86 B	35
	9	0.522	166	220	177	116	49
Species Richness	1	≤ 0.001	43 A	29 C	35 B	35 B	2
	2	0.019	58 A	45 B	50 B	48 B	3
	3	0.324	39	33	36	36	2
	4	0.108	23	23	30	20	3
	5	0.176	21	17	23	19	2
	6	0.240	36	30	33	29	3
	7	0.023	38 AB	35 B	46 A	31 B	3
	8	≤ 0.001	55 A	43 B	48 AB	32 C	4
	9	≤ 0.001	55 A	40 B	58 A	40 B	3

[1] Treatments with the same letter do not differ significantly ($P > 0.05$).

Table 1.11 Least square means (SE) of understory plant biomass in intensively managed, mid-rotation pine stands treated with prescribed burning and imazapyr in Kemper County, Mississippi, winter 2000-2008.

			Burn	Herbicide	Burn + herbicide	Control	
Forage Class	Year	P-value	$\bar{x}$	$\bar{x}$	$\bar{x}$	$\bar{x}$	SE
Grasses	1	0.002	2 B	3 B	0 B	30 A	6
	2	0.076	2	0	3	1	1
	3[1]	0.008	34 AB	56 A	67 A	5 B	12
	4	0.182	0	15	3	1	5
	5	0.170	6	7	13	0	4
	6	0.019	10 B	4 B	29 A	6 B	6
	7	0.117	0	3	1	1	1
	8	0.003	8 A	1 B	7 A	1 B	2
Grass-like	1	0.292	2	12	0	21	9
	2	0.889	7	2	1	10	9
	3	$\leq$ 0.001	107 A	66 B	114 A	36 C	9
	4	1.000	0	0	0	0	9
	5	0.961	2	4	7	0	9
	6	0.998	2	0	0	0	9
	7	1.000	0	0	0	0	9
	8	0.983	3	2	5	0	9
Semi-woody Vine	1	$\leq$ 0.001	5 B	120 A	1B	193 A	33
	2	0.347	144	102	72	109	27
	3	0.730	685	645	579	519	111
	4	0.112	102	145	245	70	51
	5	0.012	63 B	213 A	232 A	93 B	40
	6	0.437	210	287	246	135	67
	7	0.436	8	51	58	9	27
	8	0.336	191	167	98	53	58
Woody Plants	1	$\leq$ 0.001	39 B	145 A	6 B	158 A	23
	2	$\leq$ 0.001	77 B	40 BC	16 C	151 A	16
	3	0.003	254 A	154 AB	63 B	230 A	34
	4	0.350	218	112	79	99	58
	5	0.178	17	10	19	0	9
	6	0.184	11	78	22	37	23
	7	0.183	26	3	0	0	11
	8	0.390	16	18	20	0	10

Table 1.11 (continued)

Forage Class	Year	P-value	Burn	Herbicide	Burn + herbicide	Control	SE
			$\bar{x}$	$\bar{x}$	$\bar{x}$	$\bar{x}$	
Woody Vine	1	0.000	2 B	88 A	1 B	114 A	18
	2	0.013	12 B	31 B	1 B	83 A	17
	3	0.101	205	117	20	190	55
	4	0.018	20 B	22 B	5 B	59 A	11
	5	0.239	3	6	0	15	5
	6	0.005	10 B	22 B	16 B	64 A	10
	7	0.058	0	4	0	3	1
	8	0.331	14	31	2	10	11
Species Richness	1	≤ 0.001	4 B	9 A	1 C	9 A	1
	2	0.004	7 A	5 B	4 B	6 A	1
	3	0.006	8 A	7 A	4 B	6 AB	1
	4	0.221	4	5	4	6	1
	5	0.115	7	6	6	5	1
	6	0.370	7	7	8	7	1
	7	0.001	1 B	4 A	1 B	3 A	1
	8	≤ 0.001	13 A	9 B	11 A	8 B	1

[1] Treatments with the same letter do not differ significantly ($P > 0.05$).

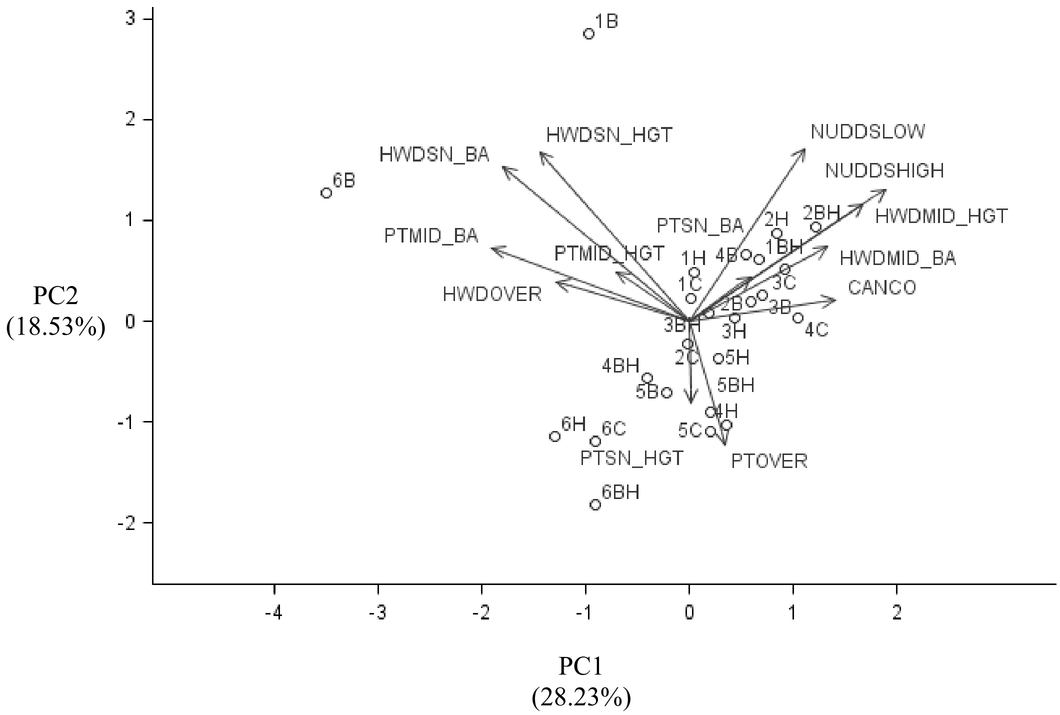
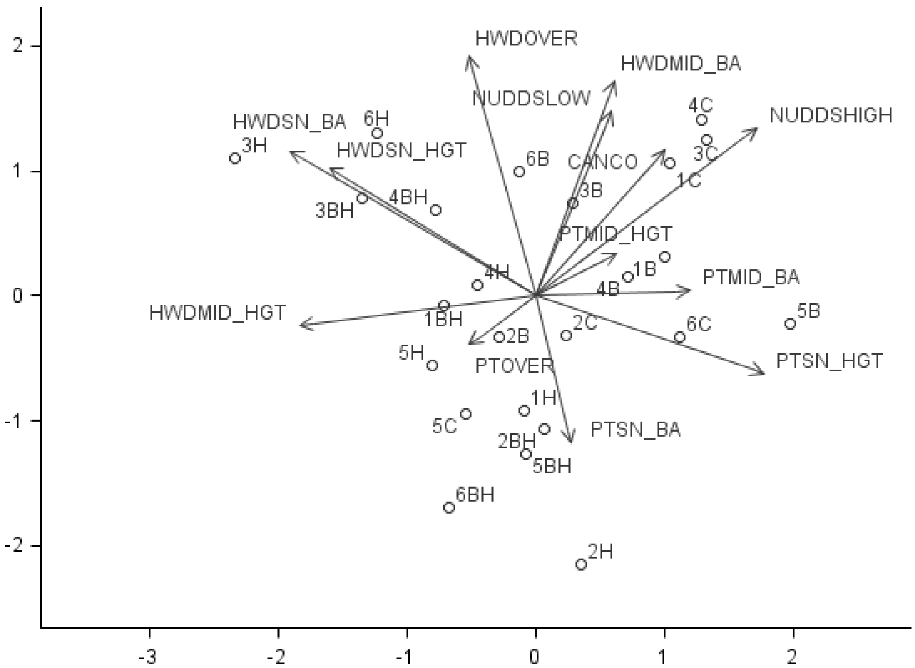


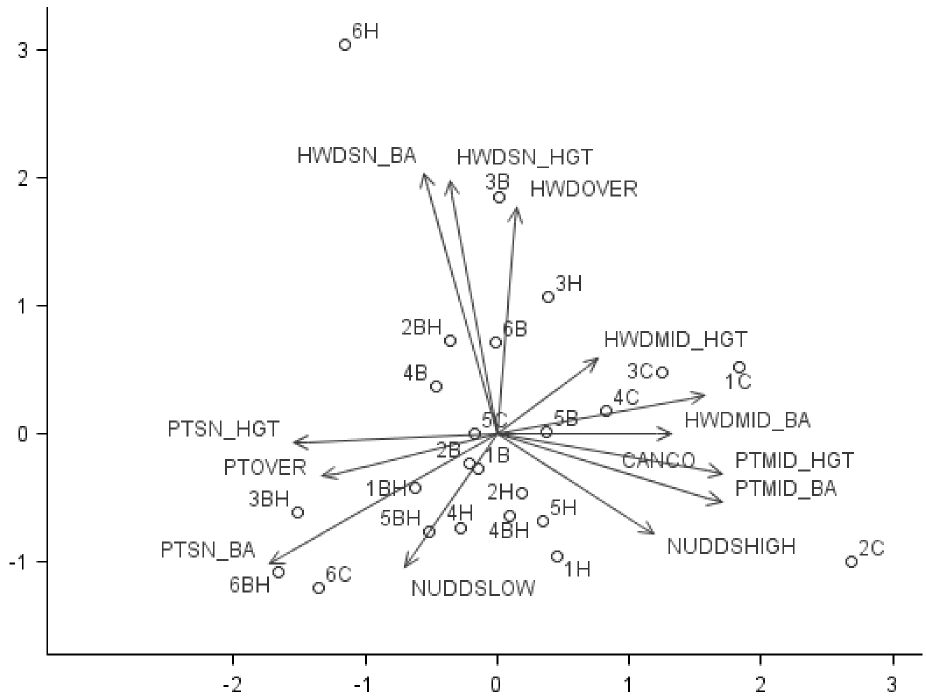
Figure 1.1 First 2 axes (percentage of variance explained by axis) of ordination by principal components analysis of vegetation structure variables and treatment plots from intensively managed, mid-rotation pine stands of Mississippi treated with prescribed fire and imazapyr, May and June 1999.



PC2
(20.19%)

PC1
(24.37%)

Figure 1.2 First 2 axes (percentage of variance explained by axis) of ordination by principal components analysis of vegetation structure variables and treatment plots from intensively managed, mid-rotation pine stands of Mississippi treated with prescribed fire and imazapyr, May and June 2001.



PC2
(18.25%)

PC1
(24.68%)

Figure 1.3 First 2 axes (percentage of variance explained by axis) of ordination by principal components analysis of vegetation structure variables and treatment plots from intensively managed, mid-rotation pine stands of Mississippi treated with prescribed fire and imazapyr, May and June 2004.

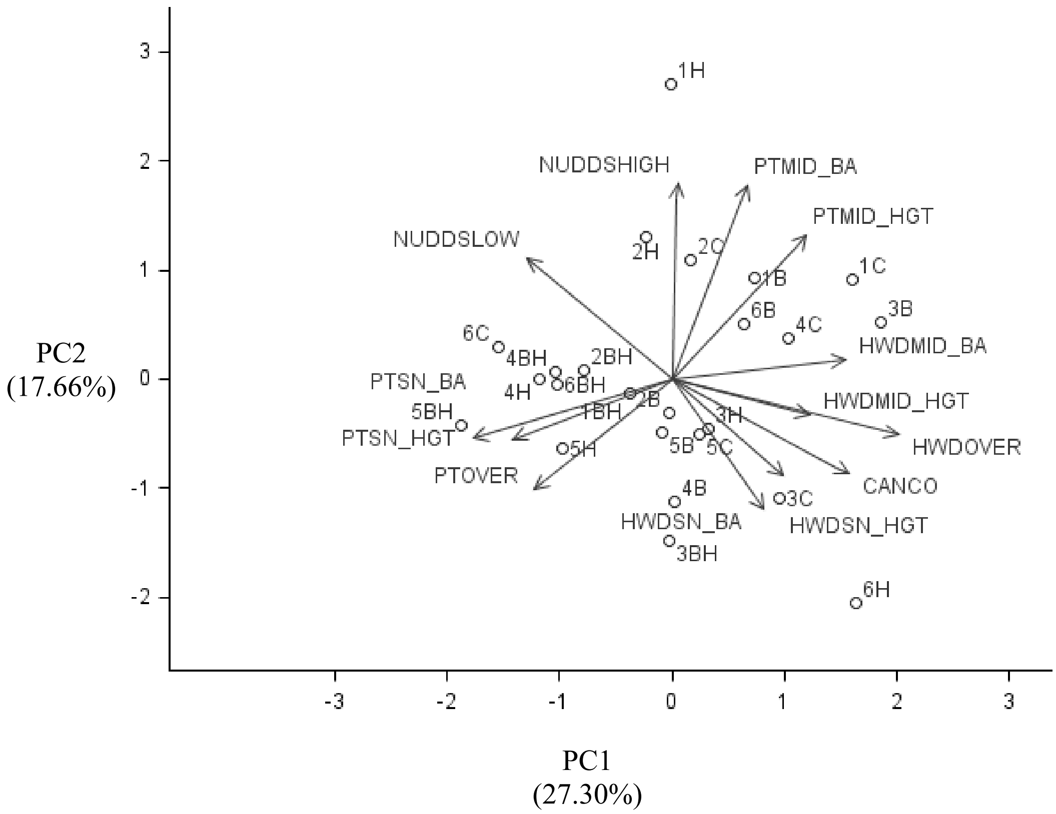


Figure 1.4 First 2 axes (percentage of variance explained by axis) of ordination by principal components analysis of vegetation structure variables and treatment plots from intensively managed, mid-rotation pine stands of Mississippi treated with prescribed fire and imazapyr, May and June 2005.

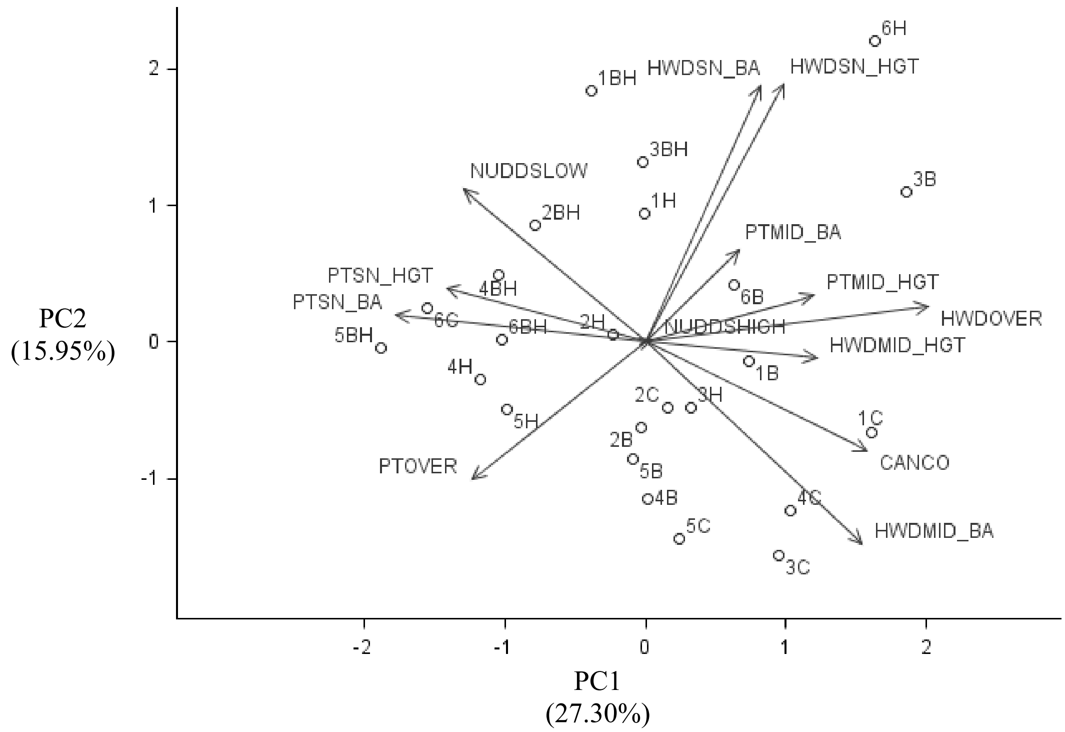


Figure 1.5 First and third axes (percentage of variance explained by axis) of ordination by principal components analysis of vegetation structure variables and treatment plots from intensively managed, mid-rotation pine stands of Mississippi treated with prescribed fire and imazapyr, May and June 2005.

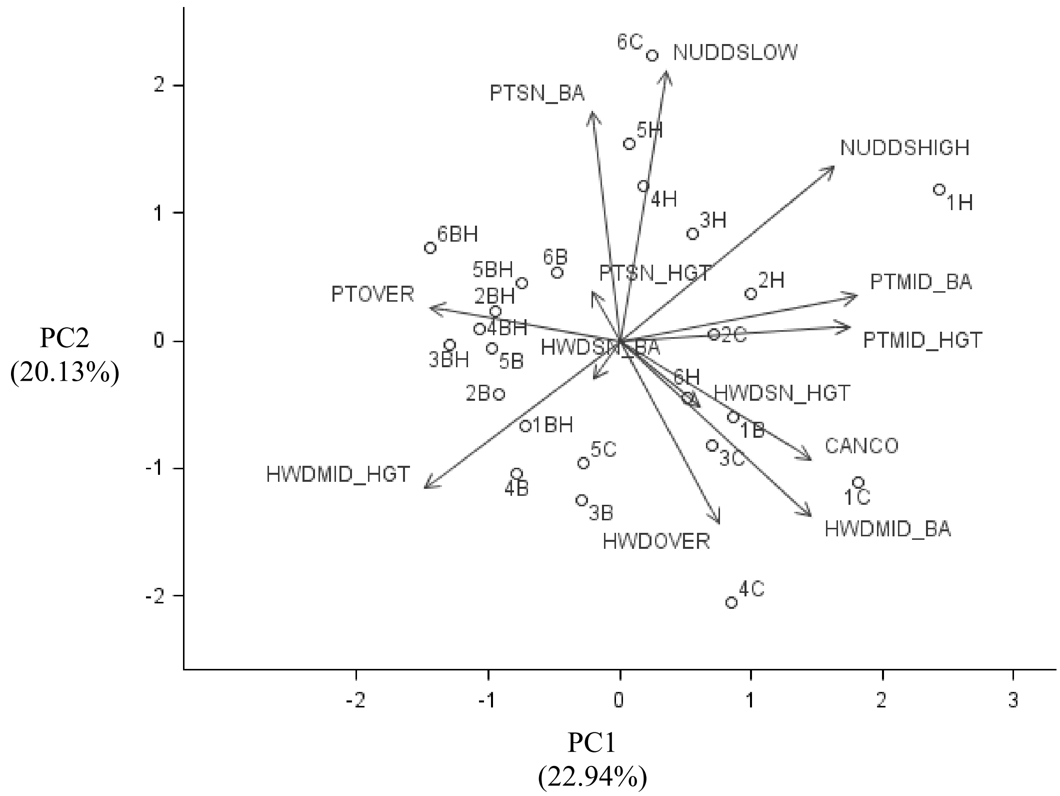


Figure 1.6 First 2 axes (percentage of variance explained by axis) of ordination by principal components analysis of vegetation structure variables and treatment plots from intensively managed, mid-rotation pine stands of Mississippi treated with prescribed fire and imazapyr, May and June 2006.

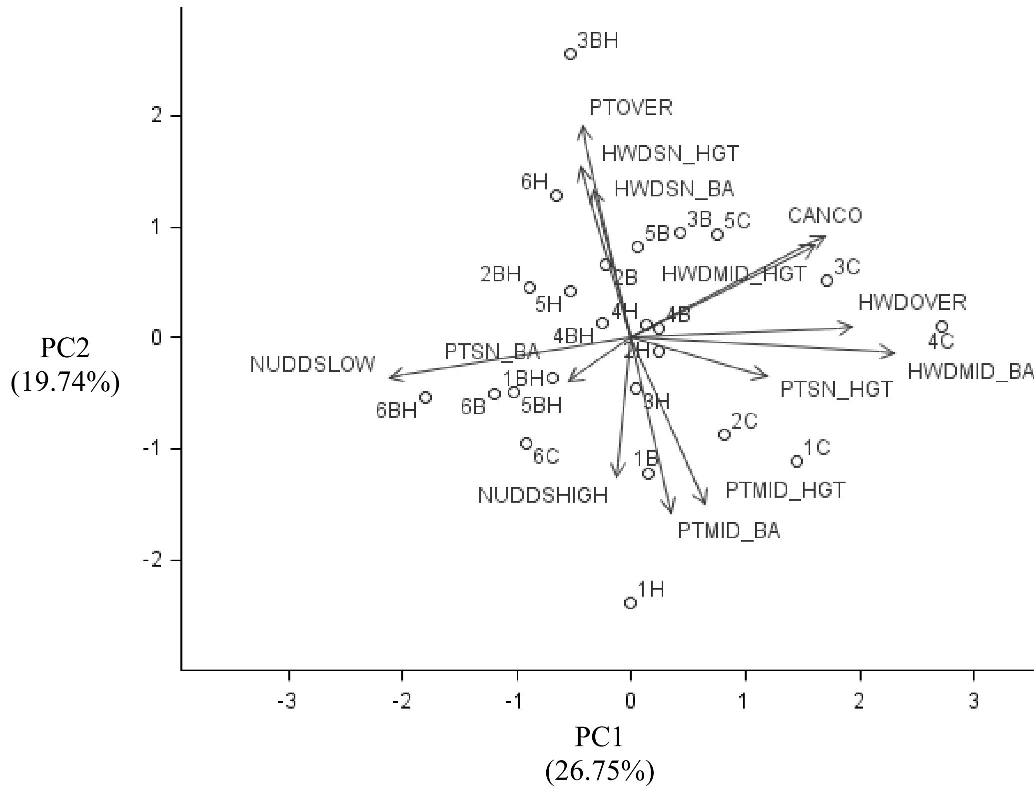


Figure 1.7 First 2 axes (percentage of variance explained by axis) of ordination by principal components analysis of vegetation structure variables and treatment plots from intensively managed, mid-rotation pine stands of Mississippi treated with prescribed fire and imazapyr, May and June 2008.

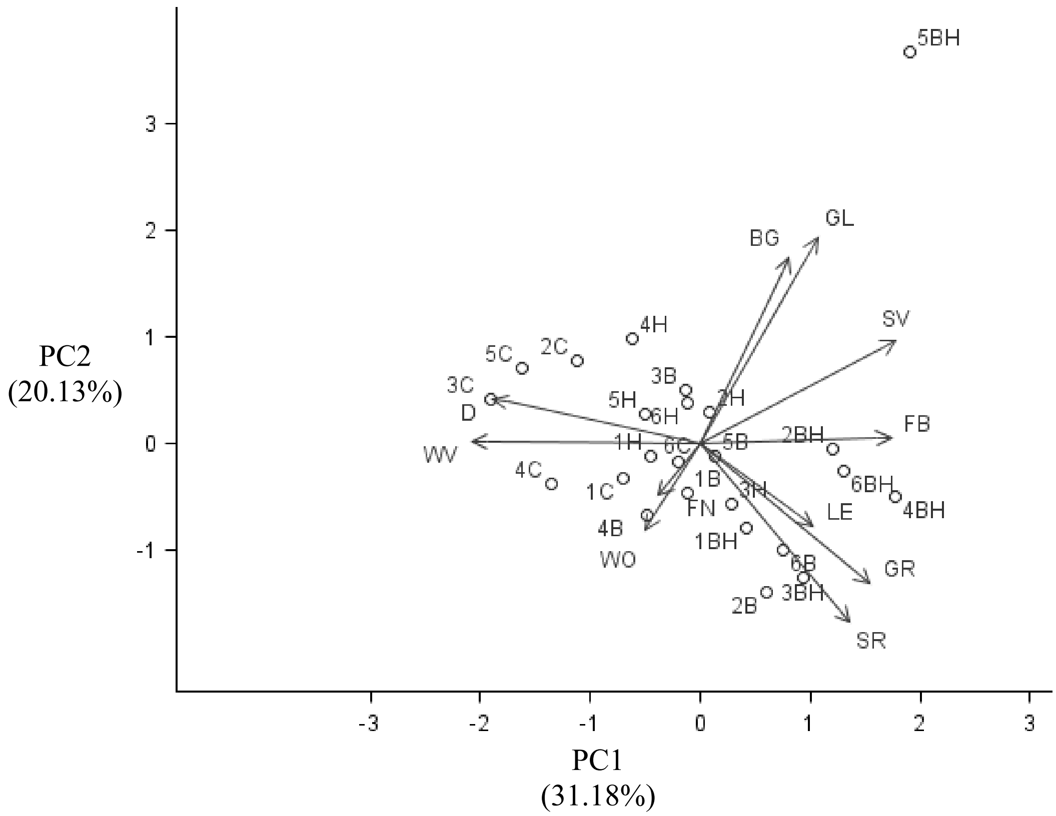


Figure 1.8 First 2 axes (percentage of variance explained by axis) of ordination by principal components analysis of vegetation coverage variables and treatment plots from intensively managed, mid-rotation pine stands of Mississippi treated with prescribed fire and imazapyr, May and June 2005.

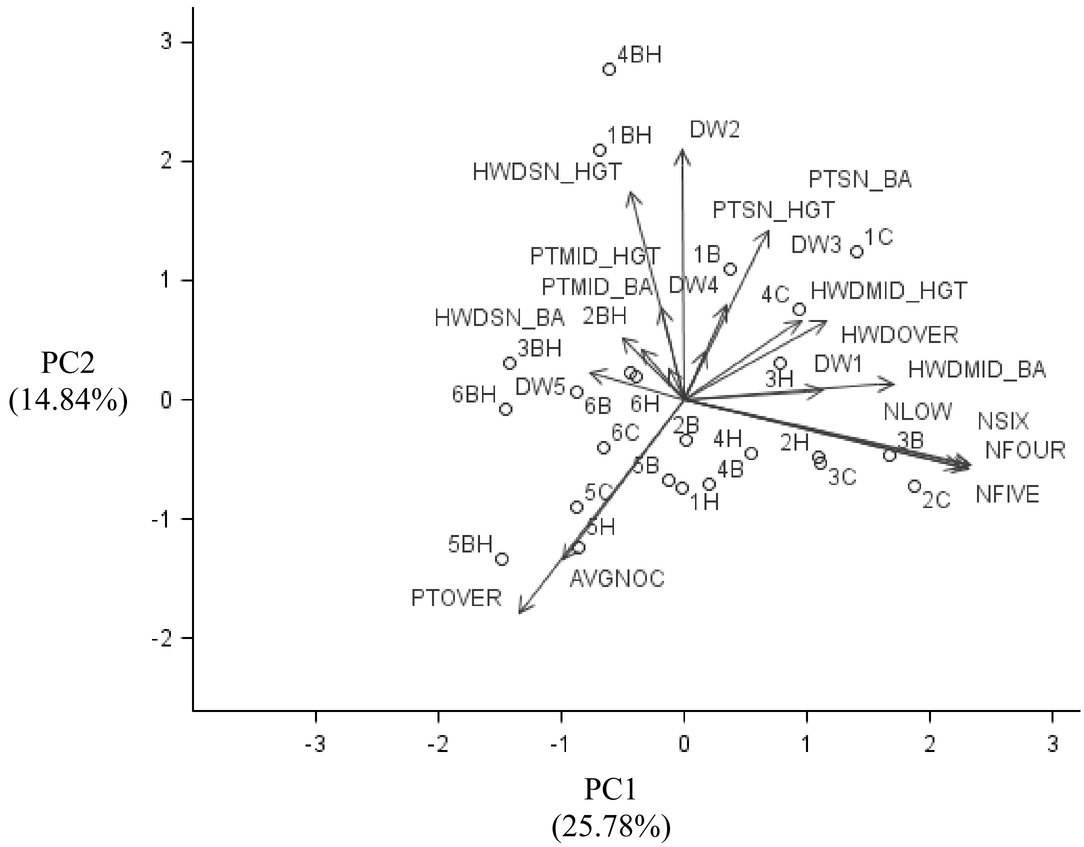


Figure 1.9 First 2 axes (percentage of variance explained by axis) of ordination by principal components analysis of vegetation structure variables and treatment plots from intensively managed, mid-rotation pine stands of Mississippi treated with prescribed fire and imazapyr, winter 2001.

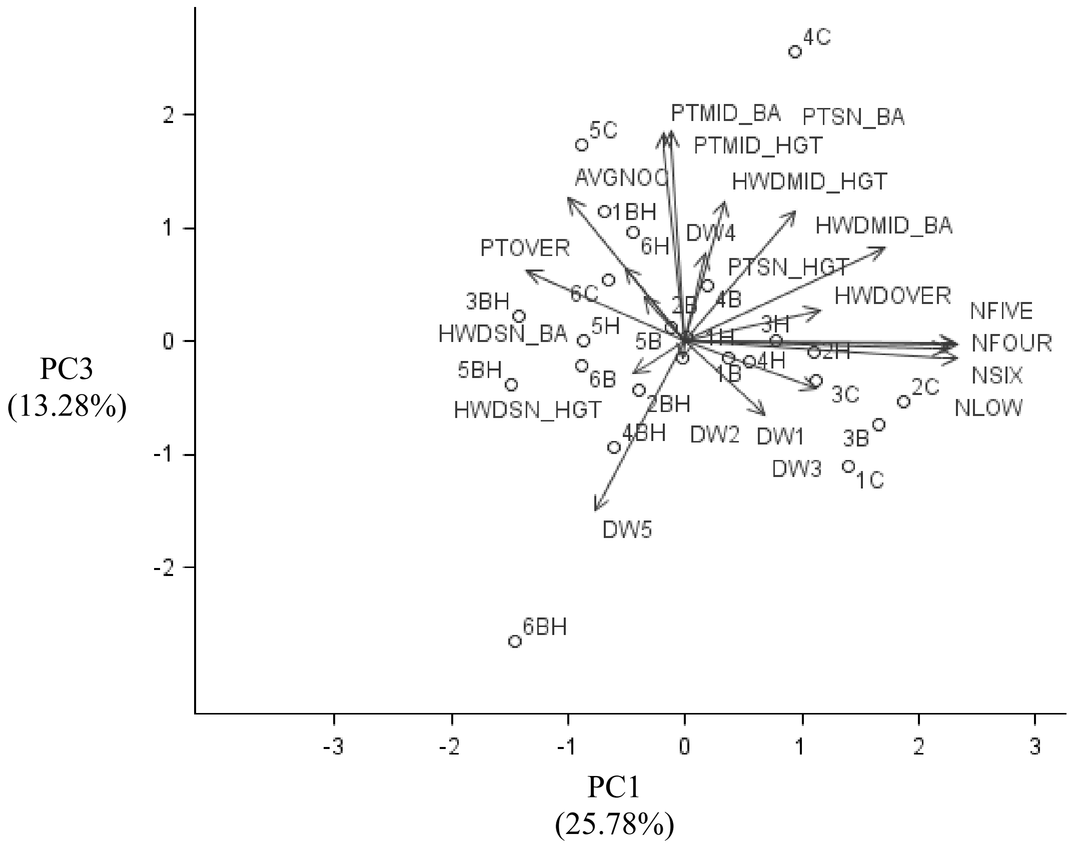


Figure 1.10 Axes 1 and 3 (percentage of variance explained by axis) of ordination by principal components analysis of vegetation structure variables and treatment plots from intensively managed, mid-rotation pine stands of Mississippi treated with prescribed fire and imazapyr, winter 2001.

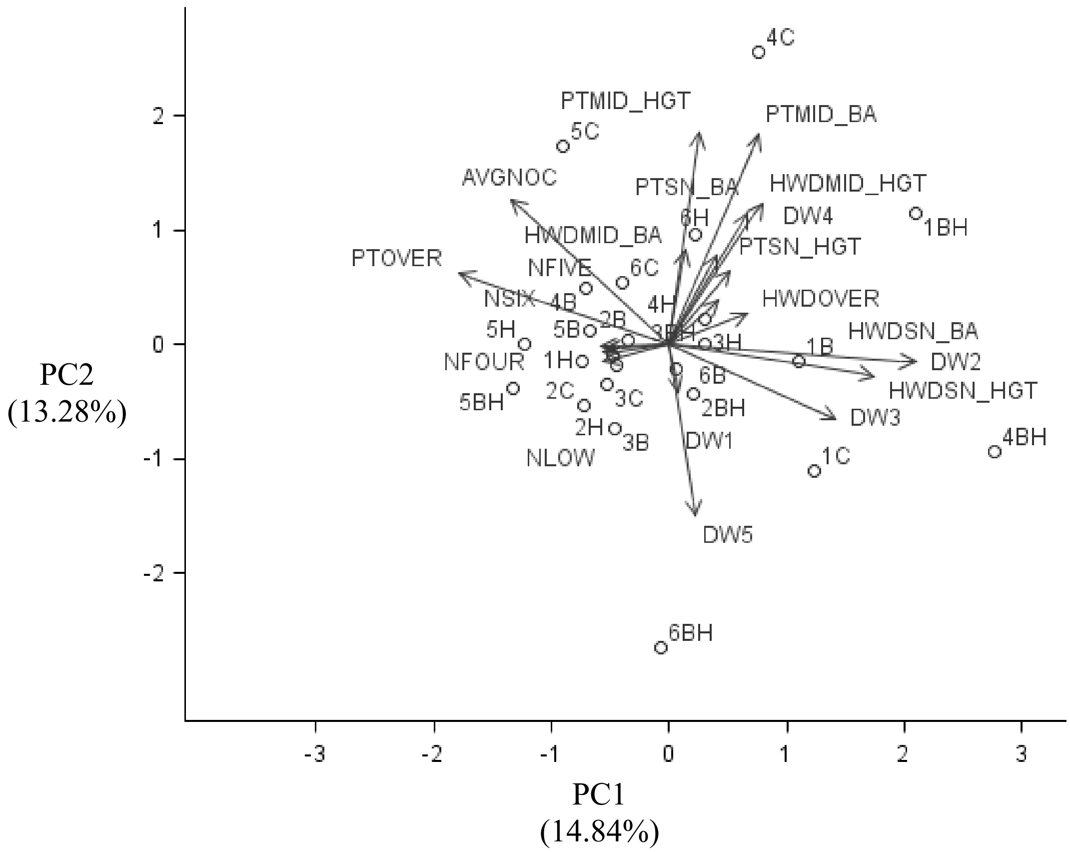


Figure 1.11 Axes 2 and 3 (percentage of variance explained by axis) of ordination by principal components analysis of vegetation structure variables and treatment plots from intensively managed, mid-rotation pine stands of Mississippi treated with prescribed fire and imazapyr, winter 2001.

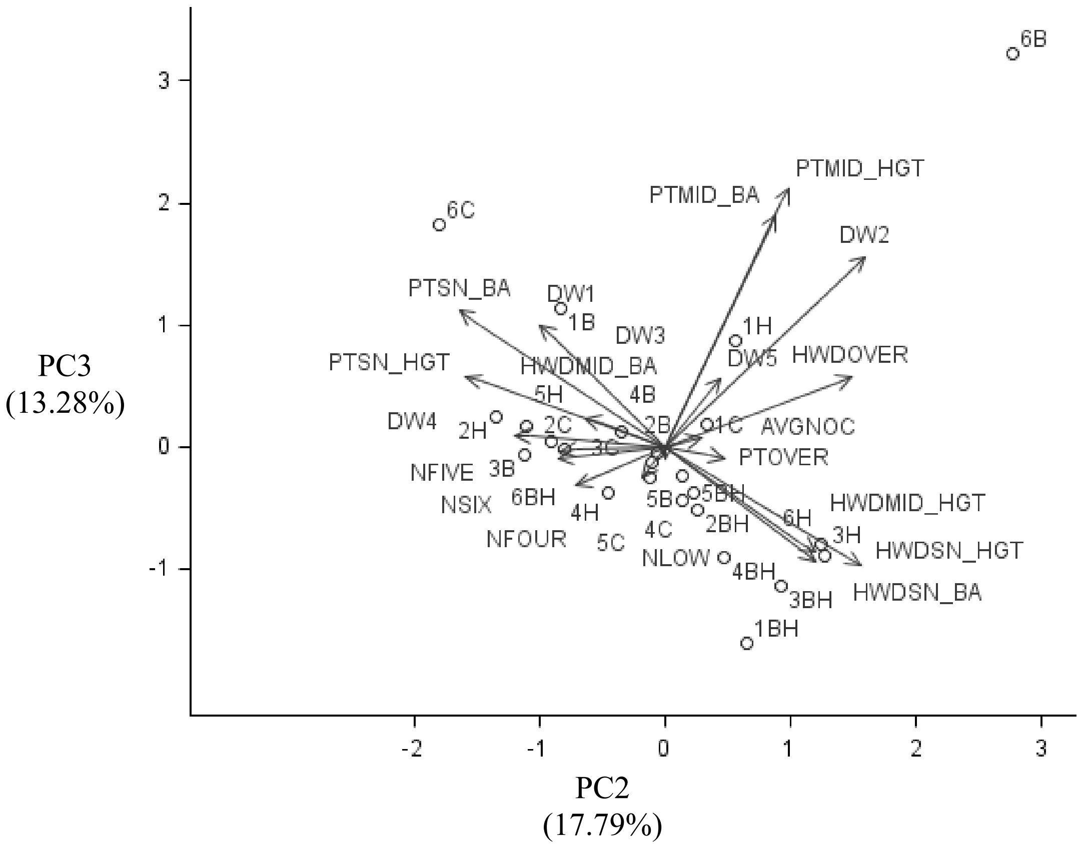


Figure 1.12 Second 2 axes (percentage of variance explained by axis) of ordination by principal components analysis of vegetation structure variables and treatment plots from intensively managed, mid-rotation pine stands of Mississippi treated with prescribed fire and imazapyr, winter 2002.

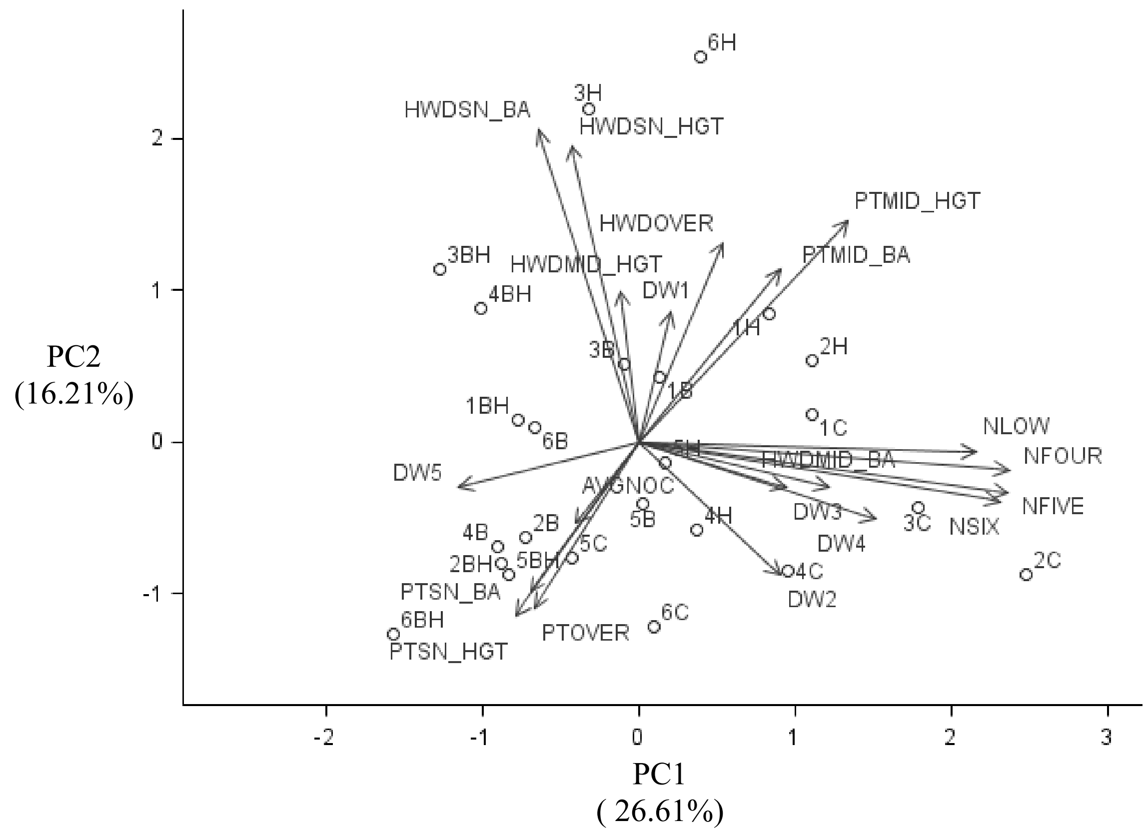


Figure 1.13 First 2 axes (percentage of variance explained by axis) of ordination by principal components analysis of vegetation structure variables and treatment plots from intensively managed, mid-rotation pine stands of Mississippi treated with prescribed fire and imazapyr, winter 2003.

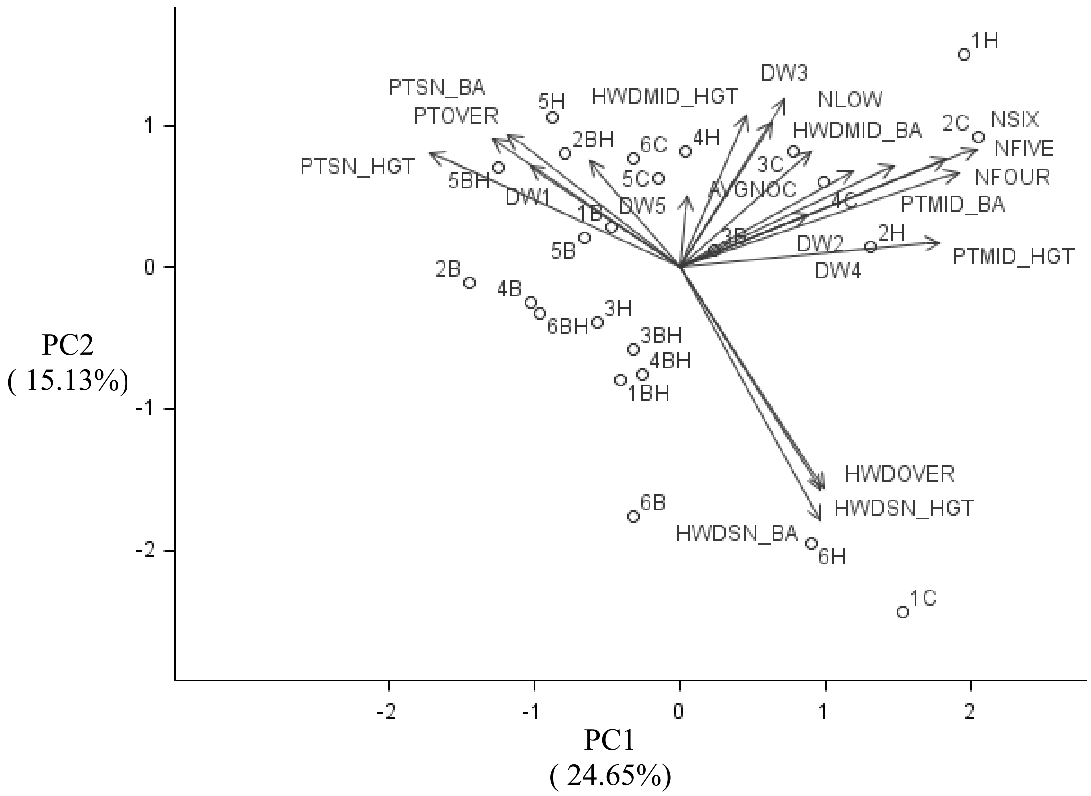


Figure 1.14 First 2 axes (percentage of variance explained by axis) of ordination by principal components analysis of vegetation structure variables and treatment plots from intensively managed, mid-rotation pine stands of Mississippi treated with prescribed fire and imazapyr, winter 2005.

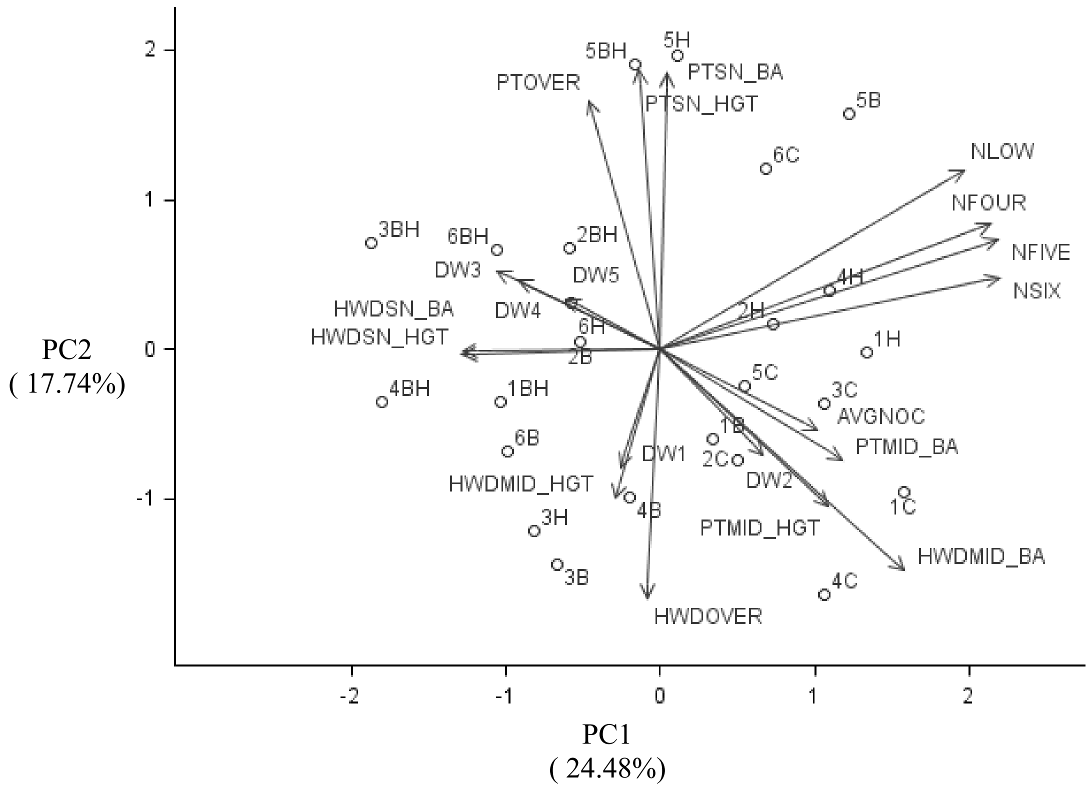


Figure 1.15 First 2 axes (percentage of variance explained by axis) of ordination by principal components analysis of vegetation structure variables and treatment plots from intensively managed, mid-rotation pine stands of Mississippi treated with prescribed fire and imazapyr, winter 2006.

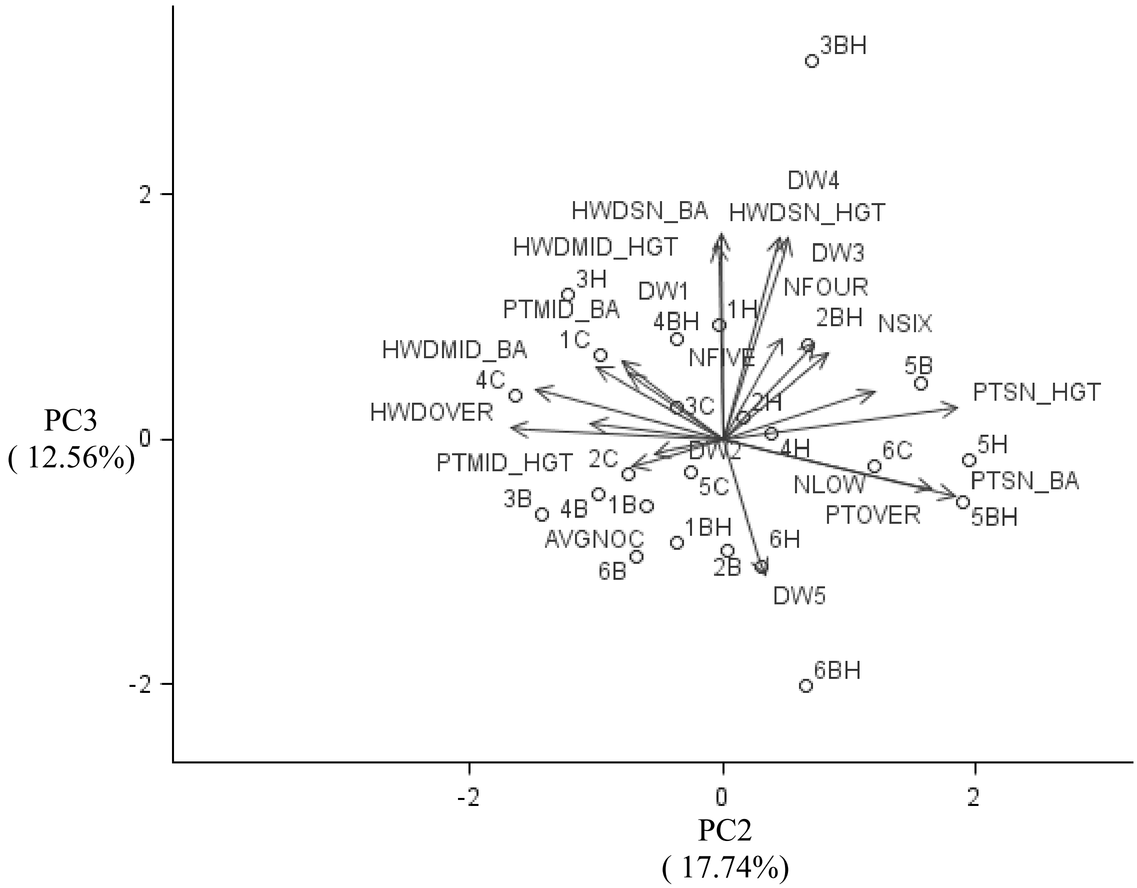


Figure 1.16 Second 2 axes (percentage of variance explained by axis) of ordination by principal components analysis of vegetation structure variables and treatment plots from intensively managed, mid-rotation pine stands of Mississippi treated with prescribed fire and imazapyr, winter 2006.

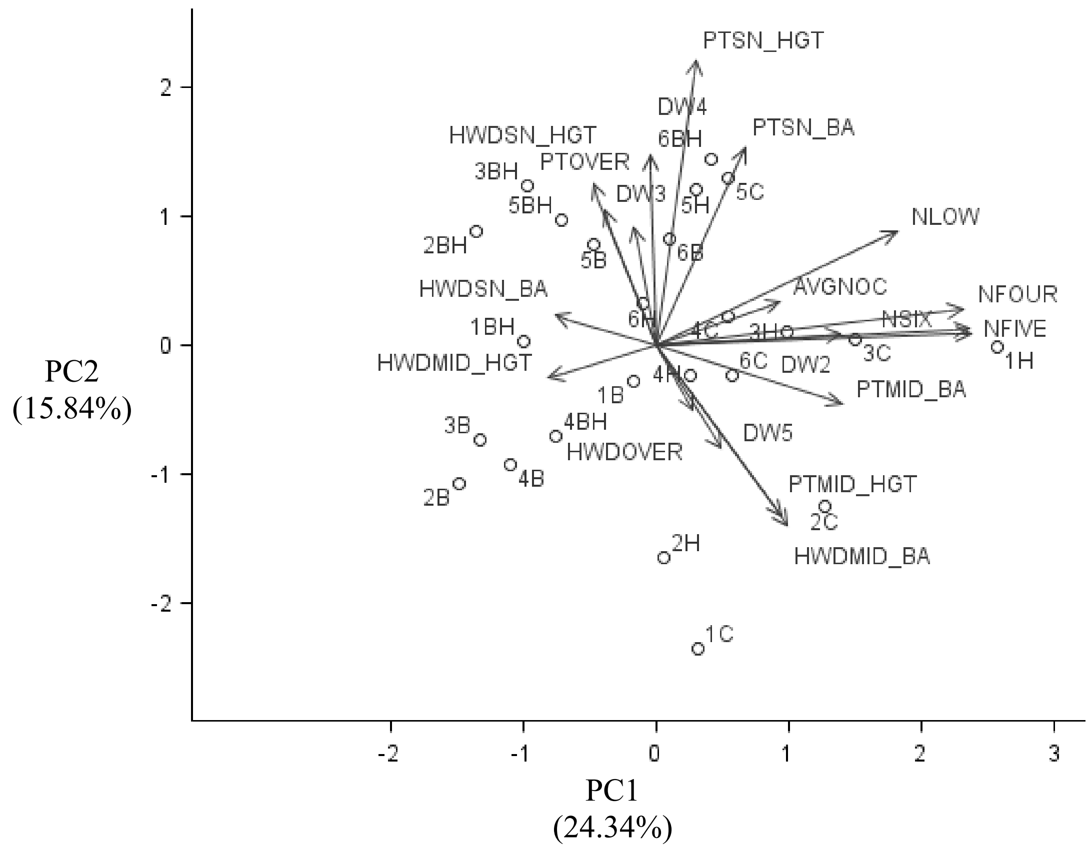


Figure 1.17 First 2 axes (percentage of variance explained by axis) of ordination by principal components analysis of vegetation structure variables and treatment plots from intensively managed, mid-rotation pine stands of Mississippi treated with prescribed fire and imazapyr, winter 2007.

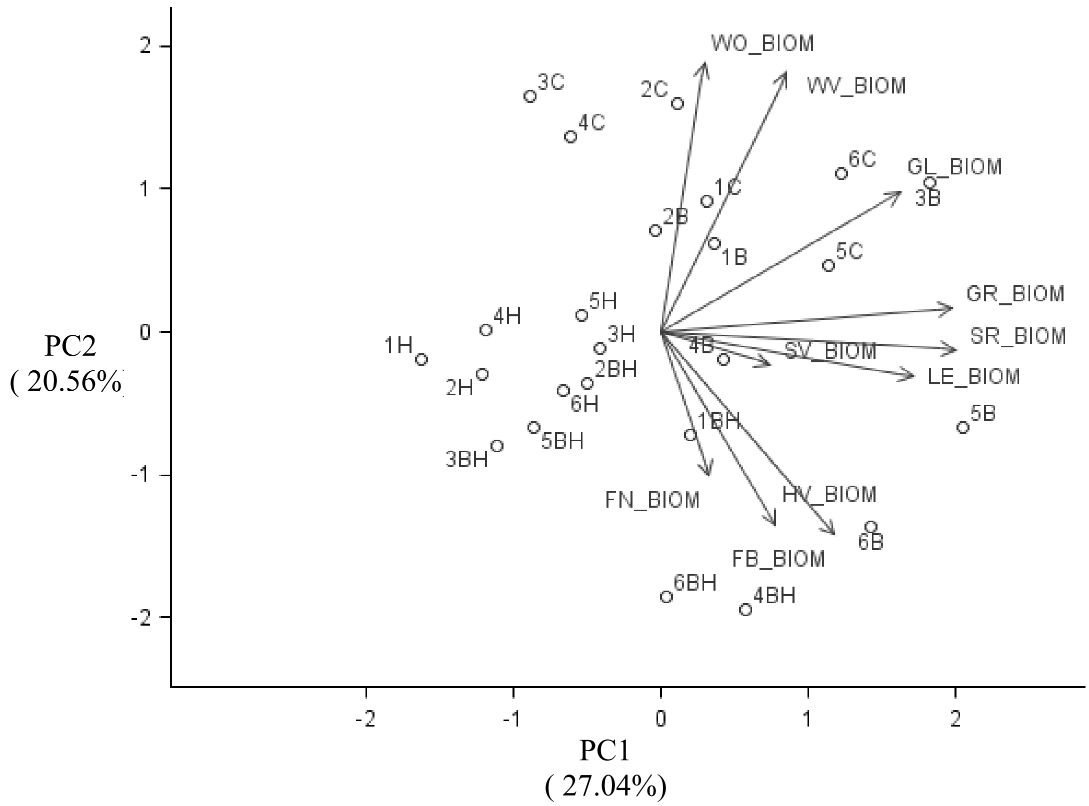


Figure 1.18 First 2 axes (percentage of variance explained by axis) of ordination by principal components analysis of biomass variables and treatment plots from intensively managed, mid-rotation pine stands of Mississippi treated with prescribed fire and imazapyr, summer 2000.

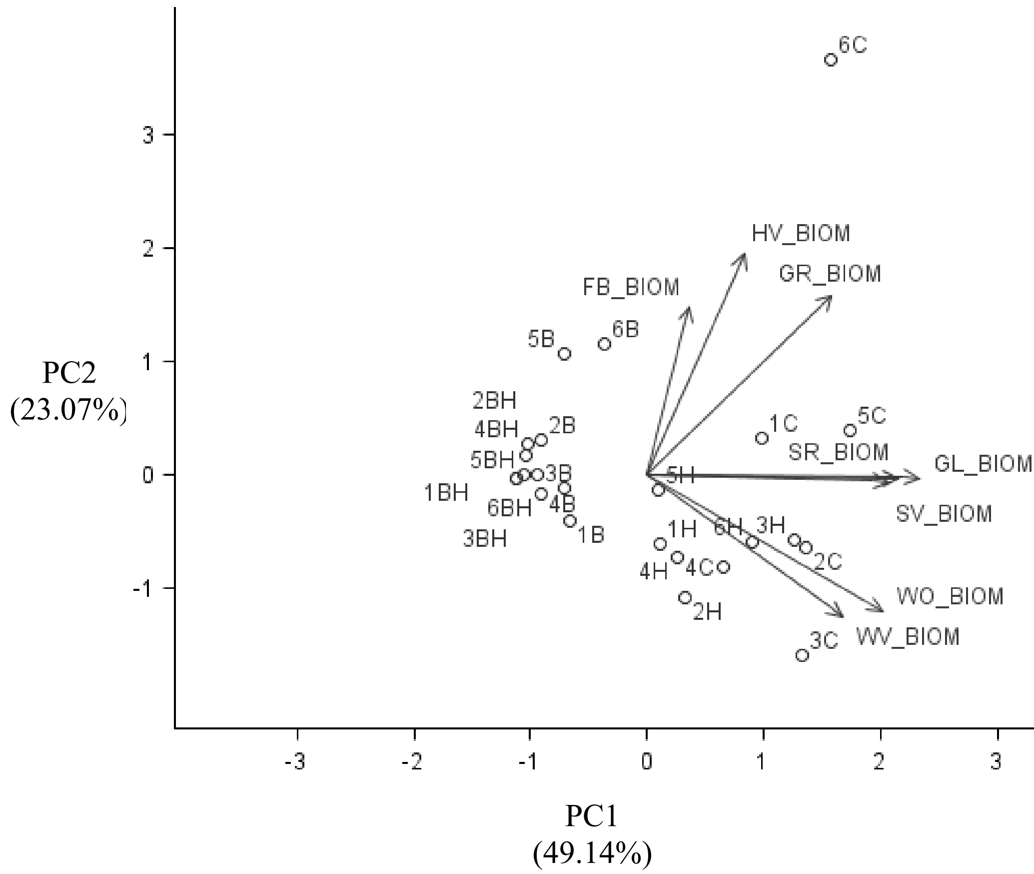


Figure 1.19 First 2 axes (percentage of variance explained by axis) of ordination by principal components analysis of biomass variables and treatment plots from intensively managed, mid-rotation pine stands of Mississippi treated with prescribed fire and imazapyr, winter 2000.

CHAPTER 2

AVIFAUNA RESPONSE TO FIRE AND IMAZAPYR APPLICATION IN INTENSIVELY MANAGED, MID-ROTATION PINE STANDS OF MISSISSIPPI

Intensively managed pine (*Pinus spp.*) forests cover 18 million ha in the southern United States, about 20% of southern forests, with 2.2 million ha in Mississippi (USDA Forest Service 2007). Forest managers are increasingly expected to incorporate management efforts contributing to the conservation of biodiversity (Sustainable Forestry Initiative 2005), but current management practices at mid-rotation such as commercial thinning and fertilizing may only provide short-term benefits for conservation (Wood 1986, Peitz et al. 1999, Iglay et al. 2010*a*). Disturbances can perpetuate biodiversity (Turner et al. 2001, White and Jentsch 2001, Rundel et al. 1998), and although frequent disturbances favor disturbance-dependent or –tolerant species, timber management incorporating biocomplexity objectives could compensate for disturbance-intolerant species by creating a mosaic of stand treatments, vegetative structures, species, and successional stages across a landscape (Hunter 1990, Franklin 1993*ab*, Heljden et al. 1998, Carey et al. 1999*ab*, Tilman 1999) . Managed forests can provide ecosystem benefits such as wildlife habitat, protection of water quality, and carbon sequestration while offering a mosaic of successional stages across the landscape (Petraitis et al. 1989, Greenberg et al. 1994, McLeod and Gates 1998, Vogt et al. 1999). Therefore, it is essential to determine optimal management approaches of intensively managed, mid-rotation pine (*Pinus spp.*) stands to meet forestry and biodiversity objectives (Wear and Greis 2002, Stein et al. 2005). Addition of dormant season prescribed fire and selective herbicide application after commercial thinnings may enhance wildlife habitat and sustain improved biodiversity until harvest supporting concepts of sustainable forestry (Hartley 2002, Carnus et al. 2003, Sustainable Forestry Initiative 2005).

Prescribed fire and selective herbicides are 2 silviculture tools used to control midstory hardwood competition and improve wildlife habitat in mid-rotation, intensively managed pine stands of the southeastern United States (Brockway and Outcalt 2000, Edwards et al. 2004, McInnis et al. 2004). Prescribed fire is similar to historical disturbances of the southeast (Brennen et al. 1998). Following specific prescriptions, dormant season prescribed fires, applied during winter, avoid detrimental effects on pine growth caused by crown scorch (Wade and Lunsford 1989, Bessie and Johnson 1995, Schimmel and Granstrom 1997). Selective herbicides, such as those containing imazapyr, offer an alternative to prescribed fire without smoke management issues or limited available burning degree days (Wigley et al. 2002). Both treatments have demonstrated abilities to reduce woody plant coverage and consequently increase herbaceous understory plant coverage (Stransky and Harlow 1981, Brockway and Outcalt 2000, Miller and Miller 2004, Chapter 1).

Past research has demonstrated treatment benefits to a myriad of game and non-game animals such as white-tailed deer (*Odocoileus virginianus*; Demarais et al. 2000, Mixon et al. 2009, Iglay et al. 2010), eastern wild turkey (*Meleagris gallopavo silvestris*;

Dickson and Wigley 2001, Miller and Conner 2007), northern bobwhite (*Colinus virginianus*; Guynn et al. 2004, Miller and Miller 2004, Welch et al. 2004), and songbirds (Sladek et al. 2008). However, research is lacking regarding avifauna response to independent and combined applications of dormant season prescribed fire and imazapyr.

Avifauna respond typically to vegetation changes associated with imazapyr application or prescribed fire, not through direct mortality (Wade and Lunsford 1989, McComb and Hurst 1987, Guynn et al. 2004, Miller and Miller 2004, Sladek et al. 2008). In intensively managed pine, a shift in food and cover availability associated with both treatments tends to favor early-successional and grassland species (Bendall 1974, Dickson 1981). Birds preferring dense midstory vegetation are not favored as midstory vegetation is altered and consequent vegetative structure changes (Bendell 1974, Dickson 1981, Brennan et al. 1995, Wilson et al. 1995, Burger et al. 1998). Increased richness and abundance of shrub and grassland bird species have been observed post-burn (Dickson 1981), but only some herbicide-treated forest stands have had greater bird species richness (Savidge 1978, Schulz et al. 1992, Easton and Martin 1998). However, declines in early-successional and shrub breeding birds across the United States (0.67% per year, 1966-2004) and in the southeastern United States (0.99% per year, 1966-2004; Sauer et al. 2005) supports application of these and similar treatments for creating habitat components of disturbance-dependent plant communities (Engstrom et al. 1984, Burger 2000, Askins 2001, Saab and Powell 2005).

Sensitivity of avifauna to environmental changes, diversity of taxa, defined habitat relationships, and ease of monitoring make them ideal for measuring responses to forest management practices at similar scales (Maurer 1993, Nuttle et al. 2003). Point counts are a common technique to estimate relative abundance and density of bird populations (Ralph et a. 1995, Rosenstock et al. 2002, Thompson 2002, Diefenbach et al. 2003). Additionally, aural detections, such as is used in point count surveys, are more frequent than visual detections in closed-canopy forests (~94% of total observations) and other densely vegetated areas (Faanes and Bystrak 1981, Richards 1981, Scott et al. 1981, DeJong and Emlen 1985), and depend on a bird's presence, probability of calling during a count, and observers' skill (Sauer et al. 1994, Farnsworth et al. 2002). Estimates of detection functions are essential for accurately estimating abundance and can be achieved via distance sampling methods (Ramsey and Scott 1979, Reynolds et al. 1980, Buckland et al. 2001).

Wildlife and intensive forest management can coexist (Miller et al. 2009). Many silvicultural practices such as thinning, mid-rotation competition control and stand-age management (e.g., even- and uneven-aged) alter vegetative communities and perpetuate structural diversity (Thompson et al. 1992, Thompson et al. 1995, Powell et al. 2000, Thompson and DeGraaf 2001). Repeated disturbances, such as multiple dormant-season prescribed fires with or without an initial imazapyr application, may maintain early successional conditions without hindering forestry practices or fragmenting forests (Robinson et al. 1995, Schulte and Niemi 1998). Information is lacking regarding combined and independent effects of prescribed fire and imazapyr on avian communities in mid-rotation, intensively managed pine but may lend significantly to understanding impacts of prescribed fire and imazapyr on conservation of biodiversity objectives. Because indirect effects (changes in vegetation) influence bird community responses, it is essential to concomitantly survey bird and vegetative communities before and after treatment application. Therefore, I investigated direct and indirect effects of dormant season prescribed fire and imazapyr on bird communities and their surrounding vegetative communities to better understand treatment potentials as tools of biodiversity conservation.

Study Area and Design

See Chapter 1 for a full description of study area and design.

Methods

Vegetation Response

See Chapter 1 for a description of vegetation sampling methods.

Avian Response

I conducted point counts for avifauna twice monthly during May and June 1999-2001 and May-July 2002-2008 from 0530-1030 hours. I designated one point per corner per treatment plot (n= 4/treatment plot) with each ≥ 75 m from plot edge and ≥ 100 m from other bird survey points for point counts. I used distance bands of < 25 m, 25 - 50 m, and ≥ 50 m to determine detection probabilities during 1999 and 2003-2008 and a fixed-radius circular point with detections of < 50 m recorded during 2000-2002. I also used 3 time brackets of 1-3, 4-5, and 6-10 minutes (Hutto et al. 1986, Ralph et al. 1995, Hamel et al. 1996). I only sampled under weather conditions stipulated by the Breeding Bird Survey (Robbins et al. 1986, Ralph et al. 1995).

I calculated mean species relative abundance, species richness, total abundance, and total avian conservation value (TACV). I calculated mean relative abundance and species richness across 4 points and 4 visits within each year. I only used species with $\geq$ 40 observations and within the adequate detection range as determined by distance sampling for species-specific relative abundance (Table 2.1). I calculated species richness as mean number of bird species per bird point count. I calculated TACV using the following equation adapted from Nuttle (1997):

$$TACV_{ij} = \sum_{k=1}^{S} (A_{ijk} * PIF_k) \tag{2-1}$$

where $TACV_{ij}$ is the avian conservation value of sampling point i in year j, S represents number of species per point, A_{ijk} is mean relative abundance for species k at point i in year j, and PIF_k is Partner's in Flight ranking (*PIF.rank*) for species k in Physiographic Area 27 of the Southeastern Coastal Plain (Nuttle et al. 2003). To calculate *PIF.rank*, I used regional Population Trend (PT-r), Threats to Breeding (TB-r), and global Breeding Distribution (BD-g) ranks because of minimum correlation (Beissenger et al. 2000: Table 1) and followed the ranking system of Nuttle et al. (2003) because it reduces influence by endangered or threatened species. I calculated a mean TACV for each treatment plot across 4 points and 4 visits per year.

Statistical Analysis

Alldredge et al. (2007) determined an effective detection radius of < 67 m for most bird point counts. Because I standardized all sampling protocols including time of day, weather conditions, and observer skill [e.g., one observer per season, except in summer 2005 (n=2 observers)], my detection rates should be similar among treatments at < 50 m from the observer. However, I used distance methods to estimate detection functions in Program Distance (Thomas et al. 2006) to ensure that bird detection was similar among treatments for observations < 50 m from the observer (Buckland et al. 2001).

I based detection function analysis on distance from observer data arranged in 3 bands (0-25 m, > 25- 50 m, > 50 m). To avoid analysis errors associated with small sample sizes, I limited analysis to the same data set used for calculating TACV (e.g., $\geq$ 40 observations of each bird species). I used model selection to compare key function and series expansion choices among global and stratified models (strata=treatment; Half-normal Cosine, Half-normal Hermite polynomial, Hazard rate Cosine, Uniform Simple polynomial, Uniform Cosine). Although pre-treatment data were available, I only used post-treatment data (2003-2008) because I was concerned with differences in detection probabilities among treatments, and pre-treatment avian and vegetation communities were similar among treatment plots (Thompson 2002). I determined the best model based on least Akaike's Information Criterion (AIC) and used it to estimate effective detection radius and detection probability. I did not attempt to determine species-specific detection functions because of inadequate observations/treatment plot/year.

I tested the hypothesis of no difference in mean relative abundance, bird species richness, and TACV among treatments within years using a mixed models, repeated measures analysis of covariance in SAS (Proc Mixed; SAS Institute Inc., Cary, North Carolina, USA) with main effects of treatment, year, and treatment year. Then, I determined a subset of explantory vegetation structure variables for all-subsets variable selection to determine relative importance of each variable for each species (Arnold 2010). For mixed models, I used 4 treatment levels (burn, herbicide, burn + herbicide, control), random effect of stand (n=6), repeated measures of year (n =9; 2000-2008), and subject of stand × treatment for each model (Littell et al. 2006). I used pre-treatment (1999) bird abundance as a baseline covariate because pre-treatment bird communities may have differed among experimental units treated alike even though they did not differ among treatments according to preliminary analysis (Thompson 2002, Milliken and Johnson 2002). For each response variable, I selected an appropriate covariance structure from the following: 8-banded Toeplitz, heterogeneous compound symmetry, heterogeneous auto-regressive, and auto-regressive. I designated the covariance structure that minimized Akaike's Information Criterion corrected for small sample size as the top candidate for analysis (Littell et al. 2006, Gutzwiller and Riffell 2007). I used Kenward-Roger correction for denominator degrees of freedom for repeated measures to avoid inflated Type I errors (Littell et al. 2006, Gutzwiller and Riffell 2007). I used LSMEANS SLICE option to identify treatment effects within years following a significant interaction (Littell et al. 2006), and LSMEANS PDIFF to conduct pair-wise comparisons (Littell et al. 2006). All year references in results refer to years post-treatment. I set the *a prior* significance level at α=0.05.

Prior to variable reduction, I used principal components analysis (PCA) to reduce dimensionality of the data to species explaining the greatest amount of variation among sites (PROC PRINQUAL; SAS Institute Inc., Cary, North Carolina, USA). Principal components analysis is an ordination method capable of representing data sets of many variables with a smaller number of composite variables (McCune and Grace 2002). It works best with data of approximately linear relationships among variables such as environmental data or transformed community data (McCune and Grace 2002). Therefore, I transformed bird species relative abundance to meet assumptions of linearity for PCA (TRANSFORM LINEAR; SAS Institute Inc., Cary, North Carolina, USA) and determined total number of non-trivial components using randomization tests of eigenvalues with 999 runs for the resulting test statistic, Rnd-Lambda, as a stopping rule or the first component without important variables. I used PC-ORD to conduct randomization tests for calculating Rnd-Lambda (version 5.10, MJM Software, Gleneden Beach, Oregon, USA; Peres-Neto et al. 2005). I used correlation matrices and considered species with eigenvector loadings of $\geq |0.5|$ important.

Because habitat associations can vary regionally (Whittingham et al. 2007) and bird communities respond to changes in vegetation structure more so than plant biomass (e.g., MacArthur and MacArthur 1961, Willson 1974, Roth 1976, Maurer 1986, Herkert 1994, Sallabanks et al. 2006), I only included vegetation structure and cover variables as explanatory variables and determined top variables for inclusion based on my data set, not past literature. I avoided examining all combinations of explanatory variables by looking for relationships among response variables (e.g., species-specific relative abundance) and explanatory variables by visually inspecting scatter plots. Although this approach may have excluded meaningful variables especially when additive or interactive relationships with other explanatory variables occurred, I felt it was a relatively unbiased approach to data reduction that considered all possible explanatory variables for each dependent variable. I always chose summary variables over independent measurements when both were chosen for a model (e.g., Nudds board high instead of Nudds board level

4, 5, or 6) and removed response variables without any noticeable relationships to explanatory variables from model comparisons. To avoid multicolinearity, I conducted pair-wise comparisons of correlation coefficients among selected explanatory variables and when very correlated ($r \geq |0.5|$), removed the vegetation variable correlated more weakly to the response variable.

I used the MIXED procedure in SAS to evaluate regression models with stand (block) as the random effect, year as a repeated measure, and treatment plot as the subject (Littell et al. 2006). I also included main effects of treatment, year, and treatment × year. Because my analysis goal was to determine relative variable importance and I ran equal number of models per parameter, I used summed model weights per parameter and model-averaged parameter estimates to determine overall impacts of each explanatory variable on bird species relative abundance (Burnham and Anderson 2002, Arnold 2010).

Results

Detection Function

Detection function analysis of the avian community included over 2,000 samples and 30,000 observations. Parameter estimation required a stratified model (Global $_{AIC}$ = 64018.75, Stratified $_{AIC}$ = 63936.01) and model selection by treatment indicated half-normal keys with Cosine adjustments for burn only and control plots and uniform keys with cosine adjustments for herbicide treatments (burn + herbicide and herbicide only). Control ($\bar{x}$ = 0.51, SE= 0.02) and herbicide only ($\bar{x}$ = 0.52, SE= 0.01) treatments had lesser detection probabilities then burn treatments (burn only $\bar{x}$ = 0.69, SE= 0.01; burn +

herbicide $\bar{x}$ = 0.63, SE = 0.01) with coefficients of variation ranging from 1.6 to 4.3 %. Effective detection radius estimates were > 50 m for each treatment with shortest radii in controls (53.8 m) and longest in burn only (62.3 m). Coefficients of variation for EDR ranged from 0.8 to 2.2 %. Because the range of detection probabilities was small among treatments and all EDRs were > 50 m, I assumed minimal influence of detectability on observations < 50 m from the observer. Therefore, I used bird observations at < 50 m from the observer as indices of relative abundance.

Avian Response to Treatments

Fifty-four bird species met analysis criteria for total observations among all years of study (Table 2.2). Mean species richness and total abundance differed among treatments within years 4-7 and 3-9, respectively (Table 2.2). Mean species richness was generally greater in treated plots than controls (Table 2.3). However, for 3 of the 4 years, ≥ 1 treatment had mean species richness similar to controls. Total abundance was generally greater in herbicide treated plots than controls. However, definitive trends were scarce. Total avian conservation values differed among treatments across all years (Table 2.2). It was greater in burn + herbicide plots than burn only and controls (12.3 vs. 10.1) with herbicide only intermediate to burn + herbicide and burn only intermediate to controls.

Eighteen species had significant treatment × year interactions (Table 2.2). Eight of these species and total relative abundance had significant differences for most years (≥ 5 years, Table 2.3). Common yellowthroat (*Geothlypis tricheas*) abundance was greatest in burn + herbicide plots from year 2-6. Eastern wood-pewee (*Contopus virens*) abundance was greatest in burn + herbicide plots from year 1-7 except for similar abundances among herbicide treated plots (burn + herbicide and herbicide only) in year 2. Eastern wood-pewee abundance was least in herbicide only and controls years 5 -7. Hooded warbler (*Wilsonia citrina*) abundance was always greater in controls than burn + herbicide plots. From years 6-9, hooded warbler abundance was least in burn treatments (burn only and burn + herbicide). Indigo bunting (*Passerina cyanea*) abundance was generally greatest in burn + herbicide plots and least in controls with abundance of independent treatments (burn only and herbicide only) intermediate to treatment extremes. Kentucky warbler (*Oporornis formosus*) abundance was usually greater in controls and independent treatments than burn + herbicide plots. During the first 4 years, abundance in herbicide only was generally similar to burn + herbicide. During years 5 and 8, abundance between burn treatments was similar. Northern cardinal (*Cardinalis cardinalis*) abundance was greater in controls and herbicide only than burn + herbicide in years 3, 4, 8, and 9. Abundance in burn only was intermediate to all treatments in years 3 and 8, and both independent treatments had abundances intermediate to treatment extremes in year 7. White-eyed vireo (*Vireo griseus*) abundance was greatest in controls initially (years 1-3), but during years 4, 5, and 7-9, white-eyed vireo abundance was always greater in herbicide only plots than burn + herbicide plots. In years 8 and 9, white-eyed vireo abundance was greatest in herbicide treatment plots. Yellow-breasted chat (*Icteria virens*) abundance was typically greater in treated plots than controls in years 5-9.

Nine species had sporadic differences among treatments across few years (< 5 years; Table 2.2). Blue-gray gnatcatcher (*Polioptila caerulea*) abundance was greatest in controls year 1 and least in burn + herbicide year 2 (Table 2.3). Gray catbird (*Dumetella* *carolinensis*) abundance was greater in herbicide treated plots in year 1 than controls, greatest in burn + herbicide plots in year 2, and greatest in controls in year 8. Mourning dove (*Zenaida macroura*) abundance was greatest in burn + herbicide in year 1 and herbicide treatments in year 6. Prairie warbler (*Dendroica discolor*) abundance was greatest in controls in year 1 and greater in burn + herbicide than burn only in year 3 with herbicide only and controls harboring intermediate abundances. Red-eyed vireo (*Vireo olivaceus*) abundance was greater in controls than herbicide treatments with burn only intermediate in year 9. Red-bellied woodpecker (*Melanerpes carolinus*) abundance was greater in burn + herbicide than herbicide only and controls in year 6 with burn only intermediate and less in herbicide treated plots than burn only in year 8. Red-headed woodpecker (*Melanerpes erythrocephalus*) abundance was greatest in burn + herbicide plots in year 3. Ruby-throated hummingbird (*Archilochus colubris*) abundance was greatest in burn + herbicide plots in years 6 and 9. Wood thrush (*Hylocichla mustelina*) abundance was greatest in burn only in year 2 but greater in controls than burn treatments in years 5, 6, and 9.

Six bird species differed among treatments across all years (Table 2.2). Black-and-white warbler (*Mniotilta varia*) abundance was greater in controls and herbicide only than burn + herbicide (0.12 birds/count vs. 0.04 birds/count) with burn only intermediate. Carolina chickadee (*Poecile carolinensis*) abundance was greater in herbicide only than burn only and controls (0.31 birds/count vs. 0.21 birds/count). Eastern towhee (*Pipilo erythrophthalmus*) abundance was greater in herbicide treated plots and least in controls (0.94 birds/count vs.0.49 birds/count) with abundance in burn only intermediate. Northern bobwhite (*Colinus virginianus*) abundance was greatest in burn + herbicide

(0.05 birds/count vs. 0.02 birds/count), and pine warbler (*Dendroica pinus*) abundance was greater in burn + herbicide plots than burn only and controls. Summer tanager (*Piranga rubra*) abundance was greatest in burn + herbicide plots (0.10 birds/ count) and least in herbicide only (0.05 birds/ count) with abundance in controls intermediate to burn only and herbicide only.

Relationships Among Avian and Vegetation Communities

Habitat associations from constructed models incorporated vegetation variables differentially affected by treatments (Table 2.4). Of 54 bird species used in ANCOVA, 18 species explained 32% of the total variance among sites and were used in model selection. Canopy coverage (n = 6 models), woody vine and plant coverage (n = 5 models each), semi-woody vine coverage (n = 4 models), and hardwood midstory basal area (n = 3 models) were predominant among models.

Carolina wrens (*Thryothorus ludovicianus*), common yellowthroats, and eastern towhees were associated with vegetation characteristics typical of treated sites (Table 2.5). Carolina wren relative abundance increased with greater hardwood snag basal. Common yellowthroat and eastern towhee relative abundances each increased as woody plant coverage decreased. Eastern towhee relative abundances also increased with greater coverage of semi-woody vines and reduced basal area of midstory hardwood trees. Indigo bunting relative abundance decreased as grass coverage increased, and Kentucky warbler relative abundance increased with decreasing debris coverage and increased woody plant coverage. Northern cardinals and white-eyed vireos were associated with greater wood plant coverage, a characteristic of controls.

Discussion

Application of prescribed fire and imazapyr in mid-rotation, intensively managed pine influences avian communities by causing a shift from common species to a community more composed of species of conservation concern. By impacting directly vegetative communities, independent and combined treatment applications created distinct vegetative structure and cover components that differentially impacted bird communities. Vegetation association models indicated key vegetative components that may have influenced avian communities such as basal area of midstory hardwoods, visual obstruction, and changes in understory vegetative cover. Ground and shrub nesting bird species such as yellow-breasted chats, indigo buntings, and eastern towhees increased in relative abundance as hardwood midstory decreased and lower level visual obstruction increased. Application of repeated, dormant season prescribed fires after a one-time application of imazapyr maintained these vegetation conditions. Closed canopy forest species tended to favor conditions of control sites where mid-rotation management was limited to thinning and fertilizing. Although thinning can reduce hardwood midstories and influence visual obstruction, their effects are generally short-lived (Peitz et al. 1999) and would most likely not maintain a primarily herbaceous vegetative structure over the long-term as is possible with repeated fires.

Imazapyr treatments favored open woodland species such as summer tanagers and indigo buntings which have been associated with similar vegetative structure of burn + herbicide sites such as an open understory (Hamel 1992, Payne 1992, Robinson 1996, Hunter et al. 2001, Sladek et al. 2008). Open area, shrub nesting preferences of common yellowthroats, yellow-breasted chats, indigo buntings, and eastern towhees supports fire and imazapyr application in mid-rotation intensively managed pine stands when *Rubus* spp. provides ample, low-level (< 0.9 m) nesting opportunities (Cooper 1996, Burger et al. 1998, Ricketts and Ritchison 2000). Open understories also can favor foraging behavior of species like eastern wood-pewees that prefer open wooded areas to sally for flying invertebrates (Hartung and Brawn 2005). Other open woodland species of greater conservation priority such as brown-headed nuthatches (*Sitta pusilla*) and Bachman's sparrow (*Aimophila aestivalis*) also may benefit from imazapyr treatments (Wilson and Watts 1999), but I was unable to detect any differences possibly due to few observations limited to the beginning and end of the study. However, these species tend to prefer sparse canopy coverage and substantial herbaceous understory cover that may be unachievable over long-term periods with current commercial tree stocking and thinning rates (Withgott and Smith 1998, Dunning and Watts 1990). Thinning may induce positive response by these species (Wilson and Watts 1999), but without hardwood control methods (e.g., prescribed fire and imazapyr herbicide), effects of thinnings are short-lived (Wood 1986, Peitz et al. 1999).

Reduction of hardwoods within pine forests can alter bird communities (Engstrom et al. 1984, Cooper 1996, Burger et al. 1998, Provencher et al. 2002). Imazapyr, with or without fire, effectively reduced hardwoods but was unable to continually support significant herbaceous understory cover other than semi-woody vines (Chapter 1). Prescribed fire alone was not as effective in reducing hardwoods but frequent fire-return intervals (3 years) eventually reduced hardwoods (burn only), always stimulated understory herbaceous growth especially forbs, and maintained open woodland conditions post-imazapyr application (burn + herbicide). Although these conditions are not ideal for many forest interior species that prefer hardwood midstory (Hamel 1992), species favoring herbicide-treated sites tended to either be higher-priority species or have greater relative abundances than species in control sites. Using Nuttle et al.'s (2003) equation for calculating TACV reduced influence of greater abundances of common species compared to other priority score calculations (Beissenger et al. 2000). Therefore, greater TACV in herbicide treated sites than controls was most likely caused by greater relative abundances of high-priority species (Table 2.1). Northern bobwhite, for example, had greater relative abundances in burn + herbicide than all other treatments, and Bachman's sparrows were only detected in treated sites (e.g., burn only, herbicide only, burn + herbicide; Table 2.1). Similar patterns were observed for mean species richness and total relative abundance concerning the benefit of imazapyr treatments to conservation of avian species. Changes in TACV over time, although not significant, revealed a distinct pattern of avian response to treatments. Immediately post-treatment, TACV was greater in control plots than treated plots (Figure 2.1). Initial treatment applications significantly altered vegetation structure and coverage to the extent of limiting use by many avian communities (Chapter 1). However, from 2 years post-treatment through the end of the study, TACV tended to follow a gradient of burn + herbicide followed by herbicide only, then burn only, and finally control plots. Lack of repeated disturbance in herbicide only plots may have favored common species of mid-rotation pine stands that prefer adequate shrub nesting habtiat.

Reductions in upper level visual obstruction and basal area of midstory hardwoods were not always evident in treated sites. As mentioned, independent treatments had opposite hardwood midstory control abilities (e.g., imazapyr with initial reduction, repeated burns with eventual reduction). Consequently, changes in visual obstruction followed temporal gradients of plant succession and disturbance frequency for herbicide only and prescribed burn only sites, respectively. Changes in relative abundances of white-eyed vireos and hooded warblers over time between controls and herbicide only sites emulated this successional change as vegetative structure of herbicide only sites assimilated to controls. Meanwhile, vegetation structure of burn only assimilated to that of burn + herbicide towards the end of the study due to repeated burns having a great impact on vegetation height than a one time application of imazapyr. Both of these species and Kentucky warblers are associated with dense mid- and understory vegetative structure (Hamel 1992). Burn only plots initially provided substantial hardwood midstory structure favoring species such as hooded warblers, but repeated burns reduced this structure and stimulated low level visual obstruction (0.0-0.9 m). Hooded warbler relative abundance eventually decreased in burn only plots. Overall, applying independent and combined treatments across the landscape would increase vegetation structure diversity, consequently influencing regional avian diversity.

Bird species with greater relative abundances in controls most likely represented common species of intensively managed, mid-rotation pine stands under typical forest management (e.g., thinnings and fertilization). Species such as hooded warbler, northern cardinals, and black-and-white warblers prefer closed canopy forests with increased upper-level visual obstruction (Hamel 1992). Wood thrushes also can benefit from similar habitat structure when adequate nesting habitat (shrub cover) is available (Hamel 1992, Hunter et al. 2001). Treated sites, especially herbicide treated plots (e.g., burn + herbicide and herbicide only, respectively), offered nesting cover (patches of *Rubus spp.*)

but lacked necessary canopy coverage for these species (Hunter et al. 2001). White-eyed vireos and occasionally red-eyed vireos also tended to have greater relative abundances with established hardwood midstories and significant visual obstruction (e.g., control plots). Although conserving these species is important, species favoring early successional, frequently disturbed forest stands similar to areas created by burn + herbicide have greater conservation demands considering current population estimates and the general lack of this type of vegetation structure and cover in the southeastern United States (Brawn et. 2001). Therefore, incorporating prescribed fire and imazapyr applications with typical mid-rotation management approaches could expand conservation capabilities of intensively managed forests.

A few species had sporadic responses to treatments [e.g., blue-gray gnatcatchers (*Polioptila caerulea*), red-bellied and red-headed woodpeckers, and ruby-throated hummingbirds] due possibly to fewer observations than previously discussed species. Minimal observations most likely decreased number of viable models available for habitat associations. Other species, such as mourning doves and prairie warblers, most likely used edges of treatment plots favoring neighboring clear-cuts, young pine stands, and firelanes. At the beginning of the study, most treated plots had less canopy coverage supporting a greater shrub component preferred by prairie warblers (Wilson et al. 1995). As the study progressed, shrub components remained but overstory canopies closed possible causing a decrease in bird abundance within treatment plots.

A few differences in species relative abundances among treatments were unexpected due to animal behavior and may be attributed to unmeasured treatment plot characteristics. Relative abundances of canopy dwellers such as pine warblers and

Carolina chickadees were greater in herbicide treated plots. However, overstory tree species were similar among treatments. Herbicide treated plots may have had greater snag densities for secondary cavity nesters such as Carolina chickadees to use versus burn only and controls (Hamel 1992). Unmeasured characteristics such as overstory invertebrate relative abundance and seed abundance may have influenced these species relative abundances among treatments.

Multiple variables such as spatial scale (i.e., area impacted versus bird perception), frequency and intensity of disturbance, and natural versus artificial disturbances all affect avian responses and should be considered when designing future projects (Southwood 1988, Alverson et al. 1994, Askins 2001, Vos et al. 2001). Within a landscape matrix of intensively managed pine, application of all treatments would create a landscape mosaic beneficial to a wide variety of species, but 10 ha treatment plots may not have been large enough to attract some individuals and may have limited number of nesting sites for territorial species. Stand-level ($\geq$ 60 ha) applications of burn + herbicide or independent treatments could attract many open woodland bird species that had few observations in this smaller scale study and that are in decline across the region (e.g., brown-headed nuthatch, Bachman's sparrow, northern bobwhite; Brawn et al. 2001). As such, it is essential for managers to use disturbance regimes capable of reducing hardwood midstories and perpetuating herbaceous understories (Plentovich et al. 1998). Burn + herbicide can create these conditions efficiently (within 2 years) and repeated burns could continually perpetuate required vegetation community components of low vegetation height and a predominately herbaceous understory. Longer fire-return intervals (> 3 years) or single imazapyr herbicide applications without fire may eventually favor greater midstory obstruction and woody plant biomass.

Use of fire in intensively managed forests may provide benefits to avian communities undetected in this study. Some declining species of the southeastern United States depend on fire for its affect on vegetative structure and diversity [e.g., red-cockaded woodpecker (*Picoides borealis*), brown-headed nuthatch, and Bachman's sparrow; Jackson 1988, Dunning and Watts 1990, Wilson et al. 1995, Plentovich et al. 1998]. Variations in fire intensity across the landscape also can benefit greater avian community diversity by creating a patchwork mosaic of different vegetation structures (Vega Rivera et al. 1999, Powell et al. 2000). Other benefits can include ecosystem benefits such as nutrient cycling, fuel reduction, and soil carbon sequestration that may help maintain habitat components necessary for a myriad of bird species (MacArthur and MacArthur 1961, Laverty and Williams 2000, USDA Forest Service 2000). However, research regarding habitat associations of avifauna within intensively managed forests is still lacking regarding habitat-centered response not treatment oriented and habitat viability in terms of nest survival and population dynamics (Saab and Powell 2005).

Information-theoretic approaches focus inward-calculating statistics based on data used and as with hypothesis testing, must be interpreted with caution (Anderson and Burnham 2002, Burnham and Anderson 2008). Minimal observations may have led to incomplete vegetation associations among avifauna and vegetative communities. However, general trends in vegetation associations among bird species with substantial observations helped construct some of the dynamics of avifauna's indirect response to treatments such as responses to decreases in hardwood midstory basal area, increases in understory herbaceous cover, and alterations to visual obstruction at upper and lower levels.

Future studies should refine their sampling approaches to focus on management practices influencing vegetation community characteristics in a habitat- and bird-centered manner similar to suggestions by Saab and Powell (2005). Determining associations among nest success and habitat characteristics across a range of management practices also could provide necessary information for effective management prescriptions. However, many of these practices require extensive sampling effort unattainable due to time and expense. Bird banding offers a cost-effective alternative that requires less sampling time and provides extensive information relevant to population survival such as age- and gender-specific survival rates and also can provide information regarding bird residency (Brownie et al. 1985). Using mist nests, researchers can gain reliable knowledge on species composition, relative abundance, population size, and demography (Dunn and Ralph 2004). Compared to point counts, mist net sampling can be easily standardized, has low observer bias, can detect species possibly missed using count methods, and provide "in-hand" information including physical characteristics and capture history (Dunn and Ralph 2004).

Future research also needs to investigate feasibility of incorporating management approaches within the current forest management framework. Positive responses of total avian conservation value and high priority species to treatments supports treatment use for the benefit of long-term sustainability of biodiversity. However, cost-benefit analyses should be performed to determine if costs of treatments can be alleviated by increased conservation value. Other benefits also should be investigated such as greater timber yield, increases in forage quality for white-tailed deer (McInnis et al. 2004, Iglay et al. 2010), and possible attraction of greater hunting lease rates due to habitat improvement (D. A. Miller, unpublished data). Past studies caution that timber yield benefits may be marginal at best especially if common practices of fertilizing occur (McInnis et al. 2004, Chapter 1). Therefore, benefits of these treatments to sustainable forestry and wildlife conservation may need to be emphasized to justify their application in southeastern pine stands.

Management Implications

Dormant season prescribed fire and imazapyr create habitat conditions that can support avian community conservation in intensively managed mid-rotation pine stands of Mississippi. Applied at the 4 levels used in this study, treatments created vegetation structure that can increase vegetation structure and coverage diversity across the landscape. Dormant season burns are ideal for pine plantations because they have minimal impact on tree growth compared to growing season burns, which have the tendency to cause crown scorch (Wade and Lunsford 1989, McInnis et al. 2004). Compared to imazapyr, dormant season burns are less effective at reducing hardwood midstory competition over the short-term (< 6 years) but may offer ecosystem benefits undetected in this study (i.e., nutrient cycling, seed scarification) which may benefit sustainable forestry in the long-term. Restricted presence of high-priority species (e.g., Bachman's sparrow) to treated plots suggests the conservation value potential of these treatments within working landscapes. Managers should consider these treatments whenever contemplating hardwood midstory control in mid-rotation pine stands of the southeastern United States.

Managers also should consider that many vegetation associations of bird species detected in this study are still relatively unknown. Current bodies of work, such as this one, are limited by space and time only providing local (i.e., site-specific) or national patterns of vegetation associations. With the high likelihood that bird vegetation associations are regional not national (Whittingham et al. 2007), more information is needed. I provided some causal relationship insights to species detected but caution on my sphere of inference. Longer-term, intensive sampling within an appropriate experimental design may help delineate vegetation associations of southeastern Neotropical migrants. Intensively managed forest matrices offer an ideal platform for such sampling efforts as they are disturbed routinely by controlled perturbations and offer a diversity of temporally segregated vegetation types (e.g., clear-cut, newly planted, pre-commercial thin, mid-rotation immediately post-thin and late-rotation). Intensive monitoring programs focused on bird-centered vegetative sampling over long temporal periods (> 10 years) in a replicated field experiment design are the key to providing new information regarding bird vegetation associations. However, such programs are costly requiring many work hours and skilled field workers. Even in research settings, these demands can limit sampling effort and even a project’s sphere of inference. Therefore, any effort set forth by landowners to monitor wildlife communities will ultimately support future conservation efforts.

As land coverage shifts from rural to urban, conservation and management of wildlife becomes ever more difficult and challenging. Managers are faced with a constantly moving target affected by land-use changes, invasive and exotic species, shifts in public perceptions of wildlife and natural places, and growing demands for efficient use of our natural resources. Landowners are faced with the same challenges and in addition, must endure economic pressures to maintain their land practices. Sustainable forestry programs such as the Sustainable Forestry Initiative (2005) are essential for supporting intensive forestry practices in these and future times. Forest industries abiding by these programs not only support sustainable forestry but also the conservation of our natural resources, such as avian communities. Dormant season prescribed fire and imazapyr are 2 silviculture tools that can help industry meet its sustainable forestry goals and help managers conserve avian communities well into the future.

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Table 2.1 Species observed while conducting point counts to investigate effects of prescribed fire and imazapyr on avian communities in intensively managed, mid-rotation pine stands of east-central Mississippi, 2000-2008. Only birds observed < 50 m from the observer are included.

	Treatment				
Species	Burn	Herbicide	Burn + herbicide	Control	Analysis[1]
Acadian Flycatcher	X	X	X	X	x
American Crow	X	X	X	X	x
American Goldfinch			X		
Bachman's Sparrow	X	X	X		x
Black and White Warbler	X	X	X	X	x
Blue-gray Gnatcatcher	X	X	X	X	x
Brown-headed Cowbird	X	X	X	X	x
Brown-headed Nuthatch	X	X	X	X	x
Blue Grosbeak	X	X	X		
Blue Jay	X	X	X	X	x
Black Vulture	X				
Brown Thrasher	X	X	X	X	
Broad-winged Hawk	X	X		X	
Carolina Chickadee	X	X	X	X	x
Carolina Wren	X	X	X	X	x
Chimney Swift	X			X	
Chipping Sparrow					
Common Grackle	X			X	
Cooper's Hawk	X				
Common Yellowthroat	X	X	X	X	x
Chuck-will's Widow	X				
Downy Woodpecker	X	X	X	X	x
Eastern Bluebird	X	X	X	X	x
Eastern Kingbird	X	X	X	X	
Eastern Phoebe	X		X		
Eastern Towhee	X	X	X	X	x
Eastern Wood-peewee	X	X	X	X	x
Eastern Tufted Titmouse	X	X	X	X	x
Eastern Wild Turkey	X	X	X	X	x
Fish Crow			X	X	
Gray Catbird	X	X	X	X	x
Great-crested Flycatcher	X	X	X	X	x
Hairy Woodpecker	X	X	X	X	x
Hooded Warbler	X	X	X	X	x
Indigo Bunting	X	X	X	X	x
Kentucky Warbler	X	X	X	X	x
Mourning Dove	X	X	X	X	x
Northern Bobwhite	X	X	X	X	x
Northern Cardinal	X	X	X	X	x

Table 2.1 (continued)

Species	Treatment				Analysis[1]
	Burn	Herbicide	Burn + herbicide	Control	
Northern Flicker	X	X	X	X	x
Northern Parula	X	X	X	X	
Painted Bunting	X				
Pine Warbler	X	X	X	X	x
Pileated Woodpecker	X	X	X	X	x
Prairie Warbler	X	X	X	X	x
Red-bellied Woodpecker	X	X	X	X	x
Red-eyed Vireo	X	X	X	X	x
Red-headed Woodpecker	X	X	X	X	x
Red-shouldered Hawk	X	X			
Red-tailed Hawk	X	X			
Ruby-throated Hummingbird	X	X	X	X	x
Scarlet Tanager	X			X	
Summer Tanager	X	X	X	X	x
Turkey Vulture	X	X			
White-eyed Vireo	X	X	X	X	x
Wood Thrush	X	X	X	X	x
Worm-eating Warbler	X	X	X	X	x
Yellow-breasted Chat	X	X	X	X	x
Yellow-billed Cuckoo	X	X	X	X	x
Yellow-bellied Sapsucker	X				
Yellow Warbler	X	X		X	
Yellow-throated Vireo	X	X	X	X	
Yellow-throated Warbler	X	X	X	X	

[1] Species used for analysis had $\geq$ 40 observations.

Table 2.2 Interaction and main effects of covariate (baseline data from pre-treatment year), treatment (burn only, herbicide only, burn + herbicide, and control), and treatment × year on avifauna sampled with point counts in intensively managed pine stands in Kemper County, Mississippi, summer 1999-2008.

	Pre-treatment		Treatment		Year		Treatment × year	
Nesting Guilds Species	*F*	*P*-value	*F*	*P*-value	*F*	*P*-value	*F*	*P*-Value
Ground-shrub nesters								
Bachman's sparrow	0.00	1.000	2.09	0.112	0.75	0.645	1.31	0.161
Black-and-white warbler	0.08	0.785	4.30	0.024	10.77	≤ 0.001	1.23	0.233
Common yellowthroat	0.20	0.661	12.52	≤ 0.001	2.57	0.015	2.59	≤ 0.001
Eastern towhee	0.01	0.921	18.77	≤ 0.001	12.65	≤ 0.001	1.26	0.211
Gray catbird	126.96	≤ 0.001	3.52	0.031	2.81	0.009	2.39	≤ 0.001
Hooded warbler	3.32	0.075	45.89	≤ 0.001	33.82	≤ 0.001	4.69	≤ 0.001
Indigo bunting	30.24	≤ 0.001	44.35	≤ 0.001	15.13	≤ 0.001	2.49	≤ 0.001
Kentucky warbler	0.26	0.616	16.89	≤ 0.001	8.62	≤ 0.001	2.36	≤ 0.001
Northern bobwhite	0.48	0.489	3.80	0.014	3.69	≤ 0.001	1.09	0.357
Northern cardinal	1.67	0.217	9.90	≤ 0.001	25.17	≤ 0.001	2.08	0.006
Prairie warbler	6.73	0.013	0.16	0.925	4.40	≤ 0.001	2.03	0.005
Wood thrush	2.41	0.128	8.82	≤ 0.001	28.49	≤ 0.001	2.16	0.004
Worm-eating warbler	18.05	≤ 0.001	0.73	0.543	88.40	≤ 0.001	0.61	0.920
Yellow-breasted chat	2.58	0.115	4.46	0.011	19.31	≤ 0.001	2.46	≤ 0.001

Table 2.2 (continued)

	Pre-treatment		Treatment		Year		Treatment × year	
Nesting Guilds Species	*F*	*P*-value	*F*	*P*-value	*F*	*P*-value	*F*	*P*-Value
Cavity nesters								
Blue-gray gnatcatcher	0.61	0.440	2.02	0.120	21.77	≤ 0.001	2.25	≤ 0.001
Brown-headed nuthatch	3.42	0.072	1.50	0.233	3.87	≤ 0.001	1.09	0.362
Carolina chickadee	0.31	0.584	3.73	0.028	10.79	≤ 0.001	1.19	0.267
Carolina wren	0.06	0.816	2.12	0.157	38.86	≤ 0.001	1.04	0.423
Downy woodpecker	0.97	0.330	5.10	0.003	13.29	≤ 0.001	1.79	0.023
Eastern tufted titmouse	0.02	0.883	1.73	0.169	5.59	≤ 0.001	1.43	0.108
Eastern bluebird	9.20	0.005	0.59	0.625	6.35	≤ 0.001	1.26	0.198
Great crested flycatcher	0.04	0.846	1.02	0.392	8.48	≤ 0.001	0.74	0.804
Hairy woodpecker	0.01	0.926	1.14	0.339	1.69	0.103	1.15	0.295
Northern flicker	0.25	0.616	1.50	0.224	11.02	≤ 0.001	0.90	0.606
Pileated woodpecker	0.15	0.703	0.89	0.462	1.44	0.191	0.79	0.737
Red-bellied woodpecker	6.87	0.014	1.72	0.189	13.36	≤ 0.001	1.89	0.014
Red-headed woodpecker	0.12	0.727	3.41	0.021	10.10	≤ 0.001	1.88	0.011
Tree nesters								
Acadian flycatcher	1.22	0.275	0.55	0.653	4.66	≤ 0.001	0.64	0.890
American crow	0.00	1.000	0.24	0.898	2.40	0.024	0.97	0.509
Blue jay	0.60	0.440	0.35	0.792	4.04	≤ 0.001	0.87	0.648
Brown-headed cowbird	2.45	0.144	0.09	0.965	17.31	≤ 0.001	1.24	0.217
Eastern wood-pewee	0.00	0.998	43.13	≤ 0.001	3.53	0.002	3.64	≤ 0.001

Table 2.2 (continued)

	Pre-treatment		Treatment		Year		Treatment × year	
Nesting Guilds Species	*F*	*P*-value	*F*	*P*-value	*F*	*P*-value	*F*	*P*-Value
Mourning dove	0.02	0.902	5.36	0.002	2.58	0.011	1.95	0.007
Pine warbler	1.38	0.278	8.20	$\leq$ 0.001	37.74	$\leq$ 0.001	1.07	0.394
Red-eyed vireo	77.91	$\leq$ 0.001	2.58	0.069	4.74	$\leq$ 0.001	1.76	0.026
Ruby-throated hummingbird	1.15	0.288	2.03	0.118	2.90	0.007	1.67	0.040
Summer tanager	8.97	0.004	4.69	0.005	7.01	$\leq$ 0.001	0.94	0.552
White-eyed vireo	7.08	0.014	13.04	$\leq$ 0.001	4.84	$\leq$ 0.001	3.76	$\leq$ 0.001
Yellow-billed cuckoo	0.88	0.355	1.93	0.156	8.12	$\leq$ 0.001	1.43	0.110
Species Richness	0.71	0.415	2.60	0.064	41.03	$\leq$ 0.001	2.31	0.002
Total Abundance	0.96	0.340	6.13	0.002	65.01	$\leq$ 0.001	2.36	$\leq$ 0.001
Total Avian Conservation Value	0.08	0.780	8.84	$\leq$ 0.001	34.93	$\leq$ 0.001	1.60	0.054

Table 2.3 Least square mean estimates (SE) of mean relative bird abundance determined with point count surveys among treatments within years in intensively managed pine stands of Kemper County, Mississippi, 2000-2008.

			Burn	Herbicide	Burn + herbicide	Control	
Variable	Year	*P*-value	$\bar{x}$	$\bar{x}$	$\bar{x}$	$\bar{x}$	SE
Blue-gray Gnatcatcher	1[1]	≤ 0.001	0.12 B	0.16 B	0.09 B	0.33 A	0.03
	2	0.003	0.20 A	0.17 A	0.06 B	0.18 A	0.03
	3	0.998	0.00	0.00	0.00	0.00	0.03
	4	0.210	0.03	0.08	0.12	0.08	0.03
	5	0.685	0.08	0.12	0.12	0.08	0.03
	6	0.998	0.00	0.00	0.00	0.00	0.03
	7	0.878	0.00	0.00	0.01	0.03	0.03
	8	0.998	0.00	0.00	0.00	0.00	0.03
	9	0.926	0.00	0.00	0.02	0.01	0.03
Common Yellowthroat	1	0.114	0.16	0.09	0.12	0.31	0.07
	2	0.006	0.12 B	0.15 B	0.38 A	0.06 B	0.07
	3	0.018	0.13 B	0.09 B	0.36 A	0.11 B	0.07
	4	0.002	0.15 B	0.17 B	0.47 A	0.13 B	0.07
	5	≤ 0.001	0.20 B	0.13 B	0.62 A	0.04 B	0.07
	6	≤ 0.001	0.18 B	0.04 B	0.64 A	0.02 B	0.07
	7	0.074	0.17	0.08	0.27	0.03	0.07
	8	0.091	0.20	0.05	0.24	0.04	0.07
	9	0.339	0.09	0.01	0.18	0.04	0.07
Eastern Wood Peewee	1	≤ 0.001	0.03 B	0.03 B	0.20 A	0.04 B	0.03
	2	0.005	0.01 B	0.10 A	0.13 A	0.03 B	0.03
	3	≤ 0.001	0.05 B	0.03 B	0.34 A	0.03 B	0.03
	4	≤ 0.001	0.08 B	0.09 B	0.49 A	0.05 B	0.05
	5	≤ 0.001	0.20 B	0.02 C	0.43 A	0.00 C	0.05
	6	≤ 0.001	0.15 B	0.03 C	0.33 A	0.01 C	0.04
	7	≤ 0.001	0.13 B	0.01 C	0.23 A	0.01 C	0.03
	8	0.358	0.09	0.01	0.08	0.02	0.04
	9	0.121	0.04	0.02	0.15	0.00	0.05
Gray Catbird	1	0.039	0.01 AB	0.04 A	0.06 A	0.00 B	0.02
	2	≤ 0.001	0.01 B	0.02 B	0.14 A	0.00 B	0.02
	3	0.713	0.01	0.01	0.01	0.00	0.02
	4	0.713	0.01	0.01	0.01	0.00	0.02
	5	0.814	0.01	0.01	0.02	0.00	0.02
	6	0.414	0.04	0.01	0.01	0.00	0.02
	7	0.138	0.01	0.01	0.05	0.06	0.02
	8	0.030	0.01 B	0.02 B	0.01 B	0.08 A	0.02
	9	0.333	0.01	0.01	0.01	0.05	0.02

Table 2.3 (continued)

Variable	Year	*P*-value	Burn $\bar{x}$	Herbicide $\bar{x}$	Burn + herbicide $\bar{x}$	Control $\bar{x}$	SE
Hooded Warbler	1	0.012	0.20 AB	0.07 B	0.06 B	0.41 A	0.08
	2	0.006	0.17 AB	0.03 B	0.04 B	0.30 A	0.06
	3	0.047	0.38 AB	0.63 A	0.27 B	0.55 A	0.09
	4	≤ 0.001	0.33 B	0.35 B	0.10 C	0.61 A	0.07
	5	≤ 0.001	0.47 BC	0.67 B	0.32 C	1.01 A	0.08
	6	≤ 0.001	0.49 B	0.94 A	0.27 B	0.81 A	0.09
	7	≤ 0.001	0.47 B	1.22 A	0.26 B	1.49 A	0.13
	8	≤ 0.001	0.32 B	1.22 A	0.42 B	1.19 A	0.10
	9	≤ 0.001	0.40 B	1.26 A	0.38 B	1.24 A	0.10
Indigo Bunting	1	≤ 0.001	1.43 AB	1.14 BC	1.70 A	0.84 C	0.13
	2	≤ 0.001	1.05 B	1.19 B	1.94 A	0.66 C	0.13
	3	≤ 0.001	0.50 B	1.02 A	1.24 A	0.52 B	0.13
	4	≤ 0.001	0.60 BC	0.84 AB	1.09 A	0.35 C	0.13
	5	0.003	0.72 A	0.70 A	1.00 A	0.30 B	0.13
	6	≤ 0.001	0.61 B	0.69 AB	1.01 A	0.22 C	0.13
	7	≤ 0.001	1.32 B	0.69 C	1.72 A	0.26 D	0.13
	8	≤ 0.001	0.64 B	0.44 BC	1.41 A	0.19 C	0.13
	9	≤ 0.001	0.74 B	0.24 C	1.10 A	0.15 C	0.13
Kentucky Warbler	1	≤ 0.001	0.21 A	0.02 B	0.03 B	0.38 A	0.06
	2	≤ 0.001	0.29 AB	0.12 BC	0.00 C	0.41 A	0.06
	3	≤ 0.001	0.31 A	0.11 BC	0.00 C	0.27 AB	0.06
	4	≤ 0.001	0.39 A	0.15 B	0.02 B	0.40 A	0.06
	5	≤ 0.001	0.24 BC	0.48 A	0.14 C	0.36 B	0.06
	6	0.004	0.50 A	0.45 A	0.25 B	0.55 A	0.06
	7	0.008	0.34 A	0.42 A	0.14 B	0.38 A	0.06
	8	0.003	0.26 AB	0.43 A	0.13 B	0.16 B	0.06
	9	0.003	0.27 AB	0.36 A	0.04 C	0.19 BC	0.06
Morning Dove	1	≤ 0.001	0.00 B	0.00 B	0.07 A	0.00 B	0.01
	2	0.854	0.00	0.00	0.01	0.00	0.01
	3	0.786	0.02	0.02	0.01	0.01	0.01
	4	0.852	0.01	0.01	0.01	0.00	0.01
	5	0.177	0.04	0.01	0.02	0.02	0.01
	6	0.025	0.00 B	0.03 A	0.03 A	0.00 B	0.01
	7	0.854	0.00	0.00	0.01	0.00	0.01
	8	0.072	0.00	0.00	0.03	0.00	0.01
	9	1.000	0.00	0.00	0.00	0.00	0.01

Table 2.3 (continued)

Variable	Year	P-value	Burn $\bar{x}$	Herbicide $\bar{x}$	Burn + herbicide $\bar{x}$	Control $\bar{x}$	SE
Northern Cardinal	1	0.078	0.44	0.33	0.32	0.58	0.07
	2	0.230	0.32	0.48	0.23	0.34	0.08
	3	0.042	0.43 AB	0.46 A	0.24 B	0.55 A	0.07
	4	0.002	0.28 A	0.28 A	0.07 B	0.26 A	0.04
	5	0.197	0.55	0.45	0.51	0.70	0.08
	6	0.074	0.61	0.54	0.50	0.79	0.08
	7	0.010	0.68 AB	0.71 AB	0.38 B	1.01 A	0.12
	8	0.004	0.57 AB	0.78 A	0.28 B	0.83 A	0.10
	9	$\leq$ 0.001	0.50 B	0.71 A	0.27 C	0.82 A	0.07
Prairie Warbler	1	0.002	0.05 B	0.07 B	0.04 B	0.16 A	0.02
	2	0.247	0.09	0.06	0.03	0.07	0.02
	3	0.014	0.01 B	0.07 AB	0.12 A	0.06 AB	0.02
	4	0.994	0.03	0.03	0.03	0.02	0.02
	5	0.252	0.06	0.00	0.05	0.01	0.02
	6	0.613	0.03	0.05	0.01	0.01	0.02
	7	0.978	0.01	0.00	0.00	0.00	0.02
	8	0.904	0.02	0.00	0.00	0.01	0.02
	9	0.995	0.00	0.00	0.00	0.00	0.02
Red-eyed Vireo	1	0.349	0.32	0.22	0.18	0.20	0.06
	2	0.686	0.14	0.22	0.26	0.23	0.07
	3	0.066	0.26	0.08	0.08	0.07	0.06
	4	0.093	0.17	0.09	0.09	0.10	0.03
	5	0.487	0.19	0.08	0.19	0.14	0.06
	6	0.130	0.20	0.05	0.18	0.07	0.05
	7	0.069	0.32	0.11	0.17	0.29	0.06
	8	0.295	0.31	0.10	0.16	0.24	0.08
	9	0.025	0.28 AB	0.11 B	0.15 B	0.40 A	0.07
Red-bellied Woodpecker	1	0.267	0.04	0.02	0.08	0.04	0.02
	2	0.155	0.01	0.03	0.00	0.01	0.01
	3	0.374	0.01	0.04	0.06	0.04	0.02
	4	0.541	0.03	0.04	0.02	0.00	0.02
	5	0.264	0.15	0.16	0.27	0.14	0.05
	6	0.025	0.20 AB	0.08 B	0.26 A	0.09 B	0.04
	7	0.199	0.06	0.13	0.17	0.07	0.04
	8	0.032	0.21 A	0.02 B	0.07 B	0.13 AB	0.05
	9	0.560	0.11	0.13	0.13	0.07	0.03

Table 2.3 (continued)

Variable	Year	*P*-value	Burn $\bar{x}$	Herbicide $\bar{x}$	Burn + herbicide $\bar{x}$	Control $\bar{x}$	SE
Red-headed Woodpecker	1	0.174	0.00	0.04	0.01	0.00	0.02
	2	1.000	0.00	0.00	0.00	0.00	0.02
	3	≤ 0.001	0.04 B	0.06 B	0.16 A	0.06 B	0.02
	4	0.800	0.03	0.01	0.02	0.02	0.02
	5	0.482	0.03	0.00	0.02	0.01	0.02
	6	0.132	0.02	0.00	0.04	0.00	0.02
	7	0.943	0.00	0.00	0.01	0.00	0.02
	8	0.155	0.02	0.00	0.04	0.04	0.02
	9	0.734	0.01	0.00	0.00	0.02	0.02
Ruby-throated Hummingbird	1	0.094	0.01	0.04	0.00	0.01	0.01
	2	0.260	0.06	0.02	0.08	0.03	0.02
	3	0.629	0.00	0.01	0.02	0.02	0.01
	4	0.434	0.00	0.02	0.01	0.01	0.01
	5	0.380	0.02	0.00	0.01	0.01	0.01
	6	0.038	0.01 B	0.00 B	0.04 A	0.00 B	0.01
	7	0.196	0.00	0.00	0.00	0.01	0.01
	8	0.287	0.02	0.00	0.03	0.00	0.01
	9	0.047	0.00 B	0.00 B	0.02 A	0.00 B	0.01
White-eyed Vireo	1	≤ 0.001	0.50 B	0.34 B	0.26 B	1.08 A	0.10
	2	≤ 0.001	0.54 B	0.30 C	0.25 C	0.96 A	0.10
	3	≤ 0.001	0.63 B	0.51 C	0.31 C	0.93 A	0.10
	4	≤ 0.001	0.48 A	0.57 A	0.10 B	0.67 A	0.10
	5	0.024	0.56 AB	0.75 A	0.32 B	0.61 A	0.10
	6	0.235	0.45	0.59	0.30	0.42	0.10
	7	≤ 0.001	0.33 C	1.20 A	0.25 C	0.77 B	0.10
	8	≤ 0.001	0.40 B	0.86 A	0.36 B	0.43 B	0.10
	9	0.009	0.34 B	0.70 A	0.32 B	0.25 B	0.10
Wood Thrush	1	0.478	0.08	0.08	0.04	0.03	0.03
	2	0.020	0.04 A	0.00 B	0.00 B	0.01 B	0.01
	3	0.908	0.27	0.34	0.31	0.31	0.06
	4	0.100	0.06	0.04	0.01	0.08	0.02
	5	≤ 0.001	0.14 BC	0.21 B	0.08 C	0.33 A	0.04
	6	0.003	0.08 BC	0.20 AB	0.02 C	0.34 A	0.06
	7	0.091	0.44	0.30	0.11	0.40	0.09
	8	0.337	0.24	0.17	0.10	0.29	0.07
	9	0.010	0.15 BC	0.29 AB	0.04 C	0.35 A	0.06

Table 2.3 (continued)

Variable	Year	P-value	Burn	Herbicide	Burn + herbicide	Control	SE
			$\bar{x}$	$\bar{x}$	$\bar{x}$	$\bar{x}$	
Yellow- breasted chat	1	0.114	1.22	1.04	0.66	1.45	0.22
	2	0.573	1.51	1.30	1.48	1.11	0.22
	3	0.256	1.24	1.39	1.47	0.93	0.20
	4	0.059	0.82	1.14	0.91	0.71	0.11
	5	0.015	1.17 A	1.29 A	1.37 A	0.73 B	0.14
	6	0.004	0.83 BC	0.92 AB	1.23 A	0.51 C	0.12
	7	≤ 0.001	0.94 A	1.24 A	1.29 A	0.33 B	0.16
	8	≤ 0.001	0.61 B	0.65 B	1.13 A	0.18 C	0.13
	9	≤ 0.001	0.29 BC	0.42 B	0.65 A	0.10 C	0.08
Species Richness	1	0.142	5.90	5.28	5.47	6.89	0.51
	2	0.970	4.99	5.05	5.17	5.20	0.35
	3	0.069	5.20	5.88	6.05	5.40	0.25
	4	0.013	4.55 BC	5.26 A	5.13 AB	4.32 C	0.22
	5	≤ 0.001	6.55 B	6.81 B	7.21 A	5.99 C	0.20
	6	0.004	6.18 AB	6.49 A	6.78 A	5.58 B	0.22
	7	0.002	6.59 AB	7.24 A	5.88 C	6.19 BC	0.24
	8	0.126	5.45	5.82	5.53	4.61	0.37
	9	0.132	5.05	6.01	5.22	5.08	0.32
Total Abundance	1	0.270	8.30	7.48	7.91	9.62	0.79
	2	0.463	7.34	7.39	8.41	7.02	0.65
	3	0.002	7.02 B	8.67 A	9.19 A	7.06 B	0.45
	4	0.020	5.33 BC	6.23 A	6.06 AB	5.02 C	0.32
	5	≤ 0.001	8.14 B	8.50 B	9.34 A	7.05 C	0.30
	6	≤ 0.001	7.74 B	8.11 AB	9.12 A	6.67 C	0.31
	7	0.039	9.74 AB	11.17 A	9.72 B	9.08 B	0.51
	8	0.034	7.13 AB	8.14 A	7.96 A	5.95 B	0.56
	9	0.023	6.38 B	8.13 A	7.36 AB	6.58 B	0.43

[1] Treatments with the same letter do not differ significantly (*P* > 0.05).

Table 2.4 Explanatory variables used to relate bird species relative abundance and community indices to vegetation structure and coverage components in intensively managed pine stands of Mississippi treated with prescribed fire and imazapyr and sampled in summer from 2000-2008.

Code	Description
PTMID_HGT	Mean height of midstory pine trees
PTOVER	Mean overstory pine basal area
HWDMID_BA	Mean midstory hardwood basal area
HWDOVER	Mean overstory hardwood basal area
HWDSN_BA	Mean hardwood snag basal area
CANCO	Percentage canopy cover
NUDDSLOW	Mean percentage cover of Nudds board sections 1-3 (0- 90cm)
NUDDSHIGH	Mean percentage cover of Nudds board sections 4-6 (90-180cm)
NFOUR	Mean percentage cover 4^{th} Nudds board section (90-120 cm)
BG	Exposed bare ground coverage
D	Exposed debris coverage
FN	Understory fern coverage
GL	Understory sedge and rush coverage
GR	Understory grass coverage
LE	Understory legume coverage
SV	Understory semi-woody vine coverage
WO	Understory woody plant coverage
WV	Understory woody vine coverage

Table 2.5 Cumulative Akaike's Criterion weights (Σw_i) of vegetation association model parameters for bird species in intensively managed, mid-rotation pine stands of Mississippi treated with prescribed burning with or without imazapyr application and sampled by point counts, summer 2000-2008.

Variable	Model parameter	Estimate[1]	Σw_i
Brown-headed cowbird	FN	1.051	1.0000
Brown thrasher	PTMID_HT	0.000	0.2605
	PTOVER	0.000	0.5165
	HWDMID_BA	-0.001	0.3945
	BG	-0.070	0.4441
Carolina wren	PTOVER	0.000	0.2054
	HWDSN_BA	0.102	0.6844
	CANCO	0.001	0.3056
	GL	0.044	0.2054
	LE	0.172	0.2891
Common yellowthroat	WO	-0.571	0.9819
	WV	0.015	0.1900
Eastern towhee	HWDMID_BA	-0.080	0.9984
	SV	0.167	0.4080
	WO	-0.318	0.5022
Hooded warbler	HWDOVER	0.004	0.2882
	CANCO	0.003	0.5101
	D	0.050	0.3000
	WV	0.032	0.2608
Indigo bunting	CANCO	-0.005	0.5302
	GR	-0.185	0.3753
	SV	0.021	0.2581
	WV	0.005	0.2415

Table 2.5 (continued)

Variable	Model parameter	Estimate[1]	Σw_i
Kentucky warbler	CANCO	0.005	0.7319
	D	-0.316	0.8252
	WV	0.000	0.1995
	WO	0.209	0.5514
Northern cardinal	HWDMID_BA	0.008	0.3592
	CANCO	-0.001	0.2914
	WO	0.457	0.8196
Northern flicker	SV	0.024	0.8293
	WV	0.004	0.3414
Pine warbler	NFOUR	0.000	1.0000
White-eyed vireo	NUDDSHIGH	0.000	0.2042
	WO	0.275	0.9920
Yellow-breasted chat	CANCO	-0.002	0.6460
	SV	-0.019	0.4719
Species richness	NUDDSLOW	0.005	1.0000

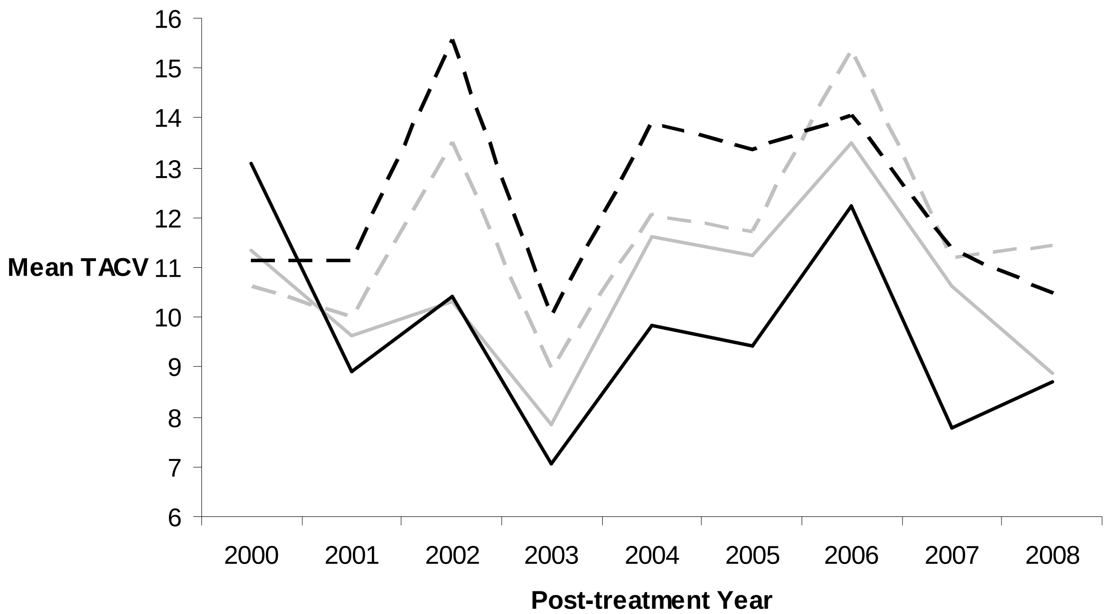

Figure 2.1 Mean total avian conservation value (TACV) in intensively managed, mid-rotation pine stands treated with factorial combinations of prescribed fire and imazapyr in east-central Mississippi, summer 2000-2008. Prescribed fire was applied via drip torch in January 2000 and 2003 and February and March 2006.

CHAPTER 3

SMALL MAMMAL RESPONSE TO FIRE AND IMAZAPYR APPLICATION IN INTENSIVELY MANAGED, MID-ROTATION PINE STANDS OF MISSISSIPPI

Intensively managed pine (*Pinus spp.*) forests cover 18 million ha in the southern United States, about 20% of southern forests, with 2.2 million ha in Mississippi (USDA Forest Service 2007). Management for economic gain and biodiversity objectives is achievable (e.g., Miller et al. 2009, Wigley et al. 2000), but current practices at mid-rotation may not achieve adequately long-term biodiversity and sustainable forestry objectives. Forest managers are increasingly expected to incorporate management efforts contributing to the conservation of biodiversity (Sustainable Forestry Initiative 2005), but current management practices at mid-rotation such as commercial thinning and fertilization may only provide short-term (< 4 years) benefits for conservation (Wood 1986, Peitz et al. 1999, Iglay et al. 2010*a*). Disturbances can perpetuate biodiversity (Turner et al. 2001, White and Jentsch 2001, Rundel et al. 1998), and although frequent disturbances favor disturbance-dependent or –tolerant species, timber management incorporating biocomplexity objectives could compensate for disturbance-intolerant species by creating a mosaic of stand treatments, vegetative structures, species, and successional stages across a landscape (Hunter 1990, Franklin 1993*ab*, Heljden et al.

1998, Carey et al. 1999*ab*, Tilman 1999). Managed forests can provide ecosystem benefits such as wildlife habitat, protection of water quality, and carbon sequestration while offering a mosaic of successional stages across the landscape (Petraitis et al. 1989, Greenberg et al. 1994, McLeod and Gates 1998, Vogt et al. 1999). Therefore, it is essential to determine optimal management approaches of intensively managed, mid-rotation pine (*Pinus spp.*) stands to meet forestry and biodiversity objectives (Wear and Greis 2002, Stein et al. 2005). Addition of dormant season prescribed fire and selective herbicide application after commercial thinnings may enhance wildlife habitat and sustain improved biodiversity until harvest supporting concepts of sustainable forestry (Hartley 2002, Carnus et al. 2003, Sustainable Forestry Initiative 2005).

Prescribed fire and selective herbicides are 2 silviculture tools used to control midstory hardwood competition and improve wildlife habitat in mid-rotation, intensively managed pine stands of the southeastern United States (Brockway and Outcalt 2000, Edwards et al. 2004, McInnis et al. 2004). Prescribed fire is similar to historical disturbances of the southeast (Brennen et al. 1998). Following specific prescriptions, dormant season prescribed fires, applied during winter, avoid detrimental effects on pine growth caused by crown scorch (Wade and Lunsford 1989, Bessie and Johnson 1995, Schimmel and Granstrom 1997). Selective herbicides, such as those containing imazapyr, offer an alternative to prescribed fire lacking smoke management issues or limited burning degree days (Wigley et al. 2002). Both treatments have demonstrated abilities to reduce woody plant coverage and consequently increase herbaceous understory plant coverage (Stransky and Harlow 1981, Brockway and Outcalt 2000, Miller and Miller 2004, Chapter 1). Past research has demonstrated treatment benefits to a myriad of game and non-game animals such as white-tailed deer (*Odocoileus virginianus*; Demarais et al. 2000, Mixon et al. 2009, Iglay et al. 2010), eastern wild turkey (*Meleagris gallopavo silvestris*; Dickson and Wigley 2001, Miller and Conner 2007), northern bobwhite (*Colinus virginianus*; Guynn et al. 2004, Miller and Miller 2004, Welch et al. 2004), and songbirds (Sladek et al. 2008). However, research is lacking regarding small mammal response to independent and combined applications of dormant season prescribed fire and imazapyr.

Small mammals (Orders: Rodentia and Soricidae) serve a variety of functional roles in forests [e.g., prey (Verts and Carraway 1998); consumers of invertebrates, vegetation, fruits, and seeds (Terry 1974, Gunther et al. 1983); dispersers of seed and fungal spores (Gashwiler 1970, Maser et al. 1978, Price and Jenkins 1986)] and demand research attention when addressing management impacts on faunal communities (Terry 1974, Gunther et al. 1983, Verts and Carraway 1998). Greater abundance of forest-dwelling small mammals in natural versus intensively managed forests have been observed (Carey and Johnson 1995, Wilson and Cary 2000), but forest management practices such as prescribed fire and selective herbicides may reduce differences by increasing habitat quality (Carey and Johnson 1995, Carey et al. 1999, Cole et al. 1998, Sullivan et al. 1998). These treatments affect disturbance-dependent small mammal food resources (Howard et al. 1959, Ahlgren 1966, Black and Hooven 1974).

Past studies have observed guild shifts post-treatment due to direct impacts on vegetation, not due to direct mortality of individuals. Positive effects of fire on understory herbaceous cover and seed-producing plants may benefit granivores (Fala 1875, Landers 1987, Masters et al. 1998). Increased herbaceous growth post-burn could harbor greater invertebrate abundance favoring insectivores (Fala 1975). Herbicide effects on small mammals may be temporary (Schulz 1997, Cole et al. 1998, Sullivan et al. 1998) supporting repeated prescribed fires every 3-6 years if early-successional habitat is favored. However, imazapyr's ability to increase coarse woody debris (CWD) and fire's tendency to reduce CWD raises questions regarding best management options between these treatments for small mammal conservation within intensively managed forest landscapes (Covington and Sackett 1984, Arno et al. 1995, Converse et al. 2006).

Difficulties in researching small mammal response include a tendency for individuals to respond to treatments more than populations (Lautenschlager 1993). Such responses may reduce observed differences among treated and untreated sites even when treatments have differential influence on small mammal presence and abundance (Clough 1987, Cole et al. 1998, Ford et al. 1999, Hood et al. 2002). Many past studies also were short-term limiting their ability to investigate long-term trends and treatments effects (e.g., Masters et al. 1998, Ford et al. 1999, and Converse et al. 2006). Therefore, I investigated small mammal response to prescribed fire and imazapyr for 9 years in intensively managed, mid-rotation pine stands of east-central Mississippi.

Study Area and Design

A full description of study area and design is provided in Chapter 1.

Methods

Vegetation Response and Climate Variables

I describe vegetation sampling methods in Chapter 1. I gathered rainfall and temperature data from in-stand gauges (1 set/stand), summer and winter 2001-2007. I checked gauges daily during trapping periods and calculated mean maximum and minimum daily temperatures. I also measured percentage cloud cover from 2001-2007 (except 2003).

Small Mammal Response

I trapped small mammals using drift fence arrays during May and June 1999-2007 and October 2000-2007 except October 2003 and Sherman live box traps (7.6 x 7.6 x 27.9-cm) baited with peanut butter and oats during January-March 1999-2007. I created drift fence arrays with four, 5-gallon buckets arranged as one center bucket with 5-m arms of 35.6 cm high aluminum flashing at 120^{o} angles from center bucket with a single bucket at arm ends. Along each arm, I placed one funnel trap of wire mesh (Enge 1997) so that each quadrant had one funnel trap. I permanently placed 3 drift fences per treatment plot diagonally with each $\geq$ 50 m from plot edge. I trapped 3 stands simultaneously for 10-days twice monthly May and June and once in October and recorded species and trap location (bucket and treatment plot) for every capture. When not in use, I closed traps by removing funnel traps, closing buckets using lids, and placing a large stick in each bucket to allow animals to escape in case lids were detached between trapping sessions. I placed Sherman live box traps randomly in a 5 X 5 grid with 20 m spacing (80 X 80 m) during 1999 and 2001-2007 ≥ 50 m from plot edge. In 2000, I used randomly placed 7 X 7 trapping grids (120 m X 120 m) instead of 5 X 5 grids to compare trapping efficiency. I trapped 2 stands (1999-2002) or 3 stands (2003-2007) simultaneously for 10 days or until a 50% recapture rate was reached with an adequate trapping history. I recorded species, gender, weight (g), and toe-clip number for mark recapture for every small mammal in the order Rodentia (Baumgartner 1940, Melchoir and Iwen 1965, Nietfeld et al. 1996). I also weighed Soricidae (shrews) but did not mark individuals. I removed Sherman traps from each site when not trapping. From these data, I calculated mean species catch-per-unit effort (CPUE) from new captures by treatment plot, species richness, and Shannon-Weaver diversity index for final analyses. I used CPUE as an index of relative abundance (Ludwig and Reynolds 1988) and therefore refer to relative abundance in results and discussion, not CPUE. I also grouped white-footed (*Peromyscous leucopus*) and cotton mice (*Peromyscous gossypinus*) as *Peromyscus* spp. due to similarities in field identification characteristics and hybridization among these species (McCarley 1954, Laerm and Boone 1994, Rich et al. 1996, Barko and Feldhamer 2002). I followed IACUC protocol #98-046 approved by Mississippi State University's IACUC for all small mammal trapping and handling.

Statistical Analysis

I tested the hypothesis of no difference in mean relative abundance, small mammal species richness, and relative abundance among treatments within years using mixed models, repeated measures analysis of covariance in SAS (MIXED procedure; SAS Institute Inc., Cary, North Carolina, USA). I used main effects of treatment, year, and treatment × year on species-specific relative abundance, species richness, and total relative abundance for summer and winter data. For each model, I used 4 treatment levels (burn, herbicide, burn + herbicide, control), random effect of stand (n = 6), repeated measures of year (n = 8; 2000-2007), and subject of stand × treatment (Littell et al. 2006). I used pretreatment (1999) small mammal relative abundance as a baseline covariate because pre-treatment small mammal communities may have differed among treatment plots treated alike even though small mammal reltaive abundances did not differ among treatments (Hood et al. 2002, Milliken and Johnson 2002). For fall trap data without pre-treatment estimates of relative abundance, I used mixed models, repeated measures analysis of variance in SAS. For each model, I selected an appropriate covariance structure from the following: 7-banded Toeplitz, heterogeneous compound symmetry, heterogeneous auto-regressive, and auto-regressive. I designated the covariance structure that minimized Akaike's Information Criterion corrected for small sample size (AIC_c) as the top candidate for analysis (Littell et al. 2006, Gutzwiller and Riffell 2007). I checked residuals and transformed data when deemed necessary to meet normality assumptions. I used Kenward-Roger correction for denominator degrees of freedom for repeated measures and small sample sizes (Littell et al. 2006, Gutzwiller and Riffell 2007). I used LSMEANS SLICE option to identify a treatment effect within years following a significant interaction and LSMEANS PDIFF to conduct pair-wise comparisons (Littell et al. 2006). All year references in results refer to years post-treatment. My *a priori* significance level was α=0.05.

I used varaible reduction and all-subsets regression models to investigate small mammal vegetation associations and response to climate variables within intensively managed, mid-rotation pine during summer and winter. Because my vegetation sampling focused on response to treatments, small mammal habitat associations would help visualize influence of within-stand treatment effects (e.g., changes in vegetation structure) on small mammal communities. Past studies have emphasized importance of plant forage crop, standing structure, and forest floor woody debris to small mammal communities (Carey and Johnson 1995, Masters et al. 1998, Kyle and Block 2000, Wilson and Carey 2000, Carey and Harrington 2001, Manning and Edge 2004, Block et al. 2005). Therefore, I used plant biomass (kg/ha), basal area (m^2/ha) of midstory and overstory trees, midstory tree height (m), Nudds board visual coverage, and volume of coarse woody debris (m^3/ha) as explanatory variables in model selection. I avoided examining all combinations of explanatory variables by looking for relationships among response variables (e.g., species-specific relative abundance) and explanatory variables by visually inspecting scatter plots. Although this approach may have excluded meaningful variables, especially when additive or interactive relationships with other explanatory variables occurred, I felt it was a relatively unbiased approach to data reduction that considered all possible explanatory variables for each dependent variable. I always chose summary variables over independent measurements and biomass over coverage variables when both were chosen for a model (e.g., Nudds board high instead of Nudds board levels 4, 5, or 6; semi-woody vine biomass instead of semi-woody vine coverage) and removed response variables without any noticeable relationships to explanatory variables from model comparisons. To avoid multicolinearity, I conducted pair-wise comparisons of correlation coefficients among selected explanatory variables, and when very correlated ($r \geq |0.5|$), removed the explanatory variable more weakly correlated to the response variable. If any global model had ≥ 6 explanatory variables, I reduced it to 5 explanatory variables (average number of variables of initial model sets) by removing the least correlated explanatory variable to avoid excessive numbers of models (i.e., 6 variables = 63 models) and inclusion of explanatory variables with minimal influence. Only 3 initial model sets had > 5 explanatory variables (n = 1 model with 7 variables, n = 2 models with 9 variables).

I used the MIXED procedure in SAS to evaluate regression models with stand (block) as the random effect, year as a repeated measure, and treatment plot as the subject (Littell et al. 2006). I also included treatment, year, and treatment × year as covariates. Because my analysis goal was to determine relative variable importance and I examined an equal number of models per parameter, I used summed model weights per parameter and model-averaged parameter estimates to determine overall impacts of each explanatory variable on small mammal species relative abundance (Burnham and Anderson 2002, Arnold 2010).

Results

I trapped 5,827 small mammals of 9 species over 32,845 trapnights (summer = 12,960 trapnights, winter = 14,865 trapnights, fall = 5,040 trapnights; Table 3.1). Least shrews (*Cryptotis parva*; 23%), short-tailed shrews (*Blarina c. carolinensis*; 21%), and *Peromyscus spp.* (25%) accounted for 69% of total captures. Only house mouse (*Mus musculus*) differed among treatments within years for summer drift fence sampling (Table 3.2) with house mouse relative abundance greatest in herbicide only plots in year 3 (Table 3.3). Short-tailed shrews ($\bar{x}$ = 0.02 vs. $\bar{x}$ = 0.01, SE = 0.002), least shrews ($\bar{x}$ =

0.03 vs. $\bar{x}$ = 0.02, SE = 0.003), and total relative abundance ($\bar{x}$ = 0.07 vs. $\bar{x}$ = 0.05, SE = 0.005) were greater in herbicide only and controls than burn treatments across all years during summer. Rice rat (*Oryzomys palustris*) relative abundance was greater in burn + herbicide than burn only ($\bar{x}$ = 0.0009 vs. $\bar{x}$ = 0.0002, SE = 0.0003) across all years during summer with relative abundance in herbicide only and controls intermediate ($\bar{x}$ = 0.0003, SE = 0.0003).

Relative abundances of 3 species, species richness, and total relative abundance differed among treatments within years in winter (Table 3.2). Short-tailed shrew relative abundance was greater in burn treatments (burn only and burn + herbicide) than herbicide only in year 1 but greater in herbicide only than all other treatments in year 6 (Table 3.3). Golden mouse (*Ochrotomys nuttali*) relative abundance was greater in controls than herbicide treatments in year 2 with burn only intermediate. In year 4, golden mouse relative abundance was still least in herbicide treatments but greater in burn only plots with relative abundance in controls intermediate. *Peromyscus* spp. relative abundance was greatest in burn + herbicide in year 2. Species richness had sporadic differences among treatments in years 1-2 and 7-8 without any definitive response to treatments. Total relative abundance was greater in burn + herbicide and controls than herbicide only in years 1-2 and greater in burn + herbicide than burn only and control in year 7. Rice rat relative abundance was greater in controls ($\bar{x}$ = 0.0006, SE = 0.00017) than burn treatments ($\bar{x}$ = 0.00004, SE = 0.00017) across all years during winter.

Short-tailed shrew and rice rat relative abundances differed among treatments within years in fall (Table 3.2). Short-tailed shrew relative abundance was greater in controls and herbicide only than burn + herbicide with burn only intermediate in year 5 and greatest in herbicide only in year 7 (Table 3.3). Rice rat relative abundance was greater in burn + herbicide and controls than independent treatments in year 5 and greater in controls than independent treatments in year 7 with burn + herbicide intermediate. Relative abundance of *Peromyscus* spp. was greater in burn + herbicide ($\bar{x}$ = 0.005, SE = 0.002) than burn only and controls ($\bar{x}$ = 0.004, SE = 0.002) with herbicide only ($\bar{x}$ = 0.007, SE = 0.002) intermediate across all years.

Nineteen of 46 explanatory variables were present in small mammal vegetation and climate associations (Table 3.4). Canopy coverage, mean rainfall, species richness of understory plants, and semi-woody vine biomass were primary explanatory variables of summer small mammal relative abundance associated with $\geq$ 4 species (Table 3.4). Midstory hardwood and hardwood overstory basal areas, woody plant biomass, minimum 24-hour temperature, upper Nudds board, and cloud coverage were associated with 2 species each. Winter small mammal abundance coincided mostly with mean rainfall, semi-woody vine biomass, and woody plant biomass. Number of rain days was the only other explanatory variable associated with > 1 species.

Discussion

Small mammal communities of intensively managed pine stands in east-central Mississippi had limited responses to dormant season prescribed fire and imazapyr treatments. Kirkland (1990) suggested the relatively minor effect of disturbance on small mammals which are inherently robust to disturbance and typically favor frequently disturbed environments. Therefore, if my treatments positively affect other faunal groups and their associated vegetative communities, applying prescribed fire and imazapyr to the landscape could support conservation of biodiversity without detrimental effects to small mammal communities.

Number of species captured in my study was similar to past studies (Dickson and Williamson 1988, Mengak et al. 1989, Daniel and Fleet 1999). Overall species richness was limited, and captures were dominated by few species (Langley and Shure 1980, Morrison and Anthony 1989, Perkins et al. 1988, Vickery et al. 1989, Daniel and Fleet 1999, Kirkland and Findley 1999, Darveau et al. 2001*)*. *Peromyscus* spp. comprised the vast majority of captures and has been described as a habitat generalist species (Miller and Getz 1977, Dueser and Shugart 1978, McComb and Rumsey 1982, Ormiston 1984, Adler and Wilson 1987). Generalist behavior of *Peromyscous spp.* and other species most likely help small mammal communities remain robust to frequent disturbances as suggested by Kirkland (1990). However, shrews, also dominating overall captures, offered some insights to treatment effects.

Based on summer captures, least and short-tailed shrews responded negatively to prescribed burning. Their dominance of total captures most likely caused total relative abundance to follow the same pattern. Both species prefer mesic litter conditions, that may be absent or less ideal post-burn (Chew 1951, Wrigley et al. 1979), and areas of abundant invertebrates (Getz 1961). Although both herbicide only and controls did not receive frequent burns, controls offered greater diversity of litter types (e.g., hardwood leaves and pine needles) and possibly more mesic conditions due to a closed canopy of pine overstory and hardwood midstory (Chapter 1). Fire reduces soil invertebrates (Pearse 1943, Metz and Farrier 1973), but I did not sample this faunal group. Previous work on my study sites investigating understory invertebrate abundance and carabid beetles found no differences among treatments other than immediate declines in some carabid beetle species post-burn (Iglay 2007). Although controls did not manipulate litter and may be beneficial for shrews, canopy closure reduces small mammal species richness of streamside management zones within intensively managed pine stands (Miller et al. 2004) and may limit beta diversity when only a single mid-rotation treatment approach is used (e.g., commercial thinning followed by fertilizing).

Past studies have found differences in small mammal communities when similar treatments were applied. Deer mice (*Peromyscus maniculatus*) densities have been shown to increase immediately post-burn in Arizona ponderosa pine (*Pinus ponderosa*; Converse et al. 2006). Ahlgren (1966) and Sullivan and Boateng (1996) attributed dramatic increases in deer mice post-burn to increased foraging ability for seeds and insects in Minnesota and British Columbia, respectively. Herbicide treatments have had positive (Kirkland 1978, Borrecco et al. 1979) and neutral effects on populations of deer mice (Sullivan and Sullivan 1982, Anthony and Morrison 1985). Santillo and colleagues (1989) observed reduced small mammal populations, especially insectivores and herbivores, for 3 years after herbicide treatment due to alterations in vegetation in Maine. Although prescribed burning can pose greater direct risks to small mammals (e.g., mortality), it is more likely to leave areas unaffected by treatments (i.e., unburned areas) than skidder-applied imazapyr. These areas can provide refuge for small mammals (Santillo et al. 1989). Ford et al. (1999) observed sufficient unburned areas post burn on Appalachian slopes providing refugia.

Increased diversity of vegetation types within and among pine stands has been suggested as important for supporting and maintaining small mammal communities

(Atkenson and Johnson 1979, Miller and Getz 1977, Clough 1987, Perkins et al. 1988, Kirkland 1990, DeGraaf et al. 1991, Bramble et al. 1992, Yahner 1992, Michael 1995). Typical intensive, short-rotation management forgoes hardwood midstory competition control creating a relatively monotypic vegetation structure among mid-rotation stands (e.g., pine overstory, hardwood midstory, shade-tolerant plant understory with expanses of exposed litter). Applying the 4 levels of treatment used in this study across the landscape may increase vegetation type heterogeneity (Chapter 1), but affects on small mammal population dynamics should be monitored. Measurements of reproductive rates, predation intensity, and survival among treated and untreated plots may help researchers describe overall treatment effects or better understand the lack thereof (Sullivan 1979, Loeb 1999).

Similar conditions at trap sites (microhabitat) among treatment plots may have limited detection of overall treatment effects on small mammal communities by influencing local capture success. Past research has observed greater diversity of microhabitats harboring greater small mammal diversity (Carey and Johnson 1995, Sullivan et al. 2001). Disturbances such as fire and imazapyr have the potential to create multiple microhabitat sites considering their abilities to perpetuate new plant growth, greater seed production and greater densities of invertebrates (Blake and Hoppes 1986, Chapter 1). If microhabitat diversity was present in my sites, similarities in microhabitat characteristics at trap-sites among treatment plots may have reduced observed differences among treatments. In addition, rodent species differ in their use of microhabitats (e.g., Dueser and Shugart 1978, McComb and Rumsey 1982) making it difficult to discern overall treatment effects at the macro-habitat level. Future studies should incorporate trap-centered vegetative sampling at the microhabitat-level to better understand small mammal response to treatments directly affecting vegetation communities. I agree with Ford et al. (1999) that short distances among various micro- and macro-sites tend to occur post-treatment providing ample, easy access to cover and food diversity for small mammals.

Vegetation structure and biomass and climate variables did not provide any information regarding indirect impacts of treatments on small mammal communities. When analyzed separately, most explanatory variables in the top models differentiated treatments (Chapter 1). However, variables important to small mammals did not differ among treatments. Coarse woody debris was one such variable, but a lack of associations with CWD in this study does not dismiss importance of CWD to small mammals. It provides nesting and travel cover in addition to insect and fungal food sources (Hayes and Cross 1987, Graves et al. 1988, Loeb 1999, Bowman et al. 2000, Carey and Harrington 2001). However, minimal affects of CWD on species-specific relative abundances are common in the literature (Loeb 1999, Menzel et al. 1999, Bowman et al. 2000). As suggested by Greenberg (2002), minor to insignificant differences in small mammal relative abundance among treatments may be due to some species not requiring it or the need for significant differences in CWD volumes among areas to reveal a population level response. Other vegetation variables such as stem density and vegetation structure height used in past studies (e.g., Masters et al. 1998) may have helped explain indirect effects of prescribed fire and imazapyr on small mammal communities. Although I did not detect many relationships among vegetation standing crop and small mammal relative abundances, positive correlations among these variables have been observed and they should be included in future research (Huntly and Inouye 1987).

Lack of differences among treatments does not simply imply minimal effect on small mammal communities as suggested by Kirkland (1990), but merits more intensive sampling incorporating research suggestions above to better understand small mammal community dynamics in intensively managed, mid-rotation pine stands of east central Mississippi. The functional roles fulfilled by small mammals elicit their incorporation in biodiversity monitoring studies and experiments investigating wildlife response to treatments (Terry 1974, Gunther et al. 1983, Gashwiler 1970, Maser et al. 1978, Price and Jenkins 1986, Verts and Carraway 1998). Long-term studies ($\geq$ 30 years) have been suggested as ideal platforms for gathering information on small mammal response to forest and wildlife management (Converse et al. 2006). However, within short-rotation, even-aged forests, mid-rotation to harvest lasts typically < 15 years, limiting study period length. Therefore, short-term (10-15 years) intensive sampling projects may offer the best solution to gaining new reliable knowledge of small mammal community response to prescribed burning and imazapyr in intensively managed, mid-rotation pine stands of east-central Mississippi.

Management Implications

Small mammals were mostly unaffected by dormant season prescribed burning and imazapyr in intensively managed, mid-rotation pine stands of east-central Mississippi. Considering importance of these species to forest ecosystem functions, managers should always consider possible treatment effects on these communities when determining management approaches for particular areas. However, considering absence of treatment effects in this long-term study and minimal detrimental effects observed by others (e.g., Masters et al. 1998, Ford et al. 1999, Converse et al. 2006), managers may want to consider these treatments for hardwood midstory control at mid-rotation in intensively managed pine. These 4 treatments can support greater biodiversity within intensively managed pine stands by creating different vegetation types across the landscape supporting diversity of vegetation types and possibly small mammal communities (Chapter 1, Atkenson and Johnson 1979, Miller and Getz 1977, Clough 1987, Perkins et al. 1988, Kirkland 1990, DeGraaf et al. 1991, Bramble et al. 1992, Yahner 1992, Michael 1995). In addition to mid-rotation applications of prescribed burning and imazapyr, managers also should consider landscape context with respect to succession of neighboring sites, streamside management zones, and untreated areas. With ever increasing pressures on wildlife from habitat loss and degradation, it is essential for wildlife managers and researchers to aid landowners in their quest for reasonable habitat management, within the constraints of landowner objectives, for harboring greater conservation value on their property such as dormant season prescribed fire and imazapyr. Practical management tools capable of perpetuating biodiversity while meeting landowner goals are essential for the manager's toolbox and the conservation of biodiversity.

Researchers need to consider difficulties involved with small mammal sampling at project onset. Able to withstand frequent disturbances, small mammal community response to wildlife and forest management practices can be difficult to discern, especially if their resilience to disturbance outweighs treatment effects. Intensive sampling incorporating measurements of underlying population dynamics of small mammal communities may offer a clearer picture of interactions among treatments, vegetation, and small mammal communities. I am confident that my research results are definitive, but skeptical that small mammals did not respond to overall changes in vegetation structure and plant biomass among treatments. Researchers also may need to consider larger scale impacts than experimental units allow for investigation. In my study, larger scale perturbations than my treatments (e.g., clear-cuts, thinnings, rotation length), landscape matrix, and mesofilters (e.g., CWD, snag retention, SMZ's) may have had a greater impact on small mammal communities than local burning and imazapyr applications (Hunter 2004). Therefore, larger scale research projects incorporating assessments of small mammal community structure and associated micro- and macro-habitat characteristics may reveal the overall impact of these treatments on small mammal communities.

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Table 3.1 Mammal species seen or sampled using drift fence arrays and Sherman-live box traps within mid-rotation pine stands treated with prescribed fire and imazapyr in Kemper County, Mississippi, 1999-2007.

Common Name	Scientific Name	Sampled
Armadillo	*Dasypus novemcinctus*	
Bobcat	*Lynx rufus*	
Cotton mouse	*Peromyscus gossypinus*	X
Coyote	*Canis latrans*	
Eastern gray squirrel	*Sciurus carolinensis*	
Eastern harvest mouse	*Reithrodontomys humulis*	X
Eastern mole	*Scalopus aquaticus*	
Eastern woodrat	*Neotoma floridana*	
Fox squirrel	*Sciurus niger*	
Golden mouse	*Peromyscus nuttalli*	X
Hispid cotton rat	*Sigmodon hispidus*	X
House mouse	*Mus musculus*	X
Least shrew	*Cryptotis parva*	X
Opossum	*Didelphis virginiana*	
Pine vole	*Pitymys pinetorum*	X
Eastern cottontail rabbit	*Sylvilagus floridanus*	
Northern Raccoon	*Procyon lotor*	
Red fox	*Vulpes vulpes*	
Rice rat	*Oryzomys palustris*	X
Southern flying squirrel	*Glaucomys volans*	
Southern short-tailed shrew	*Blarina carolinensis*	X
White-footed mouse	*Peromyscus leucopus*	X
White-tailed deer	*Odocoileus virginianus*	

Table 3.2 Interaction and main effects of covariate (baseline data from pre-treatment year), treatment (burn only, herbicide only, burn + herbicide, and control), and treatment × year on small mammals sampled with drift fences (summer and fall) and Sherman live box traps (winter) in intensively managed pine stands in Kemper County, Mississippi, 1999-2007.

	Pre-treatment		Treatment		Year		Treatment × year	
Season and Variable	*F*	*P*-Value	*F*	*P*-Value	*F*	*P*-Value	*F*	*P*-Value
Summer								
Blarina carolinensis	5.45	0.022	7.68	≤ 0.001	27.66	≤ 0.001	1.57	0.073
Cryptotis parva	0.94	0.335	5.27	0.005	19.11	≤ 0.001	1.45	0.112
Microtus pinetorum	0.81	0.373	0.97	0.419	3.35	0.004	0.85	0.654
Mus musculus	2.50	0.118	1.87	0.140	4.05	≤ 0.001	1.96	0.010
Ochrotomys nuttalli	0.06	0.810	0.31	0.821	3.36	0.004	1.54	0.081
Oryzomys palustris	0.03	0.866	2.73	0.050[1]	5.53	≤ 0.001	1.25	0.230
Peromyscus spp.	3.29	0.076	1.06	0.372	16.24	≤ 0.001	1.31	0.192
Reithrodontomys humulis	0.00	0.986	1.03	0.386	16.78	≤ 0.001	0.97	0.501
Sigmodon hispidus	1.14	0.289	0.76	0.524	2.25	0.041	1.02	0.448
Species richness	1.65	0.202	1.41	0.248	16.73	≤ 0.001	1.04	0.424
Total CPUE	0.38	0.543	4.22	0.012	25.67	≤ 0.001	1.49	0.100
Winter								
Blarina carolinensis	0.02	0.896	0.70	0.561	3.93	≤ 0.001	1.95	0.015
Cryptotis parva	0.00	0.000	2.51	0.065	8.09	≤ 0.001	1.09	0.365
Microtus pinetorum	0.20	0.661	0.52	0.673	5.37	≤ 0.001	1.27	0.215
Mus musculus	0.00	0.000	0.08	0.970	1.82	0.097	1.43	0.120
Ochrotomys nuttalli	0.99	0.334	2.94	0.064	13.14	< 0.001	2.10	0.008
Oryzomys palustris	0.02	0.887	2.75	0.047	2.33	0.027	0.85	0.655
Peromyscus spp.	2.88	0.096	1.18	0.328	11.70	≤ 0.001	1.85	0.024
Reithrodontomys humulis	1.90	0.178	1.24	0.312	6.89	≤ 0.001	1.21	0.258
Sigmodon hispidus	1.10	0.305	0.76	0.527	6.63	≤ 0.001	1.47	0.107
Species richness	10.29	0.004	0.85	0.481	30.63	≤ 0.001	2.57	≤ 0.001
Total CPUE	1.34	0.252	0.60	0.616	20.22	≤ 0.001	2.09	0.009
Fall								
Blarina carolinensis	-	-	4.76	0.009	34.11	≤ 0.001	2.78	0.001
Cryptotis parva	-	-	0.36	0.786	6.88	≤ 0.001	1.35	0.182
Microtus pinetorum	-	-	0.17	0.913	1.85	0.105	0.97	0.497
Mus musculus	-	-	0.28	0.836	18.51	≤ 0.001	1.47	0.139
Ochrotomys nuttalli	-	-	1.41	0.252	4.85	≤ 0.001	1.18	0.292
Oryzomys palustris	-	-	2.92	0.041	3.99	≤ 0.001	1.54	0.084
Peromyscus spp.	-	-	3.48	0.022	17.37	≤ 0.001	1.35	0.170
Reithrodontomys humulis	-	-	0.68	0.567	11.70	≤ 0.001	0.82	0.676
Sigmodon hispidus	-	-	0.09	0.960	6.40	≤ 0.001	0.89	0.590
Species richness	-	-	0.80	0.507	37.69	≤ 0.001	1.38	0.166
Total CPUE	-	-	1.27	0.306	38.66	≤ 0.001	1.29	0.216

[1] *P* value before rounding was 0.0496.

Table 3.3 Least square mean estimates (SE) of small mammal relative abundance (catch-per-unit-effort) among treatments within years in intensively managed pine stands of Kemper County, Mississippi, 2000-2007.

Season and Variable	Year	*P*-value	Burn $\bar{x}$	Herbicide $\bar{x}$	Burn + herbicide $\bar{x}$	Control $\bar{x}$	SE
Summer							
House Mouse *Mus musculus*	1	1.000	0.000	0.000	0.000	0.000	0.001
	2	0.210	0.001	0.001	0.001	0.000	0.001
	3	≤ 0.001	0.001 B	0.004 A	0.001 B	0.000 B	0.001
	4	0.817	0.000	0.001	0.000	0.000	0.001
	5	0.817	0.000	0.001	0.000	0.000	0.001
	6	1.000	0.000	0.000	0.000	0.000	0.001
	7	0.714	0.001	0.000	0.000	0.000	0.001
	8	0.110	0.000	0.000	0.001	0.001	0.001
Winter							
Southern Short-tailed Shrew *Blarina carolinensis*	1	0.022	0.01 A	0.00 C	0.00 AB	0.00 BC	0.00
	2	0.059	0.00	0.00	0.00	0.01	0.00
	3	0.427	0.00	0.00	0.00	0.00	0.00
	4	0.389	0.00	0.00	0.00	0.00	0.00
	5	0.626	0.00	0.00	0.00	0.00	0.00
	6	0.043	0.00 B	0.00 A	0.00 B	0.00 B	0.00
	7	0.356	0.00	0.00	0.00	0.00	0.00
	8	0.309	0.00	0.00	0.00	0.00	0.00
Golden Mouse *Ochrotomys nuttalli*	1	0.333	0.01	0.00	0.00	0.01	0.00
	2	0.024	0.01 AB	0.00 B	0.00 B	0.01 A	0.00
	3	0.075	0.01	0.01	0.00	0.01	0.00
	4	0.028	0.03 A	0.02 B	0.01 B	0.02 AB	0.00
	5	0.865	0.00	0.00	0.00	0.00	0.00
	6	0.991	0.00	0.01	0.01	0.01	0.00
	7	0.975	0.00	0.00	0.00	0.00	0.00
	8	0.975	0.00	0.00	0.00	0.00	0.00
Peromyscus spp.	1	0.161	0.01	0.00	0.01	0.01	0.00
	2	≤ 0.001	0.01 B	0.01 B	0.04 A	0.01 B	0.01
	3	0.446	0.01	0.03	0.02	0.02	0.01
	4	0.898	0.05	0.04	0.04	0.05	0.02
	5	0.171	0.04	0.02	0.05	0.02	0.01
	6	0.410	0.03	0.02	0.01	0.03	0.01
	7	0.093	0.00	0.01	0.01	0.00	0.00
	8	0.477	0.01	0.01	0.01	0.01	0.00
Species Richness	1	0.003	4.1 A	1.8 B	2.6 AB	3.3 A	0.4
	2	0.036	1.8 B	2.2 AB	1.8 B	3.1 A	0.4
	3	0.255	2.3	3.0	2.8	2.3	0.3
	4	0.595	3.9	3.5	4.8	3.1	0.9
	5	0.429	2.3	1.8	2.6	2.8	0.4
	6	0.641	2.3	2.8	2.8	2.3	0.4
	7	0.022	0.9 B	1.0 B	1.9 A	0.5 B	0.3
	8	0.043	1.1 A	1.2 A	0.4 B	0.7 AB	0.2

Table 3.3 (continued)

Season and Variable	Year	P-value	Burn $\bar{x}$	Herbicide $\bar{x}$	Burn + herbicide $\bar{x}$	Control $\bar{x}$	SE
Total Relative Abundance	1	0.006	0.03 A	0.01 B	0.02 A	0.02 A	0.00
	2	0.009	0.02 B	0.02 B	0.04 A	0.04 A	0.01
	3	0.183	0.04	0.06	0.05	0.04	0.01
	4	0.969	0.09	0.07	0.09	0.09	0.02
	5	0.359	0.06	0.03	0.06	0.04	0.01
	6	0.725	0.04	0.04	0.03	0.04	0.01
	7	0.064	0.00	0.01	0.02	0.00	0.00
	8	0.806	0.02	0.02	0.01	0.01	0.01
Fall							
Southern Short-tailed Shrew *Blarina carolinensis*	1	0.329	0.00	0.00	0.00	0.00	0.00
	2	0.240	0.01	0.02	0.02	0.02	0.01
	3	0.381	0.00	0.00	0.01	0.01	0.00
	5	0.001	0.03 B	0.04 A	0.02 B	0.04 A	0.00
	6	0.149	0.00	0.02	0.01	0.01	0.01
	7	0.007	0.00 B	0.02 A	0.00 B	0.00 B	0.00
	8	0.472	0.00	0.00	0.01	0.01	0.00
Rice Rat *Oryzomys palustris*	1	1.000	0.00	0.00	0.00	0.00	0.00
	2	1.000	0.00	0.00	0.00	0.00	0.00
	3	1.000	0.00	0.00	0.00	0.00	0.00
	5	$\leq$ 0.001	0.00 B	0.00 B	0.00 A	0.00 A	0.00
	6	1.000	0.00	0.00	0.00	0.00	0.00
	7	0.015	0.00 B	0.00 B	0.00 AB	0.00 A	0.00
	8	1.000	0.00	0.00	0.00	0.00	0.00

Table 3.4 Explanatory variables used to relate small mammal species relative abundance (CPUE) and species richness to vegetation structure and biomass and climate variables in intensively managed, mid-rotation pine stands of Mississippi treated with prescribed fire and imazapyr and sampled from 2000-2007.

Code	Description
CANCO[1]	Percentage canopy cover
HWDMID_BA	Mean midstory hardwood basal area
HWDMID_HGT	Mean midstory hardwood tree height
HWDOVER	Mean overstory hardwood basal area
HWDSN_HGT	Mean hardwood snag tree height
NLOW	Mean percentage cover of lower half of Nudds board (0.0-0.9m)
NHIGH	Mean percentage cover of top half of Nudds board (0.9-1.8 m)
PTOVER	Mean overstory pine basal area
PTSN_BA	Mean pine snag basal area
GL_BIOM[1]	Mean sedge and rush biomass
SR_BIOM	Mean species richness biomass
SV_BIOM	Mean semi-woody vine biomass
WO_BIOM	Mean woody plant biomass
WV_BIOM	Mean woody vine biomass
CCOV[2]	Mean cloud coverage
MAX	Mean maximum temperature
MIN	Mean minimum temperature
RAIN	Mean rainfall
RDAYS	Number of rain days

[1] Vegetation structure and biomass means were derived from season-specific sampling.

[2] Climate variables were measured during each trapping period.

Table 3.5 Cumulative Akaike's Criterion weights (Σw_i) of vegetation and climate association model parameters for small mammal species in intensively managed, mid-rotation pine stands of Mississippi treated with prescribed burning with or without imazapyr application and sampled by drift fence arrays (summer) and Sherman live box traps winter, 2000-2007.

Season	Variable	Model parameter	Estimate[1]	Σw_i
Summer	*Peromyscus* spp.	HWDMID_HGT	1.03E-14	0.000
		HWDOVER	-1.00E-04	1.000
		CANCO	9.04E-17	0.000
		GL_BIOM	-1.85E-18	0.000
		SR_BIOM	-1.63E-19	0.000
		CCOV	-1.78E-18	0.000
		RDAYS	-1.80E-04	1.000
	Eastern harvest mouse	CANCO	-1.00E-04	0.690
	Reithrodontomys humulis	NHIGH	1.33E-05	0.465
		SV_BIOM	7.31E-10	0.005
		SR_BIOM	-2.65E-07	0.007
		MIN	-1.17E-03	0.995
		RAIN	-4.56E-04	0.527
	Hispid cotton rat	HWDOVER	2.44E-04	1.000
	Sigmodon hispidus	CCOV	1.10E-05	1.000
	Least shrew	HWDMID_BA	-3.24E-04	0.298
	Cryptotis parva	CANCO	1.49E-06	0.201
		NHIGH	1.04E-04	0.817
		SV_BIOM	3.54E-08	0.014
		WO_BIOM	2.21E-06	0.088
		MAX	-8.72E-05	0.215
		MIN	-3.04E-03	0.987
		RAIN	-1.18E-03	0.524

Table 3.5 (continued)

Season	Variable	Model parameter	Estimate[1]	Σw_i
Summer	Southern Short-tailed Shrew *Blarina carolinensis*	PTOVER	7.12E-12	0.000
		HWDMID_BA	-6.74E-09	0.000
		CANCO	-1.26E-10	0.000
		SV_BIOM	1.49E-12	0.000
		SR_BIOM	-1.29E-06	0.032
		RAIN	-1.30E-04	1.000
	Species Richness	CANCO	-2.19E-03	0.191
		SV_BIOM	5.56E-04	0.608
		WO_BIOM	1.77E-04	0.308
		SR_BIOM	-1.94E-03	0.301
		RAIN	-1.10E-01	1.000
Winter	*Peromyscus* spp.	PTOVER	-9.75E-05	0.250
		HWDMID_BA	3.30E-04	0.232
		HWDMID_HGT	3.41E-05	0.171
		CANCO	-5.54E-04	0.876
		GL_BIOM	-8.35E-06	0.199
		MAX	-7.09E-04	0.828
		RAIN	-5.94E-04	0.465
		RDAYS	-9.19E-05	0.247
	Golden Mouse *Ochrotomys nuttalli*	PTSN_BA	-2.40E-04	0.089
		HWDSN_HGT	-1.42E-04	0.843
		NLOW	1.25E-05	0.116
		SV_BIOM	1.47E-06	0.887
		WO_BIOM	3.68E-05	0.986
		RAIN	2.35E-04	0.720
		RDAYS	-8.44E-05	0.658
	Eastern harvest mouse *Reithrodontomys humulis*	RAIN	-1.60E-04	1.000

Table 3.5 (continued)

Season	Variable	Model parameter	Estimate[1]	Σw_i
Winter	Hispid cotton rat	SV_BIOM	3.85E-06	0.907
	Sigmodon hispidus	WO_BIOM	-2.85E-07	0.239
		RAIN	2.60E-05	1.000
	Southern Short-tailed Shrew	SV_BIOM	-1.08E-07	0.322
	Blarina carolinensis	WO_BIOM	3.59E-06	0.689
		WV_BIOM	-3.56E-07	0.344
		SR_BIOM	1.63E-05	0.550
	Species Richness	WO_BIOM	2.13E-03	0.878
		WV_BIOM	7.78E-05	0.263

[1] Model-averaged estimate.

CHAPTER 4

HERPETOFAUNA RESPONSE TO FIRE AND IMAZAPYR APPLICATION IN INTENSIVELY MANAGED, MID-ROTATION PINE STANDS OF MISSISSIPPI

Intensively managed pine (*Pinus spp.*) forests cover 18 million ha in the southern United States, about 20% of southern forests, with 2.2 million ha in Mississippi (USDA Forest Service 2007). Management for economic gain and biodiversity objectives is achievable (e.g., Miller et al. 2009, Wigley et al. 2000), but current practices at mid-rotation may not achieve adequately long-term biodiversity and sustainable forestry objectives. Forest managers are increasingly expected to incorporate management efforts contributing to the conservation of biodiversity (Sustainable Forestry Initiative 2005), but current management practices at mid-rotation such as commercial thinning and fertilization may only provide short-term (< 4 years) benefits for conservation (Wood 1986, Peitz et al. 1999, Iglay et al. 2010 *a*). Disturbances can perpetuate biodiversity (Turner et al. 2001, White and Jentsch 2001, Rundel et al. 1998), and although frequent disturbances favor disturbance-dependent or –tolerant species, timber management incorporating biocomplexity objectives could compensate for disturbance-intolerant species by creating a mosaic of stand treatments, vegetative structures, species, and successional stages across a landscape (Hunter 1990, Franklin 1993*ab*, Heljden et al.

1998, Carey et al. 1999*ab*, Tilman 1999). Managed forests can provide ecosystem benefits such as wildlife habitat, protection of water quality, and carbon sequestration while offering a mosaic of successional stages across the landscape (Petraitis et al. 1989, Greenberg et al. 1994, McLeod and Gates 1998, Vogt et al. 1999). Therefore, it is essential to determine optimal management approaches of intensively managed, mid-rotation pine (*Pinus spp.*) stands to meet forestry and biodiversity objectives (Wear and Greis 2002, Stein et al. 2005). Addition of dormant season prescribed fire and selective herbicide application after commercial thinnings may enhance wildlife habitat and sustain improved biodiversity until harvest supporting concepts of sustainable forestry (Hartley 2002, Carnus et al. 2003, Sustainable Forestry Initiative 2005).

Prescribed fire and selective herbicides are 2 silviculture tools used to control midstory hardwood competition and improve wildlife habitat in mid-rotation, intensively managed pine stands of the southeastern United States (Brockway and Outcalt 2000, Edwards et al. 2004, McInnis et al. 2004). Prescribed fire is similar to historical disturbances of the southeast (Brennen et al. 1998). Following specific prescriptions, dormant season prescribed fires, applied during winter, minimize opportunity for detrimental effects on pine growth caused by crown scorch (Wade and Lunsford 1989, Bessie and Johnson 1995, Schimmel and Granstrom 1997). Selective herbicides, such as those containing imazapyr, offer an alternative to prescribed fire lacking smoke management issues or limited burning degree days (Wigley et al. 2002). Both treatments have demonstrated abilities to reduce woody plant coverage and consequently increase herbaceous understory plant coverage (Stransky and Harlow 1981, Brockway and Outcalt 2000, Miller and Miller 2004, Chapter 1). Past research has demonstrated treatment benefits to a myriad of game and non-game animals such as white-tailed deer (*Odocoileus virginianus*; Demarais et al. 2000, Mixon et al. 2009, Iglay et al. 2010*b*), eastern wild turkey (*Meleagris gallopavo silvestris*; Dickson and Wigley 2001, Miller and Conner 2007), northern bobwhite (*Colinus virginianus*; Guynn et al. 2004, Miller and Miller 2004, Welch et al. 2004), and songbirds (Sladek et al. 2008). However, research is lacking regarding herpetofauna response to independent and combined applications of dormant season prescribed fire and imazapyr.

Herpetofauna comprise the vast majority of vertebrate species in forest ecosystems (Burton and Likens 1975, Petranka and Murray 2001) and are globally declining (Vitt et al. 1990, Gibbons et al. 2000, Stuart et al. 2004). Prescribed fire and selective herbicides affect herpetofauna in various ways (Howard et al. 1959, Russell et al. 1999). Caged snakes can be affected by fire's heat if in excess of 62^{o}C (Howard et al. 1959), but sub ground-level temperatures during prescribed fires are usually $\leq$ 60^{o}C (Howard et al. 1959, Kahn 1960). Suffocation may be the most influential direct mortality cause (Chew et al. 1958, Lawrence 1966). Animal behavior, specifically an animal's ability to evade fire, and physiological characteristics such as permeable skin may cause greater mortality for amphibians than reptiles (Stebbins and Cohen 1995, Russell et al. 1999, Renken 2006). However, most past studies have not observed fire as an influential mortality factor of herpetile communities (Komerak 1963, Means and Campbell 1981, Russell et al. 1999). Synergistic toxicity of herbicide mixtures to American toads (*Bufo americanus*) is a concern for herbicide application versus fire (Howe et al. 1998), but Tatum's (2004) overview of herbicide toxicity suggests minimal probability of toxicity but does warn of unknown side-effects due to herbicide synergy in multiple herbicide tank mixtures or multiple available surfactants (Shepard et al. 2004).

Changes in vegetation structure post-burn or imazapyr application poses a greater influence to herpetile communities than direct mortality (Lillywhite 1977, Greenberg et al. 1994, Bamford et al. 1995, Ford et al. 1999, Russell et al. 1999). Treatment frequency and intensity ultimately shape herpetile communities as treatments directly influence vegetation structure and diversity altering habitat conditions (Means and Campbell 1981, Mushinsky 1985, DeMaynadier and Hunter 1995, Harpole and Haas 1999, Greenberg 2001, Pilliod et al. 2003, Russell et al. 2004). Invertebrate responses to treatments also may influence post-treatment herpetile communities (McComb and Hurst 1987). A mosaic of vegetation structure diversity and successional stages can support greater herpetofauna diversity with increased diversity of macro- and microhabitats (Greenberg et al. 1994, McLeod and Gaites 1998). However, combining fire and imazapyr may be too intensive for overall species management as many species of herpetofauna, specifically amphibians, rely on dense and moist understory structure that would be limited by reduced canopy coverage (i.e., less hardwood midstory) and leaf litter (DeMaynadier and Hunter 1995, Harpole and Haas 1999, Russell et al. 2004, Greenberg and Waldrop 2008).

More information is required to assess adequately herpetile response to prescribed fire and imazapyr in intensively managed, mid-rotation pine stands (Chen et al. 1977, Hood et al. 2002). Responses of other wildlife communities are better documented (e.g., Cole et al. 1995, Brockway and Outcalt 2000), but forest and wildlife management planning would benefit from understanding herpetile response to treatments for developing long-term sustainable forestry management approaches (Russell et al. 2002, Iglay et al. 2010*ab*). Therefore, I investigated herpetile community responses to prescribed fire and imazapyr in intensively managed, mid-rotation pine stands of east-central Mississippi over 9 years to better understand effects of these treatments in lieu of sustainable forestry efforts.

Study Area and Design

A full description of study area and design is provided in Chapter 1.

Methods

Vegetation Response and Climate Variables

I describe vegetation sampling methods in Chapter 1. I measured daily rainfall, temperature extremes, and percent cloud coverage at stand-level in summer 2001-2007 (no cloud coverage for all of 2003) using graduated rain gauges, thermometers, and ocular estimates of cloud coverage at 10% incremenets, respectively. I then calculated mean maximum and minimum daily temperatures, mean percent cloud coverage, mean precipitation, and number of rain days per treatment plot (n=4 plots/stand).

Herpetofauna Response

I trapped herpetiles using drift fence arrays May and June 1999-2007 and October 2000-2007 except October 2003. I created drift fence arrays with 4 5-gallon buckets arranged as one center bucket with 5-m arms of 35.6 cm high aluminum flashing at 120^{o}

angles from center bucket with a single bucket at arm ends. I buried each arm $\geq$ 5 cm, drilled holes in the bottom of each bucket for drainage, and ensured bucket lips were even with or slightly below ground level. From 2004-2007, I placed a piece of woody debris in each bucket for captured animals to use as cover or for flotation. Along each arm, I placed a funnel trap of wire mesh (Enge 1997) so that each quadrant had one funnel trap. I permanently placed 3 drift fences per treatment plot diagonally with each $\geq$ 50 m from plot edge. I trapped 3 stands simultaneously for 10-days twice monthly May and June and once in October and recorded species and trap location (bucket and treatment plot) for every capture. When not in use, I closed traps by removing funnel traps, closed buckets using lids, and placing a large stick in each bucket for animals to use to escape in case lids were detached between trapping sessions. From these data, I calculated mean species catch-per-unit effort (CPUE) from new captures and mean species richness by treatment plot for final analyses. I used CPUE as an index of relative abundance (Ludwig and Reynolds 1988) and therefore refer to relative abundance in results and discussion, not CPUE. I also grouped all toads in the American toad (*Bufo americanus*) complex including American toads, Fowler's toads (*Bufo fowlerii*), and southern toads (*Bufo terrestris*) as *Bufo* spp. due to similarities in field identification characteristics and hybridization among these species (Blair 1941, Cory and Manion 1955, Volpe 1956, Blair 1959, Meacham 1962, Blair 1963). I followed IACUC protocol #98-046 approved by Mississippi State University's IACUC for all herpetile trapping and handling.

Statistical Analysis

I tested the hypothesis of no difference in mean species-specific relative abundance, species richness, and total relative abundance among treatments within years for summer data using mixed models, repeated measures analysis of covariance in SAS (GLIMMIX procedure; SAS Institute Inc., Cary, North Carolina, USA. For fall data, I used mixed models, repeated measures analysis of variance in SAS to test the same hypothesis. For each model, I used 4 treatment levels (burn, herbicide, burn + herbicide, control), random effect of stand (n = 6), repeated measures of year (n = 8; 2000-2007), and subject of stand × treatment (Littell et al. 2006). I used pre-treatment (1999) herpetile relative abundance as a baseline covariate because pre-treatment herpetile communities may have differed among treatment plots treated alike regardless of overall treatment effect (Hood et al. 2002, Milliken and Johnson 2002). I limited species-specific analyses to species contributing $\geq$ 10% of total captures for amphibians or reptiles, but species richness and total relative abundances included all captured species, only counting species groups (e.g., *Bufo* spp.) as one species. For each model, I selected an appropriate covariance structure from the following: 7-banded Toeplitz, heterogeneous compound symmetry, heterogeneous auto-regressive, and auto-regressive. I designated the covariance structure that minimized Akaike's Information Criterion corrected for small sample size (AIC_c) as the top candidate for analysis (Littell et al. 2006, Gutzwiller and Riffell 2007). I used Kenward-Roger correction for denominator degrees of freedom for repeated measures and small sample sizes (Littell et al. 2006, Gutzwiller and Riffell 2007). I used LSMEANS SLICE option to identify a treatment effect within years following a significant interaction and LSMEANS PDIFF to conduct pair-wise comparisons (Littell et al. 2006). All year references in results refer to years post-treatment. My *a prior* significance level was α=0.05.

My first approach to investigating indirect effects of treatments on herpetiles (e.g., changes in vegetation and climate variables) was to use a series of steps including restricting explanatory variables to those biologically or ecologically associated with herpetiles according to past literature, visual inspection of scatter plots to reduce number of initial candidate explanatory variables, and use summed model weights per parameter and model-averaged parameter estimates to determine relative importance of each variable from all-subsets regression models (Burnham and Anderson 2002, Arnold 2010). Although my analysis approach may have excluded meaningful variables especially when additive or interactive relationships with other explanatory variables occurred, I felt it was a relatively unbiased approach to data reduction and modeling that considered all possible explanatory variables for each dependent variable. I used basal area of midstory and overstory trees (m^2/ha), midstory tree height (m), Nudds board visual coverage, downed woody debris, and line intercept coverage data as a set of possible explanatory variables because herpetile communities can be influenced by changes in vegetative structure and coverage (Melbourne 1999, Russell et al. 2002). However, upon visual inspection, I did not observe any noticeable patterns or relationships. Therefore, instead of taking an exploratory approach to modeling habitat associations, I decided to use significant results from vegetation structure and coverage analysis provided in Chapter 1 as guidelines for describing possible indirect effects of treatments on herpetile communities.

Results

I captured 4,734 individuals (933 reptiles and 3,801 amphibians) among 35 species and 17,388 trapnights (summer = 12,384 trapnights, fall = 5,004 trapnights; Table 4.1). Summer total CPUE was greater than fall CPUE (0.364 individuals/trapnight vs. 0.045 individuals/trapnight) and amphibian CPUE (0.298 individuals/trapnight) was greater than reptile CPUE (0.066 individuals/trapnight) in summer but similar in fall (0.024 reptiles/trapnight vs. 0.021 amphibians/trapnight). Summer amphibian captures were dominated by eastern narrowmouth toads (*Gastrophryne carolinensis*; 77.9%) and *Bufo spp.* (15.8%). Ground skink (*Scincella lateralis*; 48.0%) and fence lizard (*Sceloporus undulatus*; 22.9%) captures dominated summer reptile captures. Green anole (*Anolis carolinensis*; 37 %) and ground skink (36.1%) dominated fall reptile captures and 4 species dominated fall amphibian captures [eastern narrowmouth toad (34.3%), bronze frog (*Rana clamitans melanota*; 19.0%), *Bufo spp.* (17.1%), and marbled salamander (*Ambystoma opacum*; 11.4%)].

Eastern narrowmouth toad relative abundance and amphibian species richness differed among treatment plots pre-treatment (Table 4.2). Eastern fence lizard relative abundance differed among treatments year 1 with least abundance in controls during summer (Tables 4.2 and 4.3). *Bufo spp.* relative abundance differed among treatments within years in fall with greatest relative abundance in controls and herbicide only in years 2 and 5, respectively. Green anole (*Anolis carolinensis*) relative abundance differed among treatments across all years in fall with greater relative abundance in controls than burn treatments ($\bar{x}$ =0.004 vs. $\bar{x}$ =0.001 SE= 0.001; Table 4.2). Green anole relative abundance in herbicide only was intermediate to all other treatments ($\bar{x}$ =0.003 SE=0.001).

Vegetation structure responses resulted in a few patterns among burn treatments versus herbicide treatments, burn + herbicide versus control, and treatments versus no treatment (Chapter 1). Burning only resulted in less effective woody plant control, particularly midstory hardwoods and fluctuations in low level visual obstruction (0.0- 0.9 m) temporally dependent on year of or time since each burn. Burning only also increased species richness of understory plants. Burn + herbicide could be considered the greatest intensity treatment as evident by significant initial reductions in low level visual obstruction, reduced basal area of midstory hardwoods, and increased volume of downed woody debris of class 2 and basal area of hardwood snags. Understory forage class coverage of burn + herbicide plots typically consisted of herbaceous plants such as forbs, grasses, legumes, and semi-woody vines. Understory plant coverage of controls was typically dominated by woody species. Finally, all treatments achieved some level of midstory hardwood control and increase in understory plant species richness compared to controls.

Discussion

Short-term responses of species used in analyses limited my abilities to model species-specific responses among measured vegetation variables across all years of study. Overall, herpetile responses to dormant season prescribed fire and imazapyr in intensively managed, mid-rotation pine stands were limited to a few species (northern fence lizards, green anoles, and *Bufo spp.*). Changes in vegetation structure and diversity were observed on my study and most likely influenced herpetile communities more than direct impacts of treatments (Komerak 1963, Means and Campbell 1981, Russell et al. 1999). However, my comparisons of species responses to changes in vegetation community should be read with caution because they are not based on analysis results (i.e., regression models), only speculation and past studies.

Greater relative abundance of northern fence lizards in burned sites has been observed in past studies (Greenberg and Waldrop 2008). Greater standing and fallen woody debris in treated sites 1 year post-treatment and ample basking areas compared to control plots may have influenced use of these areas by northern fence lizards (Wilson 1995, Paraker 1994, Greenberg and Waldrop 2008). Greater overall relative abundance of green anoles in controls than burn treatments (burn only and burn + herbicide) coincides with an affinity for shrub perching sites (Schaefer et al. 2009). Greater hardwood midstory tree basal area of controls and herbicide only plots and increased visual obstruction at higher levels in herbicide only (0.9-1.8 m) may have offered ample perching sites for juvenile and adult green anoles (Schaefer et al. 2009). However, managing for closed canopy forests only, such as control plots provided, can limit herpetofauna diversity (Campbell and Christman 1982, Greenberg et al. 1994). *Bufo* spp. was the only group of amphibians with differences among treatments. Each year of significant differences was 1 year post-burn (summer after winter burn), suggesting an avoidance to burned stands the year after treatment. However, this behavior has not been observed in past studies which attributed ideal thermal conditions for toads in recently burned versus unburned sites (Kirkland et al. 1996, Hossak et al. 2009).

Drift fence proximity to aquatic areas [i.e., large beaver (*Castor candensis*) slough] and dense vegetation structure may have obscured observed treatment effects for amphibians (Greenberg 1993, Alford and Richards 1999, Greenberg 2002, Schurbon and Fauth 2003, Perry et al. 2009). Three treatment plots contributed ~30% total captures of *Bufo spp.* and eastern narrowmouth toads across all years in summer. Ancillary observations found ephemeral pools in or near each of these treatment plots, including a beaver slough created soon after project initiation and immediately adjacent to a drift fence array.

Many amphibian species, such as ground-dwelling woodland salamanders, did not seem to be affected by prescribed fire in this study. Observations from past studies are similar regarding minimal differences in salamander and other amphibian relative abundance among burned and unburned sites (Means and Campbell 1981, Greenberg et al. 1994, Ford et al. 1999, Mosley et al. 2003, Keyser et al. 2004**,** Ford et al. 2010). However, I did not capture many salamanders throughout my study (n=51). Treatment buffers and streamside-management zones (SMZ) within each of my stands could have offered refuge for these and other herpetile species during prescribed burns (Goldstein et al. 2005). Greater understory plant coverage in burn treatments also may have offered adequate canopy coverage for ground-dwelling amphibians such as woodland salamanders (Perry et al. 2009) again reducing differences in amphibian response among treatments.

Reptile response to treatments in my study also was limited. Past studies vary in their conclusions regarding reptile response to fire including positive, negative, and neutral responses (Means and Campbell 1981, Lunney et al. 1991, McLeod and Gates

1998, Spellerberg 1975, Greenberg and Waldrop 2008, Perry et al. 2009). Many reptiles have a greater ability to evade fire and are dependent less on ephemeral wetlands than amphibians supporting greater mobility (Komarek 1969, Means and Campbell 1981, Jackson and Milstrey 1989). Immediate increase of northern fence lizard relative abundance in treated plots attests to such mobility.

Lack of differences among treatments does not imply an absence of treatment effects on herpetile communities. Numerous past studies regarding prescribed fire have reported minimal effects on herpetofauna communities and sometimes benefits to herpetofauna diversity (Means and Campbell 1981, Campbell and Christman 1982, Stout et al. 1988, Greenberg et al. 1994, Ford et al. 1999, Greenberg and Waldrop 2008, Perry et al. 2009). Monitoring herpetiles in intensively managed forests is important to understand management impacts on local communities (Russell et al. 2002). Herpetiles are used commonly as indicator species because of their sensitivity to changes, especially amphibians (Welsch and Droege 2001). However, within a landscape of repeated, large-scale disturbances (e.g., stand rotations, thinnings, clearcutting, etc.), small scale disturbances such as treatments applied to 10-ha experimental units may not cause fluctuations in disturbance-tolerant populations such as many species of the southeastern U.S. (Means and Campbell 1981, Greenberg et al. 1994). Some species even depend on pyric disturbances [e.g., Gopher tortoise (*Gopherus polyphemus*); Auffenberg and Franz 1982, Ernst et al. 1994, Breininger et al. 1994].

Microhabitat characteristics and seasonal climate differences also may have influenced overall captures. Similarities in trap site characteristics such as downed woody structure, woody cover, plant coverage, and litter depth could have reduced impact of stand-level differences among treatments. Coarse woody debris volume (CWD) had minimal, short-term differences among treatments (Chapter 1). High volumes of CWD are uncommon in pine forests of the southeastern United States especially post-burn but also due to high decomposition rates (Moorman et al. 1999). Researchers implementing manipulative studies of CWD volume have observed few differences in herpetile responses even though CWD contributes a structural necessity for many reptiles and amphibians (deMaynadier and Hunter 1995, Owens et al. 2008). However, adequate litter depth may provide sufficient cover (Moseley et al. 2004, Greenberg and Tanner 2005, Rothermel and Luhring 2005, Baughman and Todd 2007) and reduced litter depth post-burn may have been compensated by increased needle fall (Niwa and Peck 2002).

Weather variables sampled during trapping periods (e.g., daily temperature extremes, cloud coverage, precipitation) did not have any apparent relationship to number of herpetile captures. However, monthly precipitation measurements among years revealed some interesting patterns of amphibian relative abundances. During wet years (2001, 2003, and 2004), *Bufo spp.* relative abundance was nearly 4 times greater than remaining drier years ($\bar{x}$ = 0.54 individuals/trapnight vs. $\bar{x}$ = 0.14 individuals/trapnight). Eastern narrowmouth toad relative abundance was nearly 5 times greater in 2001 than remaining years ($\bar{x}$ = 4.7 individuals/trapnight vs. $\bar{x}$ = 0.989 individuals/trapnight). Additional ephemeral pools may have been available during wet years offering ample breeding sites for amphibians. These sharp fluctuations underline importance of using long-term studies to better understand faunal response to treatments as suggested by Greenberg and Tanner (2008) in reference to ephemeral pools.

The few species dominating overall captures were most likely efficiently sampled by pitfall trapping. Remaining species had few captures across the entire 9-year sampling period (12% and 30% of summer and fall captures, respectively). As part of a large-scale, multi-faunal, replicated field experiment, time and effort constraints limited sampling intensity for herpetofauna and other faunal communities. Pitfall trapping alone has been suggested as insufficient for monitoring community responses to treatments especially when considering rare species (Corn and Bury 1991, Cole et al. 1997). Funnel traps attached to drift fences were used because pitfall traps alone inadequately sample snakes (Greenberg et al. 1994). However, I implemented other sampling methods (e.g., cover boards, PVC pipes, anuran call counts) but had limited and variable success. Time constrained area searches and dip net samplings for amphibian larva were other methods available. However, efficacy of searches may have differed among vegetation types, and this study's design did not incorporate sampling of ephemeral pools. Although pitfall traps may limit the types of species sampled, it is still a main sampling tool used for herpetofauna (e.g., Greenberg and Waldrop 2008, Perry et al. 2009). If multiple characteristics of greater influence than treatments occurred across the landscape as suggested above, detecting herpetofauna response to prescribed fire and imazapyr would be just as challenging at greater sampling intensities.

Management Implications

Prescribed fire and imazapyr applied at mid-rotation seem to have minimal effects on herpetile communities. Similar to past studies, herpetile responses tend to reflect greater influences than small-scale treatment applications emphasizing management beyond stand-level to landscape-level (Buskirk 2005, Werner et al. 2007). Specifically, forest managers should consider vegetation structure and diversity among mid-rotation stands and protection of sensitive areas such as ephemeral wetlands and SMZs (Gibbons 2003, Goldstein et al. 2005, Perry et al. 2009). The 4 treatments I used can create a mosaic of vegetation structure under a pine canopy assisting forest managers in meeting these goals.

Herpetile response to treatments still remains poorly understood (Pilliod et al. 2003). Long-term research data is required to better understand herpetile community responses to treatments (Greenberg and Tanner 2008) but could be unattainable when focused on a specific period of stand rotation in intensively managed forests (e.g., mid-late rotation). With the potential for landscape-level characteristics to have greater impacts on herpetile communities than local perturbations, researchers are challenged with designing innovative studies to better understand interactions of treatments and herpetile populations over larger spatial scales and longer study periods. Therefore, researchers should consider studies across entire rotations with stands as experimental units, not smaller scale experimental units (e.g., 10-ha units used here). Stratifying sampling among stand edges, riparian and ephemeral wetland areas, and in-stand locations should help researchers better understand treatment differences as opposed to landscape characteristic influences.

Prescribed fire and imazapyr may be safe management tools for mid-rotation release in a sustainable forestry setting as they most likely have minimal direct effects on herpetofauna and can improve sustainable forestry objectives of improved biodiversity (Chapters 1 and 2, Iglay et al. 2010*a*). As fire suppression continues to persist, it is essential for managers to implement prescribed fire wherever plausible and for researchers to continue providing information necessary to better understand this management tool (Means and Moler 1979, Sharitz et al. 1992, Brennan et al. 1998). The succession of many forests to closed canopy is a cause for concern for many herpetofauna species of the southeast reliant on open understories of past (Means and Moler 1979, Means and Campbell 1981, Fellers and Drost 1993, Nowacki and Abrams 2008). Prescribed fire and imazapyr can help alleviate this concern by reducing hardwood midstory competition and perpetuating open forest landscapes within an intensively managed pine matrix.

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Table 4.1 Herpetofauna seen or sampled using drift fence arrays with funnel traps in intensively managed, mid-rotation pine stands treated with prescribed fire and imazapyr in Kemper County, Mississippi, 1999-2007.

Common Name	Scientific Name	Sampled
American toad	*Bufo americanus*	X
Central newt	*Notophthalmus viridescens louisianensis*	X
Corn snake	*Elaphe guttata*	
Cottonmouth	*Agkistrodon piscivorus*	X
Eastern diamondback rattlesnake	*Crotalus adamanteus*	
Eastern garter snake	*Thamnophis sirtalis sirtalis*	
Eastern hognose snake	*Heterodon platirhinos*	
Eastern mud snake	*Farancia abacura*	X
Eastern narrow-mouthed frog	*Gastrophryne carolinensis*	X
Five-lined skink	*Eumeces fasciatus*	X
Fowler’s toad	*Bufo woodhousii fowleri*	X
Gray treefrog	*Hyla versicolor/chrysocelis*	X
Green anole	*Anolis carolinensis*	X
Green treefrog	*Hyla cinerea*	X
Ground skink	*Scincella lateralis*	X
Marbled salamander	*Ambystoma opacum*	X
Midland brown snake	*Storeria dekayi wrightorum*	X
Mississippi ringneck snake	*Diadophis punctatus stictogenys*	X
Mississippi slimy salamander	*Plethodon Mississippi*	X
Northern fence lizard	*Sceloporus undulatus hyacinthinus*	X
Pickerel frog	*Rana palustris*	
Red-eared slider	*Trachemys scripta elegans*	
Rough green snake	*Opheodrys aestivus*	
Smallmouth salamander	*Ambystoma texanum*	X
Snapping turtle	*Chelydra serpentina*	
Southeastern five-lined skink	*Eumeces inexpectatus*	X
Southern black racer	*Coluber constrictor priapus*	X
Southern copperhead	*Agkistrodon contortrix contortrix*	X
Southern cricket frog	*Acris gryllus gryllus*	X
Southern leopard frog	*Rana sphenocephala utricularius*	X
Speckled kingsnake	*Lampropeltis getula holbrooki*	X
Spring peeper	*Pseudacris crucifer*	X
Timber rattlesnake	*Crotalus horridus*	X
Upland chorus frog	*Pseudacris triseriata feriarum*	X
Western pigmy rattlesnake	*Sistrurus miliarius streckeri*	

Table 4.2 Interaction and main effects of covariate (baseline data from pre-treatment year), treatment (burn only, herbicide only, burn + herbicide, and control), and treatment × year on herpetiles sampled with drift fences (summer and fall) in intensively managed pine stands in Kemper County, Mississippi, 1999-2007.

	Pre-treatment		Treatment		Year		Treatment × year	
Season and Variable	*F*	*P*-value	*F*	*P*-value	*F*	*P*-value	*F*	*P*-Value
Summer								
Bufo spp.	0.56	0.458	2.18	0.098	4.23	≤ 0.001	0.49	0.970
Gastrophryne carolinensis	8.24	0.006	0.54	0.655	17.00	≤ 0.001	0.73	0.799
Sceloporus undulatus	0.00	0.995	7.34	≤ 0.001	18.92	≤ 0.001	2.72	≤ 0.001
Scincella lateralis	0.13	0.781	2.22	0.095	5.46	≤ 0.001	0.64	0.888
Species richness								
Amphibian	4.20	0.050	0.21	0.888	26.82	≤ 0.001	1.33	0.173
Reptilian	0.44	0.512	0.62	0.607	7.69	≤ 0.001	1.25	0.219
Overall	2.50	0.128	0.58	0.639	5.74	≤ 0.001	1.39	0.144
Total CPUE								
Amphibian	2.16	0.152	0.65	0.589	11.97	≤ 0.001	1.08	0.383
Reptilian	0.06	0.843	0.72	0.545	10.36	≤ 0.001	1.05	0.418
Overall	0.10	0.754	0.21	0.890	7.19	≤ 0.001	1.34	0.176
Fall								
Ambystoma opacum	-	-	0.33	0.802	2.86	0.012	0.45	0.973
Anolis carolinensis	-	-	2.81	0.047	2.05	0.063	1.31	0.190
Bufo spp.	-	-	2.14	0.102	4.06	≤ 0.001	3.22	≤ 0.001
Gastrophryne carolinensis	-	-	0.23	0.878	8.89	≤ 0.001	0.69	0.816
Rana clamitans melanota	-	-	0.48	0.698	5.51	≤ 0.001	0.82	0.673
Scincella lateralis	-	-	0.59	0.625	4.56	≤ 0.001	1.51	0.097
Species richness								
Amphibian	-	-	0.53	0.661	15.10	≤ 0.001	1.59	0.070
Reptilian	-	-	1.31	0.281	2.04	0.073	0.90	0.582
Overall	-	-	1.55	0.214	5.33	≤ 0.001	1.09	0.379
Total CPUE								
Amphibian	-	-	1.91	0.135	14.32	≤ 0.001	1.53	0.088
Reptilian	-	-	1.96	0.146	2.93	0.013	0.97	0.500
Overall	-	-	2.73	0.058	4.16	≤ 0.001	1.27	0.225

[1]*P*-value before rounding was 0.0496.

Table 4.3 Least square mean estimates (SE) of herpetofauna relative abundance (catch-per-unit-effort) among treatments within years in intensively managed pine stands of Kemper County, Mississippi, 2000-2007.

Season			Burn	Herbicide	Burn + herbicide	Control	
Species	Year	*P*-value	$\overline{x}$	$\overline{x}$	$\overline{x}$	$\overline{x}$	SE
Summer							
Sceloporus undulatu	1[1]	0.002	0.013 A	0.012 A	0.020 A	0.002 B	0.003
	2	0.367	0.003	0.006	0.009	0.004	0.002
	3	0.536	0.004	0.001	0.006	0.006	0.002
	4	0.404	0.002	0.001	0.006	0.001	0.001
	5	0.270	0.001	0.001	0.003	0.000	0.001
	6	0.069	0.002	0.003	0.001	0.001	0.001
	7	0.325	0.008	0.000	0.003	0.000	0.002
	8	1.000	0.000	0.000	0.000	0.000	0.000
Fall							
Bufo spp.	1	1.000	0.000	0.000	0.000	0.000	0.001
	2	≤ 0.001	0.002 B	0.000 B	0.000 B	0.009 A	0.001
	3	0.877	0.000	0.000	0.001	0.000	0.001
	5	≤ 0.001	0.001 B	0.010 A	0.000 B	0.001 B	0.001
	6	0.843	0.000	0.001	0.000	0.000	0.001
	7	1.000	0.000	0.000	0.000	0.000	0.001
	8	1.000	0.000	0.000	0.000	0.000	0.001

[1] Treatments with the same letter do not differ significantly ($P > 0.05$).

CHAPTER 5

CARABID RESPONSE TO FIRE AND IMAZAPYR APPLICATION IN INTENSIVELY MANAGED, MID-ROTATION PINE STANDS OF MISSISSIPPI

Intensively managed pine (*Pinus spp.*) forests cover 18 million ha in the southern United States, about 20% of southern forests, with 2.2 million ha in Mississippi (USDA Forest Service 2007). Management for economic gain and biodiversity objectives is achievable (e.g., Miller et al. 2009, Wigley et al. 2000), but current practices at mid-rotation may not achieve adequately long-term biodiversity and sustainable forestry objectives. Forest managers are increasingly expected to incorporate management efforts contributing to the conservation of biodiversity (Sustainable Forestry Initiative 2005), but current management practices at mid-rotation such as commercial thinning and fertilization may only provide short-term (< 4 years) benefits for conservation (Wood 1986, Peitz et al. 1999, Iglay et al. 2010*a*). Disturbances can perpetuate biodiversity (Turner et al. 2001, White and Jentsch 2001, Rundel et al. 1998), and although frequent disturbances favor disturbance-dependent or –tolerant species, timber management incorporating biocomplexity objectives could compensate for disturbance-intolerant species by creating a mosaic of stand treatments, vegetative structures, species, and successional stages across a landscape (Hunter 1990, Franklin 1993*ab*, Heljden et al.

1998, Carey et al. 1999*ab*, Tilman 1999). Managed forests can provide ecosystem benefits such as wildlife habitat, protection of water quality, and carbon sequestration while offering a mosaic of successional stages across the landscape (Petraitis et al. 1989, Greenberg et al. 1994, McLeod and Gates 1998, Vogt et al. 1999). Therefore, it is essential to determine optimal management approaches of intensively managed, mid-rotation pine (*Pinus spp.*) stands to meet forestry and biodiversity objectives (Wear and Greis 2002, Stein et al. 2005). Addition of dormant season prescribed fire and selective herbicide application after commercial thinnings may enhance wildlife habitat and sustain improved biodiversity until harvest supporting concepts of sustainable forestry (Hartley 2002, Carnus et al. 2003, Sustainable Forestry Initiative 2005).

Prescribed fire and selective herbicides are 2 silviculture tools used to control midstory hardwood competition and improve wildlife habitat in mid-rotation, intensively managed pine stands of the southeastern United States (Brockway and Outcalt 2000, Edwards et al. 2004, McInnis et al. 2004). Prescribed fire is similar to historical disturbances of the southeast (Brennen et al. 1998). Following specific prescriptions, dormant season prescribed fires, applied during winter, avoid detrimental effects on pine growth caused by crown scorch (Wade and Lunsford 1989, Bessie and Johnson1995, Schimmel and Granstrom 1997). Selective herbicides, such as those containing imazapyr, offer an alternative to prescribed fire lacking smoke management issues or limited burning degree days (Wigley et al. 2002). Both treatments have demonstrated abilities to reduce woody plant coverage and consequently increase herbaceous understory plant coverage (Stransky and Harlow 1981, Brockway and Outcalt 2000, Miller and Miller 2004, Chapter 1). Past research has demonstrated treatment benefits to a myriad of game and non-game animals such as white-tailed deer (*Odocoileus virginianus*; Demarais et al. 2000, Mixon et al. 2009, Iglay et al. 2010*b*), eastern wild turkey (*Meleagris gallopavo silvestris*; Dickson and Wigley 2001, Miller and Conner 2007), northern bobwhite (*Colinus virginianus*; Guynn et al. 2004, Miller and Miller 2004, Welch et al. 2004), and songbirds (Sladek et al. 2008). However, research is lacking on responses of many invertebrates to prescribed fire and selective herbicide, especially carabid beetles (Coleoptera: Carabidae).

Carabids are well-suited for diversity studies because they are well-known taxonomically in the Northern Hemisphere (e.g., Lindroth 1961-69, Hutcheson and Jones 1999, Niemelä et al. 2000), fulfill a variety of functional roles (fungivores, predators, herbivores; Hutcheson and Jones 1999), and are relatively easy to sample (Hutcheson and Jones 1999, Werner and Raffa 2000). Their sensitivity to habitat changes makes them useful for indicating forest management impacts (e.g., Warriner et al. 2002, Pearce et al. 2003, Vance and Nol 2003). Decreases in carabid abundance have been observed post-burn (French and Keirle 1969, Richardson and Holliday 1982), but assemblages can reestablish (Holliday 1991*a, b*). Ability of carabid beetles to disperse quickly also may reduce direct mortality by fire (Pippin and Nichols 1996, Paquin and Coderre 1997). For herbicides, direct toxicity is likely not an issue for carabids (see Tatum 2004). Therefore, microhabitat changes and vegetation structure and biomass heterogeneity may be the most influential factors influencing carabid community relative to prescribed fire and herbicide use (Epstein and Kuhlman 1990, Niemelä et al. 1992, Niemelä et al. 1996, Rykken et al. 1997). Changes in vegetation types across a landscape from treatment applications could enhance carabid diversity by providing a diversity of vegetation structure (Lenski 1982, Parry and Rodger 1986, Niemelä et al. 1988, Baguette and Gerard 1993, Buse and Good 1993, Halme and Niemelä 1993, Niemelä et al. 1993. Beaudry et al.1997).

Past research regarding carabid responses to fire was predominantly in boreal forests of the Pacific Northwest (e.g., McCullough et al. 1998, Gandhi et al. 2001, Niwa and Peck 2002) with recent studies emerging in the southeastern United States (e.g., Greenberg and Thomas 1995, Hanula and Wade 2003, Iglay 2007, Hanula et al. 2009, Ulyshen et al. 2010). Few studies have investigated effects of forest herbicide application on arthropods with most limited to site preparation applications (Salminen et al. 1996, Bell et al. 1997, Duchesne et al. 1999). However, to my knowledge, information regarding carabid community responses to fire with or without herbicide is limited to a couple years of this project (2004-2005; Iglay 2007) and post-fire salvage logging in boreal forest (Cobb et al. 2007). Because carabid beetle communities can help elucidate forest management impacts (Villa-Castillo and Wagner 2002, Rainio and Niemelä 2003), I investigated carabid community response to combined and independent applications of prescribed fire and imazapyr herbicide in intensively managed, mid-rotation pine stands of Mississippi. My objectives were to determine carabid beetle response to prescribed burning and imazapyr herbicide application and model associations of carabid beetles with surrounding vegetation characteristics

Study Area and Design

A full description of study area and design is provided in Chapter 1.

Methods

Vegetation Response

Vegetation sampling methods are described in Chapter 1.

Carabid Response

I sampled carabid diversity using 2 randomly located, 80 m transects established in 1999 $\geq$ 50 m from plot edge and each other. I used transects for trapping all invertebrates of the soil surface prior to sampling for and identifying carabid species. I placed traps, a 0.47L container with 3 sheet metal barriers at 120^{o} angles from center container, every 20 m along each transect per treatment plot (n = 8). I filled containers with equal amounts of propelyne glycol and 70% ethanol and placed the lip of each container at or slightly below ground level (Morrill 1975, Southwood 1966, Housewart et al. 1979, Murkin et al. 1994, Ausden 1996). I sampled each treatment plot once monthly for one week of continuous trapping May-October 2004-2007. I sorted, pinned and labeled specimens with trap locality data to serve as voucher specimens in the Mississippi Entomological Museum. I identified all specimens to species using taxonomic keys and confirmation for 2004 and 2005 carabids from Drew Hildebrand, research associate of the Mississippi Entomological Museum. I calculated catch-per-unit-effort (CPUE; species abundance/number of active traps/treatment plot) for each species, mean species richness, and total relative abundance. I used CPUE as an index of relative abundance (Ludwig and Reynolds 1988) and therefore refer to relative abundance in results and discussion, not CPUE. I limited species-specific analysis to species with $\geq$ 40 observations.

Statistical Analysis

I tested the hypothesis of no difference in mean relative abundance, carabid species richness, and relative abundance among treatments within years using mixed models, repeated measures analysis of variance in SAS (Mixed procedure; SAS Institute Inc., Cary, North Carolina, USA) to examine main effects of treatment, year, and treatment × year on species-specific relative abundance, species richness, and total relative abundance. For each model, I used 4 treatment levels (burn, herbicide, burn + herbicide, control), random effect of stand (n=6), repeated measures of year (n =4; 2004-2007), and subject of stand × treatment (Littell et al. 2006). For each model, I selected an appropriate covariance structure from the following: 3-banded Toeplitz, heterogeneous compound symmetry, heterogeneous auto-regressive, and auto-regressive. I designated the covariance structure that minimized Akaike's Information Criterion corrected for small sample size (AIC_c) as the top candidate for analysis (Littell et al. 2006, Gutzwiller and Riffell 2007). I checked residuals and transformed data when deemed necessary to meet normality assumptions. I used Kenward-Roger correction for denominator degrees of freedom for repeated measures and small sample sizes (Littell et al. 2006, Gutzwiller and Riffell 2007). I used LSMEANS SLICE option to identify a treatment effect within years following a significant interaction and LSMEANS PDIFF to conduct pair-wise comparisons (Littell et al. 2006). All year references in results refer to years post-treatment. My *a prior* significance level was α=0.05.

For vegetation associations of carabid beetles, I first reduced my overall variable set, ran all-subsets regression models, and finally determined relative variable importance using model-estimated parameters and associated summed weights. Because variability in carabid beetle assemblages can be a function of vegetation structure, species richness, and volume of coarse woody debris (Cobb et al. 2007), I initially limited explanatory variables to these variables with understory species richness derived from plant biomass sampling. I avoided examining all combinations of explanatory variables by looking for relationships among response variables (e.g., species-specific relative abundance) and explanatory variables by visually inspecting scatter plots. Although this approach may have excluded meaningful variables especially when additive or interactive relationships with other explanatory variables occurred, I felt it was a relatively unbiased approach to data reduction that considered all possible explanatory variables for each dependent variable. I always chose summary variables over independent measurements when both were chosen for a model (e.g., Nudds board high instead of Nudds board level four, five, or six) and removed response variables without any noticeable relationships to explanatory variables from model comparisons. To avoid multicolinearity, I conducted pairwise comparisons of correlation coefficients among selected explanatory variables and, when highly correlated ($r \geq |0.5|$), removed the vegetation variable correlated more weakly to the response variable.

I used the MIXED procedure in SAS to evaluate regression models with stand (block) as the random effect, year as a repeated measure, and treatment plot as the subject (Littell et al. 2006). I also included treatment, year, and treatment × year as covariates. Because my analysis goal was to determine relative variable importance, and I conducted equal number of models per parameter, I used summed model weights per parameter and model-averaged parameter estimates to determine overall impacts of each explanatory variable on carabid species relative abundance (Burnham and Anderson 2002, Arnold 2010).

Results

I captured 1,900 carabid beetles of 41 species with 7 species used for analysis (Table 5.1). *Cyclotrachelus convivus* (55%) and *Brachinus alternans* (15%) comprised most captures followed by *Cyclotrachelus brevoori* (6%), *Galerita bicolor* (6%), *Semiardistomis puncticollis* (5%). *Calathus opaculus* (3%) and *Abacidus atratus* (2%). Relative abundances of *Cyclotrachelus convivus*, *Galerita bicolor*, *Abacidus atratus*, and species richness differed among treatment across all years (Table 5.2). *Cyclotrachelus convivus* relative abundance was greatest in controls and least in treated plots ($\bar{x}$=3.07 individuals/trapnight vs. $\bar{x}$=1.34 individuals/trapnight, SE=0.393). *Galerita bicolor* relative abundance was greater in controls than burn treatments ($\bar{x}$=0.312 individuals/trapnight vs. $\bar{x}$=0.050 individuals/trapnight, SE=0.058) with relative abundance in herbicide only ($\bar{x}$=0.201 individuals/trapnight, SE=0.058) intermediate to controls and burn only. *Abacidus atratus* relative abundance was least in herbicide treatments ($\bar{x}$=0.007 individuals/trapnight vs. $\bar{x}$=0.121 individuals/trapnight, SE = 0.030). Species richness was greatest in controls ($\bar{x}$=8.50 species vs. $\bar{x}$=6.25 species, SE=0.639) and greater in herbicide only than burn + herbicide plots ($\bar{x}$=6.75 species vs. $\bar{x}$=5.42 species, SE=0.639) with burn only intermediate ($\bar{x}$=6.57 species, SE=0.639).

I used 6 of 41 explanatory variables to explain indirect effects of treatments on 2 carabid species, species richness, and total relative abundance. *Cyclotrachelus convivus* was associated positively with greater mean hardwood midstory basal area ($\bar{x}$ =0.227, Σw_i =0.9) and canopy coverage ($\bar{x}$ =0.005, Σw_i =0.4). *Galerita bicolor* was associated negatively with increased upper level visual obstruction (0.9-1.8 m; $\bar{x}$ = -0.0001, Σw_i = 1.0). Carabid species richness decreased with greater pine snag basal area ($\bar{x}$ = -0.270, Σw_i =0.5) and greater species richness of understory plants ($\bar{x}$ = -0.032, Σw_i =0.6). Total relative abundance decreased with greater biomass of semi-woody vines ($\bar{x}$ =-0.001, Σw_i =1.0).

Discussion

Carabid responses to prescribed fire and imazapyr herbicide were species-specific but limited to a few dominant species. Most interactions with treatments were negative, especially with greater treatment intensity (i.e., burn + herbicide vs. burn only or herbicide only). Carabids had a distinct aversion to prescribed burning similar to past studies (Richardson and Holliday 1982, Hanula and Wade 2003, Niwa and Peck 2002, Saint-Germain et al. 2005). Vegetation associations also emphasized negative impacts of prescribed fire with vegetation characteristics of controls preferred over characteristics of treated sites. Instead of providing a better picture of indirect responses to treatments, vegetation associations may have indicated direct impacts on carabids (Niwa and Peck 2002). However, repeated burns are necessary to maintain habitat management goals of reduced hardwood midstory basal area and increased coverage of herbaceous understory plants (Chapter 1, Waldrop et al. 1987, 1992, Greenberg and Thomas 1995, Iglay et al. 2010*a*).

Reduced hardwood midstory basal area, canopy coverage, and increased semi-woody vine coverage were vegetation structure characteristics of sites treated with prescribed fire with or without imazapyr (Chapter 1). Although prescribed burning can have negative effects on carabid communities, areas of refuge provided by unburned patches, treatment buffers, and ground litter may have protected source populations (Gandhi et al. 2001, Saint-Germain et al. 2005). If so, carabid species recolonizing burned areas could be common species of mid-rotation pine stands under typical forest management (e.g., commercial thinning and fertilizing; Koivula and Niemelä 2002). Greenberg and Thomas (1995) observed a similar pattern with an absence of forest specialists in sand pine scrub (*Pinus c. calusa*) subject to low-frequency, high intensity wildfire. However, Wikars and Schimmel (2001) observed immediate (< 24 h) colonization of burned soil by pyrophilous species. If quick recovery occurred on my sites and sampling effort was adequate for detecting true differences, then observed carabid communities may reflect typical assemblages of intensively managed forest landscapes regardless of mid-rotation management approaches (Holliday 1991*ab*, Spence and Niemelä 1994).

Dominant species used for analyses were present in most treatment plots throughout the study. *Cyclotrachelus convivus*, *Cyclotrachelus brevoori*, and *Galerita bicolor* were sampled all years across all treatments, but *Abacidus atratus* and *Semiardistomis puncticollis* were observed rarely in all treatments each year. *Semiardistomis puncticollis* was absent from samples during 2006 and 2007. Pitfall trapping does have some inherent bias towards species sampled and influence of surrounding vegetation structure (Greenslade 1964, Greendberg and Thomas 1995). Active, highly mobile fauna on forest floors such as many carabid species have greater probabilities of capture than monophagous herbivores, xylophages, and mature forest-specialists (Greenslade 1964, Baars 1979, Greenberg and Thomas 1995). Vegetation structure immediately surrounding traps also can influence catch (Greenslade 1964, Refseth 1980). Despite these influences, pitfall trapping is one of the most effective sampling methods for ground-dwelling Coleoptera even when comparing among different habitat management techniques and vegetation types (Baguette and Gerard 1993, Buse and Good 1993, Niemelä et al. 1993, Spence and Niemelä 1994). Differences in vegetative structure among treatment plots may have altered macrohabitat, but microhabitat characteristics of trap sites may have been similar and drastic measures of removing all vegetation surrounding pitfall traps would have biased my sampling approach (Greenslade 1964).

Habitat characteristics of greater influence may exist beyond vegetation structure and biomass measured in this study. Volume of coarse woody debris (CWD) can increase carabid species abundance, richness, and diversity (Hanula et al. 2006, Latty et al. 2006, Nittérus and Gunnarsson 2006, Ulyshen and Hanula 2009), but I had few differences in CWD among treatments (Chapter 1). Microhabitat characteristics including ground surface temperature, soil moisture, and litter characteristics (e.g., depth, composition, moisture) also can influence carabid communities but could have been similar among treatment plots (Niemelä et al. 1988, Rushton et al. 1991, Niemelä et al. 1992, Holmes et al. 1993). Within treated sites, additional understory herbaceous plant coverage may have offered adequate shade similar to shade provided by hardwood midstory trees in controls and offered additional litter diversity for prey species (Chapter 1, Bultman and Uetz 1984). Increased pine needle debris post-burn also may have compensated for litter removed by fire always ensuring adequate litter depth for carabids (Niwa and Peck 2002). However, limited information from past studies suggested no significant differences in species richness or diversity among various litter depths (Koivula et al. 1999). Therefore, microhabitat differences would need to be extreme to elicit a significant response.

Although it has been argued that combined disturbance (wildfire and forestry practices) may be too intense for sustaining carabid diversity (Niemelä et al. 1993, Cobb et al. 2007) and my results support greater species richness in controls, overall carabid species diversity may benefit from vegetation structure and biomass diversity created by prescribed fire with or without imazapyr. Although fire had a significant negative effect on carabid relative abundances, 19 and 21 species were observed in burn only and burn + herbicide treatment plots across all years, respectively; compared to 25 and 26 species in controls and herbicide only plots, respectively. Four species were exclusive to burn only, 6 species each for controls and herbicide only, and one species for burn + herbicide. Cobb et al. (2007) concluded that combining fire and forestry-related disturbance (i.e., salvage and herbicide) may simplify and even homogenize ground beetle assemblages by reducing compositional variation among groups especially when fire is applied at the landscape level. However, my results show a greater contribution to carabid diversity in intensively managed, mid-rotation pine stands when prescribed fire and imazapyr are applied compared to typical mid-rotation management practices of thinning and fertilizing. Time-lags for species recovery also could diminish observed species diversity response to treatments because of short intervals for carabid reestablishment (< 3 years; French and Kerile 1969, Richardson and Holliday 1982, Holliday 1991*ab*). Therefore, future research should entail long-term studies of carabid response to wildlife management tools and assess contributions to biodiversity at local (e.g., site) and regional (e.g., landscape) scales.

Management Implications

Carabid beetles had limited responses to prescribed fire and imazapyr application in intensively managed, mid-rotation pine stands of Mississippi. Prescribed fire reduced a few dominant species but is essential for perpetuating vegetation structure and biomass heterogeneity among mid-rotation pine stands that is beneficial to many vertebrate species. Carabids may benefit most from a stratified burning approach offering variations in burn frequencies, timing (season), and fire intensities (Howe 1994). However, dormant season prescribed burns are the safest approach to avoid crown scorch in intensively manage pine (Wade and Lunsford 1989).

Although it can be argued that old-growth forests support greater biodiversity than plantation forests due to presence of diverse tree species, uneven aged management, and vertical structural heterogeneity (Lindenmayer and Hobbs 2004, Humphrey 2005, Ohsawa 2005), plantation forests play a pivotal role in maintaining and contributing to biodiversity, especially in southeastern United States (Lindenmayer and Hobbs 2004, Nelson and Halpern 2005, Miller et al. 2009). Therefore, I emphasize that although assemblages constructed from samples were “simple”, they only pertain to alpha diversity. Beta diversity including assemblages in treatment plots, intensively managed forests of all ages, and buffer zones (e.g., streamside-management zones, other wetland areas, firelanes, etc.) could be far greater and demonstrate the true conservation potential of managed forests relative to carabid beetles. Managers implementing a variety of forest management regimes will most likely help conserve biodiversity and promote carabid diversity at a regional scale as demonstrated by treatment-specific species assemblages in this study (Niemelä et al. 1987, Solheim et al. 1987).

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Table 5.1 Carabid species sampled using pitfall traps in intensively managed, mid-rotation pine stands treated with prescribed fire and imazapyr in Kemper County, Mississippi, 2004-2007.

Species	Analysis[1]
Abacidus atratus	X
Agonum punctiforme	
Anisodactylus opaculus	
Apenes sinuatus	
Aspidog lossa	
Badister notatus Haldeman	
Brachinus alternans	X
Calathus opaculus	X
Chlaenius amoenus	
Chlaenius erythropus	
Chlaenius laticollis	
Chlaenius nemoralis	
Clivina bispustulata	
Clivina postica	
Clivina rubicunda (LeConte)	
Cyclotrachelus brevoori	X
Cyclotrachelus convivus	X
Dichaelus ambiguus	
Dichaelus furvus	
Diplochaetus	
Galerita bicolor	X
Harpalus fulgens	
Harpalus pennsylvanicus/texanus	
Helluomorphoides nigripennis	
Lebia ornata	
Loxandrus agilis (DeJean)	
Loxandrus vitiosus,duryi group	
Myas coracinus	
Oodes amaroides	
Olisthopus micans	
Oodes americanus	
Panageus fasciatus	
Piesmus submarginatum	
Pseudaptinus lecontei (Dejean)	
Pterostichus punctiventris	
Scaphinotus e. elevantus	
Scarites subterraneus	
Semiardistomis puncticollis	X
Semiardistomis viridus	
Stenomorphus californicus	
Trichotichnus fulgens	

[1] Species used for analysis based on $\geq$ 40 observations.

Table 5.2 Interaction and main effects of treatment (burn only, herbicide only, burn + herbicide, and control) and treatment × year on carabids sampled with pitfall traps in intensively managed pine stands in Kemper County, Mississippi, summer 2004-2007.

	Treatment		Year		Treatment × year	
Species	*F*	*P*-value	*F*	*P*-	*F*	*P*-value
Abacidus atratus	5.64	0.006	1.07	0.373	1.22	0.303
Brachinus alternans	0.82	0.494	1.80	0.165	1.40	0.213
Calathus opaculus	0.79	0.507	3.49	0.020	0.73	0.684
Cyclotrachelus brevoori	2.54	0.085	5.78	0.002	0.93	0.507
Cyclotrachelus convivus	4.85	0.008	14.14	$\leq$ 0.001	1.29	0.265
Galerita bicolor	4.87	0.007	8.27	$\leq$ 0.001	1.10	0.380
Semiardistomis	0.26	0.850	13.58	$\leq$ 0.001	0.13	0.999
Species Richness	10.24	$\leq$ 0.001	48.12	$\leq$ 0.001	1.74	0.108
Total Relative	2.25	0.102	21.42	$\leq$ 0.001	1.56	0.154

CHAPTER 6

BIODIVERSITY RESPONSE TO PRESCRIBED FIRE AND IMAZAPYR APPLICATION IN INTENSIVELY MANAGED, MID-ROTATION PINE STANDS OF MISSISSIPPI

Numerous challenges face wildlife managers in the 21st century such as destruction and fragmentation of habitats, increasing urban sprawl, greater demands for commodities (e.g., food and timber), and climate change (MacArthur and Wilson 1967, Harris 1984, Reid 1994, Pimm and Gilpin 1989, Vitousek et al. 1997). Because biodiversity stabilizes ecological functions and enhance ecosystem performance (Naem et al. 1999, McCann 2000), wildlife management techniques increasing and conserving biodiversity are essential for sustaining our natural resources. Forest managers are increasingly expected to incorporate management efforts contributing to the conservation of biodiversity (Sustainable Forestry Initiative 2005), but the most commonly applied management practices at mid-rotation, such as commercial thinning and fertilizing, may only provide short-term (< 4 years) benefits for conservation (Wood 1986, Peitz et al. 1999, Iglay et al. 2010*a*). Because species diversity is a criterion for sustainability (Grumbine 1990, Hunter 1990, Bourgeron and Jensen 1994, Kaufmann et al. 1994, Sustainable Forestry Initiative 2005), management tools that enable conservation of biological diversity within intensively managed forests would benefit conservation of biodiversity, sustainable forestry objectives, and social expectations and needs (Hunter 1990, Burton et al. 1992, Hartley 2002, Carnus et al. 2006, Sustainable Forestry Initiative 2005). Without proper management for biodiversity, managed forests may become increasingly biologically simplified causing declines in habitat quality for wildlife (Perry 1994, Carey et al. 1999), ecosystem function (Canham et al. 1990, Franklin 1993, Tilman 1999), site productivity, watershed quality, and carbon sequestration (Harmon et al. 1996, Ponge et al. 1998, Vogt et al. 1999).

Effects of forest management on biological diversity are a concern among forest ecologists, environmentalists, and the general public (Hunter 1990, Burton et al. 1992). Disturbances can perpetuate biodiversity (Turner et al. 2001, White and Jentsch 2001, Rundel et al. 1998), and although frequent disturbances favor disturbance-dependent or –tolerant species, timber management incorporating biocomplexity objectives could compensate for disturbance-intolerant species by creating a mosaic of stand treatments, vegetative structures, species, and successional stages across a landscape (Hunter 1990, Franklin 1993, Heljden et al. 1998, Carey et al. 1999*a*, Carey et al. 1999*b*, Tilman 1999) . Managed forests can provide ecosystem benefits such as wildlife habitat, protection of water quality, and carbon sequestration while offering a mosaic of successional stages across the landscape (Petraitis et al. 1989, Greenberg et al. 1994, McLeod and Gates 1998, Vogt et al. 1999). Therefore, it is essential to determine optimal management approaches of intensively managed, mid-rotation pine (*Pinus spp.*) stands to meet forestry and biodiversity objectives (Wear and Greis 2002, Stein et al. 2005).

Intensively managed pine forests cover 18 million ha in the southern United States, about 20% of southern forests, with 2.2 million ha in Mississippi (USDA Forest Service 2007). Management for economic gain and biodiversity objectives is achievable (e.g., Miller et al. 2009, Wigley et al. 2000), but current practices at mid-rotation may not achieve adequately long-term biodiversity and sustainable forestry objectives. Addition of dormant season prescribed fire and selective herbicide application after commercial thinning may enhance wildlife habitat and sustain improved biodiversity until harvest supporting concepts of sustainable forestry (Hartley 2002, Carnus et al. 2006, Sustainable Forestry Initiative 2005).

Prescribed fire and selective herbicides are 2 silviculture tools used to control midstory hardwood competition and improve wildlife habitat in mid-rotation, intensively managed pine stands of southeastern United States (Brockway and Outcalt 2000, Edwards et al. 2004, McInnis et al. 2004). Prescribed fire is similar to historical disturbances of the region (Brennen et al. 1998). Following specific prescriptions, dormant season prescribed fires, applied during winter, avoid detrimental effects on pine growth caused by crown scorch (Wade and Lunsford 1989, Bessie and Johnson 1995, Schimmel and Granstrom 1997). Selective herbicides, such as those containing imazapyr, offer an alternative to prescribed fire lacking smoke management issues or limited burning degree days (Wigley et al. 2002). Both treatments have demonstrated abilities to reduce woody plant coverage and consequently increase herbaceous understory plant coverage (Stransky and Harlow 1981, Brockway and Outcalt 2000, Miller and Miller 2004, Chapter 1). Faunal species are generally impacted indirectly by prescribed fire and imazapyr herbicide through changes in vegetation structure and diversity (Lillywhite

1977, Greenberg et al. 1994, Bamford et al. 1995, Ford et al. 1999, Russell et al. 1999, Wade and Lunsford 1989, McComb and Hurst 1987, Guynn et al. 2004, Miller and Miller 2004, Sladek et al. 2008). Past research also has demonstrated treatment benefits to a myriad of game and non-game animals such as white-tailed deer (*Odocoileus virginianus*; Demarais et al. 2000, Mixon et al. 2009, Iglay et al. 2010*b*), eastern wild turkey (*Meleagris gallopavo silvestris*; Dickson and Wigley 2001, Miller and Conner 2007), northern bobwhite (*Colinus virginianus*; Guynn et al. 2004, Miller and Miller 2004, Welch et al. 2004), and songbirds (Sladek et al. 2008). Enhanced habitat quality for some game species may also increase economic gains from hunting leases in addition to treatment benefits to forest health (Guynn and Marsinko 2003, Iglay et al. 2010*b*).

Monitoring biodiversity response to prescribed fire and imazapyr herbicide can be achieved through observing a subset of communities (Sullivan et al. 2001). Sullivan et al. (2001) suggested monitoring multiple faunal groups such as small mammals, herpetofauna, birds, and insects for several decades. Small mammals (Orders: Rodentia and Soricidae) serve a variety of functional roles in forests [e.g., prey (Verts and Carraway 1998); consumers of invertebrates, vegetation, fruits, and seeds (Terry 1974, Gunther et al. 1983); dispersers of seed and fungal spores (Gashwiler 1970, Maser et al. 1978, Price and Jenkins 1986)] and demand research attention when addressing management impacts on faunal communities (Terry 1974, Gunther et al. 1983, Verts and Carraway 1998). Herpetofauna comprise the vast majority of vertebrate species in forest ecosystems (Burton and Likens 1975, Petranka and Murray 2001) and are globally declining (Vitt et al. 1990, Gibbons et al. 2000, Stuart et al. 2004). Sensitivity of avifauna to environmental changes, diversity of taxa, defined habitat relationships, and ease of monitoring make them ideal for measuring responses to forest management practices (Maurer 1993, Nuttle et al. 2003). Carabid beetle communities (Coleoptera: Carabidae) also are sensitive to habitat changes making them useful for indicating forest management impacts (Werner and Raffa 2000, Castillo and Wagner 2002, Heybourne et al. 2003, Koivula 2002, Niwa and Peck 2002, Warriner et al. 2002, Pearce et al. 2003, Vance and Nol 2003).

Information is lacking on biodiversity response to prescribed fire and imazapyr in intensively managed pine stands of the southeastern United States. Although biodiversity can occur at multiple scales (i.e., spatial and temporal scales), stand level management is most compatible with the scale of forestry operations on intensively managed landscapes (Franklin 1993*b*). However, increased species richness at this smaller scale would inherently influence landscape-level diversity (Carnus et al. 2006). Although Sullivan et al. (2001) suggest monitoring faunal communities for several decades, short-rotation management limits sampling periods for investigating mid-rotation treatments.

Biodiversity monitoring programs can be valuable for providing vital information for forest managers regarding sustainable forestry practices (Ferris and Humphrey 1999, Sustainable Forestry Initiative 2005). However, conducting large scale assessments of multiple fauna and floral communities is extremely expensive and time consuming and difficult to support over multiple years (Lawton et al. 1998, Smith et al. 2008). Indicator species can be beneficial for monitoring and research programs but may depend on group faithfulness (McCune and Grace 2002). Therefore, I sampled floral (e.g., vegetation structure and biomass) and faunal communities (e.g., birds, rodents, reptiles, amphibians, and carabid beetles) for 10 years in intensively managed pine stands after commercial thinning to investigate biodiversity response to factorial combinations of prescribed fire and imazapyr in east-central Mississippi. I also determined indicator species for future research investigating similar treatments and when applicable based on species faithfulness, for forest monitoring programs. Although my study was performed in east-central Mississippi, I used a replicated field experiment supporting a larger sphere of inference than past studies without replicates or using a measurative approach. My 3 objectives were to investigate biodiversity response among treatments, determine if treatments created distinct communities, and determine indicator species relevant to treatments.

Study Area and Design

A full description of study area and design is provided in Chapter 1.

Methods

Vegetation Community

In July 1999-2008, I measured plant productivity using 20, 1-m^2 clip hoops/plot (10 hoops/plot 1999 and 2000) placed randomly diagonally across each treatment plot. I increased number of hoops/plot to reduce variation. I clipped all plants $\leq$ 1.3 cm diameter, separated leaves and growing tips from stems, and placed them in brown paper lunch bags marked with hoop number, treatment plot, collection date, and plant species acronym. I dried all samples at 80^{o}C until constant weight was obtained and extrapolated biomass estimates (kg/ha).

Avian Community

I conducted point counts for avifauna twice monthly during May and June 1999-2008 from 0530-1030 hours. I designated one point per corner per treatment plot (n= 4/treatment plot) with each $\geq$ 75 m from plot edge and $\geq$ 100 m from other bird survey points for point counts. I used distance bands of < 25 m, 25 - 50 m, and $\geq$ 50 m during 1999 and 2003-2008 and a fixed-radius circular point with detections of < 50 m recorded during 2000-2002. I also used 3 time brackets of 1-3, 4-5, and 6-10 minutes to compare among sampling approaches (Hutto et al. 1986, Ralph et al. 1995, Hamel et al. 1996). I only sampled under weather conditions stipulated by the Breeding Bird Survey (Robbins et al. 1986, Ralph et al. 1995). I limited species to those detected < 50 m from the observer and with $\geq$ 40 observations across all years. I calculated mean relative abundance across 4 points and 4 visits within each year.

Small Mammal Community

I trapped small mammals using drift fence arrays during May and June 1999-2007 and Sherman live box traps (7.6 x 7.6 x 27.9-cm) baited with peanut butter and oats during January-March 1999-2007. I created drift fence arrays with 4, 5-gallon buckets arranged as one center bucket with 5-m arms of 35.6 cm high aluminum flashing at 120^{o} angles from center bucket with a single bucket at arm ends. Along each arm, I placed one funnel trap of wire mesh (Enge 1997) so that each quadrant had one funnel trap. I permanently placed 3 drift fences per treatment plot diagonally with each $\geq$ 50 m from plot edge. I trapped 3 stands simultaneously for 10-days twice monthly May and June and once in October and recorded species and trap location (bucket and treatment plot) for every capture. When not in use, I closed traps by removing funnel traps, closing buckets using lids, and placing a large stick in each bucket to allow animals to escape in case lids were detached between trapping sessions. I placed Sherman live box traps randomly in a 5 X 5 grid with 20 m spacing (80 X 80 m) during 1999 and 2001-2007 $\geq$ 50 m from plot edge. In 2000, I used a randomly placed 7 X 7 trapping grid (120 m X 120 m) to compare trapping efficiency to a 5 X 5 grid. I trapped 2 stands (1999-2002) or 3 stands (2003-2007) simultaneously for 10 days or until a 50% recapture rate was reached with an adequate trapping history. I recorded species, gender, weight (g), and toe-clip number for mark recapture for every small mammal in the order Rodentia (Baumgartner 1940, Melchoir and Iwen 1965, Nietfeld et al. 1996). I also weighed shrews (Family: Soricidae) but did not mark individuals. I removed Sherman traps from each site when not trapping. From these data, I calculated mean species catch-per-unit effort (CPUE) from new captures by treatment plot, species richness, and Shannon-Weaver diversity index for final analyses. I also grouped white-footed (*Peromyscus leucopus*) and cotton mice (*Peromyscus gossypinus*) as *Peromyscus* spp. due to similarities in field identification characteristics and hybridization among these species (McCarley 1954, Laerm and Boone 1994, Rich et al. 1996, Barko and Feldhamer 2002).

Herpetofauna Community

I trapped herpetiles using drift fence arrays May and June 1999-2007. I created drift fence arrays with 4 5-gallon buckets arranged as one center bucket with 5-m arms of 35.6 cm high aluminum flashing at 120^{o} angles from center bucket with a single bucket at arm ends. I buried each arm $\geq$ 5 cm, drilled holes in the bottom of each bucket for drainage, and ensured bucket lips were even with or slightly below ground level. From 2004-2007, I placed a piece of woody debris in each bucket for captured animals to use as cover or for flotation. Along each arm, I placed a funnel trap of wire mesh (Enge 1997) so that each quadrant had one funnel trap. I permanently placed 3 drift fences per treatment plot diagonally with each $\geq$ 50 m from plot edge. I trapped 3 stands simultaneously for 10-days twice monthly May and June and once in October and recorded species and trap location (bucket and treatment plot) for every capture. When not in use, I closed traps by removing funnel traps, closing buckets using lids, and placing a large stick in each bucket for animals to use to escape in case lids were detached between trapping sessions. From these data, I calculated mean species catch-per-unit effort (CPUE) from new captures and mean species richness by treatment plot for final analyses. I also grouped all toads in the American toad (*Bufo americanus*) complex including American toads, Fowler's toads (*Bufo fowlerii*), and southern toads (*Bufo terrestris*) as *Bufo spp.* due to similarities in field identification characteristics and hybridization among these species (Blair 1941, Cory and Manion 1955, Volpe 1956, Blair 1959, Meacham 1962, Blair 1963). I followed IACUC protocol #98-046 approved by Mississippi State University's IACUC for all herpetile and small mammal trapping and handling.

Carabid Community

I sampled carabid diversity using 2 randomly located 80 m transects established in 1999 $\geq$ 50 m from plot edge and each other. I placed traps, a 0.47 L container with 3 sheet metal barriers at 120° angles from center container, every 20 m along each transect per treatment plot (n = 8). I filled containers with equal amounts of propelyne glycol and 70% ethanol and placed the lip of each container at or slightly below ground level (Morrill 1975, Southwood 1966, Housewart et al. 1979, Murkin et al. 1994, Ausden 1996). I sampled each treatment plot once monthly for one week of continuous trapping May-October 2004-2007. I sorted, pinned and labeled specimens with trap locality data to serve as voucher specimens in the Mississippi Entomological Museum. I identified all specimens to species using taxonomic keys and confirmation for 2004 and 2005 carabids from Drew Hildebrand, research associate of the Mississippi Entomological Museum. I calculated catch-per-unit-effort (species abundance/number of active traps/treatment plot) for each species, mean species richness, and total relative abundance. I used CPUE as an index of relative abundance (Ludwig and Reynolds 1988) and therefore refer to relative abundance in results and discussion, not CPUE.

Statistical Analysis

I used 3 analyses to examine biodiversity response to prescribed fire and imazapyr herbicide in intensively managed pine stands of Mississippi. First, I compared diversity responses among treatments using mixed models, repeated measures analysis of covariance in SAS (MIXED procedure; SAS Institute Inc., Cary, North Carolina, USA) to examine main effects of treatment, year, and treatment × year on community indices. I used a repeated measures analysis of variance in SAS for carabid data because I did not have pretreatment data for a covariate. I used Simpson’s index for a measure of dominance rather than diversity to emphasize overall evenness of each community (Simpson 1949), and I used Shannon-Wiener index because it incorporates species richness and evenness (MacArthur and Macarthur 1961, McCune and Grace 2002). I calculated a separate index for each community. For each model, I used 4 treatment levels (burn, herbicide, burn + herbicide, control), random effect of stand (n=6), repeated measures of year [n=9 (2000-2008; birds), n =8 (2000-2007; small mammals and herpetofauna), n=4 (2004-2007; carabids)], and subject of stand × treatment (Littell et al. 2006) to test the hypothesis of no difference in Simpson's index and Shannon-Weiner diversity index among treatments within years or overall treatment effects. I used pre-treatment (1999) diversity indices as baseline covariates because pre-treatment diversity may have differed among treatment plots treated alike (Milliken and Johnson 2002). For each model, I selected an appropriate covariance structure from the following: (n-1 years)-banded Toeplitz, heterogeneous compound symmetry, heterogeneous auto-regressive, and auto-regressive. I designated the covariance structure that minimized Akaike's Information Criterion corrected for small sample size (AIC_c) as the top candidate for analysis (Littell et al. 2006, Gutzwiller and Riffell 2007). I checked residuals and transformed data when deemed necessary to meet normality assumptions. I used Kenward-Roger correction for denominator degrees of freedom for repeated measures and small sample sizes (Littell et al. 2006, Gutzwiller and Riffell 2007). I used LSMEANS SLICE option to identify a treatment effect within years following a significant interaction and LSMEANS PDIFF to conduct pair-wise comparisons (Littell et al. 2006). All year references in results refer to years post-treatment. My *a prior* significance level was α=0.05.

To test for no difference in floral and faunal species composition among treatments, I used blocked multi-response permutation procedures (MRBP) in PC-ORD

5.1. I constructed separate matrices for each taxa and year. To avoid saturation of community data with minimal contributing species, I limited plant species to those contributing $\geq$ 0.01% total plant biomass and bird species to those with $\geq$ 40 observations for all years. Although my cut-off points were arbitrary, they accounted for 99.5% of total plant biomass (210 of 387 species) and 99% of observed bird species (41 of 63 bird species) and also avoided including plant species unknown to observers or uncharacteristic bird sightings (i.e., non-residents). Other taxa did not have nearly as many non-contributing species. Prior to making my final data set, I grouped species of each community by treatments and blocked by stand. Unlike parametric procedures such as MANOVA, MRBP does not require distributional assumptions such as multivariate normality and homogeneity of variances uncommon to community data (McCune and Grace 2002). Using MRBP, I first calculated a weighted mean of within-group distance (δ) where smaller values of δ indicated tighter clustering of groups. Then, I calculated the test statistic, T, which is the difference of observed and expected δ divided by standard deviation of expected δ. The T-statistic describes separation between groups where more negative T values denote greater group separation. I used Euclidean distance as the distance measure and used median alignment within blocks for analyzing within-block differences (Mielke 1984, McCune and Grace 2002). Because MRBP can not handle unbalanced designs, I excluded stand 2 from summer 2003 plant biomass analysis due to missing samples in one plot. A test of T's significance provides the probability of observing a $\delta \geq \delta_{observed}$ given the Pearson type III distribution (e.g., p-value) and chance-corrected within-group agreement (A) describes within-group homogeneity [e.g., effect size; complete homogeneity within groups (A = 1), heterogeneity within groups equals expectation by chance (*A* = 0), less agreement within groups than expected by chance (*A* < 0)]. I conducted pair-wise comparisons of groups when $0 < A \leq 1$ and *p*-value < 0.05 to differentiate communities among treatments.

Last, I determined indicator species per treatment when MRBP indicated differences in species composition among treatments ($0 < A \leq 1$ and *p*-value < 0.05; McCune and Grace 2002). Indicator species analysis is an extension of MRBP by differentiating groups at species level and describing how well each species contributes to group separation. Indicator species analysis calculates an indication value (IV) for each species in each group (e.g., species A in treatment burn + herbicide) based on the species' relative abundance or biomass by group (exclusiveness) and relative frequency. Greatest indicator values per species among groups (IV_{max}) denotes the species overall indicator value. A Monte Carlo test of 1,000 permutations tests the statistical significance of IV_{max} (e.g., hypothesis of IV_{max} no larger than expected by chance). I designated species with IV ≥ 30 as top contributors to differentiating communities among treatments.

I used Principal Components Analysis (PCA) with correlation matrices to ordinate significant indicator species ($P < 0.05$) among treatments and years in PC-ORD (version 5.10, MJM Software, Gleneden Beach, Oregon, USA; Taylor et al. 1993, Morrison et al. 1998, ter Braak and Smilauer 1998, McCune and Grace 2002). Because indicator species analysis provided indicator values at treatment level not plot level, I used treatments as my sampling units and labeled them according to year to help visualize how indicator species relate to treatments. I constructed matrices of all significant indicator species using indicator values instead of relative abundances or biomass among burn only (B), herbicide only (H), burn + herbicide (BH), and control (C) treatments. I used randomization tests of eigenvalues with 999 runs and resulting test statistic, Rnd-Lambda, as a stopping rule to determine total number of non-trivial components (Peres-Neto et al. 2005). I used Rnd-Lambda because it restricts introducing additional noise into analysis results and is robust with non-normal data and uncorrelated variables (Peres-Neto et al. 2005). I deemed eigenvector loadings ≥ |0.3| significant for interpretation and used variables present in ≥ 2 years to minimize sampling units to variable ratios as suggested by Pillar (1999). When > 1 component was considered non-trivial, I graphed components for a visual representation of treatment plot differentiation.

Results

I collected 1,224 samples (12,272 subsamples) with 24 samples/year/community except small mammals with 48 samples/year (*vegetation biomass* = 240 samples, *birds* = 240 samples, *herpetofauna* = 216 samples, *small mammals* = 432 samples, *carabid beetles* = 96 samples). I collected or observed 387 plant species, 63 bird species, 32 herpetile species (15 amphibians, 17 reptiles), 11 small mammal species, and 41 carabid species. Final data sets for plants and birds included 210 and 41 species, respectively. Shannon-Weiner Diversity Index covariate for birds and Simpson's Index covariate for birds and reptiles differed among treatment plots (Table 6.1). Shannon-Weiner Indices of small mammal and vegetation communities differed among treatments within years. Small mammals had greater diversity in herbicide treatments (herbicide only and burn + herbicide) in year 3 ($\bar{x}$ = 1.55 vs. $\bar{x}$ = 1.35), and vegetation community had greater diversity in burn only and controls ($\bar{x}$ = 2.30) than herbicide treatments ($\bar{x}$ = 1.70) in year 1 and greatest diversity in burn only in year 6 ($\bar{x}$ = 2.5 vs. $\bar{x}$ = 2.03; Table 6.2). Bird diversity was greater in burn only than herbicide treatments across all years with controls intermediate ($\bar{x}$ =2.64 vs. $\bar{x}$ = 2.54, SE = 0.03). Simpson's Index of vegetation community followed the same trend as vegetation community's Shannon-Weiner Index in years 1 and 6 (Table 2). However, Simpson's Index of vegetation community also was greater in burn only than burn + herbicide in year 2 with herbicide only and control intermediate.

Vegetation and bird species composition differed among treatments all post-treatment years (Table 6.3). Vegetation species composition of herbicide treatments (herbicide only and burn + herbicide) differed from those of controls for most (8 of 9) post-treatment years and also differed from vegetation species composition of burn only 5 of 9 post-treatment years. Vegetation species composition also differed between herbicide treatments in 4, 7, and 9 years post-treatment. Bird species composition differed among each treatment one year prior and 2 years after the second prescribed burn and again for all 3 years post third prescribed burn.

Small mammal species composition differed among treatments 4 years post-treatment but amphibian, reptile, and carabid species compositions differed by treatments for only one year post-treatment each (Table 6.3). Small mammal species composition differed among all treatments one year post-treatment. Six years post-treatment small mammal species composition of burn treatments (burn only and burn + herbicide) differed from herbicide only and controls. Small mammal species composition of burn only and controls differed from burn + herbicide 2 years post-treatment with species composition of herbicide only intermediate to all treatments. Five years post-treatment small mammal species composition burn treatments differed from controls with species composition of burn + herbicide similar to independent treatments (burn only and herbicide only) and species composition of herbicide only similar to burn + herbicide and controls. Amphibian species composition differed among burn only and herbicide only 6 years post-treatment, and reptile species composition in treated sites differed from those of controls one year post-treatment. Carabid beetle species composition differed in independent treatments from controls with communities of burn + herbicide intermediate 7 years post-treatment.

Indicator species of vegetation and bird communities varied across years within treatments with distinct patterns but no perfect indicators ($IV_{max} = 100$; Table 6.4). Burn + herbicide had the greatest number of vegetation community indicators across all years (n = 26 species) followed by burn only (n = 15 species) and then by herbicide only and controls (n = 8 species each). Grass and forb species comprised most indicators for vegetation communities of burn treatments (burn only = 10 of 15 species, burn + herbicide = 18 of 26 species). Indicators of herbicide only varied among forage classes (e.g., 1 forb, 1 fern, 1 grass, 2 semi-woody vines, 2 woody plants, 1 woody vine). Woody plants (e.g., shrubs, trees) and vines contributed most (6 of 8 species) to indicators of controls. Variable panicgrass (*Dichanthelium commutatum*) was the predominate indicator of burn + herbicide plots (n = 5 years) followed by openflower rosette grass (*Dichanthelium laxiflorum*; n = 3 years). Seedbox (*Ludwigia alternifolia*), Carolina horsenettle (*Solanum carolinense*), Canada goldenrod (*Solidago canadensis*), wrinkleleaf goldenrod (*Solidago rugosa*), hirsute sedge (*Carex complanata*), slender lespedeza (*Lespedeza virginica*), and Virginia creeper (*Parthenocissus quinquefolia*) were indicators of burn + herbicide for 2 years each. Winged sumac (*Rhus copallinium*) and

Carolina vetch (*Vicia caroliniana*) were indicators of burn only plots ≥ 2 years. Ebony spleenwort (*Asplenium platyneuron*) was the only indicator of herbicide only plots for > 1 year (n = 2 years), and poison ivy (*Toxicodendron radicans*) was the predominate indicator of control plots (n = 4 years) followed by black gum(*Nyssa sylvatica*; n = 3 years) and southern red oak (*Quercus falcata*; n = 2 years).

Similar to vegetation communities, bird communities of burn + herbicide had the most indicator species across all years (n = 12 species; Table 6.5). Burn only had 2 species [Red-bellied woodpecker (*Melanerpes carolinus*), Indigo bunting (*Passerina cyanea*)]. Herbicide only followed burn + herbicide with 7 indicator species, and controls had 6 indicator species. Indigo buntings (n = 9 years), eastern wood-pewees (*Contopus virens*; n = 7 years), and common yellowthroats (*Geothlypis trichas*; n = 6 years) were the predominate indicators of burn + herbicide plots. Hooded warblers (*Wilsonia citrina*) were predominate indicators of herbicide only (n = 4 years) and controls (n = 6 years) followed by white-eyed vireos (*Vireo griseus*) for 3 and 4 years in herbicide only and controls, respectively. Red-bellied woodpeckers and indigo buntings were the only indicators of burn only plots, each for only one year.

Only 4 indicator species were apparent for amphibians, small mammals, and carabid communities for 4 of 9 post-treatment years. Reptile communities had no significant indicator species. Eastern narrowmouth toads (*Gastrophryne carolinensis*) were indicators of burn only 5 years post-treatment (Table 6.6). Golden mice (*Ochrotomys nuttalli*) were indicators of independent treatments (burn only and herbicide only) one year post-treatment and controls 2 years post-treatment. *Peromyscus* spp. were indicators of burn + herbicide small mammal communities 2 years post-treatment.

Cyclotrachelus convivus was the only indicator of carabid communities for control 7 years post-treatment.

Two non-trivial principal components explained 32.37% of the total variance among indicator species and treatments (18.49% for PC1 and 13.99% for PC1). Twenty-five variables were included in the PCA matrix of which 20 were significant according to eigenvector loadings (Table 6.7). Axis 1 separated treatment extremes (burn + herbicide and controls) with independent treatments intermediate (Figure 6.1). Axis 2 did not segregate treatments but separated years 1 and 2 from years 3-6 post-treatment. According PC1 eigenvector loadings, relative abundance of common yellowthroats, eastern wood pewees, indigo buntings, eastern towhees, and yellow-throated warblers and biomass (kg/ha) of Hirsute sedge (*Carex complenata* Torr. and Hook), Canada goldenrod (*Solidago canadensis* L.), and variable panicgrass [*Dichanthelium commutatum* (Shult.) Gould] were all associated with burn + herbicide. Relative abundances of white-eyed vireos, wood thrush, hooded warblers, and northern cardinals and biomass of poison ivy were associated with controls. During initial post-treatment years according to PC2 eigenvector loadings, white-eyed vireos, Carolina vetch (*Vicia caroliniana*), slender woodoats [*Chasmanthium laxum* (L.) Yates], rice button aster (*Aster dumosus* L.), longleaf woodoats [*Chasmanthium sessiliflorum* (Poir.) Yates], openflower rosette grass [*Dichanthelium laxiflorum* (Lam.) Gould], and golden mouse were main indicators among all treatment plots. From years 3-6 post-treatment, eastern towhee, yellow-throated warbler, hooded warbler, yellow-breasted chat, wooth thrush, Hirsute sedge, and poison ivy were main indicator species of treatment plots.

Discussion

Prescribed fire and imazapyr applied to mid-rotation, intensively managed pine stands can increase biodiversity supporting sustainable forestry objectives (Sustainable Forestry Initiative 2005). Vegetation and bird communities had greatest responses to treatments whereas small mammal, herpetofauna, and carabid communities each had limited responses to treatments. Few observations of these communities throughout the study when compared to vegetation and bird communities may have limited observed responses. Vegetation communities were influenced directly by treatments whereas faunal communities most likely responded to changes in vegetation structure, plant coverage, and platn biomass (Lillywhite 1977, Greenberg et al. 1994, Bamford et al. 1995, Ford et al. 1999, Russell et al. 1999, Wade and Lunsford 1989, McComb and Hurst 1987, Guynn et al. 2004, Miller and Miller 2004, Sladek et al. 2008, Chapter 1). Although combining prescribed fire (i.e., 3 year, fire-return interval) and a one time application of imazapyr prior to fire had the most dramatic affect on wildlife communities, independent applications of fire and imazapyr also increased biodiversity by increasing landscape-level heterogeneity of vegetation types.

Imazapyr application ensured control of hardwood midstory competition and alleviated competition pressures on wildlife preferred forages such as sawtooth blackberry (*Rubus argutus*), legumes (family: Fabaceae), and panic grasses (*Dichanthelium* spp.) by reducing coverage of woody understory plants and increasing sunlight on the forest floor (Peitz et al. 1999, BASF 2006, Iglay et al. 2010*a*). Repeated prescribed burns on a 3 year, fire-return interval maintained reduced canopy coverage, stimulated new plant growth, augmented litter layers possibly providing seed catchment opportunities, and also contributed possibly to ecological functions such as nutrient cycling and fuel reduction (Brennan et al. 1998, Iglay et al. 2010*a*). Without prescribed fire, imazapyr's selectivity and strong influence on vegetative communities limited plant community response and perpetuated dominance by a few species released from competition (Iglay et al. 2010*a*). Therefore, vegetation species composition were similar herbicide-treated sites immediately post-treatment for 3 years and then segregated from controls for remaining years suggesting prolonged residual effects of imazapyr. Burn only vegetation specie composition did not differ from controls until the last year of observation, suggesting a delayed response from vegetation communities to prescribed fire compared to imazapyr application. However, fire sustained increased plant species richness and with imazapyr, harbored the greatest number of indicator species across all years (Iglay et al. 2010*a*, Chapter 1).

Vegetation biomass community indices did not vary among treatments for many years but were always greater in burn only plots than most other treatments. Because Shannon-Weiner Diversity is robust to rare species in large sample sizes, greater diversity of burn only plots indicates a greater number of frequently observed species. Greater Simpson's Index in burn only plots such as 6 years post-treatment suggests frequently observed species in burn only also were dominant species. However, recent results from this project revealed greater plant dominance in herbicide treated sites (Iglay et al. 2010*a*). Species unaffected by imazapyr were able to remain well-established throughout the entire study because they were released from competition (Iglay et al. 2010*a*). When prescribed burns were applied, however, species dominance was reduced (Iglay et al. 2010*a*). Although burn only plots had greater species dominance 6 years post-treatment, the diversity of species dominating sites was most likely greater than diversity of species in other treatments indicating greater species richness (Chapter 1).

Similar to past studies, plant indicator species per treatment emphasized expected responses of understory plant community to treatments but did not reveal species of great enough faithfulness and exclusiveness to merit use as significant indicators of each treatment. Burn + herbicide promoted herbaceous understories with only 2 indicator species of woody plants and vines. Remaining 24 indicators were forbs (13 species), grasses (5 species), herbaceous and semi-woody vines (4 species), sedges and rushes (1 species), and legumes (1 species). Burn only also promoted herbaceous understory communities indicated by 6 forb and 4 grass species but also had a woody plant component (4 woody plant species, 1 woody vine species). Past studies also have observed reduction of woody plants and increased herbaceous, early successional species post-burn (Masters et al. 1993, Sparks et al. 1998, Brockway and Outcalt 2000). Imazapyr has a greater initial impact on woody species (Jones and Chamberlain 2004, Miller and Miller 2004, Iglay et al. 2010*a*). However, imazapyr's residual effects on understory species composition are short-lived, and its ability to encourage plant dominance restricts overall diversity compared to prescribed fire (Iglay et al. 2010*a*). Across all post-treatment years, vegetation species composition of herbicide only showed greater similarity to controls in terms of species richness and diversity than either burn treatment (Iglay et al. 2010*a*, Chapter 1).

Diversity of bird communities among mid-rotation pine stands also can be increased by prescribed fire and imazapyr herbicide application. Some trends in bird species compsition differed than those of vegetation species composition as expected because most faunal communities respond to changes in vegetation structure, not biomass (MacArthur and MacArthur 1961, Willson 1974, Roth 1976, Maurer 1986, Herkert 1994, Sallabanks et al. 2006). Each treatment used created unique vegetation structure characteristics (Chapter 1). Most vegetation structure differences were between treatment extremes (e.g., burn + herbicide and control) with independent treatments similar to one of the extremes. Bird species composition differed among treatment extremes similar to vegetation structure with similarities among independent treatment applications 4 of 9 post-treatment years. However, for remaining post-treatment years, bird species composition were grouped by treatment suggesting greater differences among vegetation structures than determined by analysis of vegetation data.

Greater number of indicator species in bird communities of burn + herbicide suggests a greater contribution to bird conservation by this treatment versus any other in this study (Chapter 2). Although indicators are not considered high priority species in the southeastern United States (Nuttle et al. 2003, Partners in Flight 2010), the breadth of species responding to this treatment emphasizes its potential as a conservation tool in intensively managed forests. Common indicators of burn + herbicide were bird species that prefer open forests (e.g., eastern wood-pewee) and open shrub nesting habitat (e.g., indigo buntings and common yellowthroats; Cooper 1996, Burger et al. 1998, Ricketts and Ritchison 2000, Hartung and Brawn 2005). Herbicide only and controls had similar indicators, specifically hooded warblers and white-eyed vireos. These species prefer closed canopies and thick shrub-nesting habitat, respectively (Hamel 1992, Hunter et al. 2001). Only burn treatments had woodpeckers as indicators, possibly due to greater access to food resources or potentially more ideal habitat conditions. Smucker et al.

(2005) observed similar trends of greater relative abundance of woodpeckers in burned versus unburned sites in Montana after a wildfire, and Breininger and Smith (1992) observed greater relative abundances of downy woodpeckers (*Picoides pubescens*) in recently burned areas in coastal scrub and slash pine Florida habitat.

Minimal treatment effects on herpetofauna, small mammal, and carabid beetle communities may have caused few observed groups among treatments and indicator species. Common species comprised most captures for each of these communities possibly supporting the lack of effective groupings by treatments for MRBP. Past studies regarding prescribed fire have reported minimal effects on herpetofauna communities and sometimes benefits to herpetofauna diversity (Means and Campbell 1981, Campbell and Christman 1982, Stout et al. 1988, Greenberg et al. 1994, Ford et al. 1999, Greenberg and Waldrop 2008, Perry et al. 2009). Herpetiles are used commonly as indicator species because of their sensitivity to changes, especially amphibians (Welsch and Droege 2001). However, within a landscape of repeated, large-scale disturbances (e.g., planting, thinning, clear cut, etc.), small scale disturbances such as treatments applied to 10-ha experimental units may not cause fluctuations in disturbance-tolerant populations such as many species of the southeastern United States (Means and Campbell 1981, Greenberg et al. 1994). Small mammals are inherently robust to disturbances especially when only observed over a brief time period with small scale perturbations (Kirkland 1990, Chapter 3). Long-term trends may reveal impacts on population dynamics and community diversity (Sullivan 1979, Loeb 1999, Converse et al. 2006), but short-rotations limit lengths of mid-rotation studies. Prescribed burning can have negative affects on carabid communities, but areas of refuge provided by unburned patches, treatment buffers, and ground litter may have protected source populations reducing observed treatment effects in this study (Gandhi et al. 2001, Saint-Germain et al. 2005). *Cyclotrachelus convivus*'s indication of controls may have been caused by reduced relative abundance in burn treatments (Chapter 5). If information about local carabid communities across the landscape were available, it could have helped disseminate whether observed communities were constructed by responses to treatments or if they resemble typical communities of Mississippi's intensively managed pine forests.

Indicator species determined through this study, particularly vegetation and bird species, may aid future monitoring and research programs. My study provides a snapshot of biodiversity response to treatments without indication of population response or long-term effects. Because intensively managed pine stands have limited temporal windows between disturbances (e.g., site preparation, thinning, harvest), within-stand species composition may be simplified by repeated disturbances (Cobb et al. 2007). However, proper management of less disturbed areas (e.g., streamside management zones, buffer zones) and implementation of mid-rotation treatments as suggested here can enhance contributions to biodiversity by intensively managed forests. Future research concerning biodiversity response to treatments may benefit from monitoring faunal community characteristics across an intensively managed forest landscape prior to and after treatment application instead of only focusing on experimental units. Investigating population dynamics of some indicator species also may increase our knowledge of wildlife community responses to treatments.

Forest management determined to support sustainable forestry can benefit from applying prescribed fire and imazapyr at mid-rotation in intensively managed pine stands of the southeastern United States. Although only 2 of 5 communities investigated had distinct groupings of species composition among treatments for multiple years, their responses emphasize impact of alternative mid-rotation management practices to biodiversity conservation objectives. None of these treatments applied after fertilizing have demonstrated significant affects on pine growth (McInnis et al. 2004, Chapter 1). However, considering beneficial treatment effects on white-tailed deer forage quality, forest managers may be able to compensate treatment costs through hunting leases (Iglay et al. 2010*b*, Chapter 1).

Management Implications

Combining prescribed fire and a one-time imazapyr application to mid-rotation, intensively managed pine stands will support greater biodiversity in the southeastern United States. When applied independently, imazapyr's ability to illicit a significant vegetative response immediately compared to delayed segregation of burn only vegetation species composition from controls highlights its effectiveness as a habitat management tool. Although the selectivity of imazapyr favors a limited suite of wildlife preferred plants (Iglay et al. 2010*a*), managers can explore multi-herbicide tank mixtures with or without prescribed fire to fine-tune habitat management approaches at local and regional scales. However, of the 2 independent treatments, prescribed fire is the single best tool for promoting diversity considering considerable increases in plant species richness. From a "bottom-up" diversity management approach, increased plant diversity would increase invertebrate and vertebrate species (Hunter 1990, Hunter and Price 1992, Power 1992).

Monitoring biodiversity response to treatments is inherently complex. Researchers should consider indicator species as foci for understanding overall treatment effects on wildlife communities (Sullivan et al. 2001). Researchers also may want to consider observing biodiversity responses among harvested (within-stand) and non-harvested areas (streamside-management zones, management buffers) to better understand overall treatment effects. My results were limited to within-stand plant and wildlife communities without concern for population dynamics possibly affected by landscape level interactions of disturbance, spatial segregation of primary resources (e.g., ephemeral pools), and cutting cycles (i.e., ages and disturbance regimes of surrounding stands).

Plantation forests play a pivotal role in maintaining and contributing to biodiversity, especially in the southeastern United States (Lindenmayer and Hobbs 2004, Nelson and Halpern 2005). Managers can improve these contributions through combined or independent applications of prescribed fire and imazapyr at mid-rotation. Brennan et al. (1998) proposed that herbicides will never replace fire due to their lack of effects on many ecosystem functions. I agree but also have shown that combining fire and imazapyr offers an ideal tool for returning fire to systems once pyric in nature (Brennan et al. 1998). As wildlife managers continue to face new conservation challenges in the 21st century, it is essential to bear in mind the potential contributions to biodiversity of commercial forests and other working landscapes. Such contributions may ensure the stability of many ecosystems for the benefit of future generations (Naem et al. 1999, McCann 2000).

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Table 6.1 Interaction and main effects of treatment (burn only, herbicide only, burn + herbicide, and control) and treatment × year on Shannon-Weaver Diversity and Simpson's indices of bird (summer 2000-2008), small mammal (summer and winter 2000-2007), herpetile (summer 2000-2007), carabid (April-October 2004-2007), and vegetation communities (summer 2000-2008) in intensively managed pine stands in Kemper County, Mississippi. Pre-treatment refers to a baseline data covariate from sampling in 1999.

	Pre-treatment		Treatment		Year		Treatment × year	
Index (*Community*)	*F*	*P*-	*F*	*P*-Value	*F*	*P*-	*F*	*P*-Value
Shannon-Weaver								
Bird	16.13	≤ 0.001	3.42	0.025	7.93	≤ 0.001	1.20	0.243
Reptile	1.38	0.244	0.78	0.511	7.76	≤ 0.001	1.21	0.249
Amphibian	1.03	0.316	1.69	0.179	22.98	≤ 0.001	1.41	0.120
Small Mammal	1.70	0.197	2.52	0.066	15.12	≤ 0.001	1.73	0.039
Carabid	-	-	1.75	0.175	20.75	≤ 0.001	0.58	0.811
Vegetation	3.45	0.073	7.61	≤ 0.001	11.93	≤ 0.001	2.08	0.006
Simpson's								
Bird	15.62	≤ 0.001	2.79	0.053	9.20	≤ 0.001	1.24	0.212
Reptile	5.49	0.023	0.89	0.453	5.52	≤ 0.001	1.38	0.136
Amphibian	0.06	0.803	1.99	0.128	14.58	≤ 0.001	1.39	0.142
Small Mammal[1]	1.37	0.246	1.16	0.332	17.44	≤ 0.001	1.90	0.014
Carabid	-	-	1.22	0.320	9.94	≤ 0.001	0.77	0.641
Vegetation	3.12	0.087	7.28	≤ 0.001	13.37	≤ 0.001	2.11	0.005

[1] Simpson's Index for small mammals differed within treatments across years.

Table 6.2 Least square mean estimates (SE) of Shannon-Weave and Simpson's indices among treatments within years in intensively managed pine stands of Kemper County, Mississippi. Small mammals were sampled by Sherman-live box traps and drift fences arrays in winter and summer 2000-2007, respectively. Vegetation biomass was sampled summer 2000-2008.

Variable *Community*	Year	*P*-value	Burn $\bar{x}$	Herbicide $\bar{x}$	Burn + herbicide $\bar{x}$	Control $\bar{x}$	SE
Shannon Weaver							
Small mammals	1	0.255	1.5	1.3	1.5	1.5	0.09
	2	0.197	1.4	1.4	1.3	1.6	0.08
	3[1]	0.005	1.4 B	1.5 A	1.6 A	1.3 B	0.05
	4	0.520	1.3	1.4	1.5	1.4	0.09
	5	0.514	1.2	1.3	1.3	1.3	0.09
	6	0.124	1.2	1.4	1.5	1.4	0.08
	7	0.107	1.1	1.2	1.0	0.8	0.12
	8	0.194	0.7	1.1	0.7	0.6	0.18
Vegetation	1	$\leq$ 0.001	2.4 A	1.7 B	1.7 B	2.2 A	0.10
	2	0.051	2.7	2.3	2.2	2.4	0.12
	3	0.305	1.6	1.6	1.7	2.0	0.15
	4	0.067	2.3	1.8	2.3	1.9	0.15
	5	0.159	2.2	1.9	1.9	2.1	0.11
	6	0.012	2.5 A	2.0 B	2.0 B	2.1 B	0.10
	7	0.083	2.3	2.1	2.3	2.0	0.10
	8	0.067	2.5	1.7	2.2	2.1	0.20
	9	0.389	2.5	2.1	2.4	2.3	0.18
Simpson's Index							
Vegetation	1	$\leq$ 0.001	8.8 A	4.3 B	3.8 B	6.1 A	0.70
	2	0.040	10.2 A	7.4 AB	6.4 B	7.8 AB	0.95
	3	0.403	3.7	3.4	3.9	4.8	0.67
	4	0.177	7.1	3.8	6.4	5.5	1.10
	5	0.155	7.1	5.2	4.9	6.7	0.84
	6	0.017	8.5 A	5.3 B	5.1 B	5.6 B	0.85
	7	0.058	7.2	5.4	6.7	5.3	0.64
	8	0.076	9.0	4.8	6.2	6.2	1.13
	9	0.794	8.1	7.1	7.6	6.7	1.09

[1] Treatments with the same letter do not differ significantly ($P > 0.05$).

Table 6.3 Faunal and floral species compositions of intensively managed, mid-rotation pine stands of Mississippi grouped by treatment according to Blocked Multi-response Permutation Procedures and samples among various years from 1999 (pre-treatment) to 2008.

				Treatment			
Community	Year	*A*	*P*-value	Burn	Herbicide	Burn + herbicide	Control
Vegetation	1999	0.00	0.543				
	2000	0.09	0.003	A[1]	B	B	A
	2001	0.08	$\leq$ 0.001	A	B	B	A
	2002	0.28	$\leq$ 0.001	A	B	B	A
	2003	0.15	0.009	ABC	C	B	A
	2004	0.15	$\leq$ 0.001	AB	BC	C	A
	2005	0.09	0.036	AB	B	B	A
	2006	0.08	$\leq$ 0.001	A	B	C	A
	2007	0.05	0.012	A	AB	B	A
	2008	0.10	$\leq$ 0.001	A	C	C	B
Birds	1999	-0.01	0.627				
	2000	0.12	$\leq$ 0.001	A	A	B	C
	2001	0.16	$\leq$ 0.001	A	A	B	C
	2002	0.18	$\leq$ 0.001	A	B	C	D
	2003	0.25	$\leq$ 0.001	A	A	B	C
	2004	0.21	$\leq$ 0.001	A	B	C	D
	2005	0.20	$\leq$ 0.001	A	A	B	C
	2006	0.25	$\leq$ 0.001	A	B	C	D
	2007	0.25	$\leq$ 0.001	A	B	C	D
	2008	0.24	$\leq$ 0.001	A	B	C	D
Amphibian	1999	0.06	0.114				
	2000	-0.02	0.709				
	2001	0.00	0.412				
	2002	-0.04	0.767				
	2003	0.01	0.335				
	2004	0.12	0.027	A	B	AB	AB
	2005	0.07	0.098				
	2006	0.01	0.350				
	2007	-0.02	0.617				

Table 6.3 (continued)

Community	Year	*A*	*P*-value	Treatment			
				Burn	Herbicide	Burn + herbicide	Control
Reptile	1999	0.00	0.439				
	2000	0.13	0.005	A	A	A	B
	2001	0.02	0.262				
	2002	0.06	0.086				
	2003	0.03	0.163				
	2004	-0.01	0.489				
	2005	-0.01	0.611				
	2006	0.02	0.216				
	2007	-0.07	0.994				
Small Mammal	1999	-0.02	0.705				
	2000	0.12	$\leq$ 0.001	A	B	C	A
	2001	0.11	0.011	A	AB	B	A
	2002	-0.01	0.586				
	2003	-0.01	0.608				
	2004	0.08	0.035	A	BC	AB	C
	2005	0.07	0.030	A	B	A	B
	2006	0.04	0.147				
	2007	0.06	0.058				
Carabid	2004	0.07	0.130				
	2005	0.01	0.256				
	2006	0.08	0.035	A	A	AB	B
	2007	0.08	0.077				

[1] Treatments with the same letter do not differ significantly ($P > 0.05$).

Table 6.4 Indicator species (IV) of vegetation communities among intensively managed, mid-rotation pine stands treated with prescribed fire and imazapyr in Mississippi, 1999-2008.

			Treatment			
Year	Species	*P*-value	Burn	Herbicide	Burn + herbicide	Control
2000	*Anisostichus capreolata*	0.014	2	1	0	72
	Aster divaricatus	0.038	8	5	51	2
	Clematis virginiana	0.035	0	0	0	50
	Cocculus carolina	0.039	4	1	58	0
	Campsis radicans	0.006	0	0	1	63
	Dichanthelium commutatum	0.002	77	1	10	9
	Erechtites hieracifolia	0.001	5	0	87	0
	Helenium autumnale	0.037	50	0	0	0
	Liquidambar styraciflua	0.032	24	1	0	49
	Oxalis stricta	0.022	12	20	54	0
	Quercus phellos	0.036	50	0	0	0
	Vicia caroliniana	0.037	48	0	1	0
2001	*Aster divaricatus*	0.001	67	1	6	15
	Berchemia scandens	0.044	7	4	7	72
	Chasmantium laxum	0.001	68	2	3	27
	Conyza canadensis	0.002	8	6	74	3
	Dichanthelium laxiflorum	0.002	13	18	57	7
	Phytolacca americana	0.026	9	70	19	0
	Rhus copallina	0.023	53	1	0	6
	Solidago gigantea	0.022	20	10	67	1
	Vicia caroliniana	0.020	58	0	1	3
2002	*Carex complenata*	0.006	0	0	65	0
	Dichanthelium commutatum	0.008	7	27	54	3
	Dichanthelium dichotomun	0.006	6	20	68	7
	Nyssa sylvatica	0.033	48	0	1	35
	Quercus stellata	0.041	50	0	0	0
	Rubus argutus	0.010	12	34	44	10
	Toxicodendron radicans	0.022	37	5	1	57
2003	*Carex complenata*	0.028	8	4	61	14
	Eupatorium serotinum	0.003	7	1	85	2
	Solidago canadensis	0.003	3	4	90	0
	Ulmas alata	0.005	0	0	79	0

Table 6.4 (continued)

Year	Species	Treatment				
		P-value	Burn	Herbicide	Burn + herbicide	Control
2004	*Dichanthelium commutatum*	0.032	7	29	54	4
	Ipomoea hederacea	0.041	8	0	49	1
	Rhus copallina	0.005	66	0	0	0
	Solidago canadensis	0.003	0	0	79	1
	Toxicodendron radicans	0.044	19	1	2	55
2005	*Oxalis stricta*	0.008	83	1	10	0
	Prunella vulgaris	0.043	50	0	0	0
	Rhus glabra	0.009	0	63	2	0
	Toxicodendron radicans	0.048	36	12	4	44
2006	*Berchemia scandens*	≤ 0.001	1	85	1	4
	Celtis laevigata	0.043	0	49	0	0
	Chasmantium laxum	0.017	18	13	57	8
	Chasmanthium sessiliflorum	0.003	21	15	63	1
	Dichanthelium commutatum	≤ 0.001	24	12	61	0
	Eupatorium atomaticum	0.042	0	0	50	0
	Ipomoeea pandurata	0.008	5	8	79	4
	Lonicera japonica	0.025	16	50	17	14
	Rhus copallina	0.005	71	0	6	1
	Solidago odora	0.015	2	0	59	0
	Tradescantia ohiensis	0.015	4	0	69	0
	Toxicodendron radicans	0.005	26	11	6	55
2007	*Andropogon virginicus*	0.007	59	1	0	1
	Dichanthelium acuminatum	0.040	0	1	44	1
	Dichanthelium commutatum	0.033	37	4	55	2
	Quercus pagodifolia	0.014	4	1	0	70
	Rhus copallina	≤ 0.001	93	0	2	0
	Solidago canadensis	0.032	22	0	63	1
	Tradescantia ohiensis	0.042	50	0	0	0
	Vaccinium ellottii	0.043	0	9	0	49

Table 6.4 (continued)

Year	Species	*P*-value	Treatment			
			Burn	Herbicide	Burn + herbicide	Control
2008	*Asplenium platyneuron*	0.045	0	50	0	0
	Chasmanthium sessiliflorum	0.037	60	5	17	1
	Dichanthelium commutatum	$\leq$ 0.001	19	11	59	7
	Dichanthelium laxiflorum	0.015	10	3	80	2
	Lespedeza virginica	0.043	0	0	50	0
	Ludwigia alternifolia	0.037	0	1	52	5
	Nyssa sylvatica	0.015	2	11	0	60
	Parthenocissus quinquefolia	0.008	13	27	47	13
	Quercus falcata	0.039	0	0	0	50
	Solanum carolinense	0.043	0	0	50	0
	Solidago rugosa	$\leq$ 0.001	14	1	74	1
	Vicia caroliniana	0.001	74	0	1	0

Table 6.5 Indicator species (IV) of bird communities among intensively managed, mid-rotation pine stands treated with prescribed fire and imazapyr in Mississippi, 1999-2008.

Year	Species	*P*-value	Treatment			
			Burn	Herbicide	Burn + herbicide	Control
2000	Common yellowthroat	0.030	15	7	12	45
	Eastern wood-peewee	≤ 0.001	3	3	66	9
	Hairy woodpecker	0.022	0	0	53	3
	Indigo bunting	0.045	29	22	35	15
	Kentucky warbler	0.032	27	1	2	49
	Mourning dove	0.039	0	0	50	0
	White-eyed vireo	0.006	23	13	10	50
2001	Common yellowthroat	0.011	8	14	54	4
	Eastern wood-peewee	0.039	1	32	46	4
	Gray catbird	0.004	0	1	68	2
	Indigo bunting	≤ 0.001	22	25	42	10
	White-eyed vireo	0.004	26	12	8	47
2002	Common yellowthroat	0.030	6	7	52	14
	Downy woodpecker	0.007	17	8	46	21
	Eastern towhee	0.017	19	28	36	17
	Eastern wood-peewee	≤ 0.001	8	3	75	2
	Indigo bunting	0.029	16	31	40	13
	Red-headed woodpecker	0.029	6	13	48	10
	White-eyed vireo	0.008	27	21	11	39
	Yellow-throated woodpecker	0.012	2	2	48	2
2003	Common yellowthroat	≤ 0.001	16	18	51	12
	Eastern towhee	0.006	23	27	33	17
	Eastern wood-peewee	≤ 0.001	6	9	68	5
	Hooded warbler	0.005	25	25	5	44
	Indigo bunting	0.005	22	29	40	9
	Yellow-breasted chat	0.011	23	32	26	19
	Yellow-throated warbler	0.027	0	3	53	0
2004	Common yellowthroat	0.006	16	11	63	2
	Eastern wood-peewee	≤ 0.001	26	1	66	0
	Hooded warbler	≤ 0.001	19	27	13	41
	Indigo bunting	0.016	27	26	39	8
	Kentucky warbler	0.003	17	40	11	29
	Pine warbler	0.020	23	23	36	18
	Wood thrush	0.009	18	23	5	44

Table 6.5 (continued)

Year	Species	*P*-value	Treatment			
			Burn	Herbicide	Burn + herbicide	Control
2005	Common yellowthroat	$\leq$ 0.001	17	2	73	0
	Eastern wood-peewee	$\leq$ 0.001	23	1	64	0
	Hooded warbler	0.018	20	37	11	32
	Indigo bunting	0.008	25	27	43	3
	Northern cardinal	0.043	25	22	20	32
	Red-bellied woodpecker	0.048	31	7	41	10
	Wood thrush	0.019	9	26	1	53
	Yellow-breasted chat	0.005	24	27	36	14
2006	Carolina wren	0.016	26	33	20	22
	Common yellowthroat	0.020	20	8	49	2
	Eastern wood-peewee	0.002	22	0	61	0
	Hooded warbler	0.008	14	35	8	43
	Indigo bunting	0.019	34	17	45	3
	White-eyed vireo	0.002	13	47	10	30
2007	Hooded warbler	0.034	11	39	13	37
	Indigo bunting	0.002	25	16	55	2
	Kentucky warbler	0.012	27	44	9	13
	White-eyed vireo	$\leq$ 0.001	20	42	14	21
	Yellow-breasted chat	0.017	23	26	45	4
2008	Hooded warbler	0.034	13	38	11	37
	Indigo bunting	0.005	28	9	52	1
	Northern cardinal	0.024	22	31	12	36
	White-eyed vireo	0.002	21	44	19	16

Table 6.6 Indicator species (IV) of amphibian, small mammal, and carabid communities among intensively managed, mid-rotation pine stands treated with prescribed fire and imazapyr in Mississippi, 1999-2008.

			Treatment			
Year	Species	*P*-value	Burn	Burn + herbicide	Control	Herbicide
2000	Golden mouse	0.0444	35	5	45	7
2001	*Peromyscus* spp.	0.0138	14	47	17	20
	Golden mouse	0.0120	19	2	51	11
2004	Eastern narrow-mouth toad	0.046	44	19	22	15
2006	*Cyclotrachelus convivus*	0.0320	15	16	47	22

Table 6.7 Codes and descriptions of indicator species used in ordination of faunal and floral communities among treatments by year in intensively managed, mid-rotation pine stands treated with prescribed fire and imazapyr in Kemper County, Mississippi, 1999-2008. Variables with eigenvector loadings ≥ |0.3| were deemed significant for interpretation.

Code	Description	Significant
COYE	Relative abundance of common yellowthroats	X
EATO	Relative abundance of eastern towhees	X
EAWP	Relative abundance of eastern wood-pewees	X
HOWA	Relative abundance of hooded warblers	X
INBU	Relative abundance of indigo buntings	X
KEWA	Relative abundance of Kentucky warblers	X
NOCA	Relative abundance of northern cardinals	X
WEVI	Relative abundance of white-eyed vireos	X
WOTH	Relative abundance of wood thrushes	X
YBCH	Relative abundance of yellow-breasted chats	X
YTWA	Relative abundance of yellow-throated warblers	X
ASD	Biomass (kg/ha) of *Aster dumosus*	X
BS	Biomass (kg/ha) of *Berchemia scandens*	
CACO	Biomass (kg/ha) of *Carex complenata*	X
CHLAX	Biomass (kg/ha) of *Chasmanthium laxum*	X
CHS	Biomass (kg/ha) of *Chasmanthium sessiliflorum*	X
DICOM	Biomass (kg/ha) of *Dichanthelium commutatum*	X
DILAX	Biomass (kg/ha) of *Dichanthelium laxiflorum*	X
NS	Biomass (kg/ha) of *Nyssa sylvatica*	
OXS	Biomass (kg/ha) of *Oxalis stricta*	
RC	Biomass (kg/ha) of *Rhus copallina*	
SOLC	Biomass (kg/ha) of *Solidago canadensis*	X
TOXR	Biomass (kg/ha) of *Toxicodendron radicans*	X
VC	Biomass (kg/ha) of *Vicia caroliniana*	X
GOMO	Relative abundance of golden mice	X
B1[1]	Burn only treatment, year 1	
H1[1]	Herbicide only treatment, year 1	
BH1[1]	Burn + herbicide treatment, year 1	
C1[1]	Control, year 1	

[1] Example of notation used to designate treatments among years.

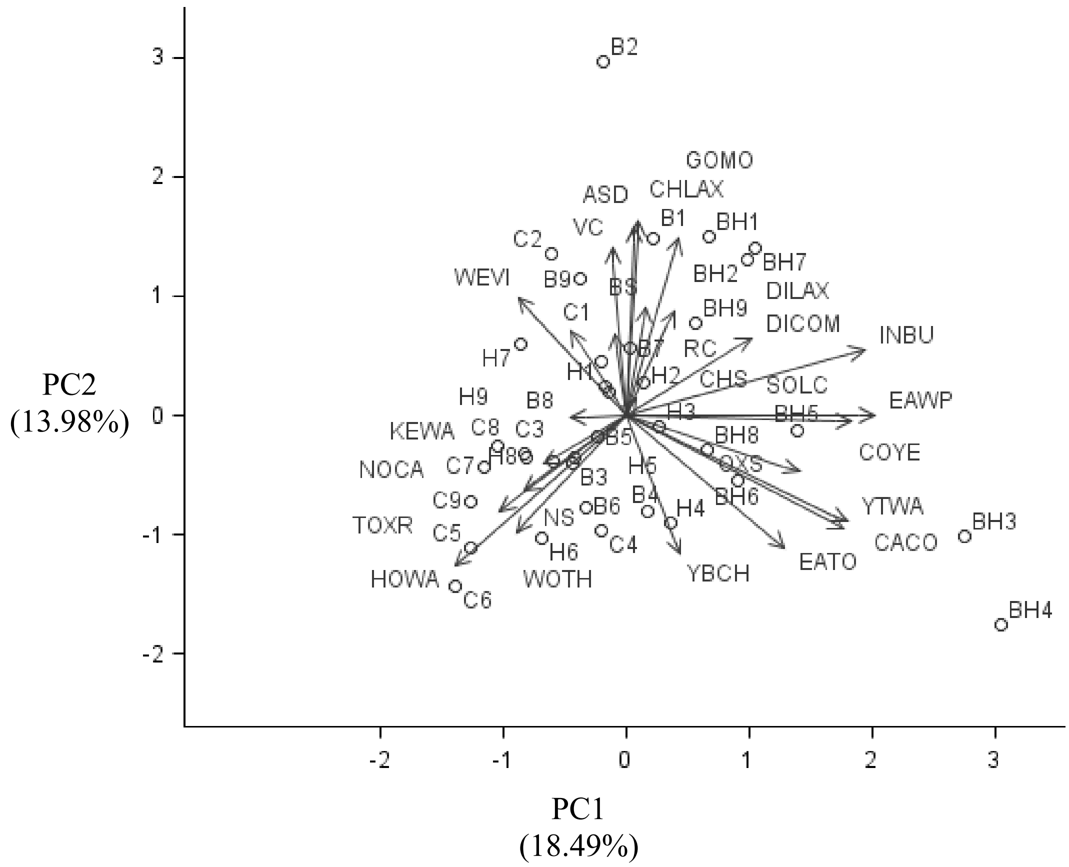

Figure 6.1 First 2 axes (percentage variance explained) of ordination using principal components analysis of indicator species and treatments by year from intensively managed, mid-rotation pine stands treated with prescribed fire and imazapyr in east-central Mississippi, 2000-2008.

CHAPTER 7

SYNTHESIS AND MANAGEMENT RECOMMENDATIONS

Prescribed fire and imazapyr are 2 silviculture tools able to reduce hardwood midstory competition and enhance biodiversity in intensively managed, mid-rotation pine stands of Mississippi. Combining treatments of dormant season prescribed fire with a 3 year fire return interval and 12 oz. Arsenal/ ac via skidder results in the greatest impact on wildlife communities, specifically vegetation and bird species compositions. Independent treatment applications offer an alternative at less cost. Applied at the 4 levels described here (burn only, herbicide only, burn + herbicide, no treatment), these treatments can promote vegetation type heterogeneity among mid-rotation pine stands improving biodiversity conservation efforts and contributing to sustainable forestry.

As with most wildlife management tools, dormant season prescribed fire and imazapyr are limited in their capabilities. Managers should be aware that imazapyr has immediate, noticeable detrimental impacts on woody plants whereas repeated burns are necessary for equivalent control. However, repeated burns eventually reduce midstory hardwood competition to levels similar to imazapyr while perpetuating increased herbaceous coverage of high-quality white-tailed deer forages (*Odocoileus virginianus*; Iglay et al. 2010*b*). Prescribed burns also enhance plant species richness more than imazapyr and remain the single best management tool for restoring pyric ecosystems (Brennan et al. 1998). Imazapyr's selectivity also limits its ability to promote plant diversity (Iglay et al. 2010*a*). Managers should consider combining imazapyr applications with prescribed fire or other herbicides to increase potential species richness by reducing plant species dominance post-treatment (Iglay et al. 2010*a*).

Vegetation structure resulting from burn + herbicide has promise for outstanding contributions to conservation of avian species in the southeastern United States. By creating open forest, two-tier structure (e.g., overstory and understory) within intensively managed, mid-rotation pine, prescribed fire and imazapyr increase total avian conservation value of commercial forests providing habitat preferred by high priority bird species (e.g., Bachman's sparrow, brown-headed nuthatch, northern bobwhite). Larger scale applications of this treatment (i.e., stand-level versus 10-ha experimental unit) may attract greater numbers of individuals and help prolong wildlife benefits of other forestry practices (e.g., thinning) while avoiding potential creation of ecological traps. Independent treatments also increased avian conservation value but did not attract a suite of high priority species, only greater relative abundances of common species. Compared to controls, any level (combined or independent) of prescribed fire or imazapyr increased avian conservation value.

Other wildlife communities had minimal responses to treatments. Each community (e.g., small mammal, herpetofauna, and carabid beetles) may have been influenced by larger scale disturbances and landscape characteristics than local treatment-based alterations (e.g., ephemeral pools, untreated buffer areas, stream-side management zones, and typical timber management practices of site preparation, herbaceous release, thinning, and fertilizing). Although responses to treatments were scarce, managers and researchers should consider potential impacts on these communities by wildlife management practices according to natural history traits of these communities such as preferences for mesic areas or benefits of coarse woody debris. Some of these communities also may be resilient to such changes increasing difficulties of developing effective research sampling design approaches when space and time are limiting factors.

Future research regarding prescribed fire and imazapyr in intensively managed, mid-rotation pine has multiple topics to consider. Multiple-herbicide tank mixtures with or without prescribed fire may offer a variety of benefits towards conservation of biodiversity with variable impacting plant species composition through expanding controlled species lists and incorporating management concerns regarding plant dominance post-treatment. Indicator species determined in this study could be used as focus species to better understand population responses to treatments. Although lists generated from indicator species analysis are not significant for future monitoring programs, investigating population dynamics of indicator species may emphasize long-term sustainability of avian communities within stands of intensively management pine. Larger-scale treatment applications and concomitant comparisons among treated sites, stream-side management zones, and other buffer areas may increase our understanding of commercial forests' contributions to diversity. Lastly, research investigating various spatial scales and filter levels may reveal mechanisms of change for species with minimal responses to treatments (Hunter 2004).

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