EFFECTS OF WILDLIFE STAND IMPROVEMENTS AND PRESCRIBED BURNING ON BAT AND INSECT COMMUNITIES: BUFFALO RANGER DISTRICT, OZARK-ST. FRANCIS NATIONAL FOREST, ARKANSAS

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Jeremy L. Jackson

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AN ABSTRACT

OF

EFFECTS OF WILDLIFE STAND IMPROVEMENTS AND PRESCRIBED BURNING ON BAT AND INSECT COMMUNITIES; BUFFALO RANGER DISTRICT, OZARK-ST. FRANCIS NATIONAL FOREST, ARKANSAS

Jeremy L. Jackson

Beginning in 2001 the managers of the Buffalo Ranger District (BRD) in the Ozark-St. Francis National Forest (ONF), began an alteration of a watershed by implementing wildlife stand improvements (WSI) and prescribed burning (PB). This strategy was adopted in an attempt to restore an oak woodland cover type. The effects of these forest management techniques were evaluated at two treatment and two reference sites. A WSI and PB were administered in the fall of 2001 and spring of 2002 respectively at treatment-site T1. Similar forest management techniques were administered at treatment-site T2 in 1999. Two reference areas were selected that had not been subjected to PB, WSI, or timber harvest. Within each site forest stand density (FSD) was estimated by calculating basal area in selected plots. Bat populations were sampled by mist nets in each of the 4 sites during between May and September of 2001 and 2002. Bat species diversity and abundance were correlated with FSD. No difference was found between these treatments and the abundance and diversity of bats. However, significant correlations were observed when abundance and diversity were compared to FSD. Such that abundance and diversity were greatest when the FSD was less than 9m^2/ha and declined as FSD increased to 20m^2/ha. These data suggest that WSI and PB are beneficial to bat communities in portions of the forest where FSD exceeds 14-16 m^2/ha.

Various aspects of the roosting ecology of female northern long-eared bats (*Myotis septentrionalis*) were evaluated in respect to treated and untreated sites and changes in the relative forest stand density. Transmitters were placed on 33 individuals and 259 roosts were located. My results indicate that these forest management strategies had no impact on bats in treated regions of the BRD. Moreover, these data suggested that bats continued and/or began to use areas where the basal area had been reduced as a result of this management strategy. Roost density and foraging distances were negatively correlated with forest stand densities indicating that bats utilized areas of the forest that were less structurally complex.

The diversity and abundance of insect orders was evaluated in relation to the treatment being administered and was correlated with changes in structural complexity of the forest. Insect assemblages peaked when the forest stand density (FSD) had a basal area of 9m^2/ha and declined as FSD increased to basal areas of 18m^2/ha. These data suggest that forest management strategies incorporating WSI and PB are beneficial to insect communities in portions of the forest that have a FSD with a basal area greater than 14-16 m^2/ha.

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CHAPTER 1

GENERAL INTRODUCTION

Structural complexity can be defined as the relative horizontal and vertical density of the forest. Areas within the forest that are more structurally complex have been reported to contain more obstacles a flying bat must detect and avoid (Fenton 1990). These obstacles, including dense stratification layers, can impact the abundance and distribution of bats in a forest by imposing physical constraints on movement (Brown 1991). In addition, stratification may create scattered patterns of resource distribution, alter microclimates (Bradshaw 1996), impede access to prey and water, fragment the distribution of roost sites, alter the exposure to predators, and influence the potential for competition (Krusic and Neefus 1996). The extent to which these physical constraints influence animal communities have been shown to vary as a function of habitat structural complexity (Brown 1991). Some species of bats appear to avoid areas with structural obstacles and preferentially forage in structurally less complex habitats (Grindal and Brigham 1998; Mackey and Barclay 1989; Pickett and White 1985). Openings within the forest have greater bat activity, and greater number of species are often found in open areas compared to relatively more dense areas (Krusic and Neefus 1996). Vertical differences in bat activities suggests that stratification layers could be viewed as edge habitats where bats benefit from the open space between vertical and horizontal planes within forested areas (Bradshaw 1996). This implies that vertical stratification in the forest may be an exploitable feature allowing bats to benefit from short vertical shifts among different horizontal layers.

Bats ability to utilize various stratification layers in the forest could be a result of wing morphology, which may be an important factor in defining exploitable niches and resource utilization by insectivorous bats (Neuweiler 1984). Bat species with short, broad wings (resulting in low wing loading and low aspect ratios) are adapted to avoiding obstacles and exhibit high degrees of maneuverability (Norberg 1981); therefore, bat species possessing such wings are likely to be adapted for foraging in the mid- and understory. Alternatively, long, narrow wings have higher aspect ratios and wing loading, and are probably adapted for foraging above the canopy or in open areas within the forest where speed and endurance are needed (Norberg and Rayner 1987). Small, interspecific differences in wing morphology could have important implications for maneuverability and agility that may be reflected in interspecific microhabitat associations (Sauders and Barclay 1992). Bradshaw, (1996) investigated the ratio of ear length to body mass, wing loading, and wing aspect ratio to determine if there were correlation's between stratification layers in which bats foraged and wing morphology. The statistical significance of correlations was not reported; however, he found that relatively more open forested habitats harbored more bat activity. Therefore, bat species adapted to forage in the mid- and understory could be detrimentally affected by an increase in forest complexity, probably because the increased structural complexity limits access to prey and water. In a less dense forest bats should be able to maneuver more efficiently which could allow greater access to alternative foraging areas and suitable roost sites.

It has been suggested that suitable roost tree availability may be limiting and the primary factor in habitat selection for bats (Kunz 1982; Humphery 1975). Roosts are a crucial resource in providing sustainable populations of forest dwelling bats, because of the amount of time allocated to roosting (>50%/day) (Brigham et al. 1997). For this reason, functionality and selectivity of roosts has received much attention; however, prior to the 1990's most bat roost data was anecdotal, derived mostly from general observations of a single tree roost, artificial structures, mines, or caves (e.g., Kurta et al. 1993a, 1993b; Parsons et al. 1986; Barclay and Cash 1985). Therefore, it is likely that a substantial number of management decisions have been based on available anecdotal evidence (Hays 2003).

Few studies have statistically addressed specific characteristics of roost trees (e.g., Barclay et al. 1988; Lunney et al. 1988; Taylor and Savva 1988) prior to the 1990's. However, technological advances allowed for a greater number of detailed bat roost studies (e.g.*,* Sasse and Pekins 1996; Crampton 1994; Crampton and Barclay 1985; Kalcounis 1994; Vonhof 1996). Using radio telemetry, probably the most important of these advances was the development of radio transmitters with a mass of less than 0.45 g (Aldridge and Brigham 1988). This is less than 5% of the body weights of most bats. This made it possible for researchers to track bats with minimal disturbance to the tagged individual (Aldridge and Brigham 1988). Although these transmitters supported more detailed studies of bat behavior, there have been few reports about the habitat requirements of forest-roosting bats (e.g., Menzel et al. 1998; Fenton 1997).

Most of these detailed studies have focused on evaluating the micro-habitat (Mager and Nelson 2001; Foster and Kurta 1999; Kalcounis and Hecker 1996; Kurta et al. 1993a; Barclay et al. 1988; Barclay 1982), food availability (Chung-

MacCoubrey 1996), behavior (Perkins 1996), predation (Kalcounis and Brigham 1994), and other general notes (e.g., Parsons et al. 1986; Cash and Barclay 1985) of forest roosting bats. These studies frequently included qualitative and quantitative measures of the characteristics of the roost cavity and/or crevice (i.e., roost tree species, temperature, and other climatic regimes within bat roost). Microhabitat usually included descriptions of the area surrounding an individual roost *i.e*., diameter at breast height, percent canopy, mid-story, understory, ground cover, and forest density of areas ≥ 0.1 ha around the roost or capture location (e.g., Menzel et al. 2001; Foster and Kurta 1999; Crampton and Barclay 1998; Menzel et al. 1998; Vonhof 1996; Sasse and Pekins 1996; Brigham 1991).

Bats select trees based on the relative diameter and height of these trees (Brigham et al. 1997, Betts 1996; and Lewis 1995). Indiana bats (*Myotis sodalis*) preferred roost trees ranging from 9.4 cm to 86 cm (mean of 33.5) diameter at breast height (DBH) (MacGregor et al. 1998). Northern long-eared bats (*Myotis septentrionalis*) also select trees within this size range (Sasse and Pekins 1996). In addition, brown bats (*Eptesicus fuscus)* and silver-haired bats (*Lasionycteris noctivagans*) also selected large-diameter (Betts 1996).

Bats are found to often utilize snags, (Kalcounis and Hecker 1996), which are dead and/or dying trees, lacking the upper canopy limbs. These trees often possess cavities and/or exfoliating bark, which provide suitable bat roost (Kalcounis and Hecker 1996). However, live trees are also utilized as roosts if they possess similar characteristics including the presence of suitable cavities and/or exfoliating bark (Kalcounis and Hecker 1996). However, due the make up of a snag, *i.e*., dead and/or decaying wood resulting from natural tree mortality, snags possess these characteristics more often than live trees. A considerable amount of intraspecific variation in little brown bat roost-site selection occurs with bats roosting in both live and dead trees (Kalcounis and Hecker 1996). Dead trees are used to a greater extent than live trees, because dead trees absorb and release heat more readily (Humphrey et al. 1977). Regardless of bats geographic ranges, several compelling functionalities of roost site selection by several temperate forests dwelling bat species. These factors are correlated with the presence of cavities and/or exfoliating bark, (Perkins 1996), relative tree size (Brigham et al. 1997; Betts 1996; Lewis 1995), and relative forest density surrounding the roost (Vonhof 1996; Sasse and Pekins 1996; Gardner 1990).

The microhabitat within an individual bat roost potentially could be the net result of habitat conditions within the area (≥ 0.1 ha) surrounding the roost (Callahan et al. 1997). Microhabitat of a forest stand is potentially defined by its structural complexity, as well as by its position within the larger landscape (i.e., watershed, hillside, ridgeline, etc.). Considering the number of temperate bat species, there have been few studies that have documented the microhabitat. However, some aspects of the microhabitat have been documented and suggest that several tree roosting temperate forest dwelling bat species prefer roost trees that are relatively distant from surrounding trees (Betts 1996; Sasse and Pekins 1996; Vonhof 1996). These observations support the hypothesis that bats tend to select areas within the forest that are relatively less structurally complex.

Because of the importance of the availability of snags and the wide range of tree height and DBH that is needed for roosts, as well as ample access to food and water, forest managers have set guidelines for snag management. In the Buffalo Ranger District (BRD), Ozark-St. Francis National Forest (ONF) these guidelines have evolved to include leaving on average five to ten snags per 0.4 ha to support bat communities and other cavity utilizing wildlife species.

The BRD in the ONF, Arkansas is located in the Boston Mountains of north central Arkansas, currently the BRD comprises approximately 1,351,348 ha of which 168,737 ha are designated wilderness areas. Dominant overstory species are deciduous hardwoods including oaks (*Quercus* spp.) and hickories (*Carya* spp.), with conifers scattered throughout the district. Small man-made wildlife openings (average size approximately 0.4 ha) exist throughout the BRD at a density of 0.002 wildlife openings/ha. The district also contains various water resources, such as man-made and natural ponds (0.003 ponds/ha), as well as several streams of various sizes.

There have been high numbers of bats captured, comprised of several species throughout the ONF (Wilhide et al. 1998). Although, capture rate is not a direct measure of abundance and diversity, this area seems to provide suitable habitat for bats, based upon capture success rate from previous studies. However, the reduction in access to resources associated with the relatively high vegetative density of some areas of the forest may be limiting the abundance and diversity of bats within their community.

Beginning in 2001 managers of the BRD began an alteration of a watershed located on the northwestern portion of the district using wildlife stand improvements (WSI) and prescribed burning (PB). Wildlife stand improvements occurring in the BRD are designed to aid restore an area to its historical pre-settlement condition of an oak woodland. These two management techniques are designed to produce reductions in forest stand density (FSD). To determine the efficacy of these techniques, FSD was estimated by “basal area”. Basal area is the cross-sectional area of all the trees at breast height per hectare and is expressed in square meters per hectare. Guidelines established for WSI in the BRD are that the midstory will be reduced to a basal area of 7-9 m^2/ha while leaving potential den trees, including live hollow trees and dead snags. Once these forest management techniques are completed, managers expect that several benefits should occur including a reduction in competition for moisture and soil nutrients among trees and an increase in sunlight to the forest floor.

According to the guidelines established by BRD managers, prescribed burning is conducted after midstory reduction to a basal area of 7-9 m^2/ha. PB is implemented on a three- to five-year rotation. Fire temperature used for the PB is regulated in an attempt to insure that the fire is hot enough to remove some of the litter layer without causing significant damage to the trees that are to remain. PB is used to stimulate growth of herbaceous plants by removing dead and decaying vegetation, so that more of the forest floor is exposed to sunlight. This PB management strategy of burning after a cutting alters the nature of plant succession (UHL et al. 1981) by reducing tree regeneration within burned areas, thereby maintaining an oak woodland habitat type (US Forest Service). Moreover, it aids in the reduction of the high forest stand densities that have resulted from fire exclusion (Kaufmann et al. 2003). Moreover, utilizing this management approach, the mid- and understory layers were significantly lower in post-burn areas, forest stand densities and basal area were also reduced (Katherine et al. 1999). Therefore, dramatic net increases in the density and diversity of herbaceous plants including grasses and forbs have been reported after a prescribed burn (Christensen 1981; Cushwa et al. 1966; Lemon 1946; Heyward 1937). Tree removal prior to PB at various landscape scales also creates gaps in the dominant canopy (Poulson and Platt 1989). A net increase in herbaceous vegetation resulting from relatively more open forested habitats that have been burned could have a direct positive impact on the insect community.

Insects are important in maintaining the health and diversity of forest ecosystems (Andrewartha and Birch 1984). Increases in the diversity of insects are correlated with plant diversity and plant structural complexity (Cornell 1986; Lawton 1978; Lawton and Schroder 1977; Murdoch et al. 1972). Phytophagous insects often attain their highest densities where food occurs in large patches (MacGarvin 1982; Denno et al. 1981; Thompson 1978). Thus, small openings in the forest support higher insect species abundance and richness than interior portions of the forest where structural complexity is greater (Gorham et al. 2002). Moreover, the abundance and diversity of insect communities are attributed to plant biomass (Hunter and Wilmer 1989; Hunter 1987; Teragushi et al. 1981). Specifically, the abundance of herbaceous plant species and/or biomass is the main factor in outbreaks of some species of Lepidoptera, especial moths and butterflies (Dempster 1983; Thompson and Price 1977; White 1974). Butterflies are often associated with open habitats and may be indicators of habitat quality (Kocher and Williams 2000).

Although there are no reports of research conducted in relation to the effects of WSI and PB on bat communities, work has been conducted on other animal communities. Substantial increases in food supply and cover for small mammals are associated with prescribed burn areas (Williams 1955). Rodent populations subsequently increase due to increases in resource availability (Williams 1955). Positive responses by small mammals are often associated with areas that are burned and with relative abundance of vegetation (Ford et al. 1999). In addition, several bird species are associated with burned areas, due to increases in available resources (Larwrence 1966).

The combination of WSI and PB management practices is expected to reduce the structural complexity of portions of the BRD by reducing amount of woody trees, shrubs, and vines resulting in an increase in herbaceous ground cover. By the selective removal of trees, WSI can reduce the midstory, while PB removes some slash and litter on the forest floor. This type of forest management leaves mature trees to provide potential roosts for bat species that prefer older, larger diameter trees. Together these management strategies may have a positive effect on bat populations by creating a more suitable horizontal and vertical stratification layers, and by creating flyways and foraging areas through which bats maneuver more efficiently. Although there are not any reports describing how WSI and PB affect bat populations, the end result is that these management practices may be to allow bats to have greater access to available food, water and roost resources.

The research reported in this document focuses on answering one basic question, *i.e.*, what is the effect of WSI and PB on bat communities in the BRD of northwestern Arkansas? These two management techniques are expected to have an impact on the overall structure of the forest in relation the stratification layers and structural complexity of portions of the forest where it has been administered. Effects of FSD on bat communities was investigated by assessing three ecological factors that could potentially have an effect on bat communities in the forest. These include: (1) the abundance and diversity of bats, (2) roosting ecology of female northern long-eared bats (*Myotis septentrionalis*), (3) insect diversity and abundance (insect availability). Sampling using mist-nets was used to estimate the abundance and diversity of bats; radio telemetry was used to evaluate the roosting ecology of female northern long-eared bats, and insect light traps were used to sample for nocturnal flying insects. Preexisting FSD data from the US Forest Service and direct measurement of FSD were used to characterize areas where bats and insects were sampled and at roost sites. The abundance and diversity of bats, insects, and the roosting ecology of female northern long-eared bat was correlated with the presence or absence of WSI and PB treatments and with FSD.

CHAPTER 2

EFFECTS OF WILDLIFE STAND IMPROVEMENTS AND PRESCRIBED BURNING ON BAT COMMUNITIES

ABSTRACT

Beginning in 2001 the managers of the Buffalo Ranger District (BRD) in the Ozark-St. Francis National Forest (ONF) began an alteration of a watershed by implementing wildlife stand improvements (WSI) and prescribed burning (PB). This strategy was adopted in an attempt to restore an oak woodland cover type. The effects of these forest management techniques were evaluated at two treatment and two reference sites. A WSI and PB were administered in the fall of 2001 and spring of 2002 respectively at treatment-site T1. Similar forest management techniques were administered at treatment-site T2 in 1999. Two reference areas were selected that had not been subjected to PB, WSI, or timber harvest. Within each site forest stand density (FSD) was estimated by calculating basal area in selected plots. Bat populations were sampled by mist nets in each site during between May and September of 2001 and 2002. Bat species diversity and abundance were correlated with FSD. No difference was found between these treatments and the abundance and diversity of bats. However, significant correlations were observed when abundance and diversity were compared to FSD. Such that abundance and diversity were greatest when the FSD was less than $9m^2$/ha and declined as FSD increased to $20m^2$/ha.

INTRODUCTION

Structural Complexity

The structural complexity of a forest has been defined as the relative density (measured separately and/or together) of its vertical and horizontal strata (e.g., Heck and Wetstone 1977; Wilson 1974; Kohn 1967). Species diversity increases as the habitat increases in complexity because areas that are more complex provide a greater number of suitable niches (Klopfer and MacArthur 1960). However, although species diversity is associated with habitat complexity (Mac Arthur et al. 1966) and complex habitats should be characterized by greater species diversity (Levins 1968; Mac Authur et al. 1962), forested habitats that are more structurally complex also have less available open space. Open space is a vital resource for flying animals, including bats. Furthermore, North American insectivorous bats depend on echolocation to navigate and forage. Therefore, areas within the forest that are more dense may contain more obstacles that a bat must detect and avoid while flying (Fenton 1990). Thus, high structural complexity might have a detrimental effect on a bats ability to maneuver through the forest.

Echolocation and wing morphology of specific bats likely reflects the habitat selected by that species. Echolocation and wing morphology exert significant energetic and mechanical limitations on where bats can forage efficiently (Aldridge and Ruatenbach 1987). According to the Optimal Foraging Theory suggesting that encounter rate divided by time is an effective estimate of prey search effort

Schoener 1971). Under conditions of constant prey density, the prey encounter rate would increase as flight speed increases (Norberg 1981). In areas of the forest where structural complexity limits maneuverability, bats would be forced to compensate by reducing their flight speed. This reduced flight speed could negatively impact an individual's search effort. Therefore, assuming equal insect availability (Zinn and Humphrey 1981), encounter rate would be decreased by the reduction in maneuverability associated with high FSD. Thus, high structural density might have the effect of increasing energy expenditures during foraging activities. For these reasons, bats that are adapted to foraging in the mid- and understory may select areas within the forest that are relatively more open. For example, bat species, such as the little brown bat (*Myotis lucifugus*) and western long-eared bat (*M. evotis*), that are forage specialist (Barclay 1991) could reduce prey search effort by utilizing more open habitats where flight speed and maneuverability would be maximized. Since wing morphology and echolocation abilities are species-specific, each bat species is likely to be impacted differently by the amount of space available as they maneuver, forage, and move within and among habitats.

Short, broad wings have low wing loading, and low aspect ratios, and are adapted to avoiding obstacles. Bats with this type of wing are highly maneuverable and are probably adapted for foraging in the mid- and understory. Long narrow wings have higher aspect ratios and wing loading. Bat species with long, narrow wings are probably adapted to forage above the canopy or in open areas within the forest where speed and endurance are needed (Norberg and Rayner 1987). Thus, differences in wing morphologies represent important adaptive mechanisms that would have important implications in determining the specific habitats with which a bat species would be associated (Sauders and Barclay 1992). In dense forest, bats tend to utilize roads, trails, and open fields as foraging areas and flight corridors (Krusic and Neefus 1996). Forest bats have been reported to preferentially select open habitat types when available (Bradshaw 1996). This implies that bat species adapted to forage in the mid- and understory could be detrimentally affected by increases in forest complexity that make access to available prey and water resources more difficult.

Forest Management

Beginning in 2001, managers of the BRD within the ONF in northwestern Arkansas (Figure 2.1) began an alteration of a watershed (Figure 2.2), by implementing wildlife stand improvements (WSI) and prescribed burning (PB) (Figure 2.3 and 2.4). Forest management used in the BRD is designed to aid in the restoration of the treated area to its pre-settlement, historical condition (i.e., an oak woodland). These strategies are also intended to reduce FSD. In this study, measurements of basal area were used to estimate FSD. “Basal area” is the cross-sectional area at breast height of all trees per hectare and is expressed as m^2/ha. Before WSI and PB were applied, treated sites had a basal area of approximately 18-21 m^2/ha (US Forest Service). Guidelines established for WSI on the BRD specified that the midstory would be reduced from a basal area of 18-21 m^2/ha to a 7-9 m^2/ha by removing selected trees. After target basal area had been achieved, PB was used to assist development of desired habitat type. To maintain this habitat, PB is scheduled to continue on a three to five-year rotation.

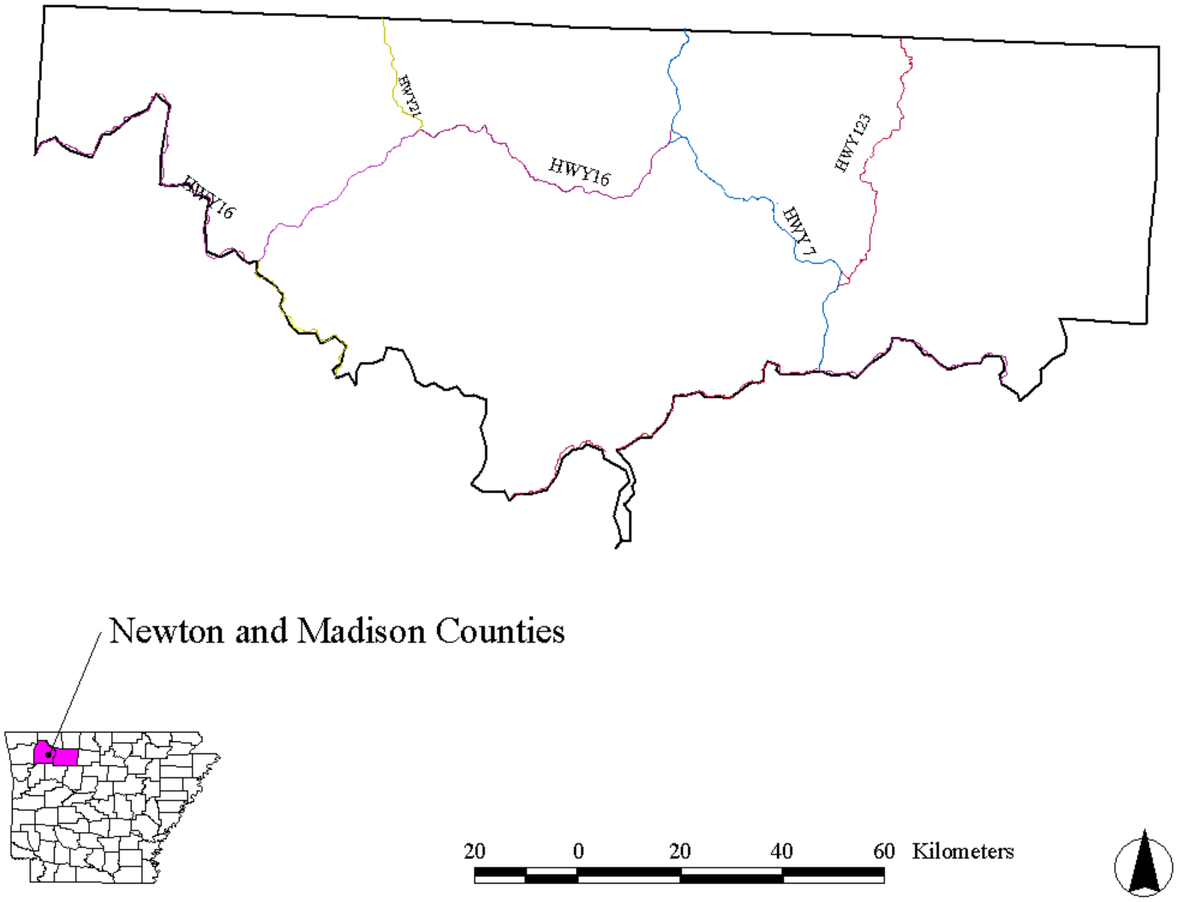

Figure 2.1. The location of the Buffalo Ranger District, Ozark-St. Francis National Forest, Arkansas.

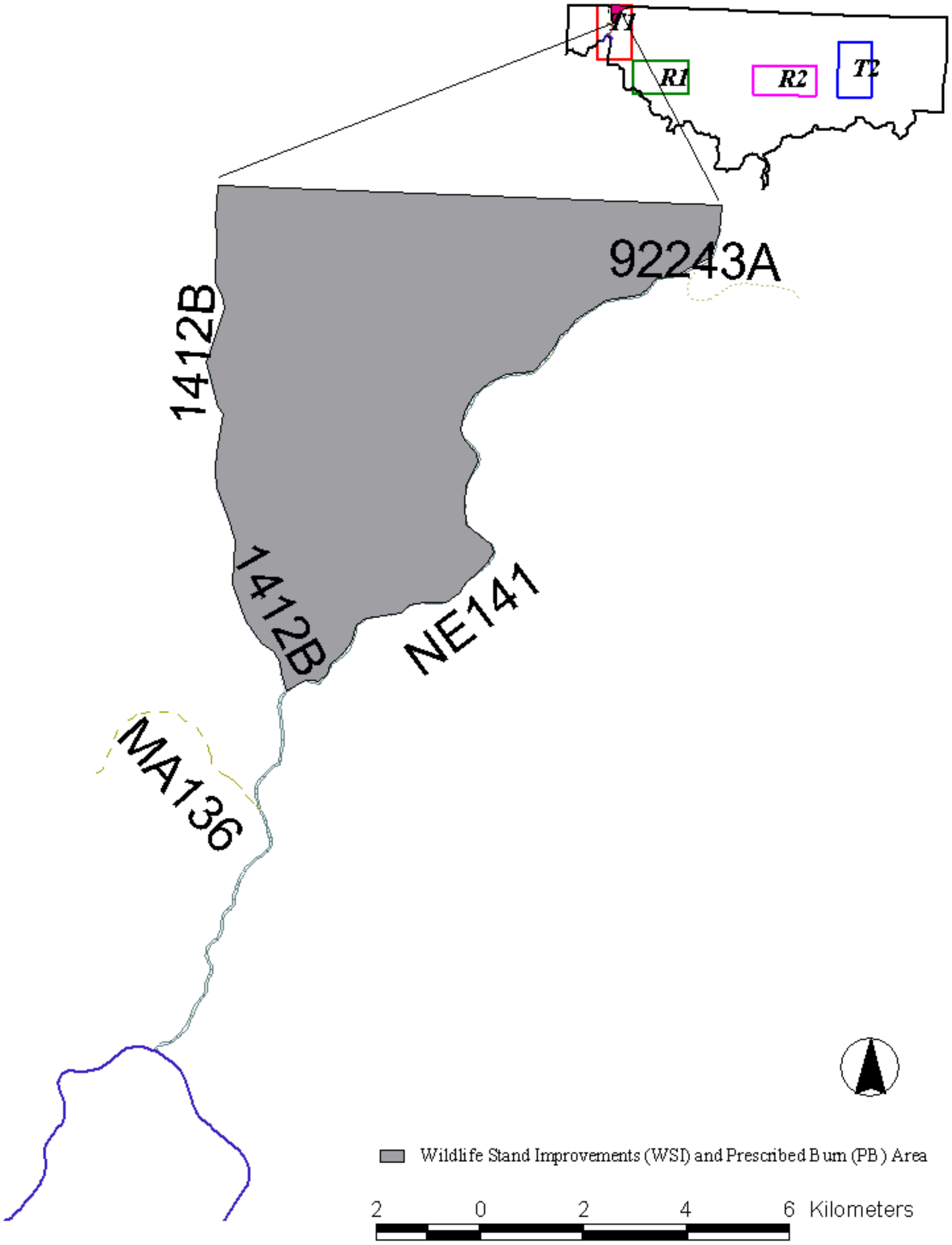

Figure 2.2. The area treated with wildlife stand improvements and prescribed burning.

Figure 2.3. Representative habitat in the Redstar site (T1) before WSI and PB were administered (23 July 2001).

Figure 2.4. Representative habitat in the Redstar site (T1) after WSI and PB were administered (15 May 2002).

Together, these management strategies may have a positive effect on bat populations by creating more open space within the horizontal and vertical stratification layers. This would create flyways and foraging areas for bats enhancing their maneuverability and allowing greater access to available food, water and roost resources and potentially reducing bats energy expenditures. Therefore, bat species abundance and diversity were predicted to be greater within and among treatment sites. Forest stand density was predicted to be relatively lower within treatment sites. Additionally, bat species abundance and diversity were predicted to be associated with forest stand density, such that as FSD decreases, the abundance and diversity of bats increase.

Purpose of the Study

The purposes of this study were to: 1. characterize the effects of WSI and PB on bat species abundance and diversity, and 2. assess the impact of changes in FSD on bat species abundance and diversity within the Buffalo Ranger District, Ozark National Forest.

METHODS AND MATERIALS

Study Area

This study was conducted on the BRD in the ONF, in the Boston Mountains of northwest Arkansas (Figure 2.1). The BRD comprises approximately 1,351,348 ha of which 168,737 ha are designated wilderness areas. Dominant overstory species are deciduous hardwoods, including oaks (*Quercus* spp.) and hickories (*Carya* spp.), with conifers (*Pinus* spp.) distributed throughout the district. The age of the forest within the

BRD rages from 0 to > 100 years (from fields and 1st year regeneration stands to mature old growth forest stands). Small wildlife openings (n = 231) averaging approximately 0.4 ha in size exist throughout the BRD at an average density of (US Forest Service). The district contains many man-made and natural ponds (n = 293) at an average density of 1 pond/50 km^2 (US Forest Service). Previous bat capture data (Wilhide et. al. 1998) in the OSNF may suggest that the current forest structure in some areas provide suitable habitat for bats. However, based on echomorphology and wing morphology, the relative high vegetative (structural complexity) density of some areas of the forest may be a limiting factor on the size of the bat communities by restricting access to available resources.

Experimental Design

To assess the effects WSI and PB on bat abundance and diversity four study sites with in the BRD were established (Figure 2.5): Redstar (T1), Fallsville (R1), Lurton (T2), and Cowell (R2). Each site was approximately 60,000 ha in size and each had similar vegetative composition (mixture of hardwoods and conifers). The age of the forest at each study site ranged from 0 to > 100 years old. Ponds less than 0.5 ha in size are distributed throughout each of the study sites at an average density of 0.0003 ponds/ha (Figure 2.6) (U.S. Forest Service).

T1 was located on the west-central side of the district. It was subject to a WSI in fall of 2001 and a PB in the spring of 2002 (Figure 2.5). Therefore, data collected from site T1 was designated as $T1^{pre}$ (collected in 2001 before treatments began) and

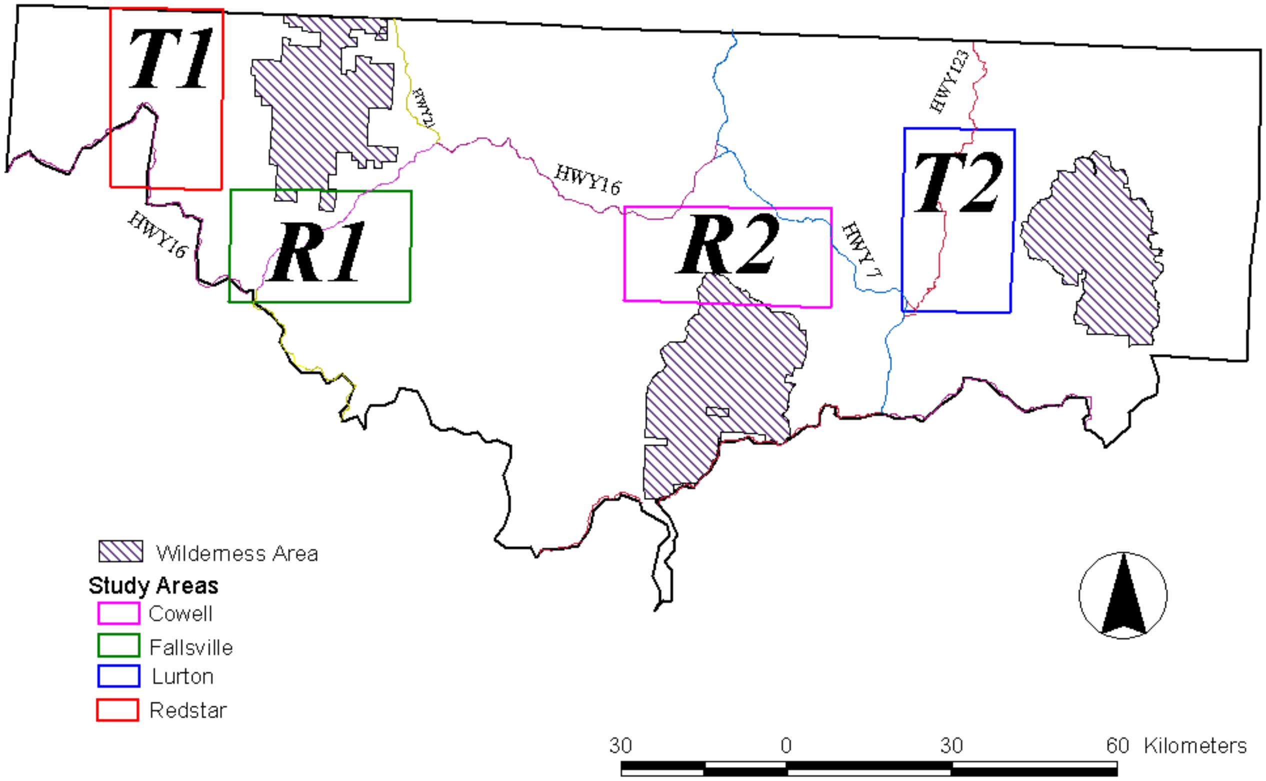

Figure 2.5. Locations of each study-site and wilderness area.

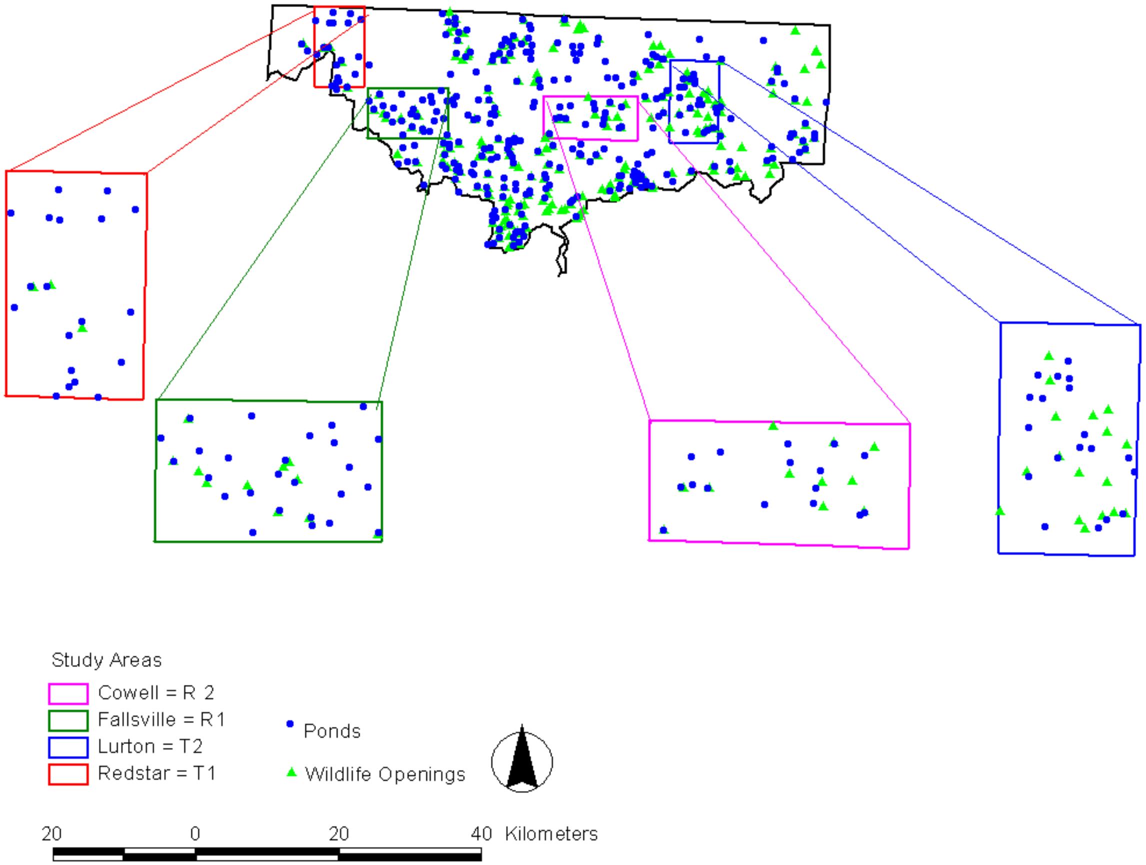

Figure 2.6. Upland pond distribution that served as sampling locations within each study site, as well as ponds that are distributed throughout the Buffalo Ranger District, Ozark-St. Francis National Forest, Arkansas

was therefore used as a reference site. After that, this site was used as a treatment site ($T1^{post}$) and was designated as $T1^{post}$ (collected in 2002 after treatments). This design allowed before and after effects of the management techniques to be evaluated at the same site. T2 is approximately 118 km east of T1 (Figure 2.5). A thinning, similar to a WSI and PB were completed at T2 during 1999, during which the basal area was reduced. Thus, treatment-site T2 was in its 4th year of regeneration when data were collected in 2002. The WSI at sites $T1^{post}$ and T2 were the same except that trees were cut and then removed from T2, while at T1 cut trees were left where they fell. R1 was approximately 2 km east of T1 (Figure 2.5). R2 was approximately 12 km to the west of T2 (Figure 2.5). Study site R2 was only sampled during 2002. At each reference site, no habitat alteration strategies were used. Reference sites were selected so they were relatively close to each treatment site and so that there were approximately equal distances between study sites and wilderness areas (Figure 2.5). Although this study designed had low site replication with treatment sites serving as replicates, this study site selection strategy was designed to minimize any local geographic and topographic variation between treatment and reference sites, as well as to reduce the effects that wilderness areas (preservation areas) may have on sampling.

Method of Capture

Bats were collected using black nylon mist nets (38 mm mesh) that were 2.6 m x 4 m, 2.6 m x 6 m, 2.6 m x 9 m, or 7.8 m x 12 m (Kunz and Kurta 1988). Bats were sampled two nights per week at a treatment and reference site with weekly rotation among the sites. Sampling was designed to insure even sampling among sites during

Figure 2.7. An example of one of the larger upland ponds that was sampled. (17 May 2001).

Figure 2.8. An example of one of the smaller ponds that was netted. (20 July 2002).

Figure 2.9. An example of one of the creeks that was sampled (14 July 2002).

Figure 2.10. An example of a typical road rut that was sampled (22 May 2001).

each year. Nets were placed over ponds, creeks, and perennial road ruts holding water (Figures 2.7 - 2.10). Nets were also placed in dry areas such as roads and in entrances to fields and ponds (Figures 2.11 - 2.12). By netting these entrance points bats were captured that were using the roads and fields as foraging areas but that might have been missed while netting over water structures. Nets were opened approximately 15 minutes before sunset and checked every 15 minutes until the rate of capture fell below one bat per two hours. While nets were open, the capture time, species, sex, and band presence of each captured bat was recorded. Bats were separated by species and placed in 100% cotton bags. When sampling was completed for the night, measurements were taken and captured bats were banded by placing plastic split ring bands, numbered in sequential order, were placed on the forearm of all bats. Data collected included: 1) species, sex, and age of each animal; 2) the reproductive condition of each animal, (males-- nonreproductive or scrotal; females--nonreproductive, pregnant, lactating, or post-lactating); 3) forearm length and weight were measured.

The specific location of each net site was determined using Global Positioning System (GPS) coordinates, date, times nets were open and closed, weather conditions, and habitat type were recorded during each sampling effort. All GPS coordinates were recorded in Universal Transverse Mercator (UTM) and North American Datum 1983 (NAD 83), Zone 15 North.

Forest Stand Density

The FSD was estimated by measuring the basal area in four plots, 0.25 ha in size (approximately 17 m in diameter). Plots were located in each of the four cardinal

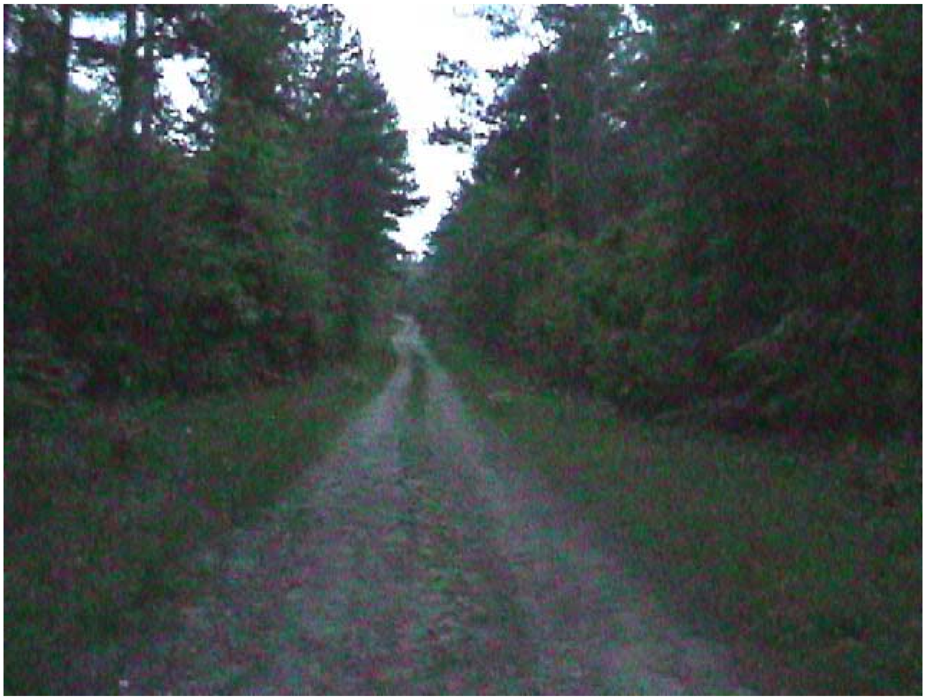

Figure 2.11. An example of an unimproved road that was sampled (23 June 2001).

Figure 2.12. An example of a small wildlife opening that was sampled (23 June 2001).

directions 50 m from the structure or area that was being sampled (Figure 2.15). All trees ≥10 cm DBH in each plot were measured. Tree basal area was calculated by multiplying 0.000078539 by the diameter at breast height squared (DBH^2) (Note: 0.000078539 = pi/40,000 where the division by 40,000 corrects for the difference in cm and diameter to radius). Stand basal area was determined dividing the total basal area for each tree in each plot by 1 ha (4 *0.25ha plots).

Data Analysis

Relative bat species abundance and diversity was estimated by calculating the number of bats captured per net-night, and bat species captured per net-night, respectively. These measures are defined as the number of bats or the number of bat species captured in a single night divided by number of nets that were used in the sampling effort that night. Additionally, species diversity of bats was calculated by using the Shanon-Weiner Diversity Index Model (Krebs 1989). Data from areas where mist nets were up less than one hour due to inclement weather were not included in analyses.

Study-sites $T1^{pre}$, R1, and R2 served as reference-sites; $T1^{post}$ and T2 served as treatment-sites in all data analysis. Minitab Software (Minitab, Inc. 1998) was used for all statistical calculations.

Since WSI and PB are designed to reduce the midstory, an Analysis of Variance (ANOVA) was used to analyze the relationship of FSD among and within study-sites. The percentage of ground cover and midstory closure were also used as additional measures of structural complexity, and therefore similarly analyzed with ANOVA.

I used a 2-sample *T*-test to assess impacts that WSI and PB have on the bat species abundance and diversity. Bat species abundance and diversity among and within sites were analyzed using ANOVA. A Tukey's HSD post hoc test was used to see where the difference occurred within and among treatments types.

I examined associations among the FSD of each sampling site location with study sites to bat species abundance and diversity using a Pearson correlation. In addition, data were combined at the forest stand level, based similarities between bat species diversity and abundance, and treatment types. A Pearson correlation was used to evaluate the relationship between FSD and bat species diversity and abundance.

For all analysis, all data were tested for normality and equal variance using the Kolmogorov-Smirnov Normality Test and Test for Equal Variance respectively. If the data were not normally distributed ($P > 0.15$), they were normalized by calculating the inverse log. An alpha value of 0.05 was selected as the maximum significance level.

RESULTS

Treatment Effects on Structural Complexity

WSI and PB had significant effects on the FSD during this study (Table 2.1). There were significant differences between the treatment and references site for FSD (Table 2.1). The mean FSD at each reference site ($T1^{Pre}$, R1, and R2) was significantly greater than treatment sites ($T1^{Post}$ and T2) (Table 2.1).

FSD was compared between years sampled (2001-2002), at each of the study-sites. The mean FSD was significantly lower in the second year (2002) of sampling at the T1 site ($T1^{post}$) than in 2001 ($T1^{pre}$) (Table 2.2). There were no significant

Table 2.1. ANOVA of the mean forest stand density (FSD) among each study-site.

Site	**# of Net Site Locations**	**Mean FSD**	**StDev**	**Difference (*a*,*b*)**	**F Value**	**P Value**
T1 Pre	14	15.9	3.65	*a*		
T1 Post	11	9.2	3.20	*b*		
T2	21	9.1	2.56	*b*	25.12	0.0001
R1	21	16.2	3.17	*a*		
R2	12	16.2	2.77	*a*		

Note: significant difference (*P*-value < 0.05) between study sites exist if subscript letters are different and study sites are similar if matching subscript letters are the same. The Family error rate = 0.05 and the Individual error rate = 0.0066

Table 2.2. ANOVA of mean forest stand density (FSD) within each study-site.

Site	**# of Net Site Locations**	**Mean FSD**	**StDev**	**Difference (*a,b*)**	***F* Value**	***P* Value**
T1 Pre, (2001)	14	15.9	3.65	a		
T1 Post, (2002)	11	9.2	3.20	*b*		
T2, (2001)	10	8.7	2.28	*b*		
T2, (2002)	11	9.4	2.85	*b*	16.53	0.0001
R1, (2001)	9	15.8	3.35	*a*		
R1, (2002)	12	16.5	3.15	*a*		
R2, (2002)	12	16.2	2.77	*a*		

*Note: significant difference (*P*-value 0.05) between study sites if subscript letters are different and study sites are similar if matching subscript letters are the same. The Family error rate = 0.05 and the Individual error rate = 0.0336 differences in the mean FSD within study site T2 and in study-sites ($T1^{pre}$, R1, and R2) (Table 2.2).

Treatment Effects on Bat Assemblages

A total of 436 bats were captured during this study, of which 203 and 233 were captured in the treatment and reference sites respectively (Table 2.3). Ten of the 14 bat species expected to occur in this region of Arkansas (Sealander and Heidt 1990) were captured during this study. Northern long-eared bats (*M. septentrionalis*) were the most frequently captured bat in each study site. Red bats (*Lasiurus borealis*) and eastern pipistrells (*Pipistrellus subflavus*) were the second and third most captured bat species respectively (Figure 2.13). Other species that were captured during this study included hoary bat (*L. cinereus*), silver-haired bat (*Lasionycteris noctivagans*), big brown bat (*Eptesicus fuscus*), gray bat (*M. grisescens*), (*M. sodalis*), eastern small footed bat (*M. leibii*), and evening bat (*Nycticeius humeralis*).

There was no significant difference in the number of bats/net night captured between treatment types (Table 2.3). Additionally, there was no detectable difference in the number bats/net night captured when compared among and within treatment types (Table 2.4 and 2.5 respectively).

There was no significant difference of bat species/net night captured between treatment types (Table 2.3), nor were any differences detected when capture rates were compared within and among study sites (Tables 2.6 and 2.7). Specifically, there were no significant differences in bat species capture rates in pre-and post treatment at study site T1 ($T1^{pre}$ and $T1^{post}$), such that near equal numbers bat species/net-night were

Table 2.3. Mean bat capture rates and species diversity between treatment types (2-sample *T*-test).

Site	***Treatment-sites (2001-02)***	***P-value***	***Reference-sites (2001-02)***
# of Species Captured	9		7
Number of Bats Captured	203		233
Net Nights	39		70
Capture Rate (Bats/Net-night)	5.6		4.2
StDev	0.79		1.00
		0.384	
Capture Rate (Bat Species/Net-night)	0.15		0.03
StDev	0.13		0.05
		0.208	
Species Diversity (H)	0.4		0.3
StDev	0.13		0.03
		0.677	

Note: Difference = [mu (treated) - mu (Untreated)] are not significant (*P*-value > 0.05).

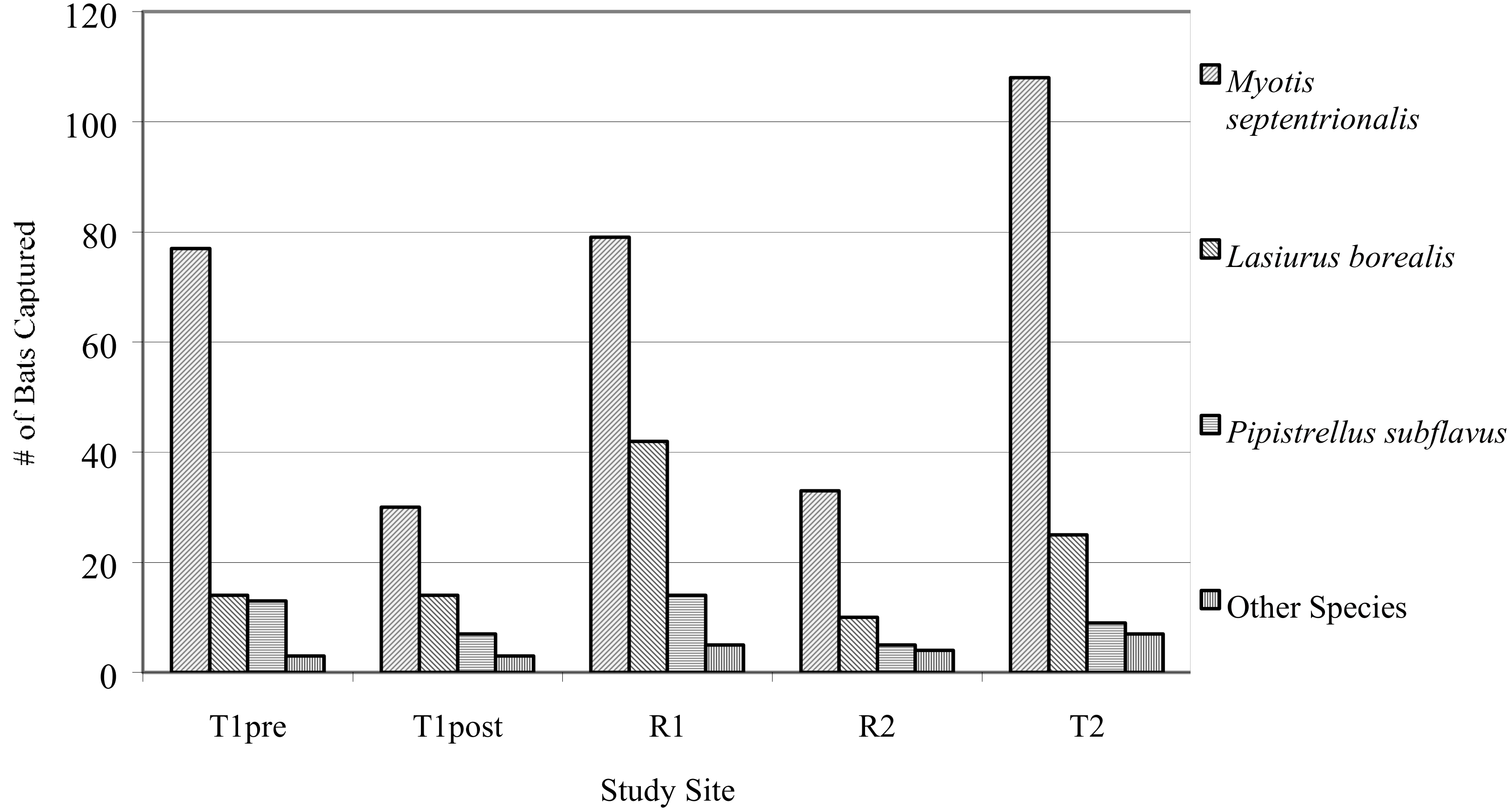

Figure 2.13. Relationships of the percentage of the most predominant bat species captured during this study. Species that are represented by the "Other Species" category, were only sampled at least once/year.

Table 2.4. Mean capture rates (bats/net night) among each study-sites (ANOVA).

Site	**# of Bats Captured**	**# Net Nights**	**Mean Capture Rate (Bats/Net-night)**	**StDev**	***F* value**	***P* value**
T1^{pre} *(2001)*	107	23	6.1	8.89	0.89	0.476
T1^{post} *(2002)*	54	13	4.9	8.00		
T2 *(2001-02)*	149	26	6.4	6.75		
R1(2001-02)	74	31	2.6	3.02		
R2 (2002)	52	16	3.7	4.21		

Note: The Family error rate = 0.05 and the Individual error rate = 0.0066

Table 2.5. Mean capture rates (bats/net night) within each study-site (ANOVA).

Site	**# of Bats Captured**	**# Net Nights**	**Mean Capture Rate (bats/net-night)**	**StDev**	***F* value**	***P* value**
$T1^{pre}$ *(2001)*	107	23	6.1	8.89	0.87	0.522
$T1^{post}$ *(2002)*	54	13	4.9	8.00		
T2 *(2001)*	86	13	7.4	7.80		
T2 *(2002)*	63	13	5.5	5.88		
R1 *(2001)*	43	19	2.8	3.22		
R1 *(2002)*	31	12	2.6	2.99		
R2 *(2002)*	52	16	3.7	4.21		

Note: The Family error rate = 0.05 and the Individual error rate = 0.0034.

Table 2.6. Mean species capture rates (bat species/net-night) among each study-site (ANOVA).

Site	**# of Species Captured**	**Mean Species Capture Rate (Bat Species/Net-Night)**	**StDev**	***F* value**	***P* value**
T1^{pre} *(2001)*	6	1.1	1.20	1.20	0.320
T1^{post} *(2002)*	6	1.2	1.77		
T2 *(2001-02)*	5	1.7	1.20		
R1(2001-02)	4	1.0	1.05		
R2 (2002)	6	1.2	1.07		

Note: The Family error rate = 0.05 and the Individual error rate = 0.0066.

Table 2.7. Mean species capture rates (bat species/net-night) for sampling sites (ANOVA).

Site	# of Species Captured	Mean Species Capture Rate (Bat Specie/Net-Night)	StDev	*F* value	*P* value
T1^{pre} *(2001)*	6	1.1	1.20	1.75	0.128
T1^{post} *(2002)*	6	1.2	1.77		
T1 *(2001)*	5	2.0	1.28		
T1 *(2002)*	5	1.5	1.15		
R1 *(2001)*	3	0.8	0.93		
R1 *(2002)*	5	1.1	1.17		
R2 *(2002)*	7	1.2	1.07		

Note: The Family error rate = 0.05 and the Individual error rate = 0.0034.

captured at sites $T1^{pre}$ and $T1^{post}$ (Table 2.6). Additionally, bat species diversity was also estimated using the Shannon-Weiner Diversity Index Model (Krebs 1989) and analysis also illustrated that there was no significant difference between bat species diversity and treatment types (Table 2.3). The Shannon-Weiner diversity index also showed that there was no significant difference in bat species diversity within and among study sites (Tables 2.8 and 2.9).

Bat Assemblages Associated with Forest Stand Densities

Since there was no significant difference in FSD detected within treatment types, associations between FSD and bat species abundance and diversity within each site were established using a Pearson's correlation. There was a negative correlation within each site replicate, such that the numbers of bats/net night, bat species/net night, and species diversity decreased as FSD increased (Figures 2.14 – 2.28).

Since similar trends were observed within each site, data were combined at the forest stand level. A Pearson's correlation was used to evaluate the relationship between bat capture rates and FSD. A significant negative correlation was observed between these variables ($r = -0.398$) ($P = 0.002$) (Figure 2.29). The mean capture rate increased as FSD decreased (less structurally complex).

Correlations between bat species/net-night and FSD were significant. Analysis of combined data showed a highly significant, but weak relationship between bat-species capture rate and FSD ($r = -0.297$, $P = 0.023$) (Figure 2.30). Similar results were obtained when bat diversity was correlated with FSD ($r = -0.316$, $P = 0.005$) (Figure 2.31).

Table 2.8. Mean species diversity (Shannon-Wiener Species Diversity Index) among each study-site (ANOVA).

Site	**# of Species Captured**	**Species Diversity (H)**	**StDev**	***F* value**	***P* value**
T1^{pre} *(2001)*	6	0.3	0.51	0.47	0.761
T1^{post} *(2002)*	6	0.3	0.46		
T2 *(2001-02)*	5	0.5	0.34		
R1(2001-02)	4	0.3	0.48		
R2 (2002)	6	0.4	0.48		

Note: The Family error rate = 0.05 and the Individual error rate = 0.0066.

Table 2.9. Mean species diversity (Shannon-Wiener Species Diversity Index) within each study-site (ANOVA).

Site	**# of Species Captured**	**Species Diversity (H)**	**StDev**	***F* value**	***P* value**
T1^{pre} *(2001)*	6	0.3	0.51	0.47	0.828
T1^{post} *(2002)*	6	0.3	0.46		
T1 *(2001)*	5	0.6	0.32		
T1 *(2002)*	5	0.4	0.32		
R1 *(2001)*	3	0.3	0.42		
R1 *(2002)*	5	0.3	0.53		
R2 *(2002)*	7	0.4	0.48		

Note: The Family error rate = 0.05 and the Individual error rate = 0.0033.

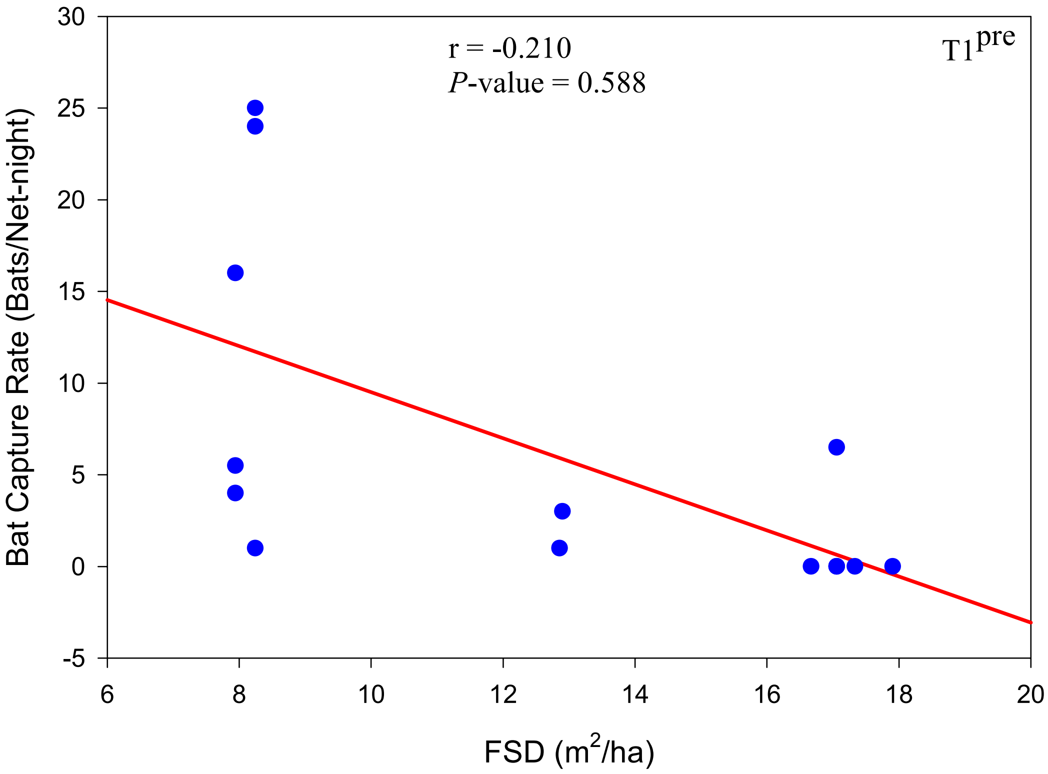

Figure 2.14. Association between the numbers of bats captured per net-night and forest stand density at each sampling location at Site T1^{pre}.

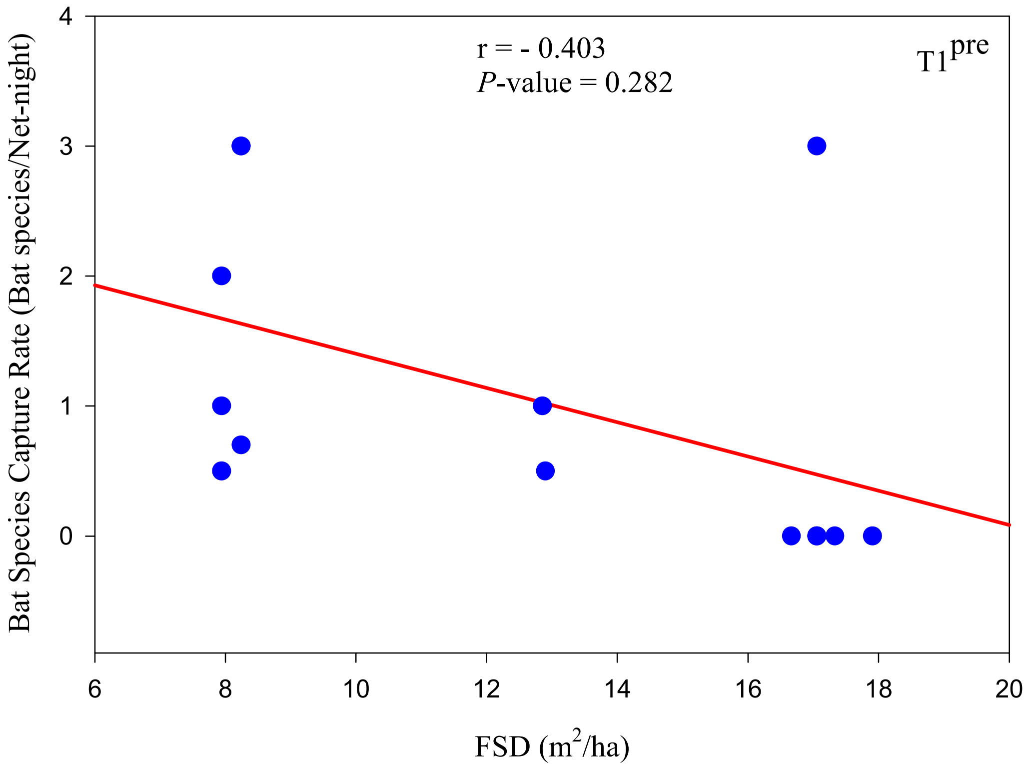

Figure 2.15. Association between the numbers of bat species captured per net-night and forest stand density at each sampling location at Site T1^{pre}.

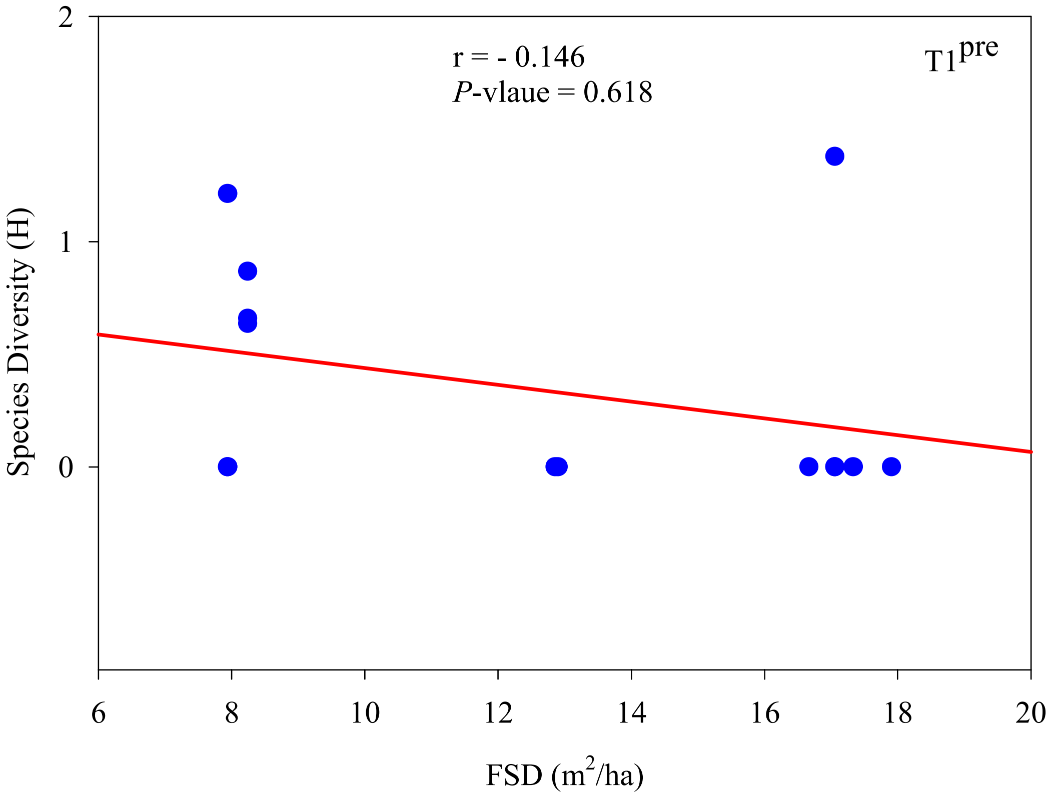

Figure 2.16. Association between species diversity and forest stand density at each sampling location at Site T1^{pre}.

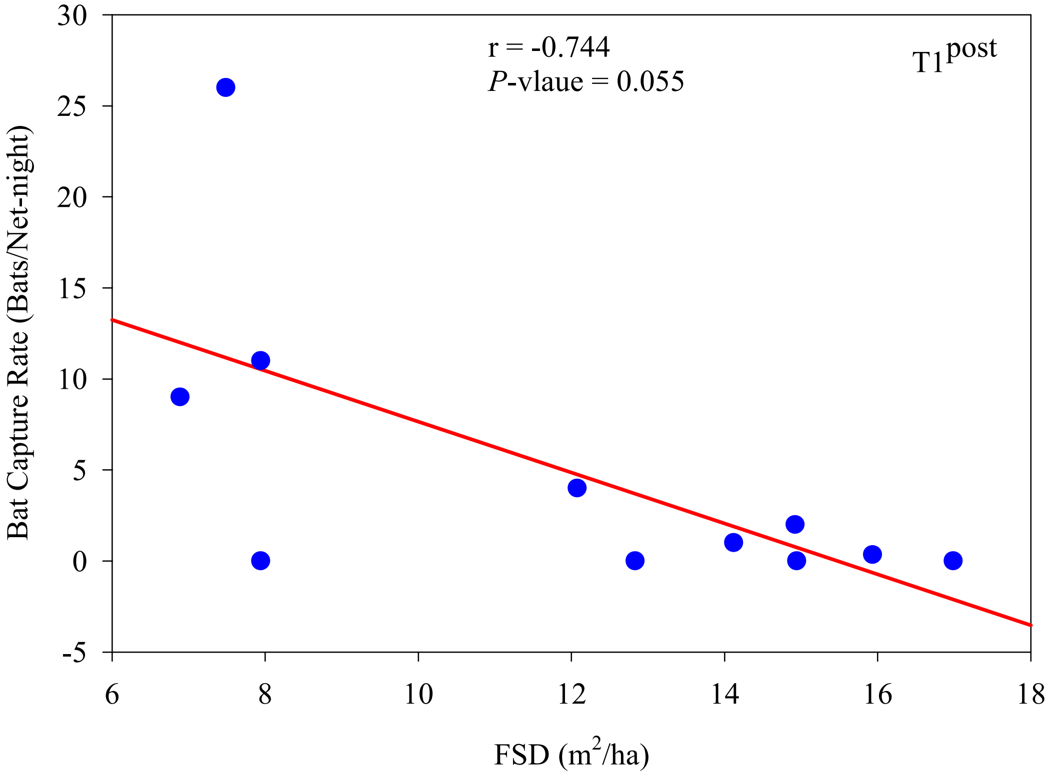

Figure 2.17. Association between the numbers of bats captured per net-night and forest stand density at each sampling location at Site T1^{post}.

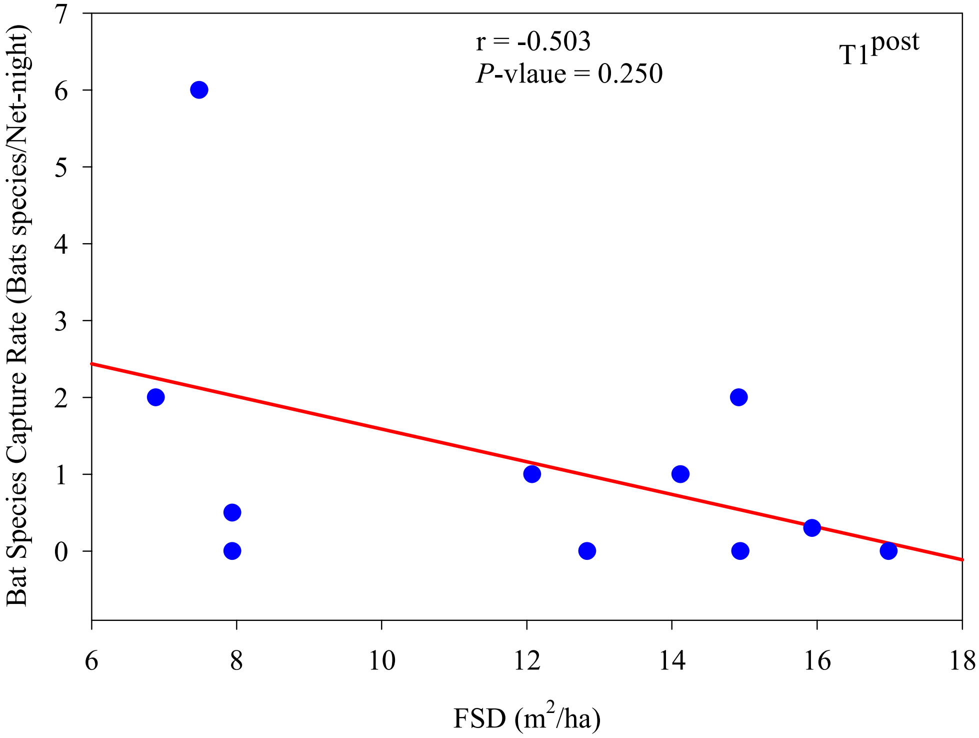

Figure 2.18. Association between the numbers of bat species captured per net-night and forest stand density at each sampling location at Site T1^{post}.

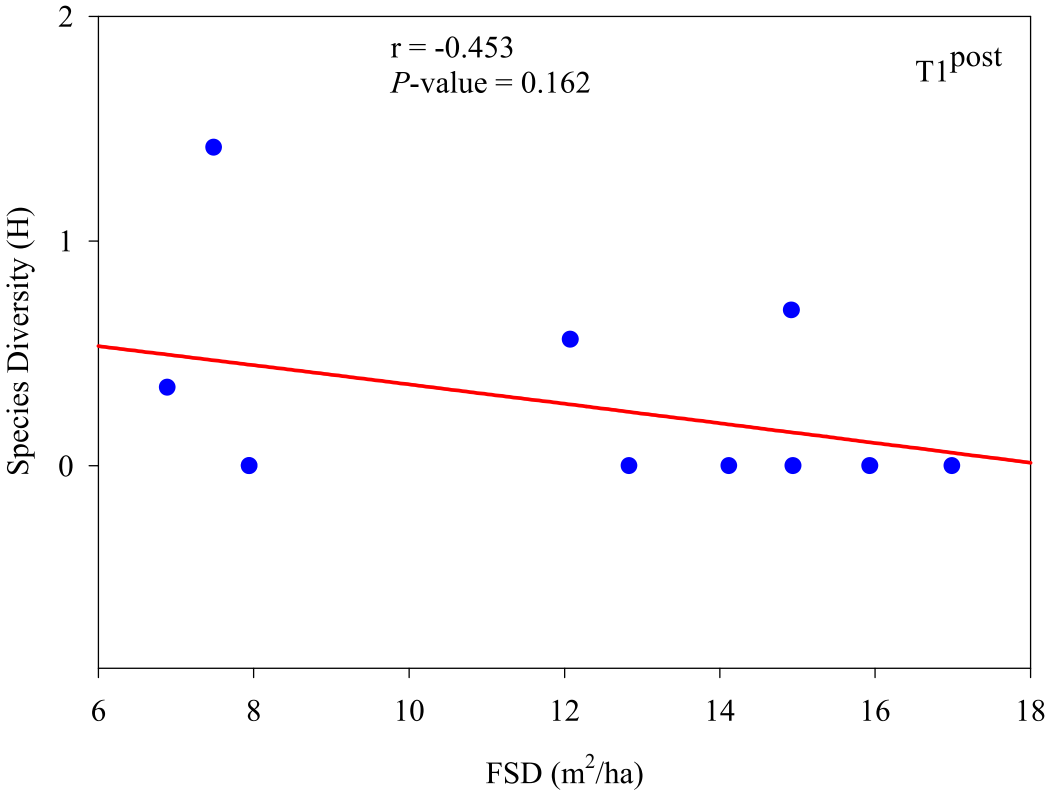

Figure 2.19. Association between species diversity and forest stand density at each sampling location at Site T1^{post}.

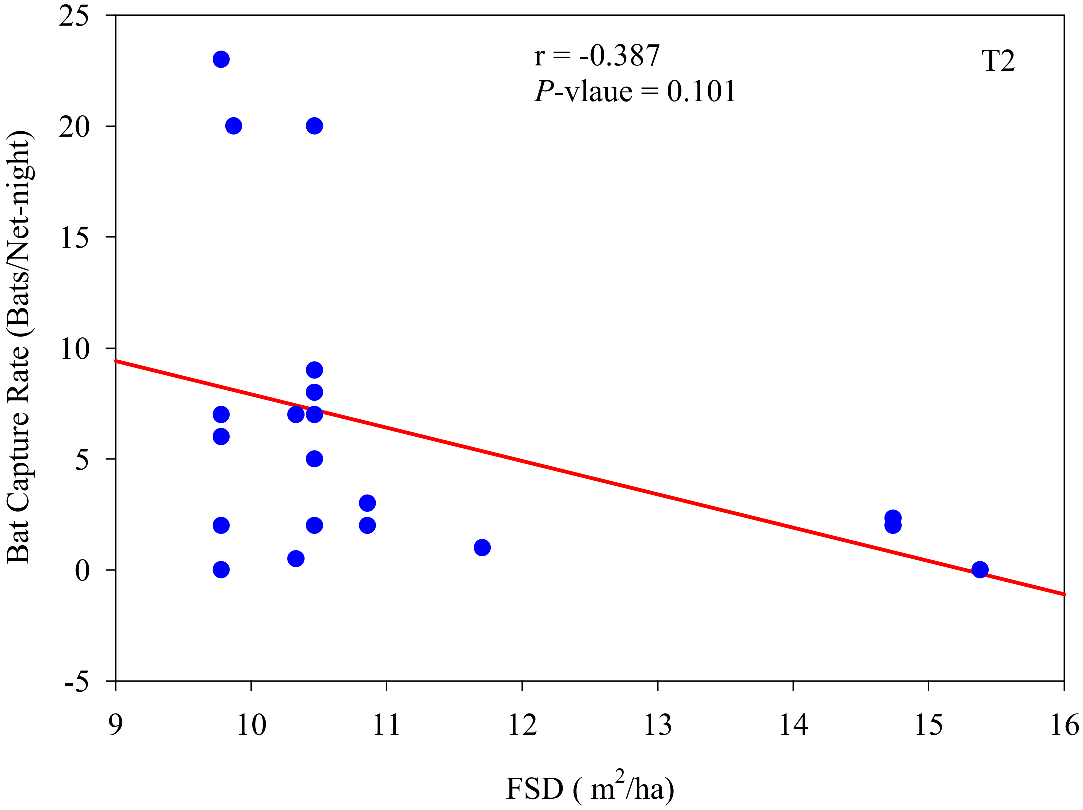

Figure 2.20. Association between the numbers of bats captured per net-night and forest stand density at each sampling location at Site T2.

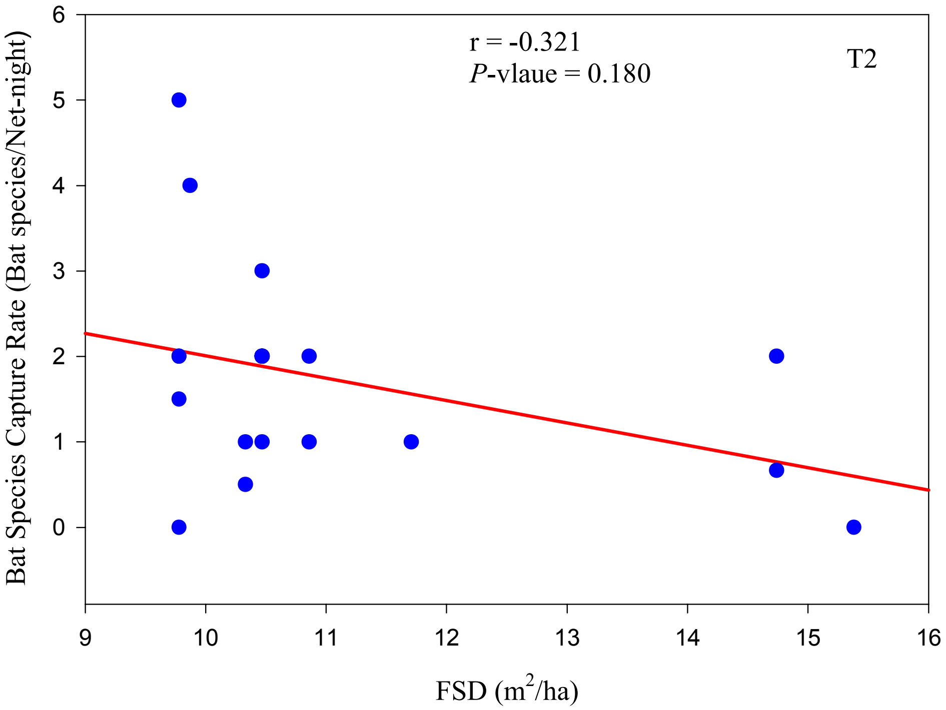

Figure 2.21. Association between the numbers of bat species captured per net-night and forest stand density at each sampling location at Site T2.

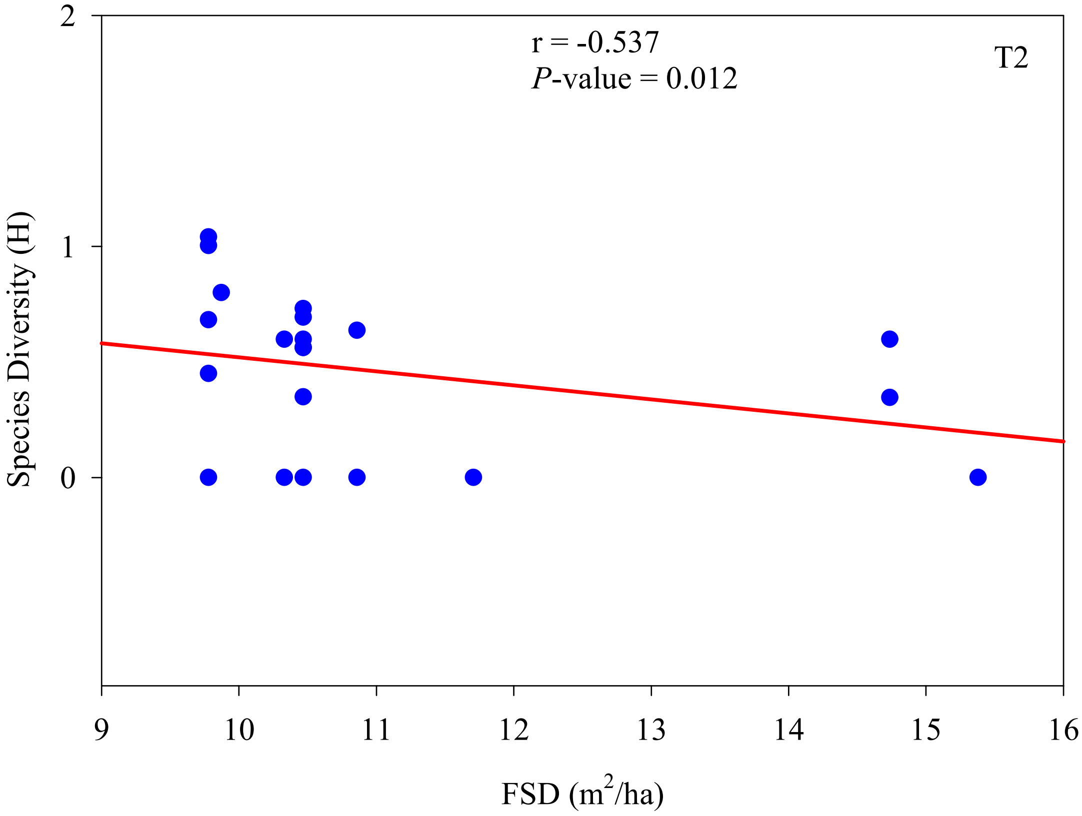

Figure 2.22. Association between species diversity and forest stand density at each sampling location at Site T2.

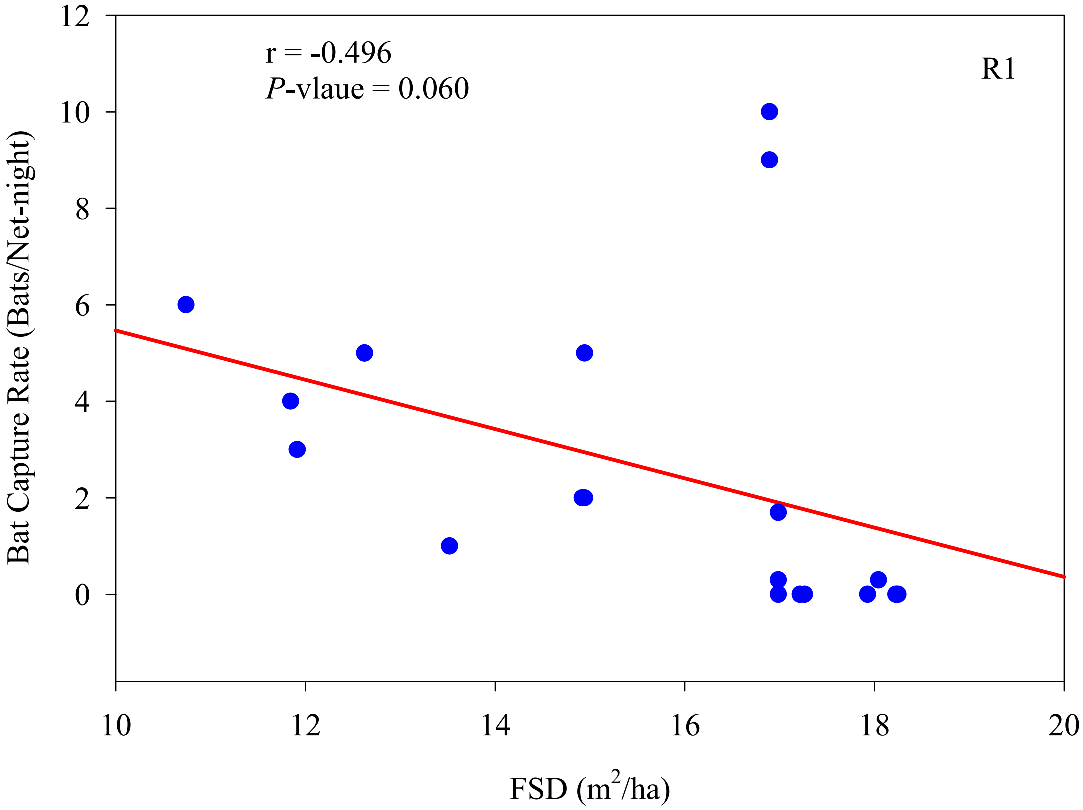

Figure 2.23. Association between the numbers of bats captured per net-night and forest stand density at each sampling location at Site R1.

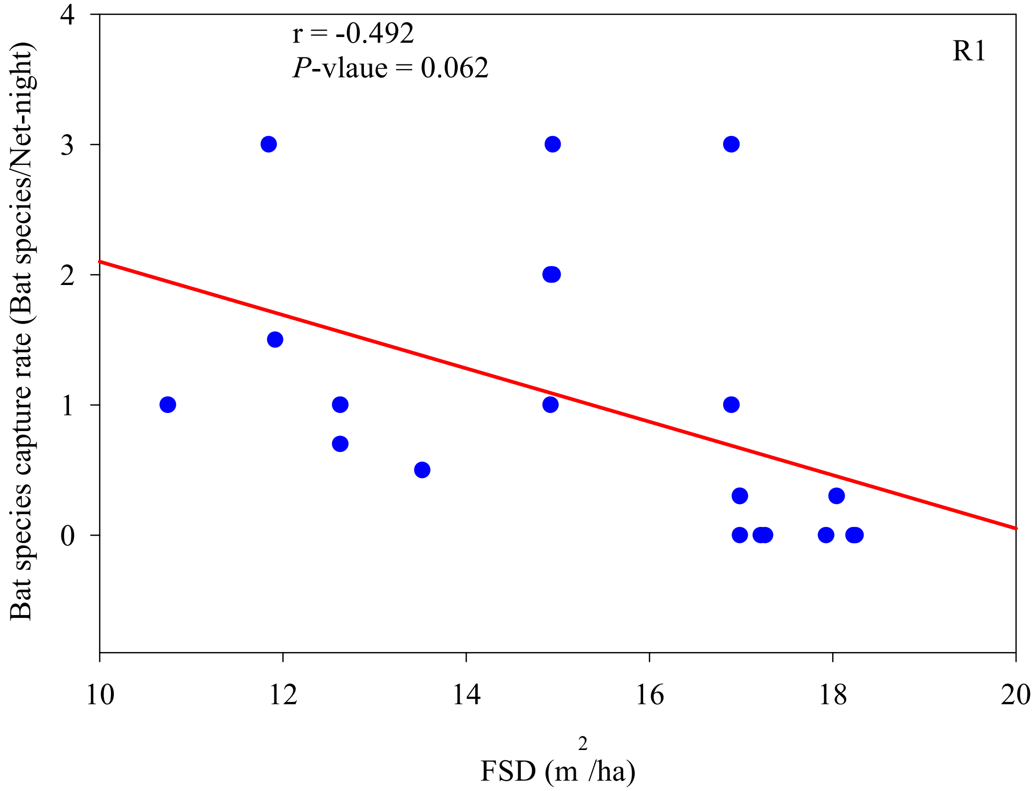

Figure 2.24. Association between the numbers of bat species captured per net-night and forest stand density at each sampling location at Site R1.

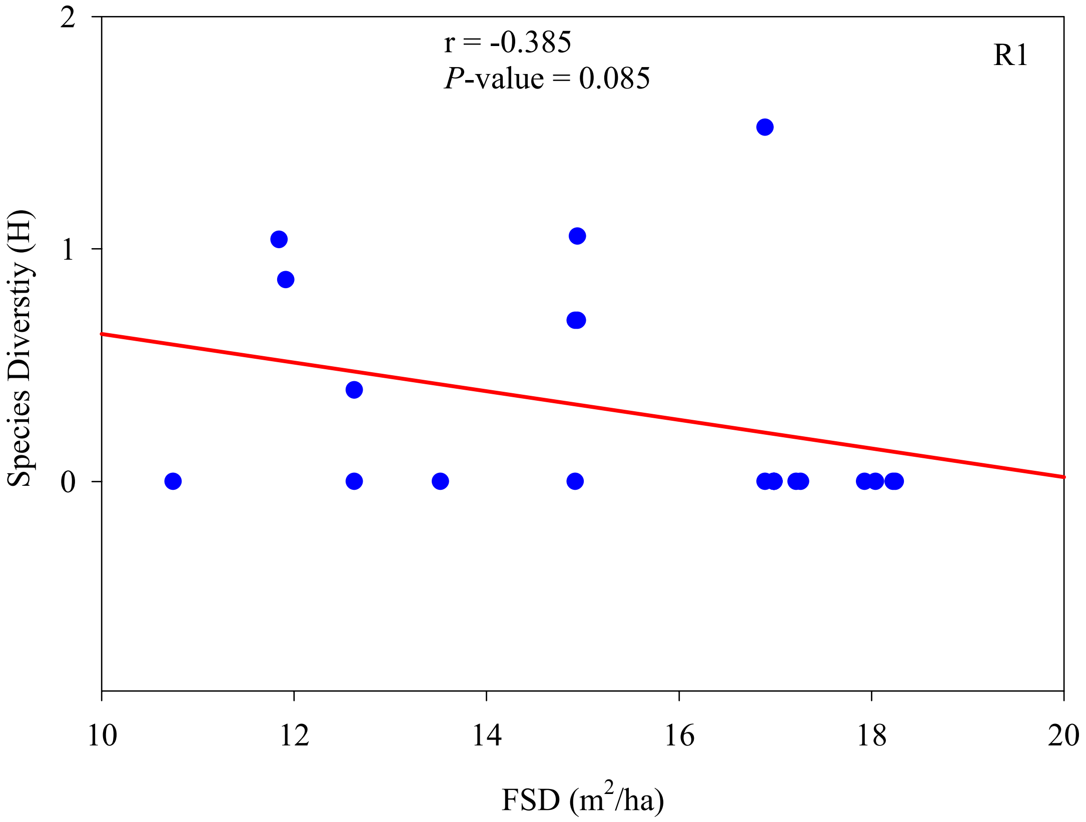

Figure 2.25. Association between species diversity and forest stand density at each sampling location at Site R1.

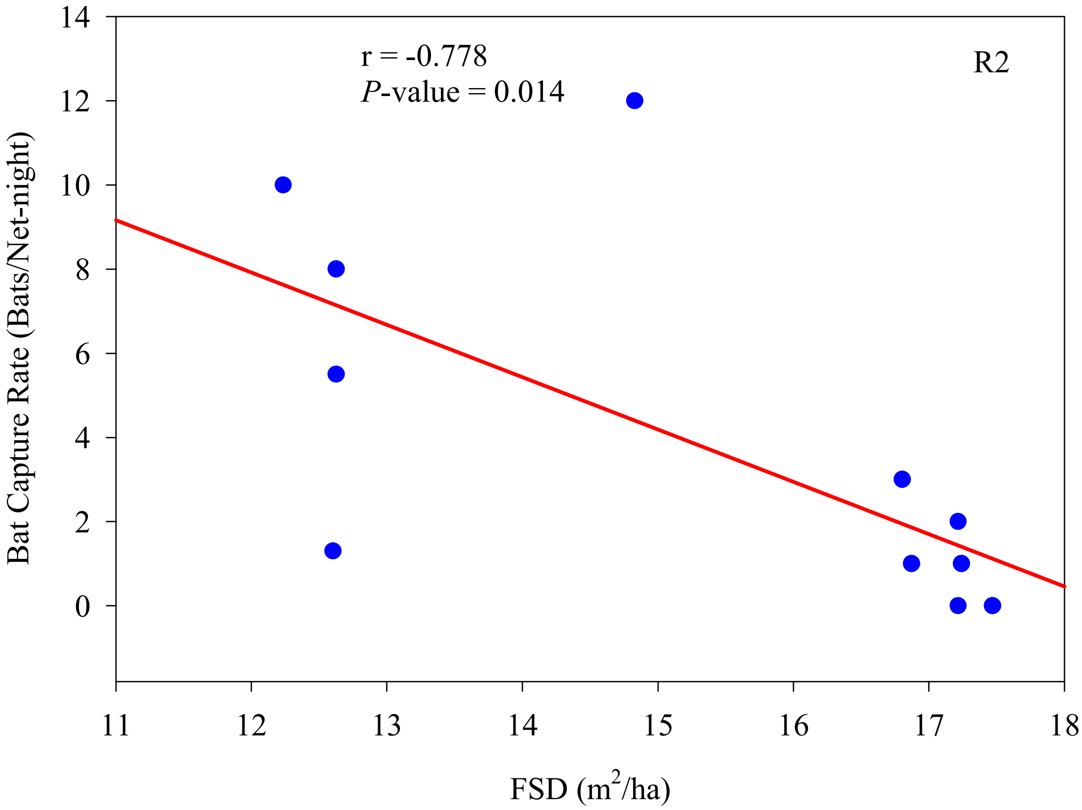

Figure 2.26. Association between the numbers of bats captured per net-night and forest stand density at each sampling location at Site R2.

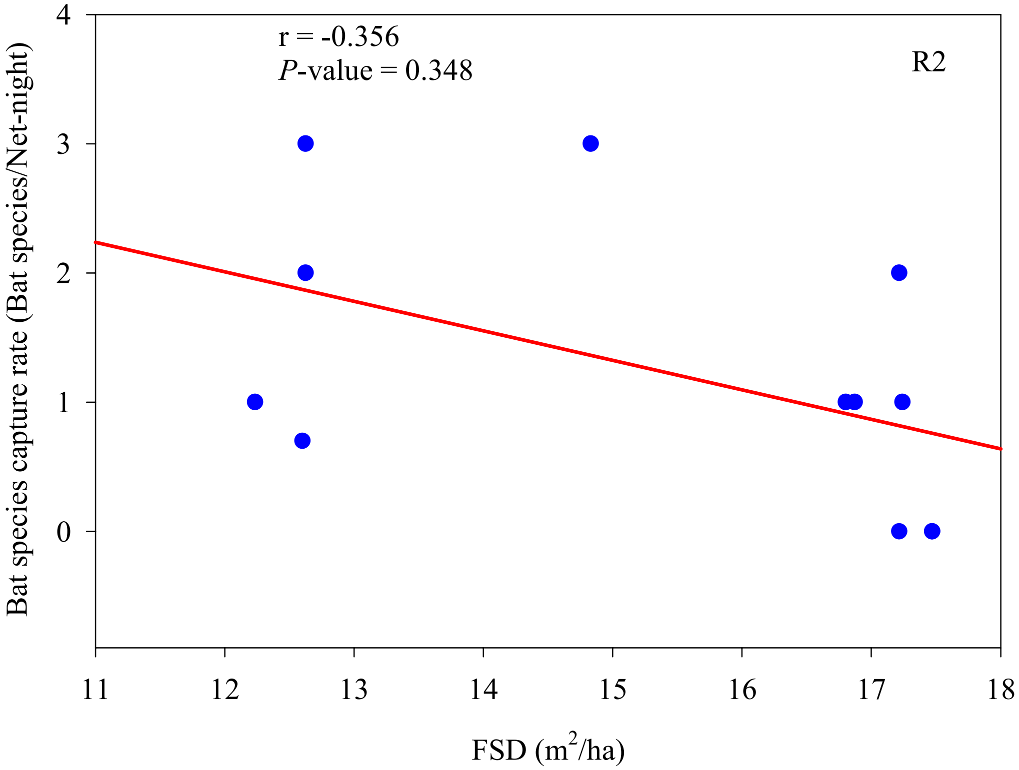

Figure 2.27. Association between the numbers of bat species captured per net-night and forest stand density at each sampling location at Site R2.

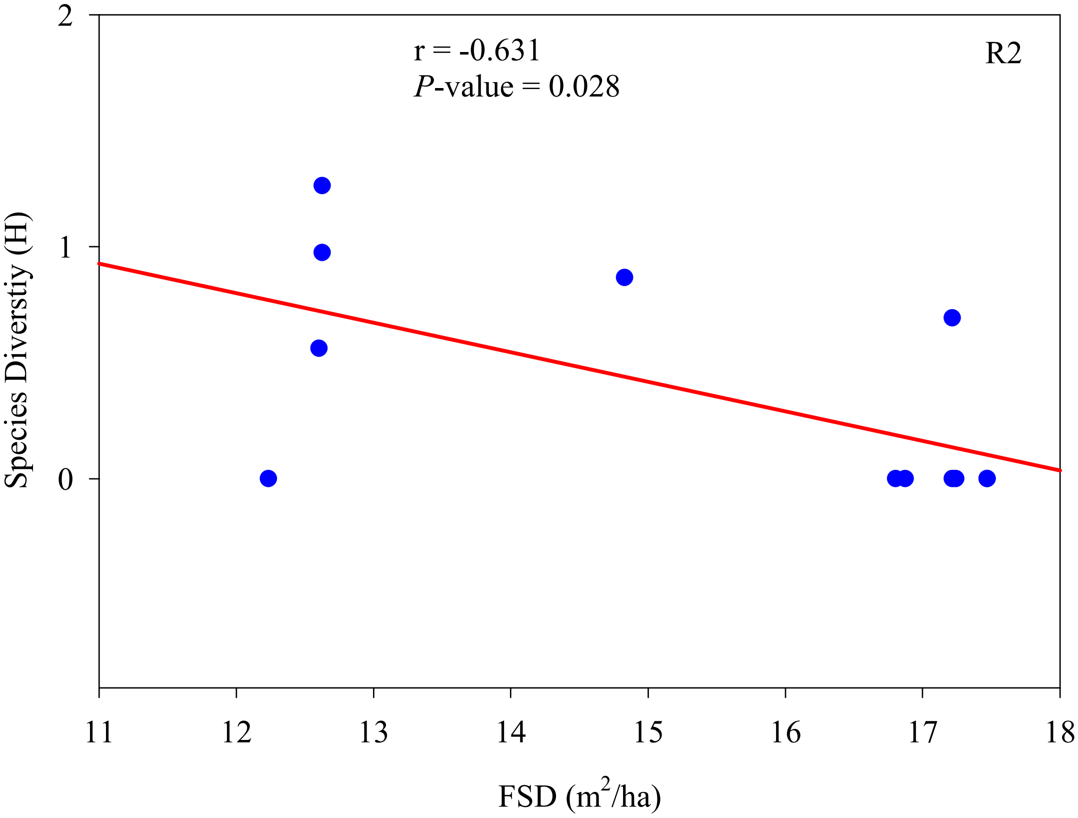

Figure 2.28. Association between species diversity and forest stand density at each sampling location at Site R2.

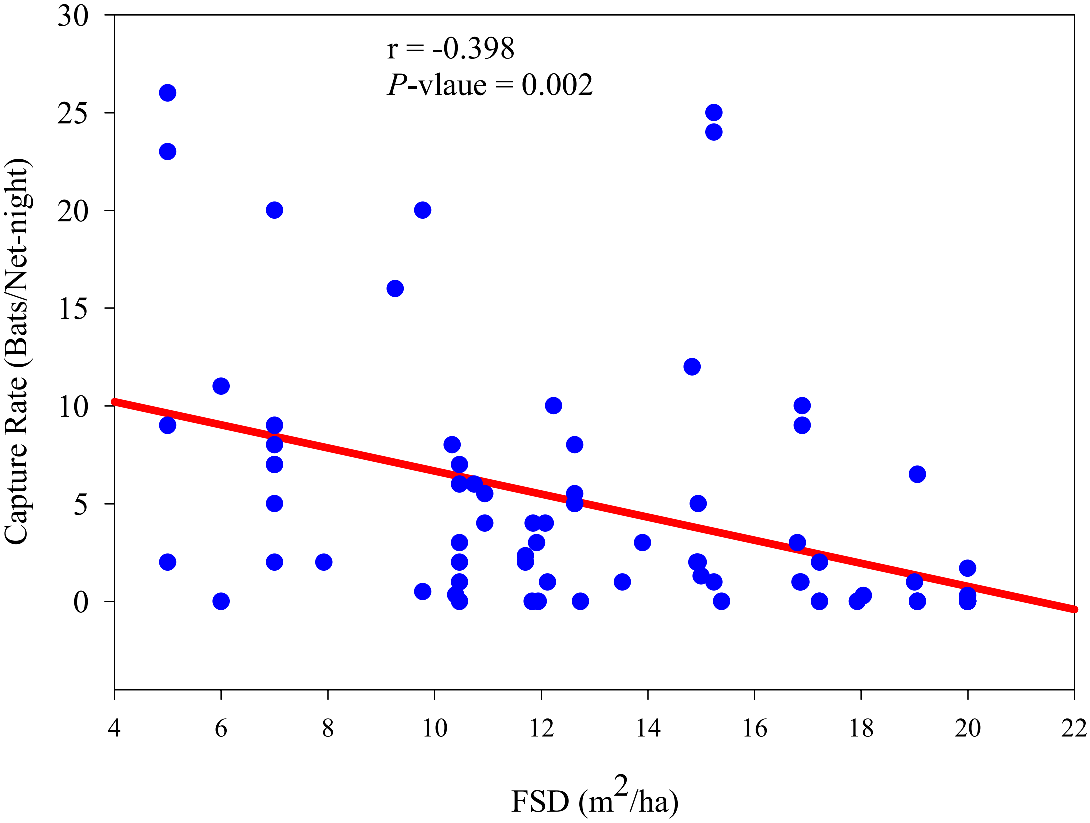

Figure 2.29. Association between the numbers of bats captured per net-night and forest stand density at each sampling site location.

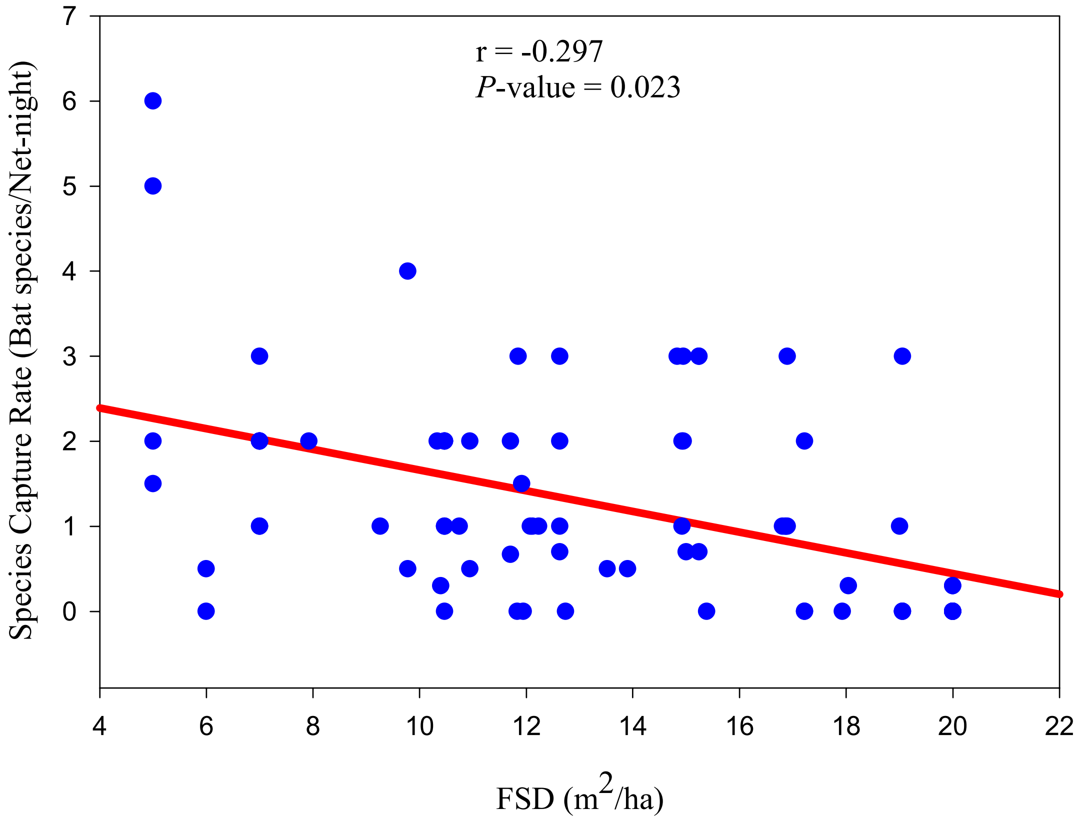

Figure 2.30. Association between bat species capture rate and forest stand density at each sampling site location.

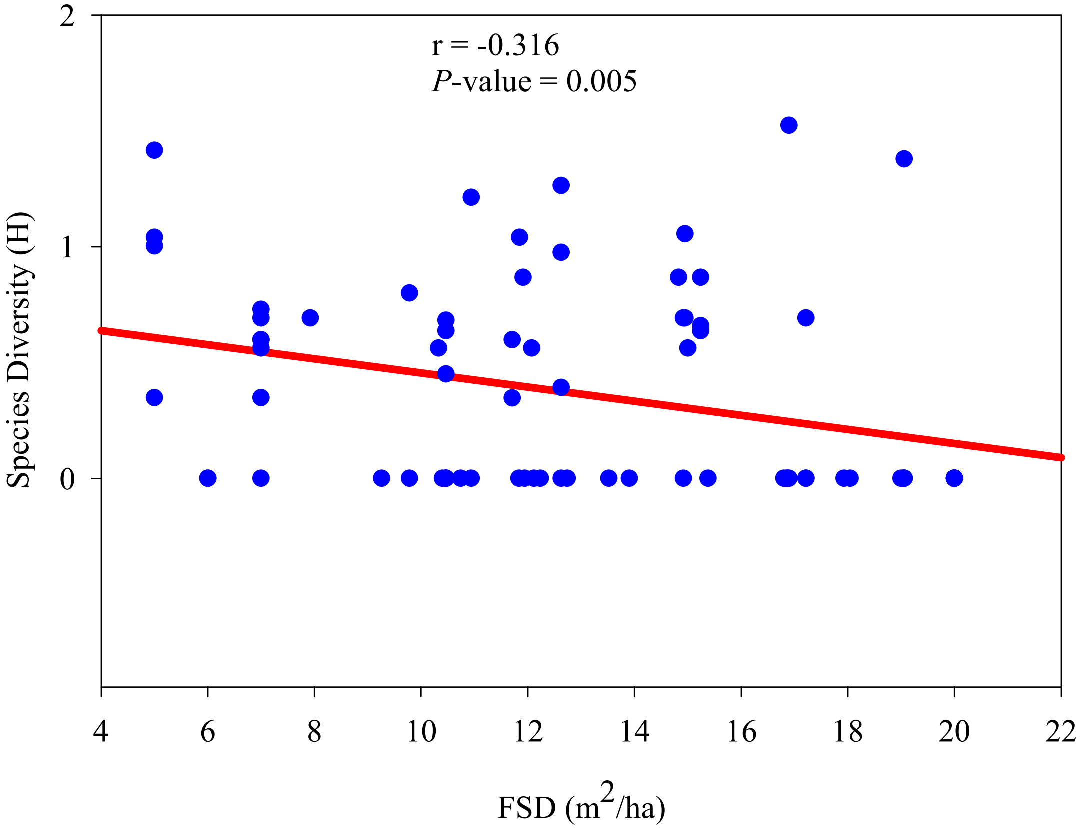

Figure 2.31. Association between bat species capture rate and forest stand density at each sampling site location.

DISCUSSION

Results show that WSI and PB can be used to provide forested areas that are less structurally complex by reducing FSD. The FSD at study site T1 decreased significantly after the treatment ($T1^{pre}$ and $T1^{post}$) was administered and was similar to the FSD at study site T2, which had a similar treatment in 1999. Additionally, FSD in $T1^{post}$ and T2 were significantly different than those in each of the reference-sites ($T1^{pre}$, R1, and R2).

Although the differences in the mean bat capture rate between treatments types were not significant, the mean capture rate for bats was higher in treatment sites than reference sites. For example, the number of bats captured per net night at the T1-site decreased by 1.2 bats/net night (10 %) during 2002. The numbers of bats captured per net-night at the T2-site was 1.9 bats/net-night less (14 %) in 2002 than in 2001. At the R1-site, slightly 3 % more bats (0.2 per bats net night) were captured in 2001 than in 2002. This lack of significance might be attributed to the low number of study sites, which resulted in low power. Therefore, any affects could be underestimated and thus it is possible that a Type II error could have been made. However, due to known home ranges and high degree of mobility by bats, each study site was relatively large (60,000 ha) and was spatially distributed so that treatment affect on bat assemblages could be documented. The relative size of each site allowed evaluating among and within each study site with net-site locations serving as the sampling unit possible.

Results on the combined data supported the predicted relationships between FSD and the bat capture rate. This relationship only accounted for 15 percent variation provided by FSD to explain changes bat capture rate. However, it is still biologically and/or ecologically significant considering the complexity of the ecosystem. In addition, this result is in agreement with a report by Saunders and Barclay, (1992) showing that *M. lucifugus* and *M. volans,* two species with similar wing and echomorphology that are adapted to dense habitats, foraged exclusively in open habitats. It is therefore plausible that other species that are also adapted to densely forested habitats benefit by the availability of more open habitats. More open areas may allow greater flight speed, which may maximize an individual's search effort while foraging and improve its ability to move to alternative foraging areas, ponds, and roosts. Thus, the advantages achieved by a preference for less dense areas within the forest seem to support foraging strategy theories (Schoener 1971).

In this investigation, mean estimates of species diversity of bats were higher in treatment sites that reference sites; however, these differences were not statistically significant and may have been affected by low site replication. As previously mention during this discussion, this lack of significance might be attributed to the low number of study sites, which resulted in low power. Therefore, any affects could be underestimated and thus it is possible that a Type II error could have been made. Diversity estimates were significant when data were combined and analyses conducted among and within study sites. Bat species captured/net-night, was negatively correlated with FSD. Similar correlations were observed when bat diversity was estimated using the Shannon-Weiner Diversity Index Model. This relationship only accounted for approximately 10 percent variation provided by FSD to explain changes both estimates of species diversity. Therefore, another 90 percent of the variation could be attributed to any number of ecological variables that may also affect species diversity. Data suggests that bat diversity was greater in areas of the forest that were relatively less dense than surrounding more dense regions. Similar results have been recorded in other studies, where increased species diversity and abundance were associated with less dense forest (Saunders and Barclay 1992; Fenton and Bell 1979).

Bat capture rates (bats/net night and bat species/net night) and bat species diversity data were combined at the forest stand level, these forest stands were selected from existing treatment and reference sites and was therefore pseudoreplicated. This study design was unavoidable, in order to evaluate the relationships between bat assemblages and FSD, while documenting any treatment affects.

Net placement could potentially exert an effect on the results. In order to reduce this possible effect, in all study sites, nets were placed at various types of water structures, roads, and small wildlife openings. Care was also taken to insure that these structures that were used as netting locations were located both on the exterior and interior portions of the forest. The emphasis on placing mist nets near or over water may also affect capture rates. Recently, Carroll et al. (2002) reported that greater numbers of northern long-eared bats and half of all Indiana bats were captured in the interior portions of the forest and suggested that bat diversity and abundance may be underrepresented when populations are sampled using traditional mist netting techniques, which emphasized placing nets over water containing structures. However, others report that higher levels of bat activity occur over or near water sites than in strictly terrestrial habitats (Lunde and Harestad 1986; Furlonger et al. 1987; Krusic et al. 1996; Grindal et al. 1999).

The data described herein indicated that WSI and PB did not have any apparent effect on bat activity in the BRD. It seems that the objectives of WSI and PB to reduce and maintain the mid- and understory thereby altering the FSD to a basal area of 7 to $9m^2$/ha provided a less structurally complex forested habitat. This habitat type could potentially harbor greater abundance and diversity of bat species.

The BRD contains several cave hibernacula that harbor colonies of Indiana bats during the winter (male and female) and summer (male) months (Harvey 1989). There are hibernacula within approximately 10 miles of each study site. The specifics of the migration of this species between winter and summer roosting areas have not been reported, however several researchers have suggested that this species migrates north towards Missouri (Harvey and Britzke pers. com.). This hypothesis is supported by the lack of Indiana bat captures during this study and the high capture rates of this species during the summer months in southern Missouri (Harvey pers. com.). This species migration could possibly be related to the habitat preferred by this species. Indiana bats prefer riparian areas with reduced woody vegetation and avoid agriculture areas (Humphrey et al. 1977) and in the upper strata of the forest (Brack 1983). Because the BRD is predominantly composed of dense mid-and understory, agriculture fields, and dense riparian areas, there is little area fitting this preferred habitat description within BRD. Since most areas within the BRD are more structurally complex, this could select against this species wing morphology and echomorphology.

MANAGEMENT IMPLICATIONS

Bats in the ONF use several types of forested and non-forested habitats. These habitats are used for foraging, predator avoidance, and protection from extreme environmental conditions (Lewis 1995). However, most of the forests in the Ozark region, as well as other National Forests throughout North America, have, over time, become increasingly dense probably because of reduced timber harvesting and fire suppression.

Forests are subjected to natural mortality events including natural tree fall resulting from age, wind, and insect damage. These events create small openings in the forest that may provide foraging areas and flight corridors for movement to other foraging areas and water resources. Using conventional timber harvesting techniques, such as selective cutting, seed tree harvesting, small group cuts, or thinning as well as WSI could potentially simulate natural mortality in temperate forest ecosystems. By creating areas within the forest that contain an increased space within the mid- and understory maneuverability during flight might be increased. Bats may also be using this habitat type because it provides stratified forest stands where different bat species are able to segregate into separate niches. This could maximize access to prey and water, provide access to available roost sites, minimize exposure to predators, and reduce potential for competition by providing more suitable foraging niches (Krusic and Neefus 1996).

Before settlement, fire was part of many of the natural ecosystems in temperate forest. Several benefits result from fires including as an increase in the nutrient cycle in forest ecosystems from the ash (Fogel and Cromak 1977). This aids in the sprouting of grasses, forbs, and shrubs as well as the regeneration of woody shoots. Regeneration of trees is vital to the forest ecosystem, as it allows for younger trees to replace older trees that have fallen due to natural mortality. Moreover, it also allows for a heterogeneous forest setting and aids in creating a stratified forest while simultaneously maintaining a relatively open midstory. Prescribed burning could potentially simulate this type of forest setting while preventing high intensity fires that occur as a result of accumulation of fuel (leaf litter and decaying wood) on the forest floor in areas where fire has been suppressed. Conducting prescribed burns on a three-year rotation after a WSI has been administered allows the mid-and understory to remain relatively open therefore preventing treated portions of the forest from becoming structurally complex enough to have a negative impact on bat communities.

Roads are developed and maintained in ONF. Many of these are used during timber harvesting and/or management and appear to be beneficial to bats. Roads provide foraging areas and flight corridors to alternative foraging areas and to drinking sites (Krusic and Neefus 1996). Therefore, forest managers should consider leaving these unimproved roads and trails; however, gates should be placed on these structures to prevent excessive vehicle traffic.

Water resources are vital for bat communities as it provides areas for water consumption and tend to attract and produce high concentrations of insects for foraging bats. Bats utilize riparian areas, as well as upland ponds for these resources (Waldien and Hayes 2001; Wilhide et al. 1998). Bat activity occurs more often in the vicinity of still water than moving water (Krusic et al. 1996). Areas of the forest that are less structurally complex than surrounding areas and that have small openings support a higher abundance and diversity of bats. Therefore, it would seem advisable to incorporate upland ponds in these areas. This would provide additional drinking sites and foraging areas for bats, an effect that could potentially reduce energy expenditures and competition for available resources.

Forest managers should incorporate small openings or clearings, upland ponds, roads, and trails in creating less structurally complex forest, *i.e.*, wildlife stand improvements and prescribed burning in their management plans. Using these management techniques as well as many others, habitat diversity and relative abundance would be created and possibly maintained. This would provide more niches that would benefit multiple floral and faunal species. Specifically, this could affect diversity and abundance of bats. Data collected during this investigation indicate that this would be beneficial to bats by creating suitable habitats that provide more niches for bats to maneuver through while foraging and migrating. Furthermore, it would encourage greater abundance and diversity of bats in these areas providing long-term sustainability of the bat species that occur in northwest Arkansas.

CHAPTER 3

EFFECTS OF WILDLIFE STAND IMPROVEMENTS AND PRESCRIBED BURNING ON THE ROOSTING ECOLOGY OF THE NORTHERN LONG-EARED BAT (*MYOTIS SEPTENTRIONALIS*)

ABSTRACT

Beginning in 2001, forest managers in the Buffalo Ranger District (BRD) of the Ozark-St. Francis National Forest (ONF) in the Boston Mountains of northwest Arkansas began an alteration of forest stands within a watershed in the northwestern portion of the district. In an attempt to restore an oak-woodland habitat, the basal area within the selected watershed was reduced by 50% by using wildlife stand improvements (WSI) and prescribed burning (PB). Impacts of this forest management strategy on bat populations were evaluated by comparing two sites treated with WSI and PB to two untreated sites. Various aspects of the roosting ecology of female northern long-eared bats (*Myotis septentrionalis*) were evaluated in respect to treated and untreated sites and changes in the relative forest stand density. Transmitters were placed on 33 individuals and 259 roosts were located. My results indicate that these forest management strategies had no impact on bats in treated regions of the BRD. Moreover, these data suggested that bats continued and/or began to use areas where the basal area had been reduced as a result of this management strategy. Roost density and foraging distances were negatively correlated with forest stand densities indicating that bats utilized areas of the forest that are less structurally complex.

INTRODUCTION

Because of the amount of time allocated to roosting (>50%/day), suitable roosts are a crucial resource for sustaining populations of forest dwelling bats (Brigham et al. 1997). The availability of suitable roost trees may be limiting and may be the primary factor in habitat selection for bats (Kunz 1982; Humphery 1975).

Most detailed studies that have focused on habitat and roost site characteristics have emphasized evaluating the physical characteristics of the roost microhabitat (Mager and Nelson 2001; Foster and Kurta, 1999; Kalcounis and Hecker, 1996; Kurta et al. 1993a; Barclay et al. 1988; Barclay 1982). Such characterizations frequently include qualitative and quantitative measures of the area within the roost cavity and include evaluation of such variables as roost tree species, temperature, and other climatic features of the bat roost. Others have focused their investigations on characterizing the microhabitat surrounding the roost (e.g., Menzel et al. 2001; Foster and Kurta 1999; Crampton and Barclay 1998; Menzel et al. 1998; Sasse and Pekins 1996; Vonhof 1996; Brigham 1991). Typically characterizations of the area surrounding an individual roost include measurements of roost tree diameter at breast height (DBH), percent coverage of the canopy, mid-story, understory, and ground cover, and, most relevant to this study, forest density.

Tree species, height, DBH of roost trees, and whether the roost tree is live or dead are examples of roost parameters that appear to be selected by forest dwelling bats. Bats preferentially select roost trees based on the relative diameter and height of these trees (Brigham et al. 1997; Betts 1996; Lewis 1995). Indiana bats (*M. sodalis*) prefer roost trees ranging from 9.4 cm to 86 cm (mean of 33.5 cm) DBH (MacGregor et al.1998). Northern long-eared bats (*M. septentrionalis*) (Sasse and Pekins 1996), big brown bats (*Eptesicus fuscus),* and silver-haired bats (*Lasionycteris noctivagans*) (Betts 1996) select trees within this size range.

With the exception of the *Lasiurus* species, which roost in the canopy, trees that are dead and/or dying (snags) provide potential suitable roost-sites for bats. However, live trees are also used for roosts by bat species if certain key characteristics including cavities and/or exfoliating bark are present (Kalcounis and Hecker 1996). Considerable intraspecific variation exists in little brown bat roost-site selection with bats roosting in both live and dead trees. Dead trees are used to a greater extent than live trees, because these trees absorb and release heat more readily (Humphrey et al. 1977). Roost site selection by several temperate forest dwelling bat species, regardless of their geographic location, is determined primarily by the presence of cavities and/or exfoliating bark, (Perkins 1996), relative tree size (Brigham et al. 1997; Betts 1996; Lewis 1995), and relative forest density surrounding the roost (Vonhof 1996; Sasse and Pekins 1996; Gardner 1990).

The microhabitat within an individual bat roost potentially is the net result of the characteristics of the habitat within approximately 0.1 ha plot surrounding the roost (Callahan et. al. 1997). One of the most important features of the area surrounding a potential roost may be its structural complexity. Structural complexity may be significant to bat species in both the region immediately surrounding the roost and in areas (i.e., watershed, hillside, ridgeline, etc.) through which the bats may travel or forage. Several tree roosting temperate forest dwelling bat species prefer roost trees that are relatively distant from surrounding trees. This maintains an open canopy in the immediate vicinity of the roost (Betts 1996; Sasse and Pekins 1996; Vonhof 1996). Therefore, bats may select roosts that are in areas of the forest that are relative less structurally complex. However, considering the number of temperate bat species, there have been few reports of studies that have investigated characteristics of the larger microhabitat. Structural complexity of a forest is defined as the relative density, separately and collectively of the vertical and horizontal strata (e.g., Heck and Wetstone 1977; Wilson 1974; Kohn 1967). In this study, forest stand density (FSD), as estimated by basal area, was used as a measure of structural complexity. By definition, forested habitats that are more structurally complex have less available open space than less complex habitats. Open space is a vital resource for animals, such as bats, that fly. It seems likely that the echolocation used by temperate insectivorous bats to navigate and forage could be detrimentally affected by clutter. Therefore, areas within the forest that are denser may contain more obstacles a flying bat must detect and avoid (Fenton 1990). Thus, forest stand density may be one of several ecological factors for which bats preferentially selected while searching for suitable roost sites.

Bats may reduce energetic expenditures by searching for roost sites in relatively more open areas in the forest (Speakman and Racey 1991). Preferentially locating roosts in more open areas of the forest could also reduce the risk of aerial predation by hawks, falcons, and owls (Fenton et al. 1994; Barclay et al. 1982) by reducing flight and thus, exposure time (Vonhof 1996). Forest stands that are relatively open may also provide female bats with thermal and energetic benefits associated with increased solar exposure (Trune and Slobodchikoff 1976). Therefore, removal of selected trees and shrubs so that FSD is reduced probably facilitates flight and maneuverability as bats travel below the dominant canopy. Such forest management strategies can also be used to maintain relative high densities of larger trees and snags needed for roosts.

Beginning in 2001, managers of the BRD (Figure 3.1) began an alteration of the forest within a watershed located on the northwestern portion of this district (Figure 3.2), by using wildlife stand improvements (WSI) and prescribed burning (PB) (Figure 3.3). Wildlife stand improvements occurring in the BRD were designed to aid in the restoration of an area to its pre-settlement, historical condition, *i.e.*, an oak woodland. This management strategy emphasizes reductions in FSD. Typically FSD is measured as basal area. Stand basal area is the sum of the cross sectional areas (DBH) of all trees in a hectare and is usually expressed in square meters per hectare. Prior to implementing this management strategy, the areas that were to be altered had a basal area of approximately 18-21 m^2/ha. Guidelines established for WSI on the BRD were that the midstory was to be reduced from a basal area of 18-21 m^2/ha to 7-9 m^2/ha. Prescribed burning occurred after the midstory reduction and, in order to maintain this habitat type and is scheduled to continue on a three to five-year rotation.

The combination of these two practices is expected to alter the age, reduce the stratification of canopy layers, and reduce the structural complexity of portions of the BRD. By selective removal of trees, WSI can reduce the midstory and understory, while prescribed burning maintains this open habitat type. Together, these management

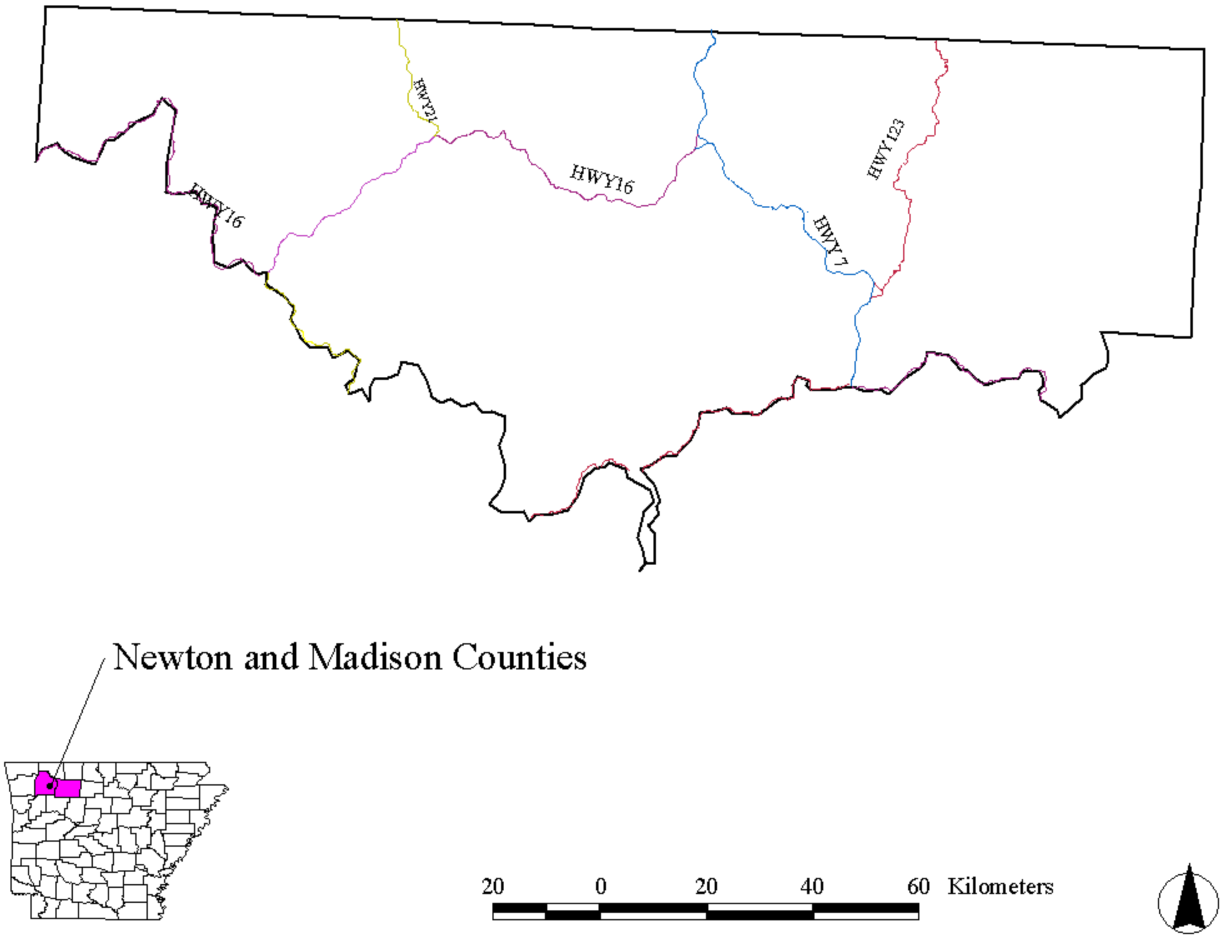

Figure 3.1. The location of the Buffalo Ranger District, Ozark-St. Francis National Forest, Arkansas.

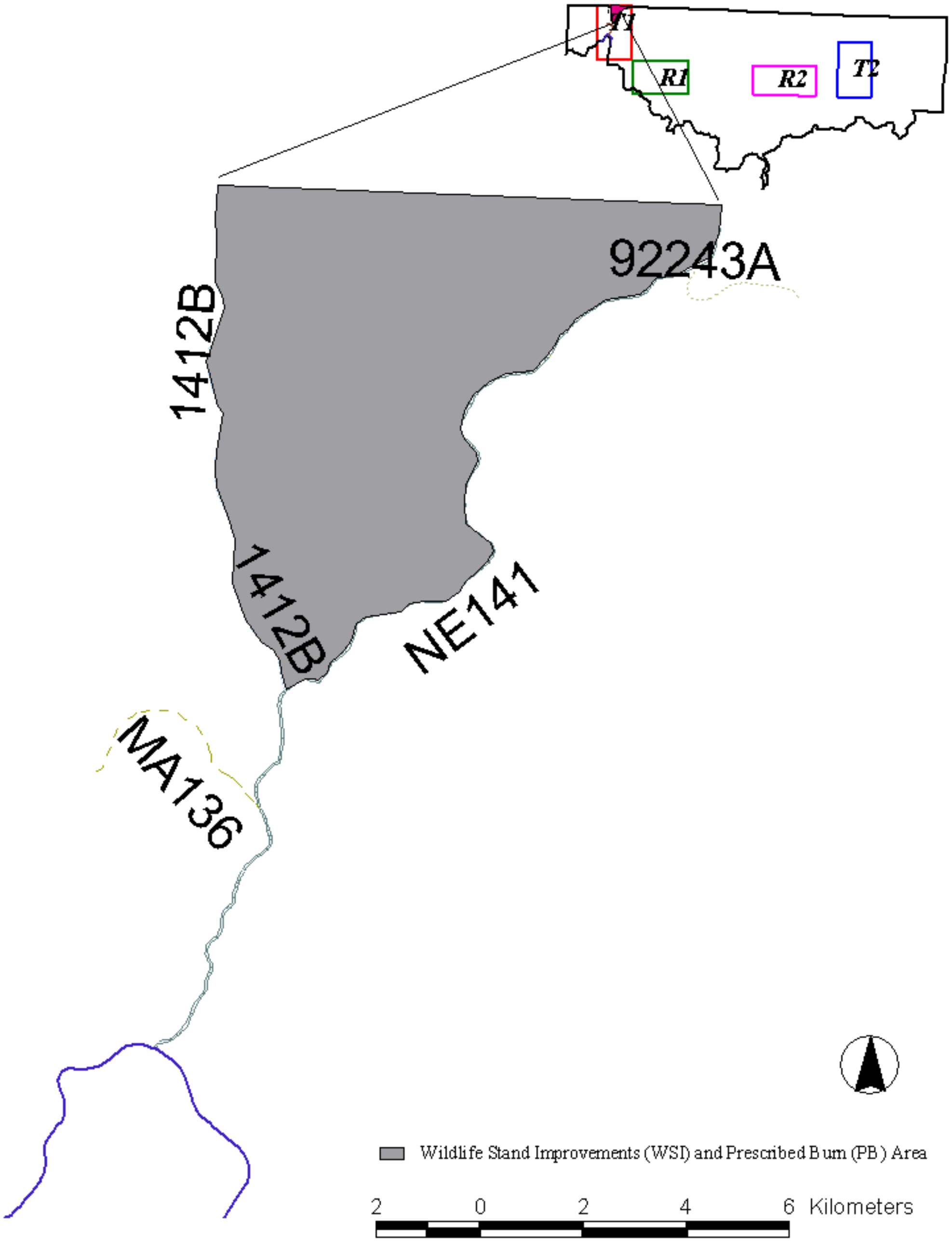

Figure 3.2. The area treated with wildlife stand improvements and prescribed burning.

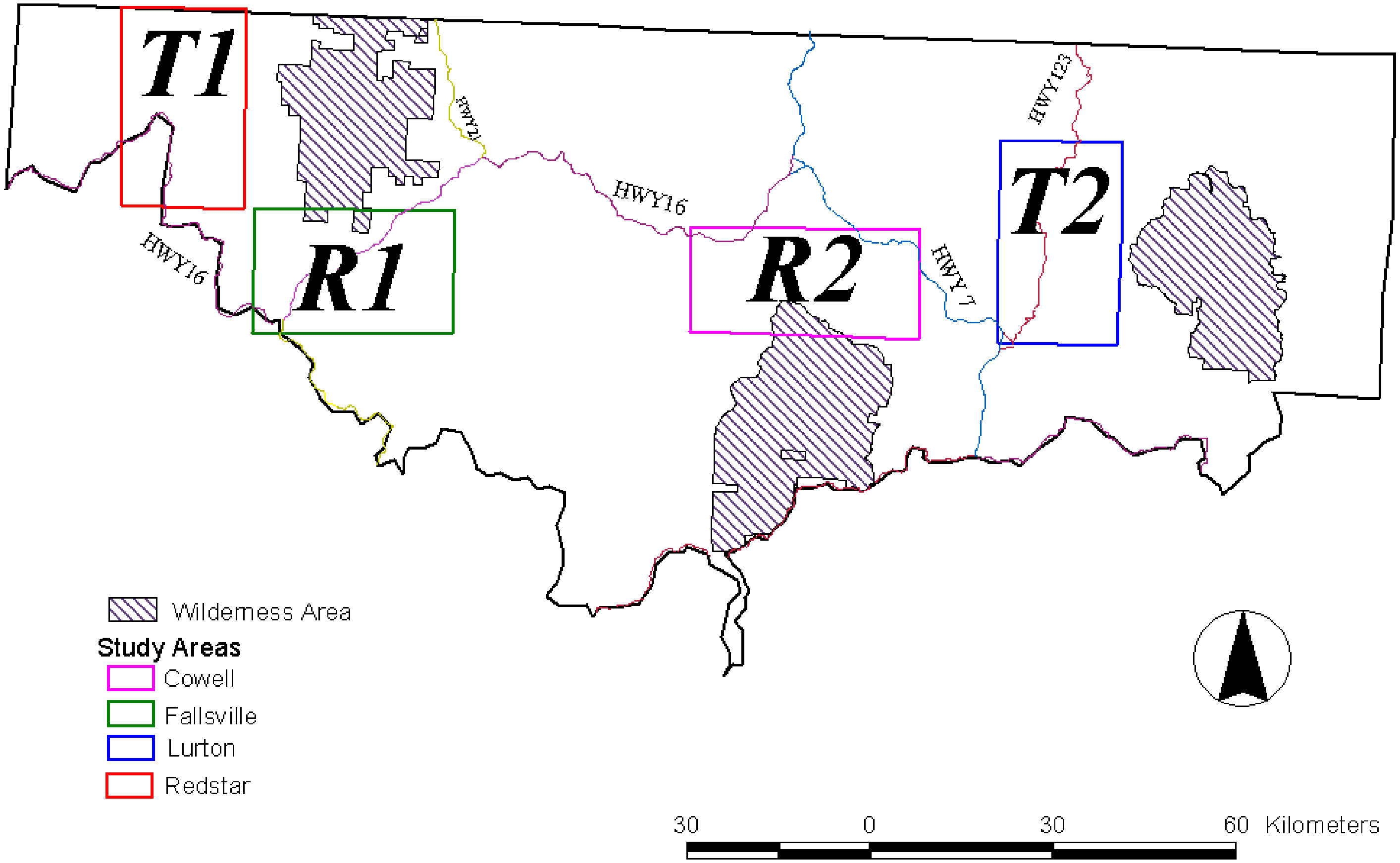

Figure 3.3 Locations of each study-site and wilderness area

strategies may have a positive effect on bat populations by creating more open horizontal and vertical stratification layers, resulting in flyways and foraging areas for bats to maneuver through more efficiently, allowing greater access to available food, water and roost resources. However, no studies have evaluated how WSI and PB affect bat populations and have not specifically addressed roost locality. Since bats ability to move through the forest would be enhanced by reductions in FSD, it was predicted that distances between roost trees (roost-tree density) for any individual bat would increase as FSD decreased. Moreover, the distance between point of capture and roost locality (minimum foraging distance) should be greater in areas of the forest that have been treated with WSI, PB, and in areas that have lower FSD.

Purpose of the Study

The purposes of this study were to assess the effects of WSI and PB on: (1) roost tree density (2) and minimum foraging distance of the northern long-eared bat, and (3) characterize the effects that WSI and PB has on the FSD within areas associated with roost site locality.

METHODS AND MATERIALS

Study Area

This study was conducted on the BRD in the ONF, in the Boston Mountains of northwest Arkansas (Figure 3.1). The BRD comprises approximately 1,351,348 ha of which 168,737 ha are designated wilderness areas (Figure 3.3). Dominant overstory species are deciduous hardwoods, including oaks (*Quercus* spp.) and hickories (*Carya* spp.), with conifers (*Pinus* spp.) distributed throughout the district. The age of the forest within the BRD rages from 0 to > 100 years (from fields and 1st year regeneration stands to mature old growth forest stands). The district contains many wildlife openings and man-made and natural ponds (n = 293) at an average density of 1 pond/50 km^2. Previous bat capture data (Wilhide et. al. 1998) in the ONF may suggest that the current forest structure in some areas provide suitable habitat for bats. However, based on echomorphology and wing morphology, the relative high vegetative (structural complexity) density of some areas of the forest may be a limiting factor on the size of the bat communities by restricting access to available resources.

Experimental Design

To assess the effects WSI and PB have on the roosting ecology of the female northern long-eared bat four study sites with in the BRD were established (Figure 3.3): Redstar (T1), Fallsville (R1), Lurton (T2), and Cowell (R2). Each site was approximately 60,000 ha in size and each had similar vegetative composition (mixture of hardwoods and conifers). The age of the forest at each study site ranged from 0 to > 100 years old (US Forest Service). Ponds less than 0.5 ha in size are distributed throughout each of the study sites at an average density of 0.0003 ponds/ha (US Forest Service).

T1 was located on the west-central side of the district. It was subject to a WSI in fall of 2001 and a PB in the spring of 2002 (Figure 3.2 and 3.3). T2 is approximately 118 km east of T1 (Figure 3.3). A thinning, similar to a WSI and PB were completed at T2 during 1999, during which the basal area was reduced. Thus, treatment-site T2 was in its 4th year of regeneration when data were collected in 2002. The WSI at sites T1 and T2 were the same except that trees were cut and then removed from T2, while at T1 cut trees were left where they fell. R1 was approximately 2 km east of T1 (Figure 3.3). R2 was approximately 12 km to the west of T2 (Figure 3.3). At each reference site, no habitat alteration strategies were used. Reference sites were selected so they were relatively close to each treatment site and so that there were approximately equal distances between study sites and wilderness areas (Figure 3.3). Although this study designed had low site replication with treatment sites serving as replicates, this study site selection strategy was designed to minimize any local geographic and topographic variation between treatment and reference sites. It was also designed to reduce the effects that wilderness areas (preservation areas) may have on bat roost distribution.

Method of Capture

Bats were collected using black nylon mist nets (38 mm mesh) that were 2.6 m x 4 m, 2.6 m x 6 m, 2.6 m x 9 m, or 7.8 m x 12 m (Kunz and Kurta 1988). Bats were sampled two nights per week at a treatment and reference site with weekly rotation among the sites. Sampling was designed to insure even sampling among sites during each year. Nets were placed over ponds, creeks, and perennial road ruts holding water. Nets were also placed in dry areas such as roads and in entrances to fields and ponds. By netting these entrance points bats were captured that were using the roads and fields as foraging areas but that might have been missed while netting over water structures. Nets were opened approximately 15 minutes until the rate of capture fell below one bat per two hours. While nets were open, the capture time, species, sex, and band presence of each captured bat was recorded. Bats were separated by species and placed in 100% cotton bags. When sampling was completed for the night, measurements were taken and captured bats were banded by placing plastic split ring bands, numbered in sequential order, were placed on the forearm of all bats. Data collected included: 1) species, sex, and age of each animal; 2) the reproductive condition of each animal, (males-- nonreproductive or scrotal; females--nonreproductive, pregnant, lactating, or post-lactating); 3) forearm length and weight were measured.

The specific location of each net site was determined using Global Positioning System (GPS) coordinates, date, times nets were open and closed, weather conditions, and habitat type were recorded during each sampling effort. All GPS coordinates were recorded in Universal Transverse Mercator (UTM) and North American Datum 1983 (NAD 83), Zone 15 North.

Forest Stand Density

The relative FSD was calculated by measuring the basal area in the areas surrounding roost site locations by establishing four sub plots that were 0.25 ha in size, (approximately 17 m in diameter) in four cardinal directions 50 m from the structure or area that was being sampled. Tree basal area was calculated by multiplying 0.000078539 by the diameter at breast height squared (DBH^2) of all trees ≥10 cm in each subplot (Note: 0.000078539 = pi/40,000) where the division by 40,000 corrects for the difference in cm and diameter to radius). Stand basal area was determined dividing the total basal area for each tree in each plot by 1 ha (4 *0.25ha plots).

Radio Tracking

Two randomly selected adult, female northern long-eared bats were chosen at three ponds selected out of 35 available ponds in each study site for tracking using radio transmitters (Figure 3.4). Individuals selected for tracking were fitted with a 0.53 g, series LB-2 transmitter with frequencies in the 218 MHz range (Holohil Systems Ltd., Ontario Canada), with a maximum transmitter life of 20 days. Transmitters were attached to the mid-sagittal region on the dorsal surface between the scapulae with surgical glue after the hair was trimmed (Figures 3.5 and 3.6).

Model TRX-1000 tracking receivers (Wildlife Material Inc., Carbondale, Illinois) with collapsible, three-element Yagi antennas (AF Antronics, Inc., White Heath, Illinois) were used to locate bats in roosts. Tracking bats began the following morning after transmitter placement and was tracked for the life of the transmitter and/or the bat could not be located.

Habitat Characterization

The dominant tree species present within stands associated with roost tree locations were identified using existing forest service data verified by ground truthing. A clinometer was used to estimate height of each roost tree.

Within each plot and subplot the percentage of canopy and midstory closure was estimated using a concave densiometer, and the percentage ground cover was estimated visually in relation to the percent canopy and midstory density.

Roost-tree density was estimated by measuring the distance (Spider Distance) between individual roost-trees for each radio-tracked bat. This was calculated in spatial

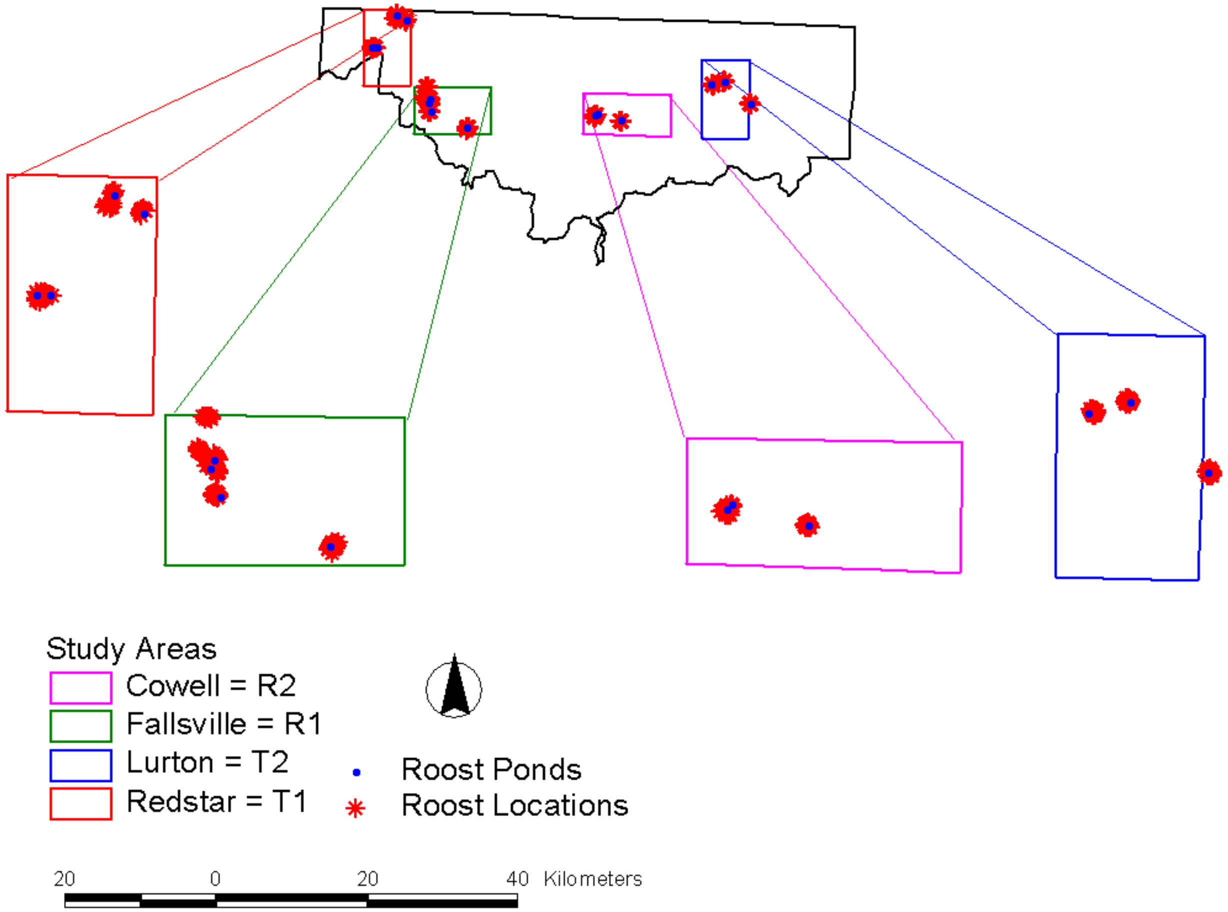

Figure 3.4. Distribution of roost and roost ponds, (sampling locations where bats were fitted with transmitters) within each study site.

Figure 3.5. Illustration transmitter preparation, i.e., soldering power wires and applying surgical glue (Photo Taken 18 May 2002).

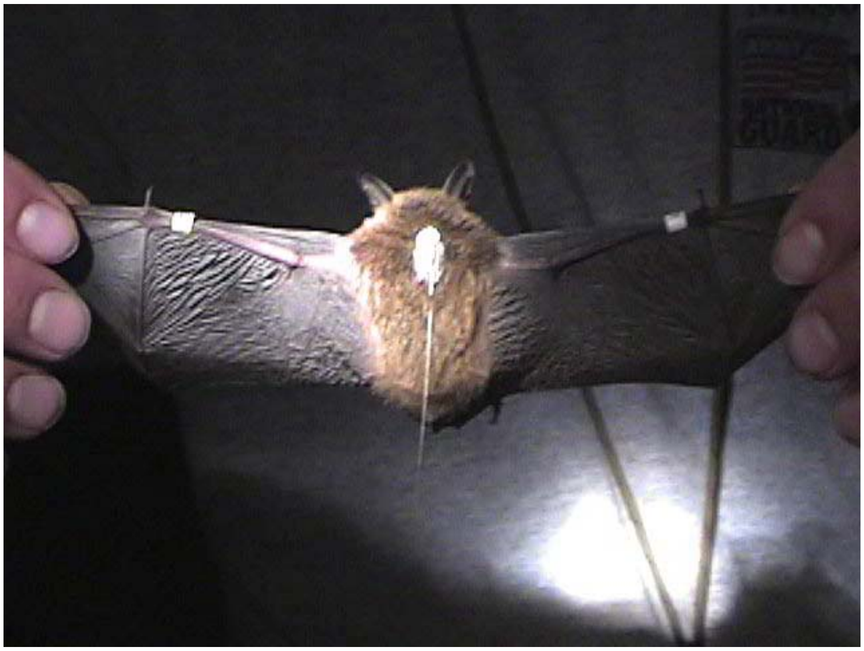

Figure 3.6. Illustration of transmitter placement, on a female northern long-eared bat (Photo taken 18 May 2002).

analysis software (ArcView 3.2), by calculating a reference point (centroid) spatially centered between all roost trees for an individual bat. Spider distance, the sum of the distances between the centroid and each roost-tree was calculated (Figure 3.7).

Distances between the nearest permanent water source and each roost tree were used in order to determine an estimated minimal foraging distances (Brigham 1991). Spatial analysis software (ArcView 3.2) was used to estimate roost distribution in close proximity to water and its association with the relative FSD. Data were imported into ArcView 3.2, and layered upon USGS topographic quadrangles, ARK (7.5-minute series) for all measurements.

Data Analysis

Study-sites R1 and R2 served as reference-sites; T1 and T2 served as treatment-sites in all data analysis. Minitab Software (Minitab Inc. 1998) was used for all statistical calculations. Data were tested for normality using the Kolmogorov-Smimov Normality Test in Minitab (Minitab Inc. 1998). If the data were not normally distributed ($P > 0.15$), it was normalized by using the inverse log of the respective data columns. A *P*-value of 0.05 was selected as the significance level.

Measures of structural complexity among and within study-sites were analyzed using an analysis of variance (ANOVA) and a Tukey's HSD post hoc test was used to see where the difference occurred between the treatments. Roost-tree locations served as sampling units.

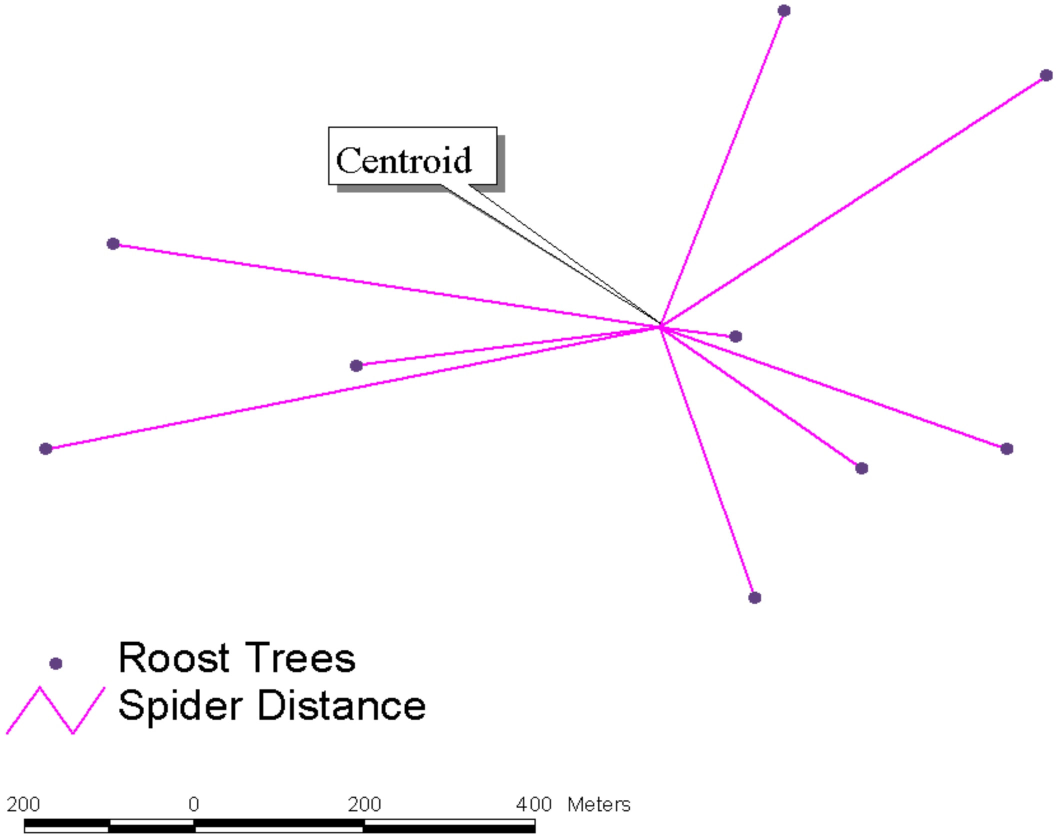

Figure 3.7. Spider distance diagram, which was the measure of roost tree density calculated in ArcView.

A 2-sample T-test was used to evaluate treatment effects on the roost-tree density, as well as the minimum foraging distance where study-sites served as sampling units. Pearson correlation analysis was used to establish the relationships between FSD and roosting ecology parameters.

RESULTS

Roost Tree Characteristics

Thirty female northern long-eared bats were radio-tracked to 259 collective roost tree localities. The majority of these roosts were located near roads, trails, and open portions within the forest interior. Briefly, these data show that this bat species tended to select white oak (55 %, range 34 – 42 trees) and red oaks (20 %, range 13 – 20 trees) at each study site. Hickories and pines were also selected, but less frequently (12 %, range 3 – 12 trees and 8 %, 1 – 12 trees respectively)(Figure 3.8).

The decay stages of roost trees were classified into three stages: including live, dead, and snag. A tree was classified as a snag if the bole remains vertical, while the dominant crown branches have fallen. Approximately 48 % of roost trees selected were snags (range n = 24 – 36 trees) and 32% were dead trees (range = 18 - 33 trees). Live trees were also selected (17 % range n = 6-20 trees); however, most of the live trees utilized by bats possessed cavities (Figure 3.9).

Roost trees tended to be either the same height as the dominant canopy (40 %, range n = 17 – 30 trees) or lower than the dominant canopy (40 %, range n = 23-29 trees) (Figure 3.10). Only 20% of the observed roosts were above the dominant canopy (range 7 – 27 trees) (Figure 3.10).

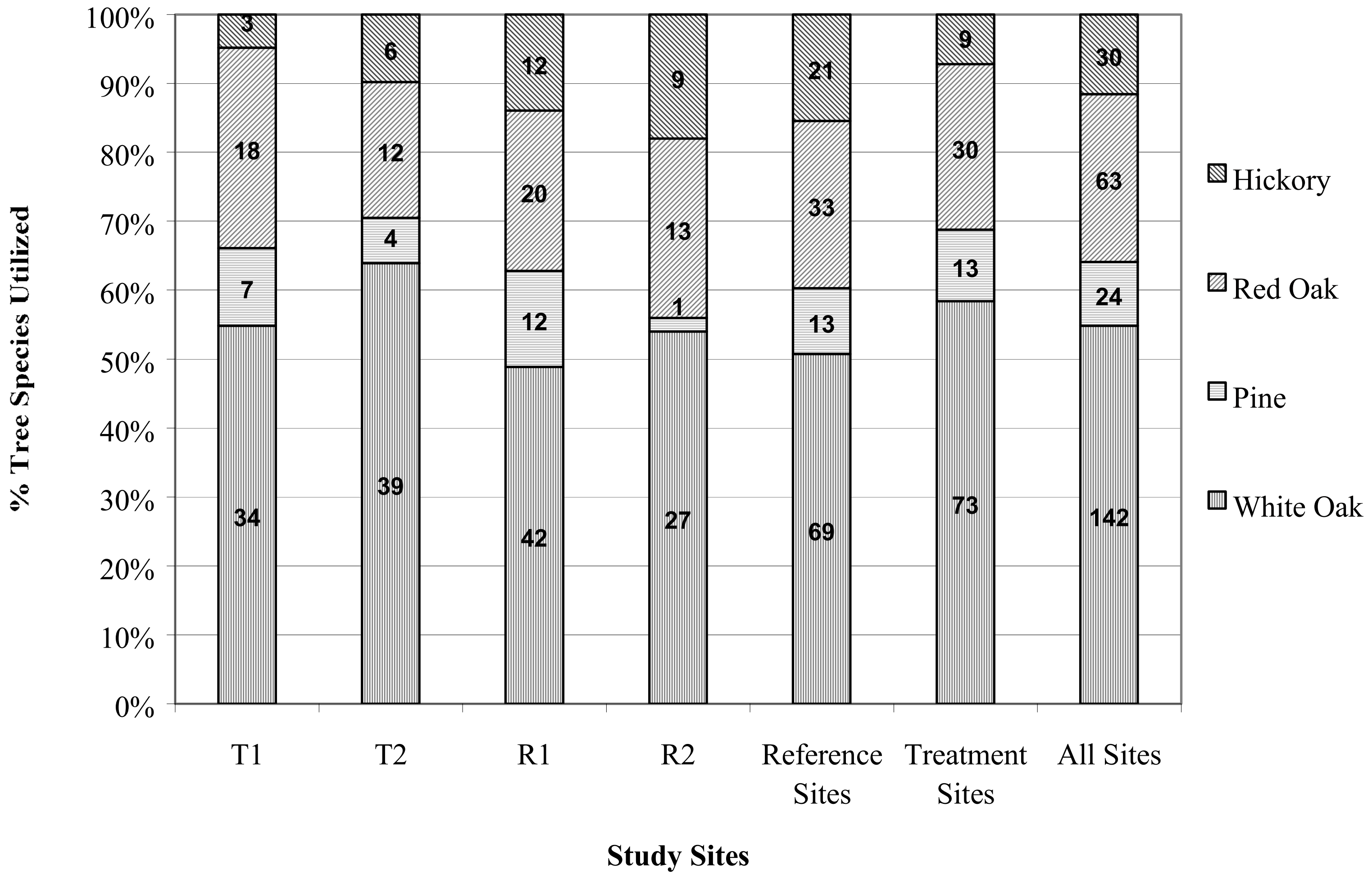

Figure 3.8. Percentage of trees used for roost-sites by individual bats within and among study sites.

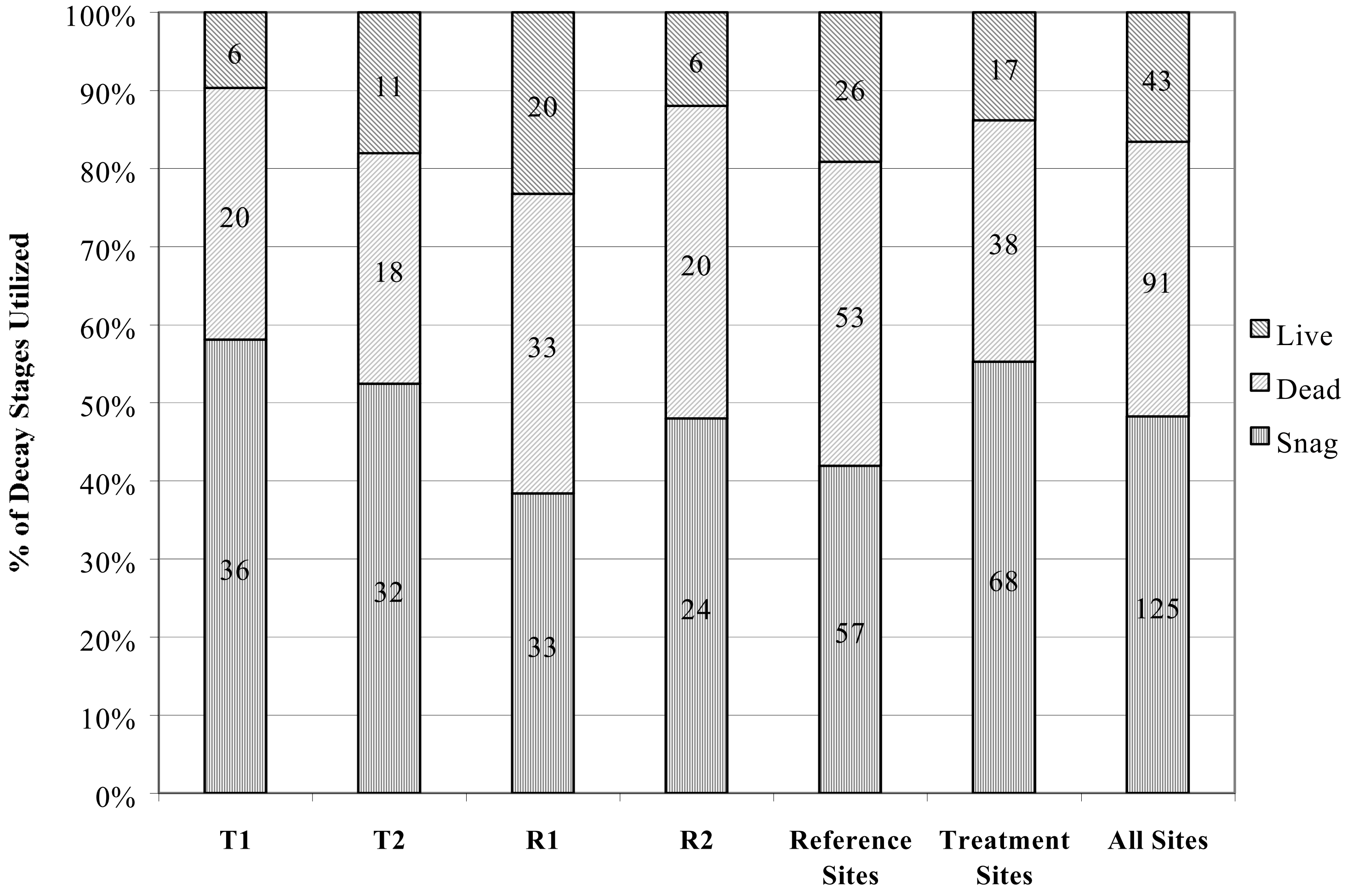

Figure 3.9. Percentage of trees at various stages of decay utilized by individual bats within and among study sites.

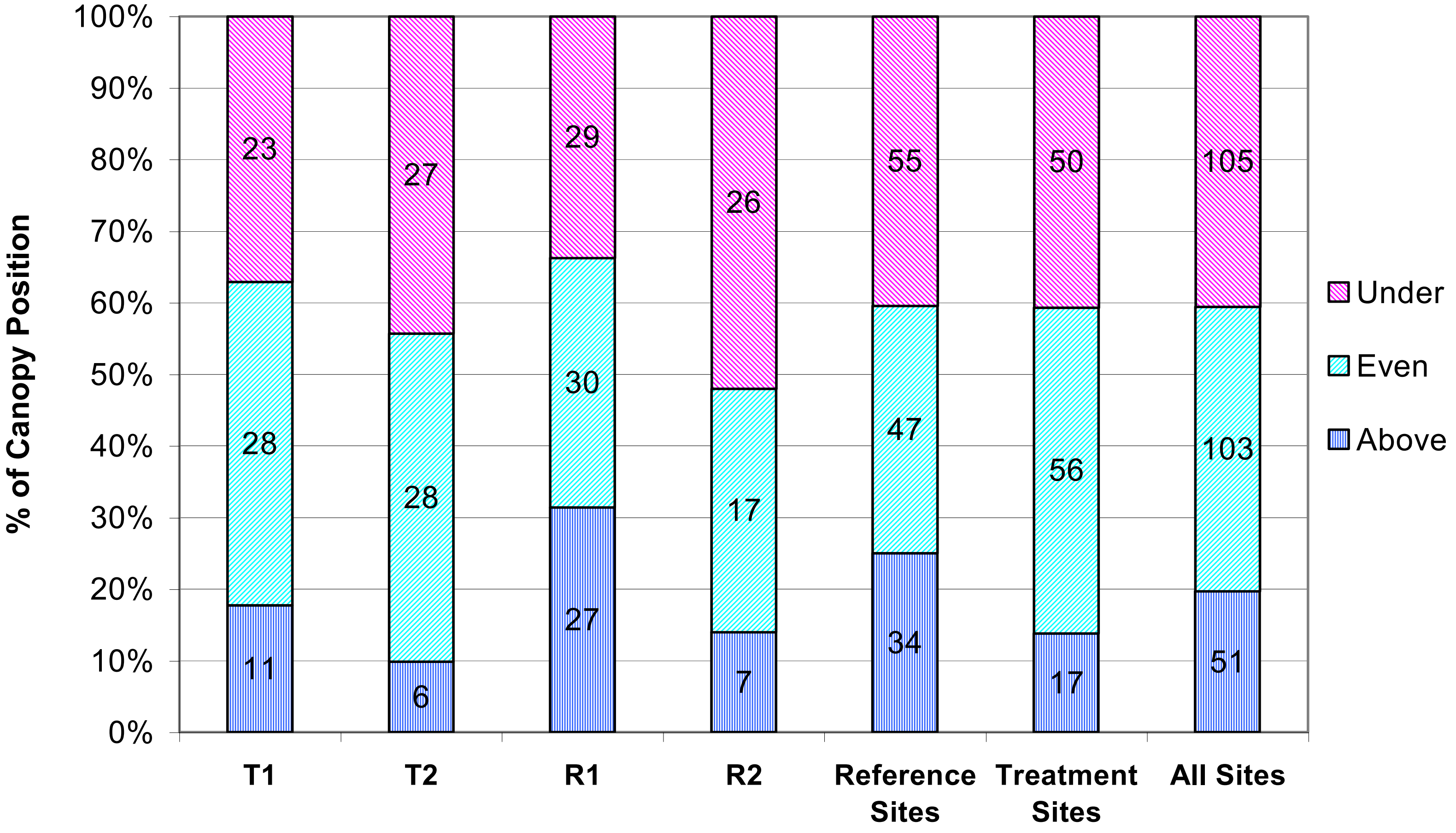

Figure 3.10. Position of roost trees of individual bats in relation to its location in the canopy based its spatial relationship to trees the immediate vicinity.

Treatment Effects on Forest Stand Density

There were significant differences between the mean FSD at roost-tree locations and study-sites. The mean FSD within each reference site (R1 and R2) was significantly greater than treatment sites (T1 and T2) (Table 3.1).

Significant differences were also observed in the horizontal strata among and within each study-site, (*P*-value< 0.05) (Table 3.1). The percentage of ground cover was significantly greater at the treatment sites (T1 and T2) compared to reference sites (R1 and R2) and was inversely proportional to the percentage of midstory and canopy closure. Study sites R1 and R2 had significantly higher percentages of midstory than the treated sites. However, no statistically significant relationship between ground cover and the percentage coverage of the midstory was observed.

Treatment Effects on Roost Site Selection

The number of roost trees for female northern long-eared bat averaged 8.6 roosts/bat (range 2 – 11 roosts/bat), during the study with different roosts being utilized each night (Table 3.2). This observation suggests there was a low degree of fidelity to a given roost tree. However, distances between individual roost trees for any given radio-tagged bat suggest that there was a high degree of fidelity to a particular roosting area. These distances ranged between 37 m to 877 m for any given radio-tagged bat (Table 3.2).

Roost trees used by an individual were relatively close to each other and were located in a close proximity to water resources. Mean flight distances (MFD) between

Table 3.1. Means of the various measures of structural complexity of the forest associated with individually radio tracked bats (ANOVA).

Structural Complexity	T1 post	T2	R1	R2
n = of radio tracked Northern Long-eared bats	6	6	8	10
FSD				
Mean FSD (m^2/ha)	8.2	9.9	12.8	16.3
StDev	2.9	1.25	1.3	1.8
Difference	*a*	*a*	*b*	*b*
Ground Cover				
Mean % of Ground Cover	45	60.8	8.5	8.6
StDev	15	17.5	5.08	16.89
Difference	*a*	*a*	*b*	*b*
Mid-story Closure				
Mean % of Mid-story Closure	10.3	11.7	30.4	25.6
StDev	10.87	2.58	9.9	5.63
Difference	*a*	a	*b*	*b*
Canopy Closure				
Mean % of Canopy Closure	53.3	85	60.7	76.3
StDev	20.25	6.32	9.17	5.82
Subscript	*a*	*b*	*c*	*b, c*

Note: significant difference (P-value < 0.05) between study sites if subscript letter is different and study sites are similar if matching subscript letter and number are the same. Tukey's Family error rate = 0.05 and Individual error rate = 0.109

Table 3.2. The distances traveled between roosts for individual radio-tagged bats. The lowest minimum and highest maximum distance traveled for any individual is in **bold** print.

Site	Bat Band I.D. #	# of Roost/ Bat	Average Distance Traveled (m)	Min Distance (m)	Max Distance (m)	StDev of Distance Traveled
R1	3642	9	179.0	71.6	276.7	68.63
	3643	10	202.2	57.3	420.2	114.70
	5152	10	465.6	209.1	620.3	160.20
	5153	10	499.6	248.8	**877.3**	196.62
	5161	8	445.8	56.7	845.7	241.15
	5162	11	461.1	59.2	814.8	200.39
	5659	10	292.7	63.9	502.3	117.99
	5660	9	185.1	**36.9**	399.7	104.71
	5669	9	315.4	72.0	616.2	186.81
R2	5187	8	537.3	208.6	747.1	201.00
	5188	8	393.6	157.9	587.6	119.17
	5197	9	291.5	125.5	456.9	114.80
	5198	7	200.5	47.6	431.7	122.81
	5342	3	81.5	72.7	72.7	15.29
	5382	2	52.3	52.3	52.3	--
	5385	8	165.1	122.9	193.9	28.38
	5386	5	126.8	59.6	174.5	50.83
T1	5035	8	359.6	77.7	570.0	153.33
	5347	8	223.6	21.5	367.0	115.42
	5348	10	408.4	135.8	831.0	206.58
	5355	7	245.0	98.2	410.0	93.06
	5357	8	219.8	126.3	462.2	114.67
	5652	10	188.7	110.5	297.0	77.47
	5653	11	421.5	90.9	734.4	174.92
T2	5311	13	265.4	94.0	624.0	146.41
	5326	10	359.2	81.1	555.3	124.29
	5327	8	373.2	161.3	552.1	137.67
	5370	10	268.8	83.6	485.3	128.73
	5374	11	298.5	63.6	449.3	103.20
	5673	9	343.5	206.9	533.5	124.44

roost trees and nearest water resources ranged from approximately 68 m to 1896 m (Table 3.3).

There were no differences in MFD between treatment and reference sites or between roost-tree density (Table 3.4). Standard deviations of distances were large, accounted for by high standard deviation for individual bats and relatively small sample size. Because the sampling unit (treatment and reference sites) was relatively large (approximately 60,000 ha in size) it was not feasible to increase the sample size. The relatively large size of the sampling unit was needed to account for the changes in FSD among and within study sites so that its affect on the roosting ecology of Northern long-eared bats could be evaluated.

Roost Site Selection Associated with Forest Stand Density

Since there were no detectable differences between treatment and roosting ecology parameters, data were combined the forest stand level using individual radio-tagged bats as a sampling unit. A Pearson correlation analysis was used to evaluate the relationship between roost tree density and FSD. This correlation was significant (r = -0370; *P*-value = 0.044), such that distance between individual roost trees increased as FSD decreased (Figure 3.11). Additionally, the relationship between MFD and FSD was significant (r = -0.620; *P*-value = 0.0001) (Figure 3.12) and thus, the distances to roost site locations from nearest water source for individual radio-tracked bats increased as FSD decreased (Table 3.3 and 3.4).

Table 3.3. The distances traveled to the nearest permanent water source for individual radio tagged bats. The lowest minimum and highest maximum distance traveled for any individual is in **bold** print.

Site	Bat Band I.D. #	# of Roost / Bat	Ave Distance to Nearest Water Source (m)	Min Distance to Nearest Water Source (m)	Max Distance to Nearest Water Source (m)	St Dev of Distance to Nearest Water Source (m)
R1	3642	9	977.26	780.19	1109.62	106.18
	3643	10	2372.35	2098.21	**2724.14**	192.23
	5152	10	830.77	319.29	1183.37	279.79
	5153	10	587.64	314.76	941.58	179.69
	5161	8	704.76	311.49	1265.85	332.87
	5162	11	570.11	**68.01**	1080.16	328.74
	5659	10	821.40	481.36	1041.23	202.12
	5660	9	492.75	260.66	846.89	175.53
	5669	9	530.49	313.34	847.24	156.78
R2	5187	8	537.26	209.76	751.27	201.62
	5188	8	434.44	253.40	633.02	116.00
	5197	9	299.02	130.87	398.91	109.98
	5198	7	292.20	201.39	439.32	84.71
	5342	3	184.19	104.96	245.81	72.06
	5382	2	118.84	117.78	119.90	1.50
	5385	8	362.17	171.14	479.52	110.23
	5386	5	371.45	277.40	465.02	78.69
T1	5035	8	641.33	293.63	853.61	204.17
	5347	8	634.10	451.15	949.86	174.02
	5348	10	656.13	232.37	1193.87	322.11
	5355	7	789.70	404.45	997.74	227.61
	5357	8	387.69	326.26	435.00	45.28
	5652	10	528.24	235.28	732.68	155.02
	5653	11	1436.85	982.34	1896.80	319.05
T2	5311	13	803.28	534.81	874.30	107.93
	5326	10	445.76	289.22	657.80	126.30
	5327	8	459.49	279.57	648.99	142.37
	5370	10	957.46	669.13	1201.07	186.63
	5374	11	383.13	81.84	720.33	212.99
	5673	9	551.59	334.09	863.89	147.73

Table 3.4. Mean roost tree characteristics at treated and untreated study-sites (2-sample *T*-test).

	Treatment Sites (*n*=2)		**Reference Sites (*n*=2)**		
	Mean	StDev	Mean	StDev	*P* - value
Roost Density					
Average Spider Distance (m)	306.7	16.20	284.8	76.00	0.729
Minimum Foraging Distance					
Distance to Nearest Water Source (m)	662.6	90.90	600	390.00	0.846

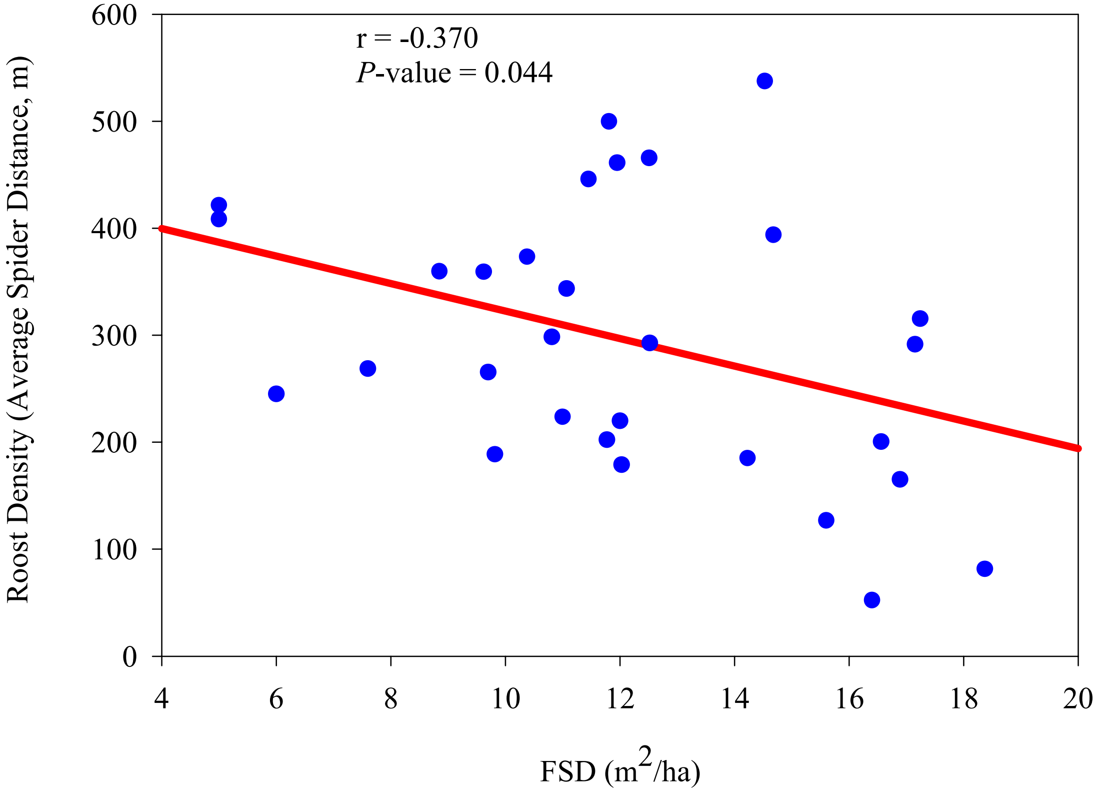

Figure 3.11. Associations between forest stand densities and roost tree density

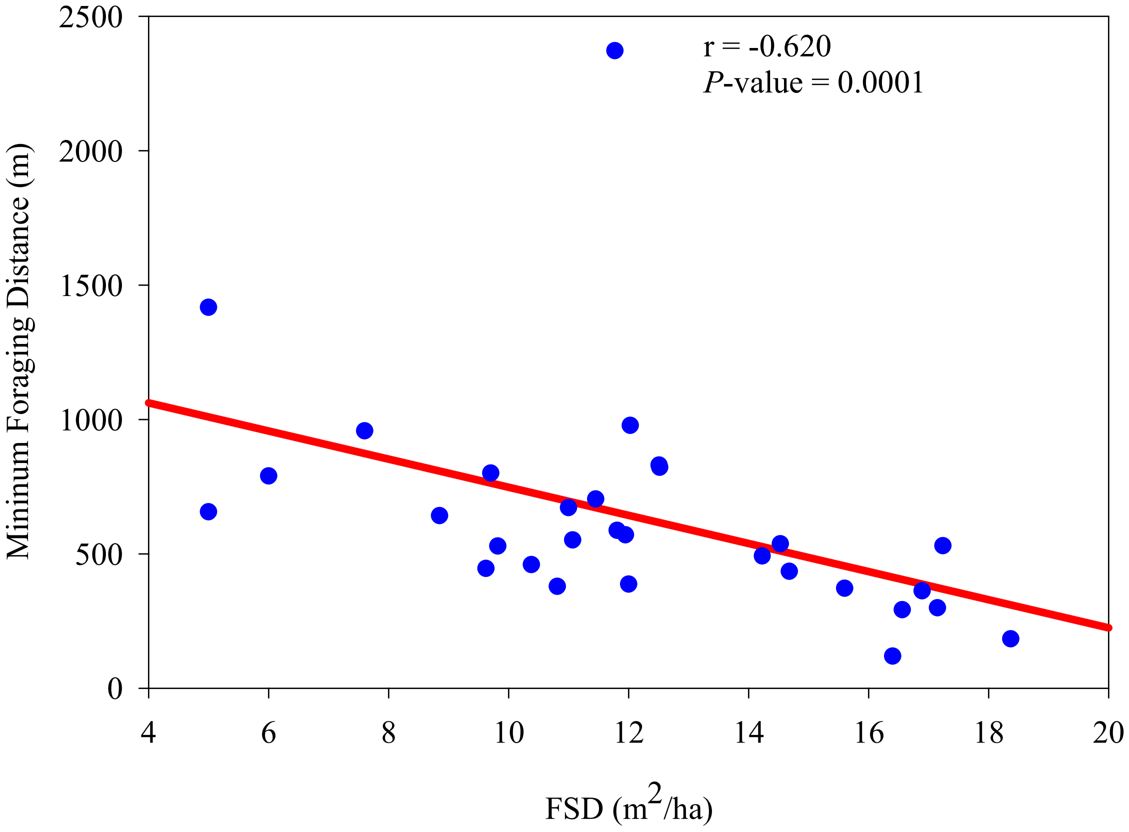

Figure 3.12. Associations between forest stand densities and minimum foraging distance.

DISCUSSION

Structural Complexity

As predicted, my treatment sites had significantly lower FSD and increase in herbaceous plants. Thus, in this respect the WSI and PB were successful. Study-sites T1 and T2 were significantly different from each of the reference-sites (R1, and R2). The percentage of midstory coverage was lower and the percentage of herbaceous plant coverage higher in the treated areas. These changes in structural complexity were expected and probably occurred because the reduction of FSD permitted more sunlight to reach the previously shaded forest floor.

Roosting Ecology

Roosts are a limiting factor in bat populations (Lewis 1995; Brigham and Fenton 1986; Kunz 1982; Humphrey and Cope 1976). In some locations bats reproductive success could be limited due to the lack of available suitable roosts (Brigham and Fenton 1986). Fidelity to a particular roost is often used as a measure of roost suitability (e.g., Humphrey 1975). During this study, northern long-eared bats changed roosts nearly every day, demonstrating that in the BRD at least, this species exhibits a high degree of roost liability. Roost availability and permanency have been proposed as factors that affect roost fidelity (Brigham 1991). Although individual females frequently changed roosts, they also showed a high degree of fidelity to particular areas, often selecting trees within close proximity (40 m) to prior roosts.

Therefore, these data suggest that the benefits of switching roost must exceed the costs associated with locating and using several difference roost trees (Lewis 1995). Causes of possible roost switching include disturbance (Kunz 1982), predation (Wilkinson 1985), foraging area (Fleming 1988), parasitism (Fleming 1988; and Wilkinson 1985), and microclimate/habitat (Fleming 1988; Kunz 1982).

Undisturbed animals should have high site fidelity and increasing predator (Wilkinson 1985) or man (Kunz 1982) induced disturbance should correlate with lower site fidelity. In this study factors that may have affected roost fidelity were not examined, but it seems likely that bats could have been affected by various disturbances, either from presence of predators. Disturbances probably not a major influence during this study because each tracked bat demonstrated the same behavior of daily roost switching throughout the study site and period. Moreover, northern long-eared bats are often observed roosting within and/or among highly disturbed buildings, bridges, and other man-made structures. Although they switched roosts daily bats were observed to stay within the same general area, sometimes only moving approximately 10 m from the previous roost. This suggests that bats may have established a territory within the forest. If this is true activities associated with administering WSI and PB and/or data collection probably would not cause bats to move to alternative area

Parasitism could be a possible cause for roost switching. Several species of animals alter their behavior in response to parasites (e.g., Mills 1990; Barclay 1988; Hausfater and Meade 1982). Bats could potentially switch roosts frequently in order to avoid parasites, however, this has not been supported by many studies (review Lewis 1995) nor was it investigated in this study.

Microclimate/habitat has been proposed to influence site fidelity (Fleming 1988; Kunz 1982). Results of several studies, including this one, indicate that bats appear to be preferentially selecting roost based the characteristics of the area within and surrounding the roost. Parameters that may be important could include the relative size of the roost tree, roost tree decay stage, and species of the roost tree, its relative position in the canopy and its distance from surrounding trees (Brigham et al., 1997; Sasse and Pekins 1996). However, microclimate/habitat of the roost may be less important than the surrounding habitat and the overall forest composition. Parameters like dominant tree species, average forest age, and stand level structural complexity may be very important in roost selection. Therefore, while bats may appear to select roost based on roost tree characteristics; they may instead be selecting roost based on stand level habitat characteristics. Although in this study no significant effects of treatment types on roost-site selection were observed, the relationship between FSD and northern long-eared bats roost tree density and their proximity to water resources are evident.

The relationships described by the analysis of combined data at the forest stand level were pseudoreplicated, because these forest stands were selected from existing treatment and reference sites. This was unavoidable in order to evaluate the relationships between roost tree density and MFD to FSD, while documenting any treatment effects. Therefore, the data herein describe the relationships between FSD, roost-tree density and the minimum foraging distance that were identified during this study. My observation that minimum foraging distance decreased as the relative FSD increased agrees with several other studies that evaluated the effects of physical structure (e.g., Krusic and Neefus 1996) and abundance of clutter (Mackey and Barclay

1989) on the distribution of foraging bats. The relationships between FSD and the roost parameters, *i.e.*, roost-tree density and minimum foraging distance, that were observed during this study also indicates that bats tend to avoid densely vegetated mid-and understories, and prefer less structurally complex habitats, consist with other studies (Betts 1996; Kalcounis 1994). Additionally, predation rates decrease in less dense forest habitats (Barclay et al. 1982). Probably because bats are able to exit and enter roost at a faster rate and because their ability to locate new alternative roost increases, while decreasing energy expenditures (Vonhof 1996).

Bats ability to locate new and alternative roost could be detrimentally effected by the structural complexity of a forest. There are more obstacles bats must detect and avoid in relatively more structurally complex forest (Fenton 1990). Therefore, dense habitats may limit bats view of suitable roost trees below the canopy by reducing acoustical and visual acuity. Flight is costly (Speakman and Racey 1991) and energetic expenditures associated with travel through more structurally complex habitats could be detrimental to bats. Therefore, bats often select trees that are high above the canopy probably to reduce impacts of dense vegetation in the understory (Betts 1996; Vonhof 1996; Kalcounis 1994). This may explain why there was a negative relationship between northern long-eared bats roost site selection and FSD during this study. Roost trees were mostly located even and/or bellow the dominant canopy where the forest is less dense. This provided bats enough open space for exiting roost.

It therefore seems feasible that clear flight path to the roost and its entrance would be advantageous (Vonhof 1996) and allow more efficient foraging in the interior portions of the forest. For this reason bats often used roads and trails that are distributed throughout the forest as flight and foraging corridors (Krusic and Neefus 1996). These roads could have provided bats that were radio-tagged within both reference areas access to portions of the forest that would otherwise been inaccessible, due to the complexity of the forest in these areas. This could potentially cause relationships between FSD, RTD and MFD associated with reference areas to be underestimated. Thus, bats may be able to forage in areas that have a relatively high FSD by utilizing relatively open linear habitats such as roads and/or trails has possible flight corridors.

MANAGEMENT IMPLICATIONS

Bats in the ONF use several types of forested and non-forested habitats. These habitats provide areas for foraging, predators avoidance and as protection from extreme environmental conditions. However, because of an emphasis on reduced timber harvesting and an increase in fire suppression many forests have become increasingly dense. Forest management incorporating wildlife stand improvements and prescribed burning appear to be beneficial for forest dwelling insectivorous bats. Using these two management techniques in concert produces a less structurally complex forest with reduced forest stand densities that potentially benefit bats.

Forests are subjected to natural tree fall resulting from aging, wind, and insect damages creating snags and small openings in the forest that may provide flight corridors for bats. Conventional timber harvesting, as well as wildlife stand improvements could potentially simulate natural mortality, while maintaining relatively high snag availability.

Fire regimes were a part of the pre-settlement natural ecosystems in this National Forest pre-settlement. One direct benefit of fires is the increase nutrients that are cycled in forest ecosystems from the ash (Fogel and Cromak 1977). This increase in soil nutrients aids in the sprouting and growth of grasses, forbs, and shrubs and the regeneration of woody shoots. Regeneration of trees is vital to forest ecosystems as it allows for younger trees to replace older trees that have died and eventually have fallen due to natural mortality. Moreover fire produces a more heterogeneous forest setting, aids in creating a stratified forest, and helps maintain a relatively open midstory. Prescribed burning and wildlife stand improvements stimulate the development of this type of forest setting. Conducting prescribed burns on a three-year rotation after a WSI has been administered allows the mid-and understory to remain relatively open. This will also prevent portions of the forest from becoming structurally complex with the associated negative impacts on bat communities.

Upland ponds should be created in areas that lack abundant water resources. Bats selected roosts trees and roosting areas within close proximity water resources and were faithful to these areas. These water resources are vital for bat communities as they provide areas for water consumption and high concentrations of insects for bats to forage (Waldien and Hayes 2001; Wilhide et al. 1998). The presence of these ponds also introduces small openings into the forest and reduces structural complexity of the area immediately surrounding the pond.

CHAPTER 4

EFFECTS OF WILDLIFE STAND IMPROVEMENTS AND PRESCRIBED BURNING ON INSECT COMMUNITIES

ABSTRACT

Beginning 2001, the managers of the Buffalo Ranger District (BRD) in the Ozark-St. Francis National Forest (ONF), located in the Boston Mountains of Northwest Arkansas, began an alteration of a watershed located on the northwestern portion of this district using wildlife stand improvements (WSI) and prescribed burning (PB). This type of forest management was evaluated at two treatment and two reference sites. A WSI was administered in the fall of 2001 at treatment site T1and and PB was completed at that site in the spring of 2002. Similar forest management was conducted at treatment site T2. Neither WSI nor PB was used within reference sites nor had any timber been harvested within the last 15 years at those sites. The diversity and abundance of insect orders was evaluated in relation to the treatment being administered and was correlated with changes in structural complexity of the forest. Insect assemblages peaked when the forest stand density (FSD) had a basal area of $9m^2$/ha and declined as FSD increased to basal areas of $18m^2$/ha. These data suggest that forest management strategies incorporating WSI and PB are beneficial to insect communities in portions of the forest that have a FSD with a basal area greater than 14-16 m^2/ha.

INTRODUCTION

Tree densities have increased in several National Forests due to fire suppression (Kaufmann et al. 2003). Increased densities could potentially affect the structural complexity of the forest. Structural complexity has been defined as the relative density of the vertical and horizontal strata, considered separately and/or collectively (e.g., Heck and Wetstone 1977; Wilson 1974; Kohn 1967). Because areas that are more complex may provide more suitable niches and thus species diversity should increase as the habitat increases in complexity (Klopfer and Mac Arthur 1960). Therefore, habitats that are more structurally complex (i.e., early successional forests) should support greater numbers of insect species (Brown 1991; Hawkins and Lawton 1987; Cornell 1986; Stinson and Brown 1983; Lawton and Schroder 1977; Murdoch et al. 1972). Although species diversity has been associated with habitat complexity (MacArthur et al. 1966), and complex habitats should support greater species diversity than less complex forested habitats, greater structural complexity also reduces available, open space or gaps in the forest canopy. This interaction between canopy closure and overall site conditions affects the relative composition of the understory (Spies 1991), thereby, effecting the relative abundance and diversity of the ground cover (Gorham et al. 2002). Thus, habitats that are relatively more open or less dense allow for more sunlight to reach the forest floor, which stimulates herbaceous plant growth.

Wildfires played a crucial role in presettlement floral and faunal communities in forest ecosystems (Grimm 1984; Davis 1979; Kline and Cottam 1979; Curtis 1959). Fire prevention strategies adopted during the 1960’s greatly altered pre-settlement forest conditions in many forested areas of the United States (McClain et al. 1993; McCune and Cottam 1985; Curtis 1959). These fire suppression strategies have contributed to the decline of numerous flora (Brockway et al. 1998). For example, forested savanna habitats that were present on the pre-settlement landscape are now greatly reduced, having become increasingly dense in the mid- and understory, and crown cover increased when fire was suppressed. This dense forest shaded the forest floor and prevented direct sunlight from reaching the forest substrates and stimulating growth of herbaceous plants. Sunlight is further reduced as large amount of dead plant material such as leaves and woody debris accumulated on the forest floor and further reduced. Thus, reductions in herbaceous plant groundcover, occurred as forests became more dense (Kaufmann et al. 2003). For these reasons, prescribed burning has been reintroduced successfully as a management tool to preserve and restore pre-settlement floral taxa (Christensen 1981; Vogl 1973; Cushwa et al. 1966; Sheppard 1953; Lemon 1946). Prescribed burning has aided in restoring plant communities in Wisconsin sedge meadows (Kost and De Steven 2000), Florida scrub (Menges and Hawkes 1998) grassland prairies (Howe 1994), Longleaf pine stands (Haywood et al. 2001) and pine/hardwoods savannas (Elliott et al. 1999).

Management strategies that emphasize decreasing forest stand densities and canopy closure have been reported to also positively affect various insect communities in temperate ecosystems, specifically Lepidoptera (Kocher and Williams 2000). Herbivorous insect species attain their highest densities where food plants occur in large monespecific patches (e.g., MacGarvin 1982; Denno et al. 1981; Thompson 1978; Cromartie 1975). Available abundance of individual plants species is the main factor that determines population composition in moth and butterfly communities (Dempster

1983; Thompson and Price 1977; Myers and Campbell 1976; White 1974). Moreover, host plants with relatively more biomass attract more insect species (Teragushi et al. 1981). A high forest stand density with low herbaceous plant resources will support fewer insects (Hunter and Wilmer 1989; Hunter 1987). Conversely, insect abundance is often greater in less dense habitats such as forest clearings (Tibbels and Kurda 2003; de Jong 1994; Lunde and Harestand 1986).

Beginning 2001, the Buffalo Ranger District (BRD) on the Ozark-St. Francis National Forest (ONF), located in Northwest Arkansas (Figure 4.1) used wildlife stand improvements (WSI) and prescribed burning (PB) as a strategy to reduce FSD in a watershed located on the northwestern portion of this district (Figure 4.2), (Figures 4.3 and 4.4). These wildlife stand improvements were designed to aid in the restoration to the area's pre-settlement, historical condition, *i.e.*, an oak woodland. These forest management strategies were designed to reduce FSD in the treated areas. Prior to this study, the treated areas had a basal area of approximately 18-21 m^2/ha. Guidelines established for WSI on the BRD were that the midstory would be reduced in treated areas from a basal area of 18-21 m^2/ha to 7-9 m^2/ha. Prescribed burning would occur after the midstory reduction and would continue on a three to five-year rotation in an attempt to maintain a more open habitat type.

These two management practices are expected to alter the age, reduce the stratification of canopy layers, and reduce the structural complexity within treated areas of the BRD. An increase in insect populations was predicted as a result of more sunlight stimulating more herbaceous plant growth. Specifically, abundance (numbers of individuals and biomass) and diversity (number of insect orders and Shannon-Wiener

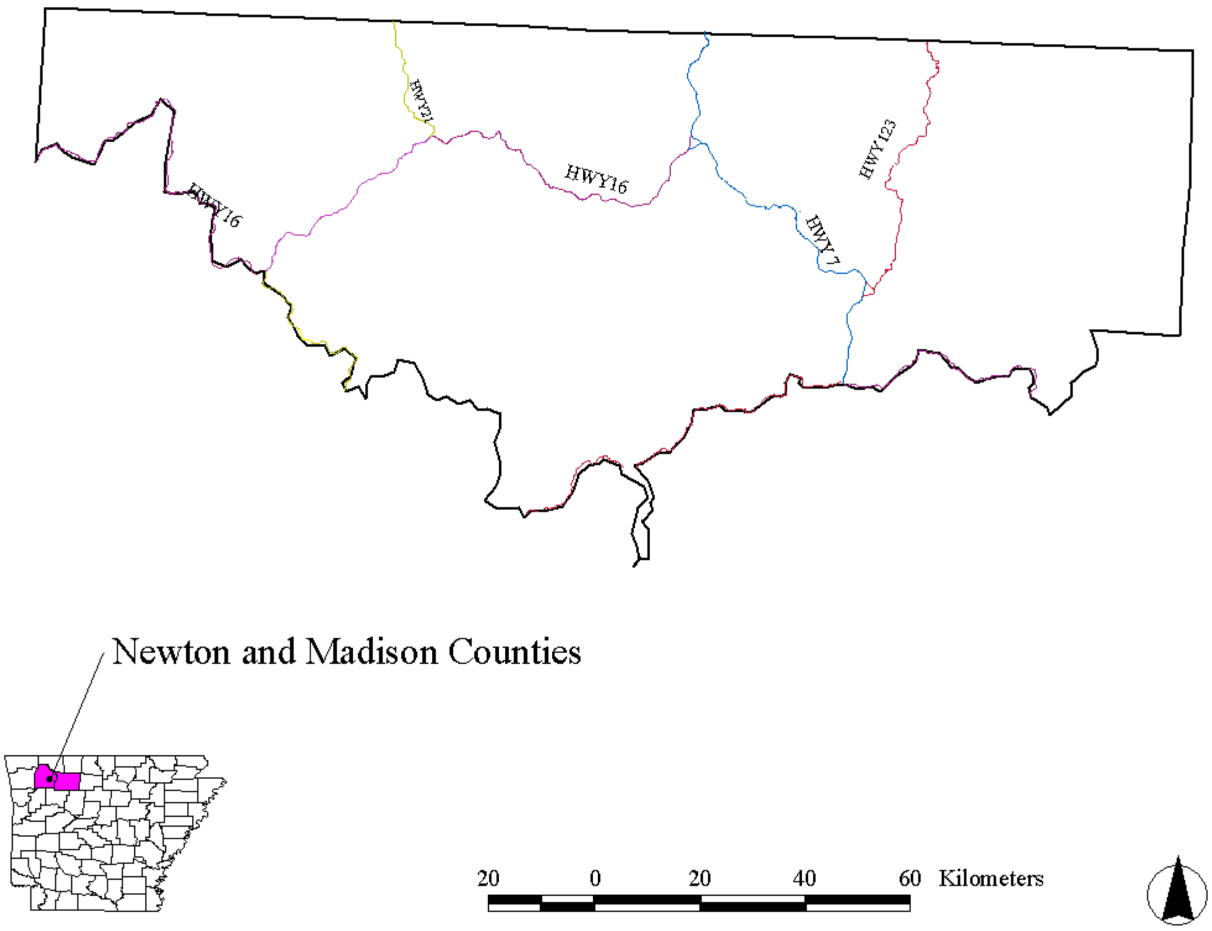

Figure 4.1. The location of the Buffalo Ranger District, Ozark-St. Francis National Forest, Arkansas

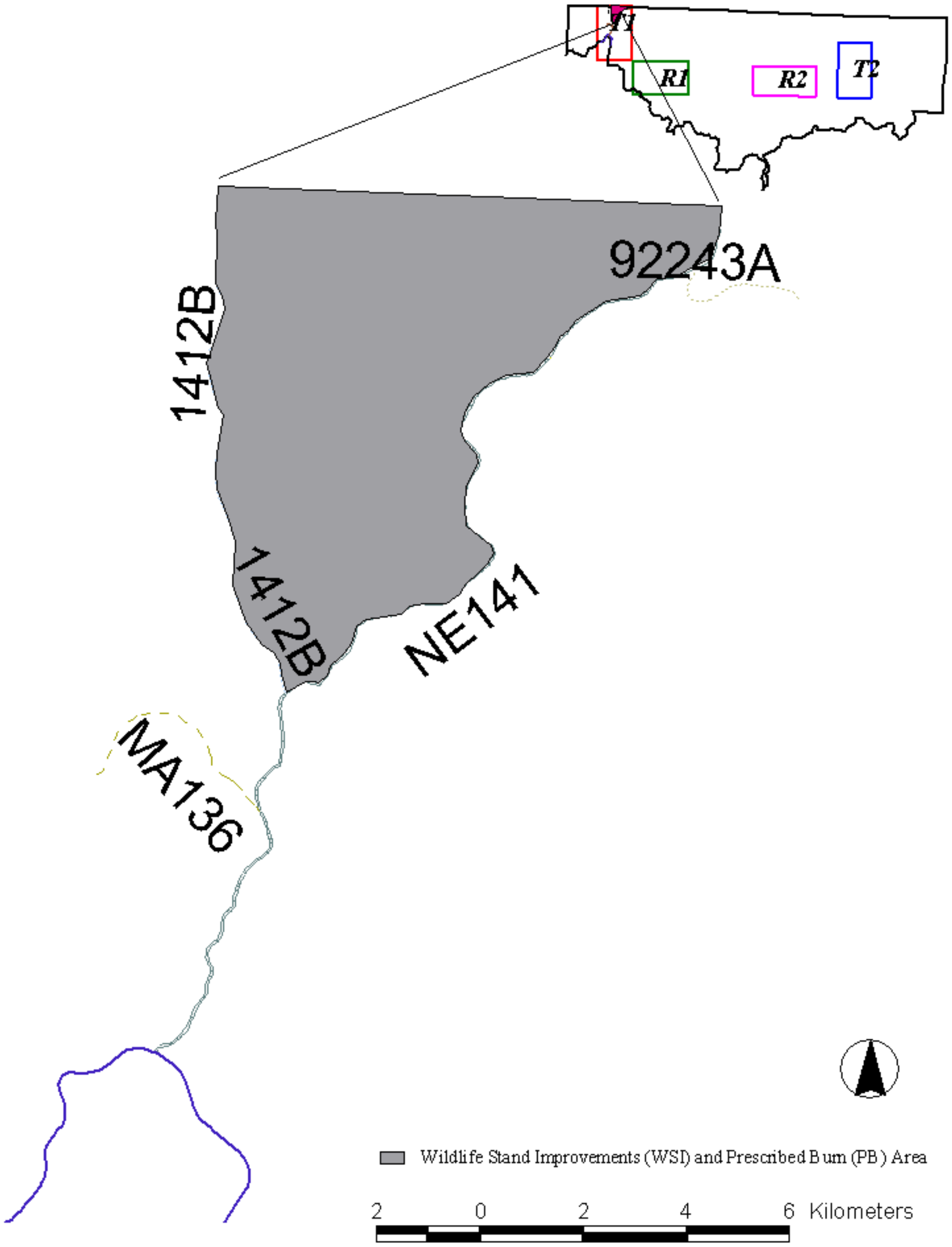

Figure 4.2. The area treated with wildlife stand improvements and prescribed burning.

Figure 4.3. Representative habitat in the Redstar site (T1) before WSI and PB were administered (23 July 2001).

Figure 4.4. Representative habitat in the Redstar site (T1) after WSI and PB were administered (15 May 2002).

diversity index) of insects was predicted to be greater within and among treatment sites. Insect abundance and diversity was predicted to be negatively associated with FSD.

Purpose of the Study

The purposes of this study were to assess the impacts that wildlife stand improvements and prescribed burning have on insect assemblages within the Buffalo Ranger District, Ozark National Forest. This study also evaluated the effects that forest stand density have on insect assemblages.

METHODS AND MATERIALSERROR! BOOKMARK NOT DEFINED.

Study Area

This study was conducted on the BRD in the ONF, in the Boston Mountains of northwest Arkansas (Figure 4.1). Currently the BRD comprises approximately 1,351,348 ha of which 168,737 ha are designated wilderness areas. Dominant overstory species are deciduous hardwoods including oaks (*Quercus* spp.) and hickories (*Carya* spp.), with conifers distributed throughout the district (US Forest Service). The age of the forest within the BRD rages from 0 to > 100 years. The BRD is comprised of habitats ranging from fields and 1st year regeneration stands to and mature old growth forest stands (US Forest Service). Small wildlife openings (approximately > 0.4 ha) exist throughout the BRD with an average density of 0.0002 wildlife openings/ha (US Forest Service). The district contains many man-made and natural ponds whose level is controlled by intermittent streams at an average density of 0.0002 ponds/ha (US Forest Service).

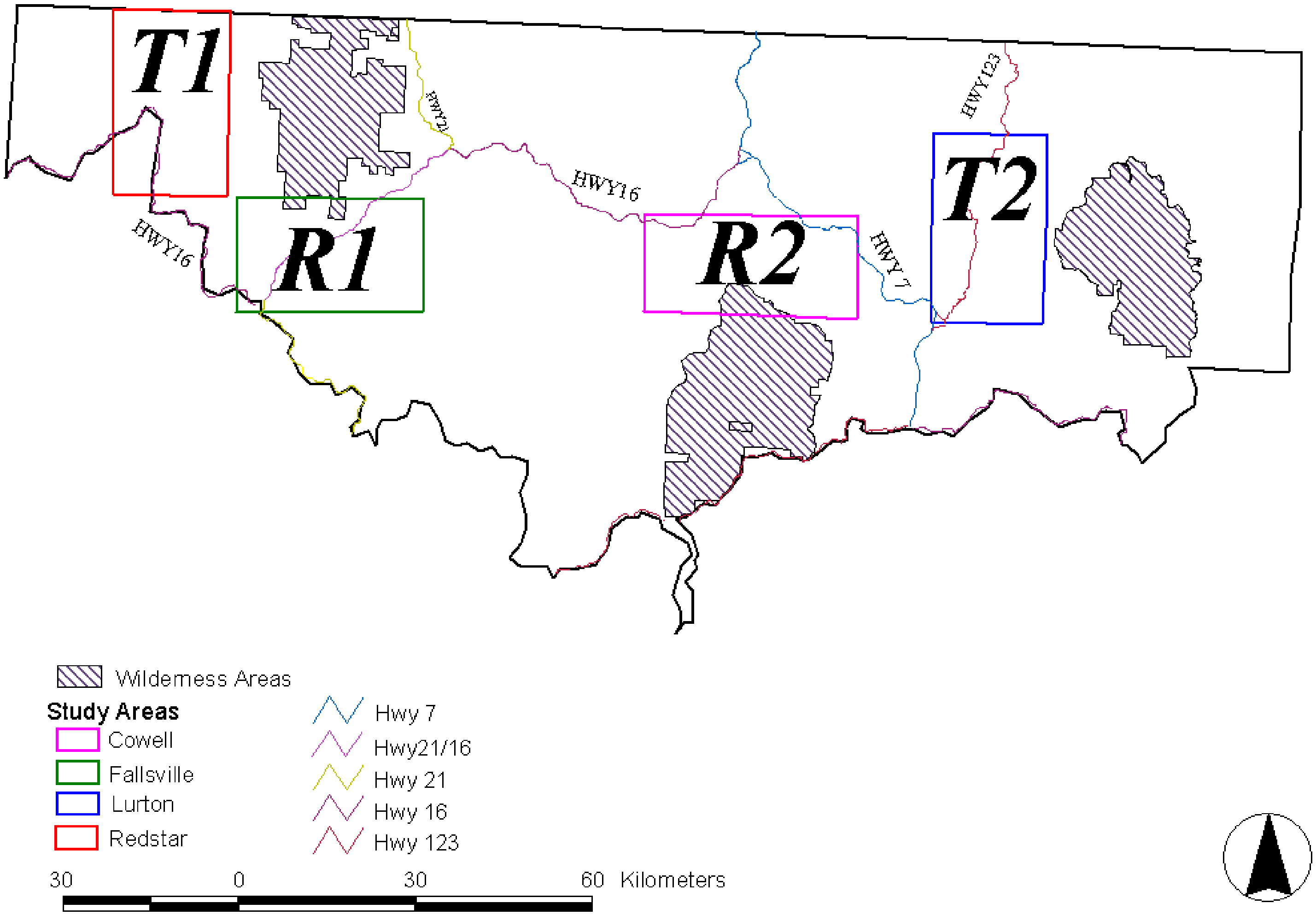

Figure 4.5. Wilderness areas and study site locations.

To assess the effects WSI and PB have on insect abundance and diversity, four study sites with in the BRD were established (Figure 4.7): Redstar, (T1), Fallsville, (R1), and Lurton, (T2), and Cowell (R2). Each site will be approximately 60,000 ha in size and each had similar vegetative composition (mixture of hardwoods and conifers) and age. Each has wildlife openings and ponds less than 0.5 ha in size are distributed throughout each study site, with an average density of 0.0002 wildlife openings/ha and ponds/ha (U.S. Forest Service).

T1 is located on the west central portion of the district. It was subject to a WSI and PB beginning in November 2001 and concluding in the spring of 2002 (Figures 4.2 and 4.5). T2, is approximately 118 km east of T1 (Figure 4.5). A thinning (similar to a WSI) and PB were completed at T2 during 1999; therefore, this study site was in its fourth year (2002) of regeneration. R1 is approximately 2 km east of T1 (Figure 4.5). R2 is approximately 12 km to the west of T2 (Figure 4.5). At each reference site no alterations that would affect FSD occurred. Reference sites were selected so they are relatively close to each treatment site and each study-site was spatially distributed approximately equal distances to wilderness areas.

Insects were sampled two nights per week at a treatment and reference site with weekly rotation among the sites (Figure 4.6) using a ground level, funnel black-light trap placed randomly within the interior portions of the forest (Figure 4.7). If a site could not be sampled for any reason (e.g. inclement weather) insects were sampled at that location as soon as possible to insure even sampling among sites. Placing light traps within the forest interior minimized any affects that sampling near water sources might have on estimates of insect numbers and species diversity.

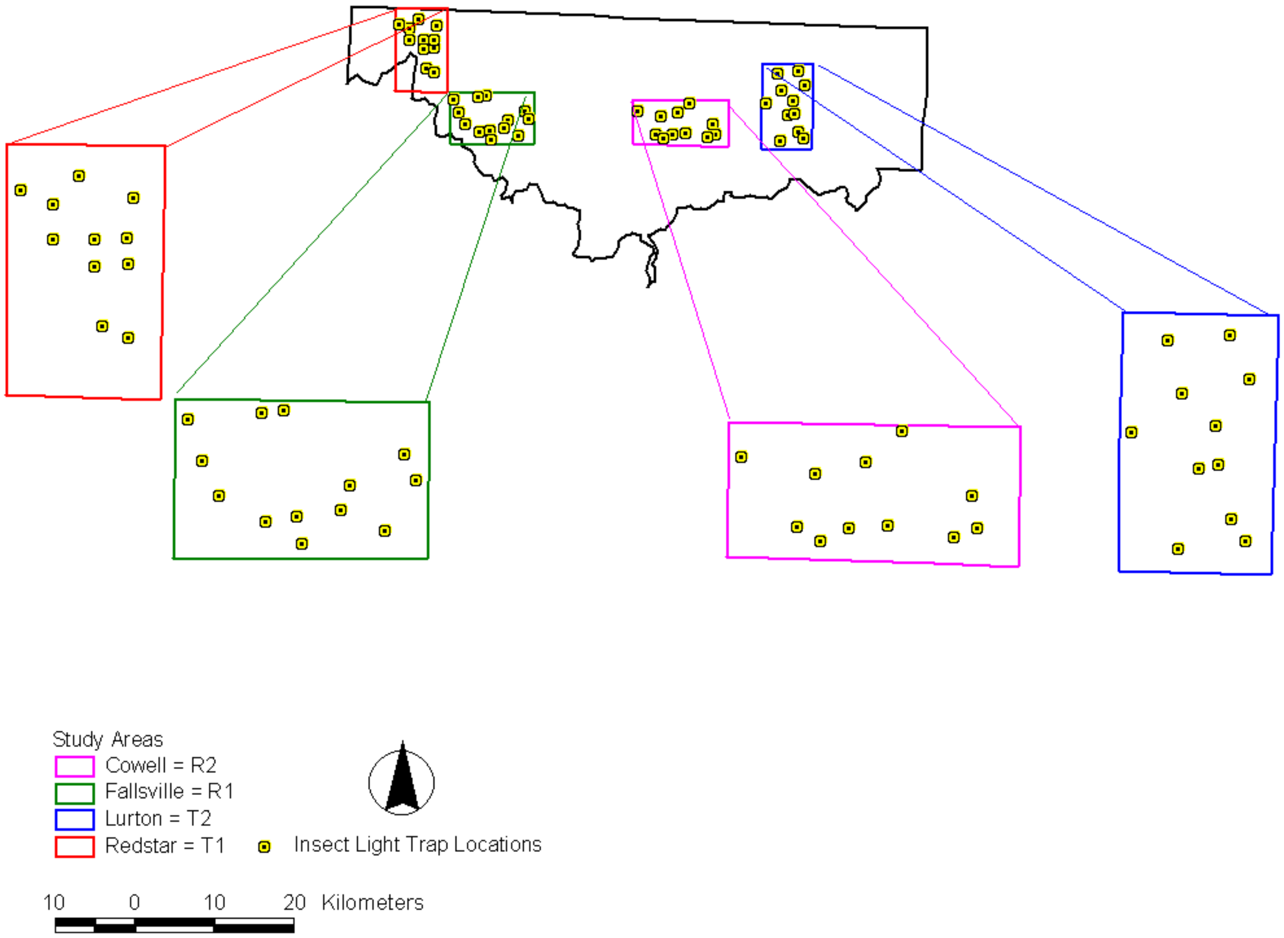

Figure 4.6. Insect trap site locations in the Buffalo Ranger District, Ozark National Forest, in northwest Arkansas (U.S. Forest Service).

Figure 4.7. Illustration of the Ground black-light trap used to sample insects (23 June 2001).

Light traps were opened approximately 30 minutes prior to sunset and were shut down at midnight. Insects were collected in containers filled with alcohol at the base of the funnel. This allowed insects to be preserved immediately after capture. All insects were identified at least Order. Voucher specimens were retained that represented each taxonomic group. The Global Positioning System (GPS) coordinates of each sampling location, date, times traps were open and closed, weather conditions, and habitat type were recorded during each sampling effort. Habitat type description consisted of characterizing major tree species and ground cover in the immediate area where the trap was set, a general description of structure being netted, and terrain characteristics. All GPS coordinates were recorded in Universal Transverse Mercator (UTM) and North American Datum 1983 (NAD 83), Zone 15 North.

Forest Stand Density

The relative FSD was calculated by measuring the basal area which is cross sectional area at breast height of all trees in a hectare expressed as square meters per hectare, in the areas surrounding each insect sampling locations by establishing four sub plots that were 0.25 ha in size, (approximately 17 m in diameter) in four cardinal directions 50 m from the structure or area that was being sampled. Tree basal area was calculated by multiplying 0.000078539 by the diameter at breast height squared (DBH^2) of all trees ≥10 cm in each subplot (Note: 0.000078539 = pi/40,000 where the division by 40,000 corrects for the difference in cm and diameter to radius). Stand basal area was determined by dividing the total basal area for each tree in each plot by 1 ha (4 *0.25ha subplots).

Data Analysis

Insect assemblages (IA) were evaluated by calculating the abundance and diversity of insects captured per night. Diversity of insect orders was also calculated with the Shannon-Weiner Diversity Index Model (Krebs 1989) and abundance was quantified by measuring the collective dry weight individuals from each taxonomic group for each trap location.

To evaluate treatment effects on IA, the mean values were analyzed using a 2 sample *T*-test (Minitab, Inc. 1998). Since WSI and PB are designed to reduce FSD, the mean values of IA were also analyzed using an ANOVA to evaluate difference and similarities within and among each study site. A Tukey's HSD post hoc tests was used to see where the difference occurred between the treatments. A Pearson correlation analysis in Minitab (Minitab, Inc. 1998) was used to establish relationships among IA, and FSD, and ground cover. All data were tested for normality using the Kolmogorov-Smimov Normality Test (Minitab, Inc. 1998). If the data were not normally distributed, they were normalized by using the inverse log of the respective data columns. A preset *P*-value of 0.05 was selected as the significance level.

RESULTS

Treatment Effects on Structural Complexity

The four study sites (T1, T2, R1, and R2) were sampled between the months of April and September in 2002, with a total of 46 sampling locations, averaging 11.5 nights/study site (Figure 4.6). Significant differences in structural complexity were observed when treatment sites were compared to reference sites. The mean FSD was significantly lower in the treated sites compared to untreated (Table 4.1). The percentage of ground cover was also significantly higher in the treated sites than in untreated sites (Table 4.1).

Treatment and Site Effects on Insect Assemblages

A total of 12 orders of insect taxa were captured during this study. Nine of the 12 insect orders were recorded within each reference site and all 12 taxa were observed within each treatment site (Table 4.2).. Differences between the mean numbers of insect orders captured in treated vs reference sites were significant when compared using a 2-sample *T*-test (Table 4.2).

The most predominant insect order captured was Lepidoptera, followed by Coleoptera (Figure 4.8). Significant differences and similarities of the mean number of insect orders captured were observed within and among each study site when analyzed using an ANOVA. The mean number of insect orders at the treated sites (T1 and T2) was similar to each other and reference sites were also similar to each other (Table 4.1). When treated sites were compared to reference sites, significant differences in the mean number of insect taxa were observed, with more insects being captured within each treatment site (Table 4.2).

Number of Individuals

There were no significant differences observed in the number of individual insects captured in treatment sites compared to references sites (Table 4.2). ANOVA

Table 4.1. The means of the structural complexity parameters and measures of insect assemblages (ANOVA).

	Study Site			
Parameter	**T1**	**T2**	**R1**	**R2**
# of Trap Site Locations	11	11	13	11
Forest Stand Density (FSD)				
Mean	7.4	8.0	17.0	16.0
StDev	0.67	1.039	1.66	2.168
Subscripts	a	a	b	b
% Ground Cover				
Mean	81.8	68.6	11.2	18.2
StDev	12.3	13.25	5.83	7.83
Subscripts	a	a	b	b
# of Insect Orders				
Mean	7.7	8.0	6.5	5.8
StDev	0.64	1.58	1.39	0.60
Subscripts	a	a	b	b
# of Individuals				
Mean	273.4	350.0	127.6	158.9
StDev	43.72	83.97	43.73	63.04
Subscripts	a	a	b	b
Diversity of Insect Orders				
Mean	0.9	1.1	0.9	1.1
StDev	0.07	0.18	0.28	0.34
Subscripts	*	*	*	*
Insect Weight				
Mean	52.31	50.28	17.69	25.9
StDev	8.72	13.74	7.01	20.48
Subscripts	a	a	b	b

Note: significant difference (*P*-value 0.05) between study sites if subscript letter is different and study sites are similar if subscript letter and number are the same. Nonsignificant differences are represented by an (*) Tukey's Family error rate = 0.05 and Individual error rate = 0.107

Table 4.2. The means of insect assemblages between treatment types (ANOVA).

	Study Site	
Parameter	**Treatment Sites**	**Reference Sites**
Number of Taxa (Order)	12	9
Mean	7.9	6.167
StDev	0.257	0.236
P-Value	0.019	
Total Number of Individuals	7062	3407
Mean	311.7	142
StDev	54.2	17.5
P-Value	0.052	
Total Diversity of Insect Orders	1.075	1.230
Mean	1.012	1.032
StDev	0.13	0.155
P-Value	0.899	
Total Insect Weight (g)	1157.619	513.87
Mean	51.26	21.41
StDev	1.43	3.77
P-Value	0.009	

Note: Difference = [mu (treated) - mu (Untreated)]

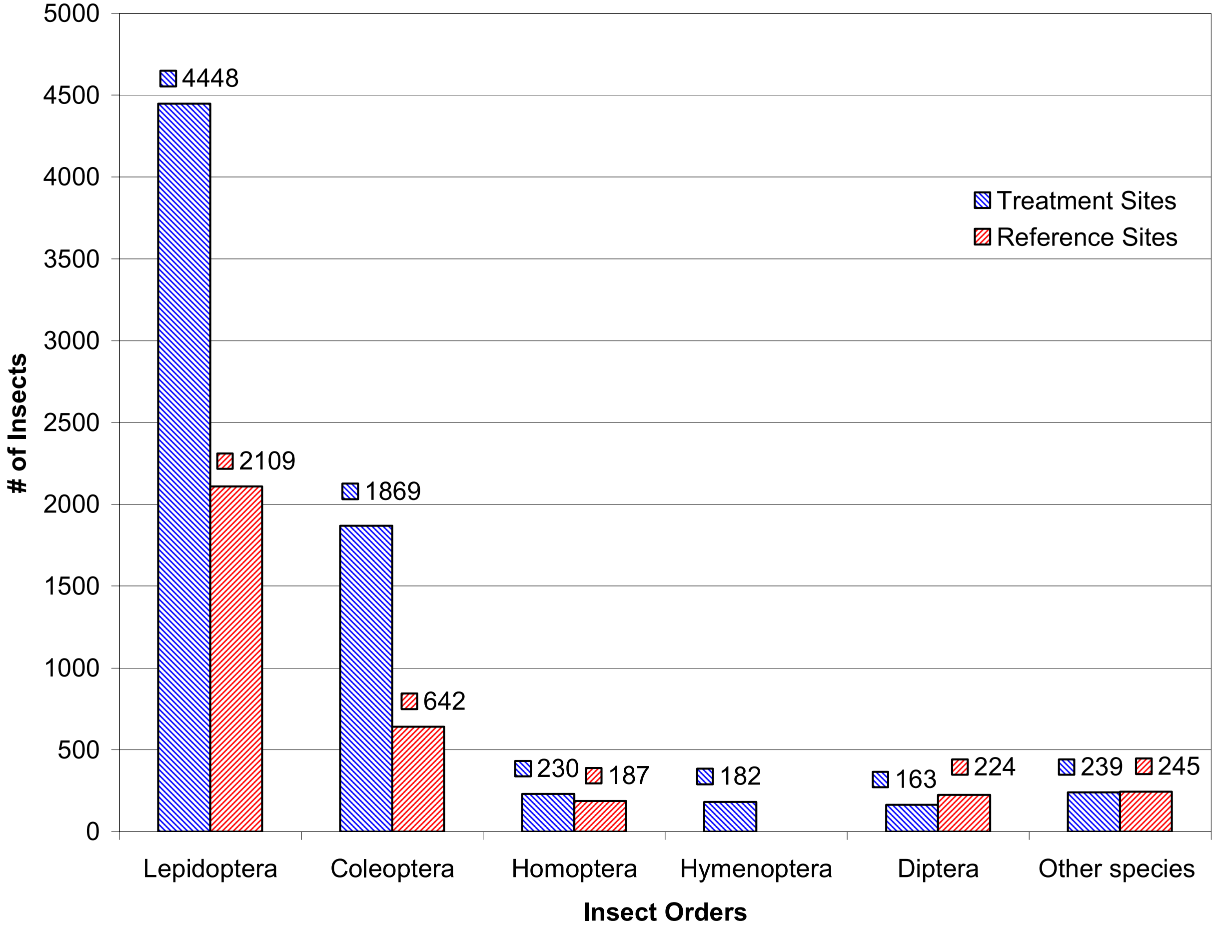

Figure 4.8. Number of individuals within each taxonomic unit (Order) captured in both treatment types.

was used to compare the mean number of individual insects captured within and among study sites (Table 4.1). This analysis revealed that the mean number of individuals captured within the two treated areas (T1 and T2) were not statistically different while the mean number of individuals captured in treated sites were significantly greater than that captured in untreated sites. The mean number of individuals captured within the reference sites (R1 and R2) was not statistically significant (Table 4.1).

Diversity of Insect Orders

Significant differences in the mean diversity of insect orders were not identified (Table 4.2 and 4.1).

Insect Weight/Biomass

Differences between the mean insect mass captured at treated and reference sites was highly significant, with higher insect bio-mass being captured at treatment sites (Table 4.2).

Additionally, significant similarities and differences were detected in the mean insect biomass captured within and among study-sites when an analysis was conducted using an ANOVA. The mean insect biomass was not statistically different between treatment sites and between reference sites (Table 4.1). There were significant differences among treatment sites and reference sites, with greater insect biomass being collected in the treated areas (Table 4.1).

Insect Assemblages Associated with Structural Complexity

Since there were significant differences and similarities in structural complexity and insect assemblages within and among study sites, the data were combined at the forest stand level using trap site locality as the sampling unit. This was done in order to evaluate the effects that structural complexity of the forest would have on the insect assemblages.

Although there were no significant correlations between structural complexity and diversity of insect orders (Figures 4.9 and 4.10), there was a highly significant correlation between the number of insects orders captured and structural complexity parameters (FSD and % of ground cover) (r = - 0.595, *P*-value = 0.0001 and 0.602, *P*-value = 0.0001 respectively). These data and the results of the analyses are shown in Figures 4.11 and 4.12. Figures 4.13 and 4.14 also show that significant correlations were observed between structural complexity and the number of individual insects captured (r = - 0.781, *P*-value = 0.0001 and r = 0.736, *P*-value = 0.0001). Additionally, there was a significant correlation between the mean biomass of insects captured per trap site location and structural complexity (r = -0.793, *P*-Value = 0.0001 and r = 0.725, *P*-value = 0.0001)(Table 4.3)(Figure 4.15 and 4.16).

DISCUSSION

During my six month investigations of insects the number of taxa, number of individuals and total weights taxa present differed among treatment types as predicted with treated areas having higher levels of each. However the total diversity of insects orders present did not differ among sites contrary to predictions. My results clearly

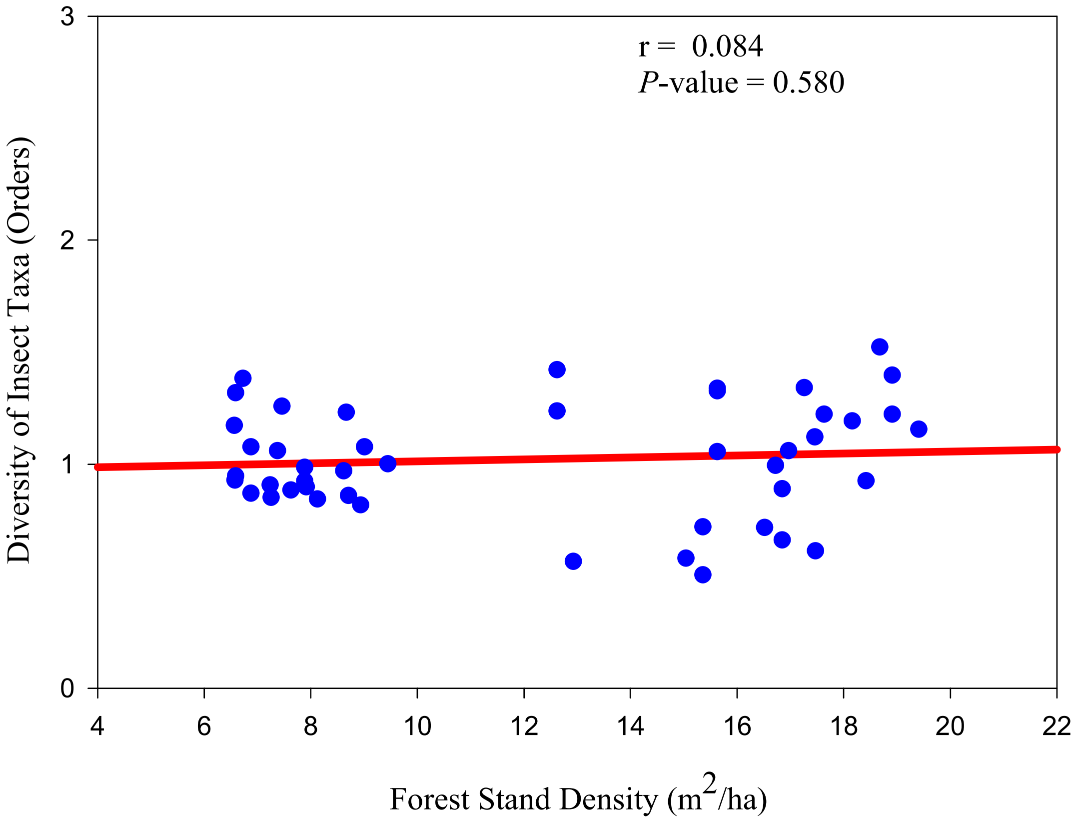

Figure 4.9. Association between forest stand density (FSD) and the diversity (Shannon-Weiner, H) of insect orders present at each trap site location.

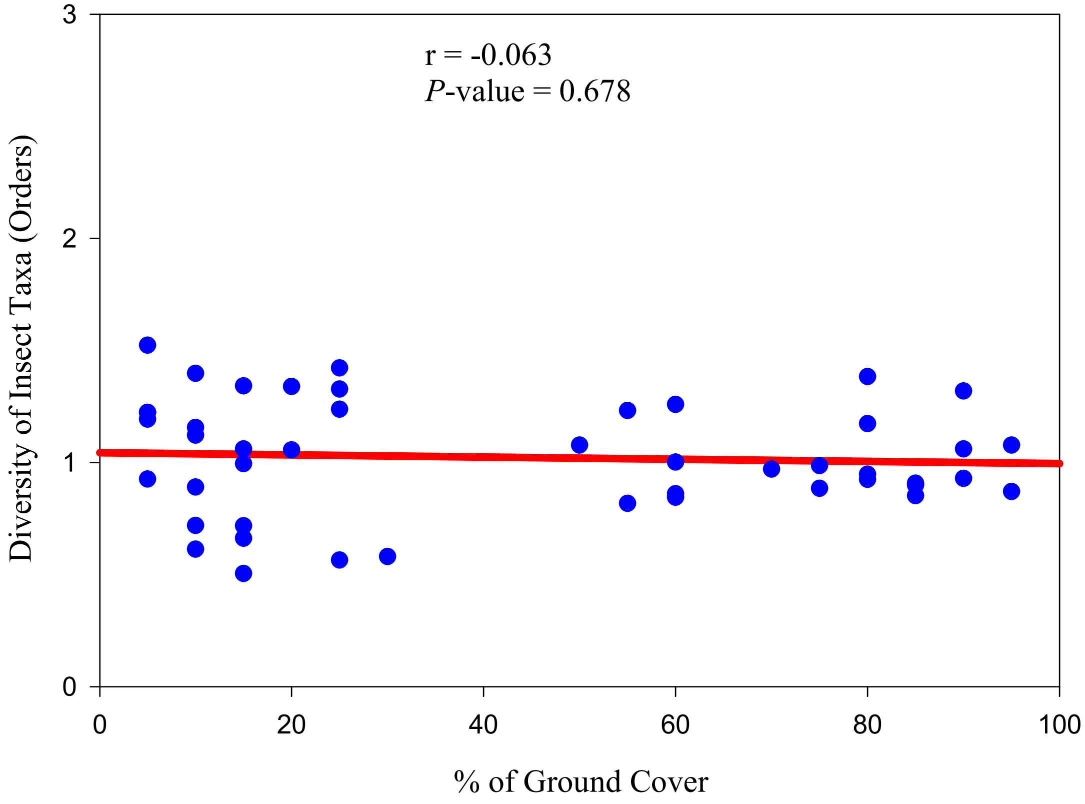

Figure 4.10. Association between the percentage of ground cover and the diversity (Shannon-Weiner, H) of insect orders present at each trap site location.

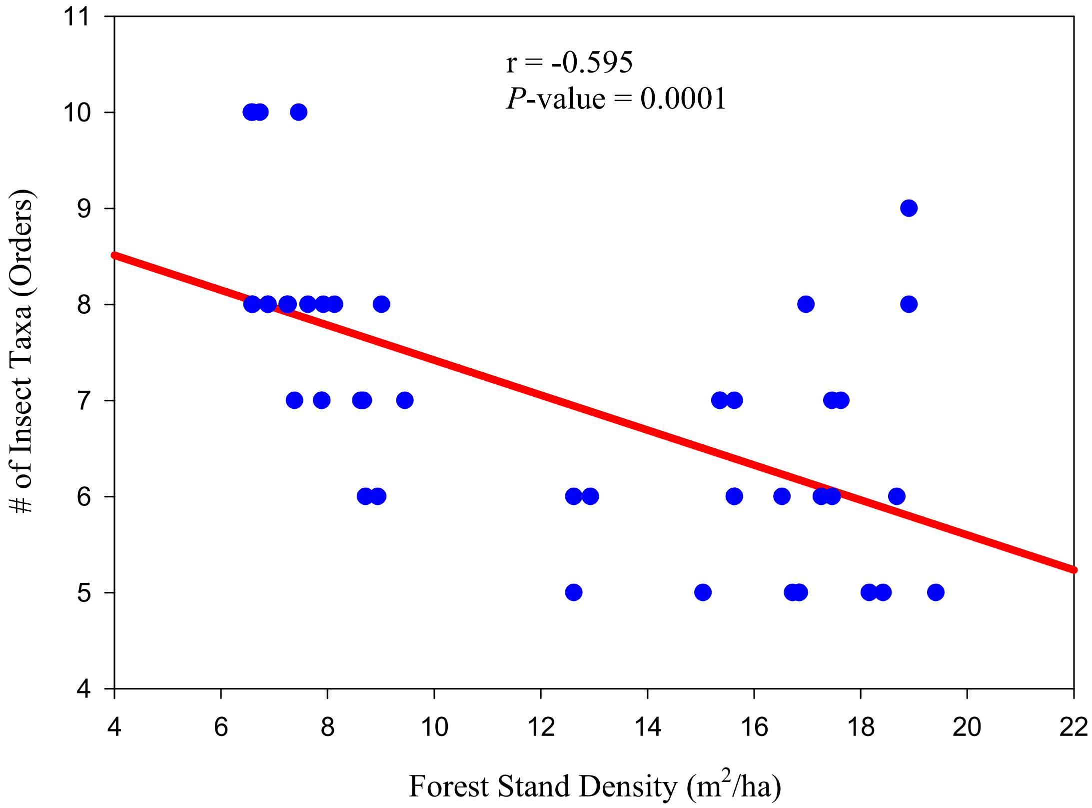

Figure 4.11. Association between forest stand density (FSD) and the number of insect taxa (Orders) present at each trap site location.

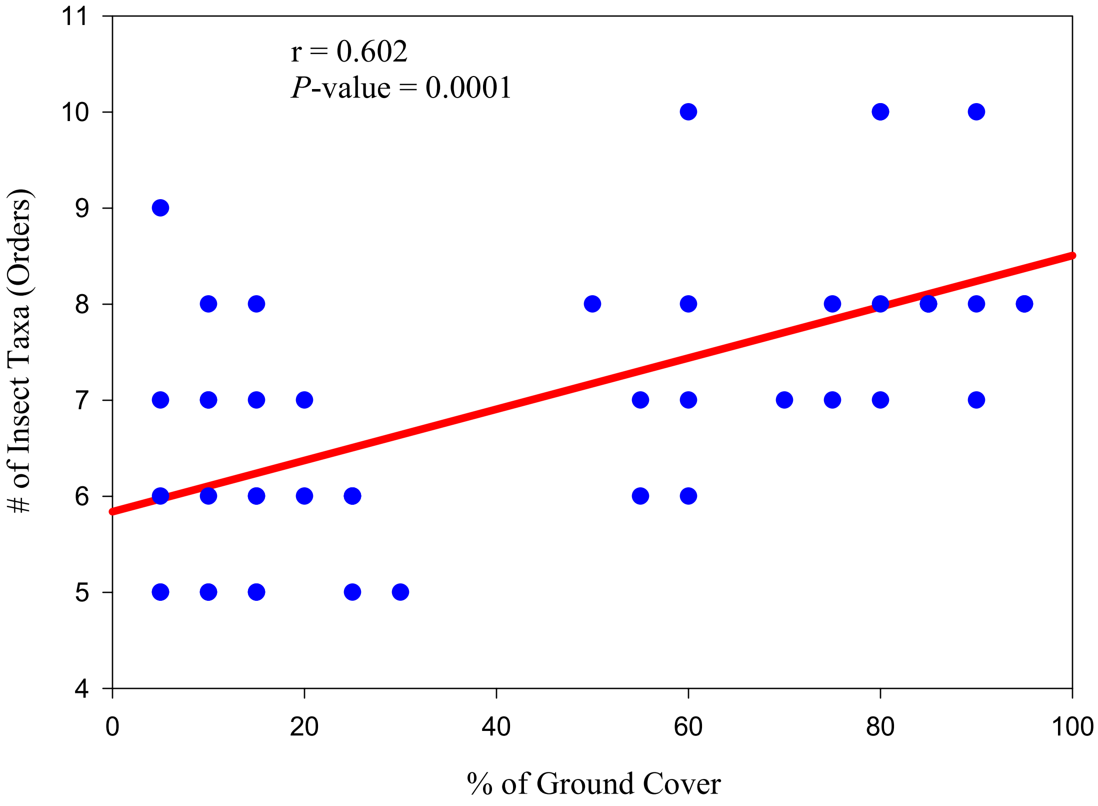

Figure 4.12. Association between the percentage of ground cover present and the number of insect taxa (Orders) present at each trap site location.

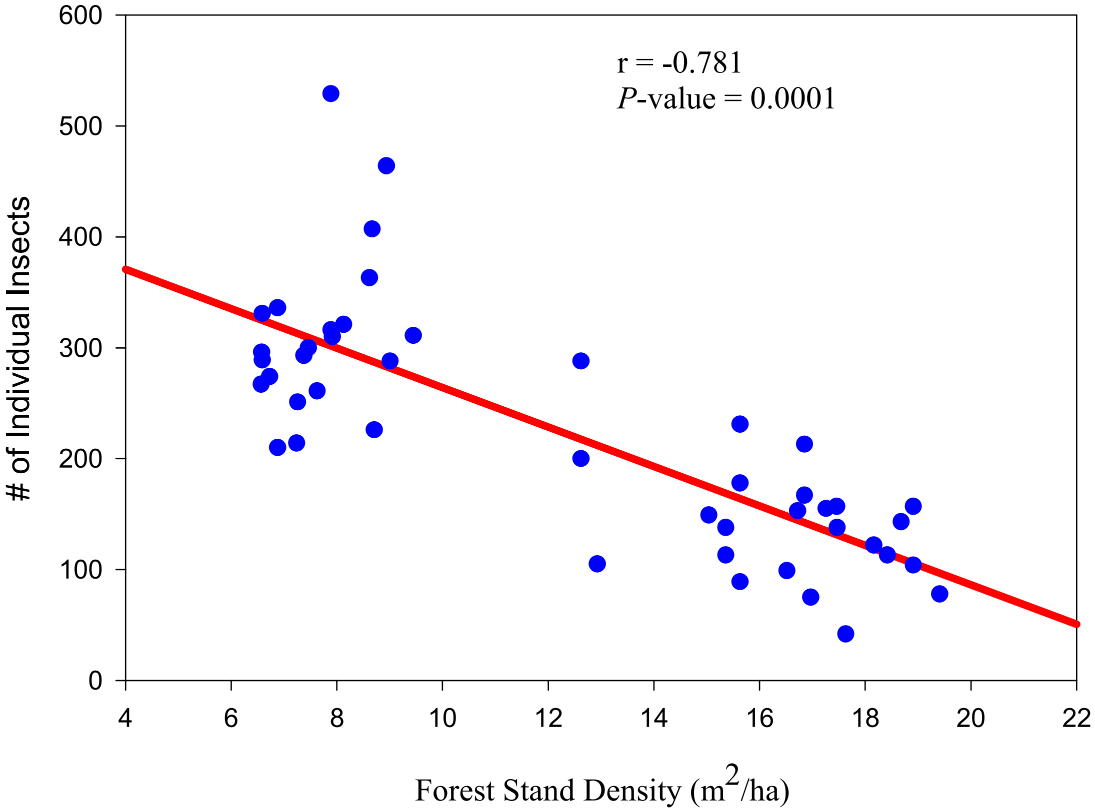

Figure 4.13. Association between forest stand density (FSD) and the number of individual insects present at each trap site location.

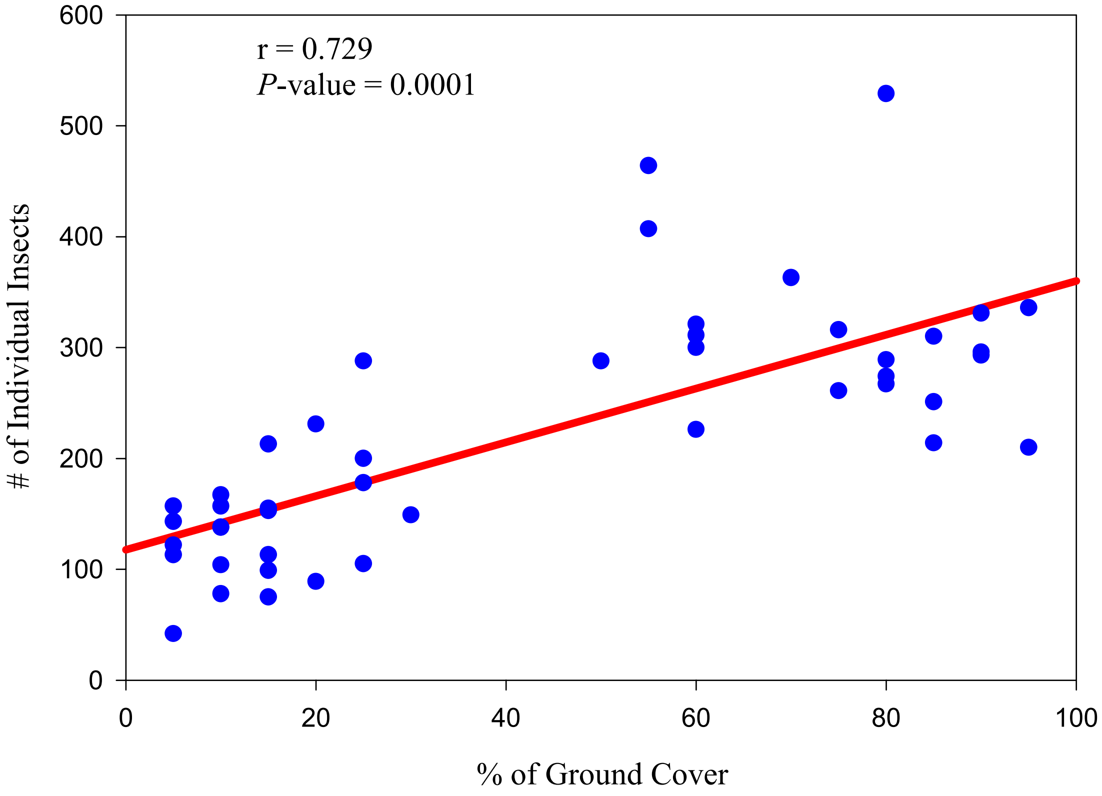

Figure 4.14. Association between the percentage of ground cover present and the number of individual insects present at each trap site location.

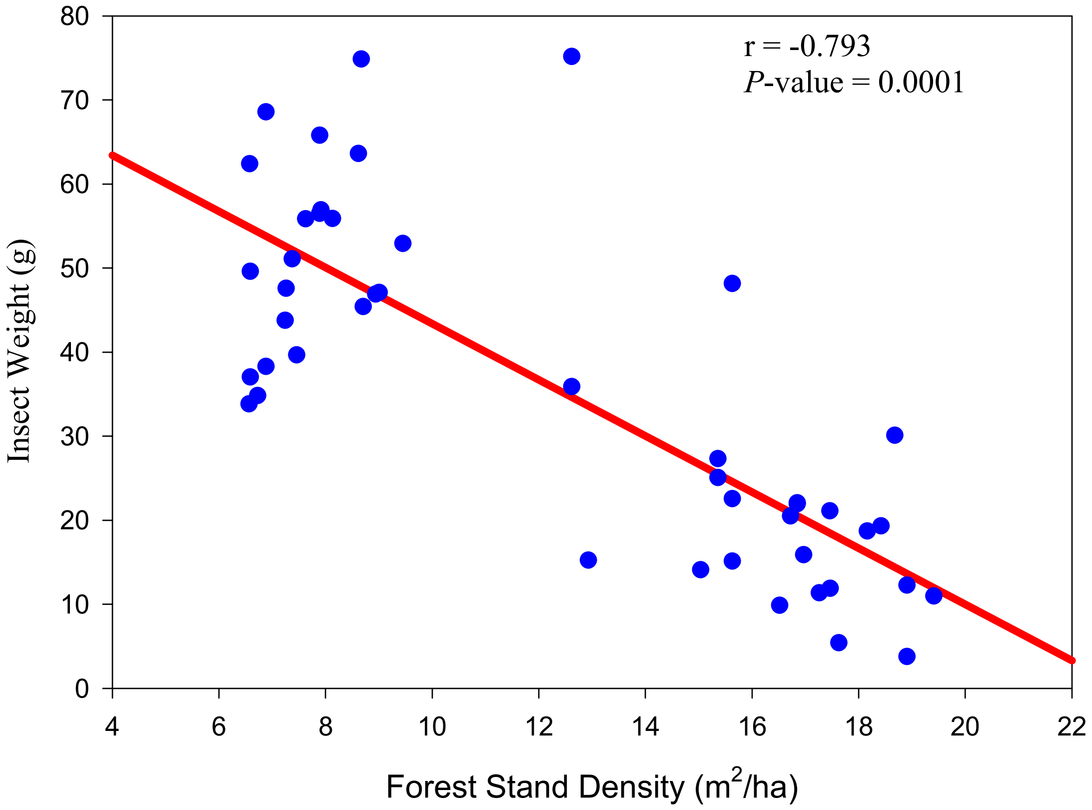

Figure 4.15. Association between forest stand density (FSD) and the cumulative weight of insects at each trap site location.

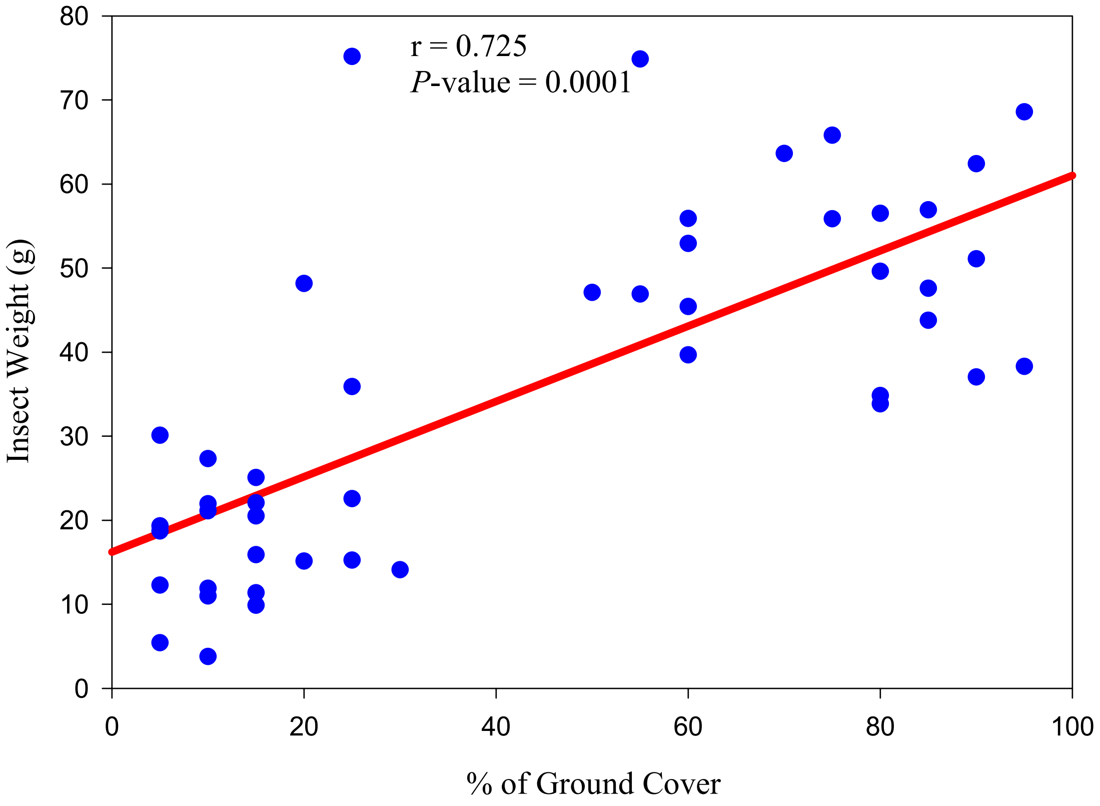

Figure 4.16. Association between the percentage of ground cover and the cumulative weight of insects at each trap site location

show that WSI and PB can be used to establish forested areas that are less structurally complex by reducing the FSD, and that changes in FSD have significant affects on insect communities. The management strategies applied to the treated sites was designed to reduce the FSD; therefore, as expected complexity was less in the treated areas than in untreated areas. My results are also consistent with previous studies that showed increases in herbaceous vegetation was observed after a reduction in forest stand density and prescribed burning (Kost and De Steven 2000; Christensen, 1981; Vogl 1973; Cushwa et al. 1966; Sheppard 1953; Lemon 1946).

Herbaceous plants are known serve as host plants for several insect taxa, and increases in herbaceous plants have been shown to benefit these species (Kocher and Williams 2000; Marques et al. 2000). These data indicate that the number of insect orders captured was greater in treatment areas, and associated with reductions in FSD and with a higher percentage of ground cover.

Species diversity of insects should increase as the percentage of herbaceous ground cover (Dempster 1983; Teragushi et al. 1981; Thompson and Price 1977; Myers and Campbell 1976; and White 1974) and available biomass (Marques et al. 2000) increases. However, we observed no significant differences in species diversity between treated and reference sites. Since specimens were only identified to the taxonomic level of an order, the species diversity reported in these data is conservative and observed differences are may be underestimated.

As expected the highest density of insects occurred in treatment areas, compared to reference areas. Moreover, higher numbers of insects were also captured within areas with reduced FSD and an increased percentage of ground coverage. This is consistent with other reports, where higher insect density was associated with less structurally complex habitats were (Tibbels and Kurda 2003; de Jong 1994; Lunde and Harestand 1986).

Insect biomass was also found to be higher in treatment areas than in reference areas and in areas of the forest with lower FSD and with a higher percentage of ground cover. This suggests that not only were more biomass in the treated areas, there more individual insects in areas of the forest that was relatively less structurally complex.

Although these data, which are represented by the Pearson's correlation analyses, are combined at the forest stand level, these forest stands were selected from existing treatment and reference sites. It was therefore, pseudoreplicated, however, it was unavoidable in order to evaluate the relationships between insect assemblages and FSD, while documenting any treatment affects.

MANAGEMENT IMPLICATIONS

Forests are subjected to natural mortality events including natural tree fall resulting from age, wind, and insect damage. These events create small openings in the forest that may provide habitat for insect species that preferentially select areas that are relatively less dense and more open (Gorham et al. 2002; Tibbels and Kurda 2003; Lunde and Harestand 1986; de Jong 1994; Gladden and Smock 1990). Using conventional timber harvesting techniques, such as selective cutting, seed tree harvesting, small group cuts, or thinning as well as WSI could potentially simulate natural mortality in temperate forest ecosystems. This may provide stratified forest stands where different insect species are able to segregate into separate niches thereby increasing insect species diversity and abundance.

Wildlife openings and upland ponds are distributed throughout the BRD, as well as across most of our National Forest. The functionality of these wildlife openings and upland ponds vary across taxonomic groups. Insects are one of these taxonomic groups that could potentially take advantage of this habitat alteration as a source of water and food. For example, sunlight exposure to the forest floor is increased by creating small gaps (wildlife openings) and/or gaps with a water source (upland ponds). This indirectly benefits the insect community, (Marques 2000; Marino and Landis1996) due to the increase of herbaceous plants, which is stimulated by the sunlight exposure (Gorhom et. al. 2002; Billups and Burke 1999). This also creates an edge effect within the forest interior, where it is reported to support the greatest species abundance and diversity (de Jong 1994). Therefore, incorporating small (e.g., 1.0 ha in size) wildlife openings and upland ponds into future management plans could potentially be beneficial to insect communities.

Before settlement, fire was part of many of the natural ecosystems in temperate forest. Several benefits result from fires including an increase in soil nutrients from the ash. This aids in the sprouting of grasses, forbs, and shrubs as well as the regeneration of woody shoots. Regeneration of trees is vital to forest ecosystems, as it allows for younger trees to replace older trees that have fallen due to natural mortality. Moreover, it also allows for a heterogeneous forest setting and aids in creating a stratified forest while simultaneously maintaining a relatively open midstory. Prescribed burning could potentially simulate this type of forest setting while preventing high intensity fires that occur as a result of accumulation of fuel (leaf litter and decaying wood) on the forest floor in areas where fire has been suppressed. Conducting prescribed burns on a three-year rotation after a WSI has been administered may allow the mid-and understory to remain relatively open, therefore preventing treated portions of the forest from becoming structurally complex enough to have a negative impact on various insect communities.

Prescribed burning increases herbaceous vegetation, which provides direct and indirect benefits to insect communities. However, it is also used as measure of control from erratic insect outbreaks. For example prescribed fire is used as a management to control populations of the red oak borer, which aided in destroying large portions of oaks distributed across the ONF (Davenport et al. 2002). Fire is also used to control populations of the southern pine beetle (*Dendroctonus frontalis*) (Swain and Remion 1981).

In summary, forest thinning and prescribed burning, as well as the creation of wildlife openings and upland ponds are beneficial to plant communities. This effect appears to also benefit insect communities, which makes up an intrinsic part of the food chain, by providing a large prey base for wildlife species such as bats (Aldridge and Rautenback 1987).

CHAPTER 5

SUMMARY AND FINAL CONCLUSION

Structural Complexity

My results clearly show that wildlife stand improvements and prescribed burning can be used to establish forested areas that are less structurally complex by reducing forest stand density. The FSD at study-site T1, decreased significantly after the treatment was administered and was similar to the FSD at study-site T2. Additionally, FSDs in $T1^{post}$ and T2 were significantly different than those in each of the reference-sites ($T1^{pre}$, R1, and R2).

One of the objectives of these management strategies by reducing the midstory by approximately 50%, was to increase the percentage of herbaceous plants of the forest floor. Increases in ground cover within treatment sites were expected because the reduction of FSD permitted more sunlight to reach the previously shaded forest floor (Kost and De Steven 2000; Christensen 1981; Vogl 1973; Cushwa et al. 1966; Sheppard 1953; Lemon 1946). Because significant increases in herbaceous plant coverage were observed, this objective was achieved.

Herbaceous plants serve as host plants for several insect taxa, including Lepidopteran species, which benefit by increases in density of herbaceous plant species (Kocher and Williams 2000; Marques et al. 2000). Similar results were observed during this study, and therefore supporting the prediction that greater numbers of insect orders would be captured in treated areas (Tibbels and Kurda 2003).

Insects Assemblages

During this study insect populations were only sampled for six months (April to September) and do not allow longer-term impacts of administered treatments to be assessed. The small number of sampled sites may also have impacted these data (2 treatment and 2 reference sites) and by seasonal changes variably affect different taxonomic groups of both flora and fauna. Therefore, it is possible that these data over or underestimate the effects of these management strategies on insect communities. However, when these factors are considered as treatment effect on insect populations are evaluated, it seems likely that our data represent conservative estimates of the impacts of WSI, PB, and FSD on the insect community so that any effects of these treatments are probably underestimated. Therefore, this study clearly shows that when WSI and PB are used to establish forested areas that are less structurally complex by reducing the FSD, they have the effect of increasing the diversity of insect communities.

Species diversity of insects increases as the percentage of herbaceous ground cover (Dempster 1983; Teragushi et al. 1981; Thompson and Price 1977; Myers and Campbell 1976; White 1974) and available biomass increase (Marques et al. 2000). However, the prediction that species diversity would be highest in treatment areas, compared to reference areas, and would increase as FSD decrease was not supported by this study. Since specimens were only identified to the taxonomic level of an order, the species diversity reported in these data is conservative. Therefore, any observable differences are most likely underestimated.

Additionally, the highest density of insects occurred in treatment areas and higher numbers of insects were observed within areas with reduced FSD and an increased percentage of ground coverage. Increases in insect abundance is associated with less structurally complex habitats and/or less dense forest, specifically in open habitats (Tibbels and Kurda 2003; Lunde and Harestand 1986), intermediate densities at forest edges, and lowest densities within the forest interior (de Jong 1994).

Insect weight and/or biomass were also observed to be higher in treated sites, and in areas with lower FSD and with a higher percentage of ground cover. This observation also suggests that not only were there more insects, but there were also significantly greater amounts of insect biomass in the treated sites and in areas of the forest that was relatively less structurally complex. These treatment-associated increases in insect numbers and biomass could greatly benefit bat communities located within treated sites.

Bats Assemblages

Differences of the mean bat capture rate between treatment and reference sites were not significant, and impacts on may not be apparent for some time after implementing treatments. However, the predicted relationships between FSD and the capture rate for the abundance of bats that is similar to the relationship between insect assemblages and FSD. There was a negative relationship between lower FSD and capture rate at each study-site, such that bat capture rate increased as FSD decreased. Other species, *M. lucifugus*, and *M. volans,* with similar wing and echomorphology adaptive to forage in dense habitats, forage exclusively in open habitats (Saunders and

Barclay 1992). It is therefore plausible that other species that are also adapted to forage in densely forested habitats benefit by in the availability of more open habitats. More open areas may allow greater flight speed, which may maximize an individual's search effort while foraging and improve its ability to move to alternative foraging areas, ponds, and roosts. Thus, the advantages achieved by a preference for less dense areas within the forest seem to support foraging strategy theories (Schoener 1971).

Estimates of bat species diversity were not statistically different. Diversity estimates may have been affected by the same study limitations that impacted estimates of bat numbers such as high standard deviation within sampling site locations. However, estimates of species diversity were significant when analysis was conducted among and within study-sites. Bat species captured/net-night was negatively associated with FSD. Similar relationships were observed when bat diversity was estimated using the Shannon-Weiner Diversity Index Model. Increased species diversity and abundance were associated with less dense forest (Saunders and Barclay 1992; Fenton and Bell 1979) and therefore suggests bat diversity was greater in areas of the forest that were relatively less dense than surrounding more dense regions.

It is acknowledged that net placement could potentially exert a significant effect on estimates of bat numbers and diversity. In order to reduce this possible effect, in all study sites, nets were placed at various types of water structures, roads, and small wildlife openings. Care was also taken to insure that these structures that were used as netting locations were located both on the exterior and interior portions of the forest. Greater numbers of northern long-eared bats and half of all Indiana bats are have been documented in the interior portions of the forest (Carroll et al. 2002). This suggest that bat diversity and abundance may be underrepresented when populations are sampled using traditional mist netting techniques that emphasize placing nets over water containing structures. Therefore, an emphasis on placing mist nets near or over water may also affect capture rates, however, bat activity more often occur over or near water sites than in strictly terrestrial habitats (Lunde and Harestad 1986; Furlonger et al. 1987; Krusic et al. 1996; Grindal et al. 1999).

The data described herein indicate that WSI and PB did not have any apparent negative effect on bat activity in the BRD. Based on my observations of FSD, insect and bat communities it seem likely that the use of WSI and PB to produce a less structurally complex forested habitat could result in a greater abundance and diversity of bat species. However, these effects could not be reliably detected within a few months of the treatment. This is not surprising and indicates the need for longer-term assessment of the effects of FSD reductions on bat communities.

The BRD contains several cave hibernacula that harbor colonies of Indiana bats during the winter (male and female) and summer (male) months (Harvey 1989). There are other hibernacula within an approximate 10 miles of each study sites. However, no Indiana bats were captured during this investigation. While the reasons for this are unknown, this species may migrate north into southern Missouri during the summer (Harvey and Britzke pers. com.). This hypothesis is supported by the lack of Indiana bat captures during this study and the high capture rates of this species during the summer months in southern Missouri (Harvey per. com.). If this species does migrate from the BRD into southern Missouri, its movement may be related to differences between the habitat preferred by this species and that is found in the BRD. The BRD is predominantly composed of forest characterized by dense mid-and understory, agriculture fields, and dense riparian areas. Indiana bats prefer riparian areas with reduced woody vegetation and avoided agriculture areas (Humphrey et al. 1977). Several other forest stand structural components could affect the presence of this species. These characteristics include a habitat characterized by, low mid- and understory densities, high overstory species richness, and low understory species richness (Brack 1983). Because most areas within the BRD are more structurally complex, these habitats common in the BRD could select against this species' wing morphology and echomorphology.

Roosting Ecology

Roosts are suggested as a limiting factor in bat populations (Lewis 1995; Brigham and Fenton 1986; Kunz 1982; Humphrey and Cope 1976). In some locations bats reproductive success could be limited due to the lack of suitable roost (Brigham and Fenton 1986). Fidelity to a particular roost is often used as a measure of roost suitability with high fidelity to a roost suggesting that it is better suited for reproduction (e.g., Humphrey 1975). During this study, northern long-eared bats changed roosts nearly every day, suggesting that in the BRD at least, this species exhibits a high degree of roost liability. Roost availability and permanency have been proposed as factors that affect roost fidelity (Brigham 1991). Although individual females frequently changed roosts, they also showed a high degree of fidelity to particular areas, often selecting trees within 40 m of prior roosts. Therefore, the benefits of switching roost must exceed the costs associated with locating and using several difference roost trees (Lewis

1995). Reasons for switching roost include disturbance (Kunz 1982), predation (Wilkinson 1985), foraging area (Fleming 1988), parasitism (Fleming 1988; and Wilkinson 1985), and microclimate/habitat (Fleming 1988; Kunz 1982).

Undisturbed animals should have high site fidelity and that increasing predator (Wilkinson 1985) or human (Kunz 1982) disturbance should correlate with lower site fidelity. In this study factors that may have affected roost tree fidelity were not examined, but it seems likely that bats could have been effected by various disturbance, either from presence of predators and/or from data collection activities. Our observations indicate that disturbances associated with data collection were probably not a major influence during this study because each radio-tagged bat exhibited daily roost switching throughout the study. Bats observed during this study remained in the same general area, sometimes only moving approximately 10m from the previous roost. Similar trends were observed among the northern long-eared bats roosting in reference-sites where there was not any habitat alterations resulting from forest management. Moreover, northern long-eared bats are often observed roosting within and/or among highly disturbed buildings, bridges, and other man-made structures. Thus it seems likely that timber management activities probably did not cause these bats to move to alternative areas.

Microclimate/habitat has been proposed to influence site fidelity (Fleming 1988; Kunz 1982). The results of several studies, including this one, have suggested that bats appear to be preferentially selecting roost based the characteristics of the area within and surrounding the roost. Parameters that may be important could include the relative size of the roost tree, roost tree decay stage, and species of the roost tree, its relative position in the canopy and its distance from surrounding trees (e.g., Brigham et al. 1997; Sasse and Pekins 1996). However, microclimate/habitat of the roost may be less important that the surrounding habitat and the overall forest composition. Parameters like dominant tree species, average forest age, and stand level structural complexity may be very important in roost selection. Therefore, while bats may appear to select roost based on roost tree characteristics, they may instead be selecting roost based on stand level habitat characteristics. Although in this study no significant effects of treatment types on roost-site selection were observed, there was a correlative trend between FSD and northern long-eared bats roosts sites.

Strong correlations between FSD and roost-tree density and the minimum foraging distance were identified during this study. My observation that minimum foraging distance decreased as the relative FSD increased is consistent with other studies that evaluated the effects that physical structure (e.g., Krusic and Neefus 1996) and abundance of clutter (Mackey and Barclay 1989) on the distribution of foraging bats. A strong correlation between FSD and the roost parameters, i.e., roost-tree density and minimum foraging distance was found. This implies that bats tend to avoid densely vegetated mid-and understories and prefer less structurally complex habitats (Betts 1996; Kalcounis 1994).

Several reasons have been proposed to explain why bats tend to avoid regions of the forest with higher stand densities. In less dense forest habitats predation rates have been shown to decrease (Barclay et al. 1982) probably because bats are able to exit and enter roost at a faster rate and because their ability to locate new alternative roost increases. As described above, lower FSD may also benefit bats by decreasing energy expenditure (Vonhof 1996).

Bats ability to locate new and alternative roost could be affected by the structural complexity of a forest, because there are more obstacles bats must detect and avoid in relatively more structurally complex forest (Fenton 1990). Dense habitats may limit bats view of suitable roost trees below the canopy by reducing acoustical and visual acuity. Bats' often select trees that are high above the canopy to reduce impacts of dense vegetation in the understory (Betts 1996; Vonhof 1996; Kalcounis 1994), which may also explain why northern long-eared bats roost-tree and the minimum foraging distance was negatively correlated with FSD during this study. Since flight is costly (Speakman and Racey 1991), energetic expenditures could be detrimental to bats in relatively more structurally complex habitat. Thus, it would seem preferable to have a clear flight path to the roost and its entrance (Vonhof 1996). This would allow for bat species that forage in the interior portions of the forest to migrate and forage more effectively. Micro-and microhabitat characteristics within and among forests may also assists bats' by enabling them in locating the same and/or alternative roost where large and/or conspicuous tees may stand out as landmarks.

It is possible that bats could have used roads and trails as flight and foraging corridors throughout the forest. These roads could have provided bats that were radio-tagged within both reference areas access to portions of the forest that would otherwise been inaccessible, due to the complexity of the forest in reference areas. This could potentially cause analyses between FSD and RTD and MFD associated with reference areas to be underestimated. Thus, bats may be able to forage in areas that have a relatively high FSD by utilizing relatively open linear habitats such as roads and/or trails has possible flight corridors. Bats may use these corridors like “highways” to migrate to day roost, which could potentially be located in less dense areas of the forest (Krusic and Neefus 1996).

In conclusion, the data herein illustrate that by reducing structural complexity of the forest at the forest stand level or at a greater landscape scale, can greatly impact both floral and faunal assemblages. By reducing the forest stand density and administering a prescribed burn treatment, a significant increase in the amount of herbaceous ground cover was achieved. Insect assemblages positively responded to the increase in herbaceous ground cover, which served as food resources for this taxonomic group. Bats possibly responded to this increase in abundance and diversity in insects’ while potentially reducing their energy expenditures when foraging in response to open space created in this ecosystem restoration, *i.e.*, an oak savanna habitat type. Northern long-eared bats were also able to increase the distance between roost trees and increased their foraging home range, which was shown in the increased distance between water resources and roost tree locality.

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