THE EFFECT OF WOODLAND RESTORATION ON BATS IN A METROPOLITAN ENVIRONMENT

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By

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ABSTRACT

Urban expansion fragments natural habitats, which can increase susceptibility of wildlife to invasive species, predation, disease, and pollution. However, these habitat fragments may benefit some wildlife species. Recently, many organizations have made efforts to maintain and restore (e.g. through prescribed burning, invasive/exotic species removal, snag recruitment, drain tile disablement, and deer population control) natural areas in metropolitan areas, which led to changes in forest structure, such as reduced tree and shrub densities and decreased canopy cover. Habitat restoration may affect how species, such as bats, use forest fragments in urban environments.

Bats are highly vagile and are able to exploit habitat patches, particularly woodlands, in urban landscapes. In North America, bats use forests for foraging, roosting, and/or rearing young. Previous studies have shown a positive relationship between bat activity and woodlands in urban environments, and have suggested the importance of microhabitat to bat activity. I determined relationships between (1) restoration efforts and general bat activity, (2) general bat activity and microhabitat charactersitics (3) interspecific variation with woodland variables, and (4) roost selection of northern myotis (*Myotis septentrionalis*) and red bats (*Lasiurus borealis*) in woodland fragments.

During the summers of 2004 and 2005, I used ultrasonic detectors to monitor both general and species-specific bat activity in 9 forest preserves that are in various stages of restoration. I identified 5,074 of 7,652 collected bat passes to species during 5,760 detector hours. Restoration variables and general bat activity were compared using linear regression and Akaike's Information Criterion (AIC) for model selection. Prescribed burning and invasive species removal were positively related to general bat activity. Relationships between general bat activity and vegetation variables were determined using mixed-effects linear regression models. Model fit was compared using AIC. General bat activity was positively related to small tree density (7.7-20 cm DBH) and inversely related to shrub density and clutter at 0-6 m heights.

I used partial canonical correspondence analysis with site and year as covariables to determine relationships between vegetation variables and bat species. Red bats were associated with small and medium tree densities and inversely related to clutter at 0-9 m. *Myotis* spp. were positively associated with canopy cover, clutter at 6-9 m, and small and medium tree densities. Silver-haired bat (*Lasionycteris noctivagans*) activity was associated with more open forests. Big brown bats (*Eptesicus fuscus*) were not strongly associated with any measured vegetation variable.

In 2005, I radio-tracked 5 northern myotis and 6 red bats to determine roost, plot, and stand characteristics that may clarify roost selection in urban environments. Northern myotis' roosts were positively associated with snags that had exfoliating bark and were located in unrestored woodlands. Red bat roosts were primarily located in unrestored woodlands and in residential lawns.

These results suggest bats may respond to some forms of woodland restoration in urban landscapes. Sites that had repeated measures of restoration had greater overall bat acitivity. However, species-specific responses to vegetation and to roost selection differed and should be taken into consideration when developing management plans.

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CHAPTER 1

ECHOLOCATION AND FORAGING BEHAVIOR OF BATS (ORDER CHIROPTERA)

Bats, the only mammals capable of powered flight, are an extremely unique taxonomic group. They form their own order, Chiroptera, which is divided into two subgroups: Megachiroptera and Microchiroptera. The main distinguishing feature between these two suborders is the ability to echolocate. Megachiropterans do not echolocate at high frequencies but navigate using large eyes that are placed forward on the head or with low frequency pulses (Findley 1993). Microchiropterans, the focus of this study, echolocate by emitting pulses (20-150 kilohertz) from their larynx and expelling the pulses through their mouths. Many microchiropterans have developed elaborate facial and aural features that are designed to maximize the quality of returning calls. Echolocating species use their radar to navigate around objects and to locate prey (Griffin 1958, Findley 1993). Echolocation calls allow bats to detect prey and to determine prey characteristics despite the presence of surrounding clutter. For example, bats can decipher surface detail, size, form, speed and direction of prey movements (Schnitzler and Kalko 1998). Bats use this information to capture optimal prey types.

To distinguish between outside factors and prey, the structure of echolocation signals may vary considerably (Fenton 1982). Microchiropteran calls are composed of constant-frequency (CF), quasi-constant frequency (QCF), and/or frequency-modulated (FM) signals (Schnitzler and Kalko 1998). CF, a narrowband signal, is typically used in the search phase of a call and has a long duration time (to 100 msec in Rhinolophid bats). It is a concentrated call that detects weak echoes, such as those produced from insect wings. Portions of CF are used to create a Doppler effect. However, CF does not accurately determine the location of objects. In contrast, broadband signals, which include uni- or multiharmonic FM signals, are unreliable for finding attenuated echoes. Instead steep or shallow signal components measure range and angles of the prey. This allows bats to specifically identify the exact location of an insect. FM signals vary from high to low frequencies and last 1-5 msec (Fenton 1982).

Bats have adapted to forage in a variety of habitat types by altering their foraging strategy (Schnitzler and Kalko 1998). Echolocating bats encounter two major problems when foraging: echolocation call interference from clutter (e.g. leaves, twigs) and the ability to locate and capture prey while avoiding obstacles. Swift and strong flying bats, such as *Lasiurus* spp., tend to forage in open spaces. Their wings are long and narrow, which enhances speed over maneuverability. Because they hunt in large areas with sparsely dispersed prey, larger bats use long pulse signals comprised of QCF or FM-QCF elements. These long narrow band frequencies increase the detection range and thus aid in hunting success. As bats approach prey, the once low repetition pulse dramatically increases, becoming what is known as the feeding buzz (Fenton 1982).

Bats that hunt in clear areas are almost guaranteed that their target is a prey item. However, bats that forage within forest interiors must contend with vegetation obstructions and extrinsic echoes. In cluttered habitats, pulse intervals emitted by bats are considerably shorter than for those observed in open spaces. Bats that frequent edges and gaps typically utilize broadband FM signals, such as FM-QCF (Schnitzler and Kalko 1998). In particular, *Pipistrellus* and *Eptesicus* spp. forage along forest edges and in spaces between canopy and subcanopy layers. These species have been observed converting to narrowband FM-QCF signals when crossing into uncluttered regions.

Fenton et al. (1995) suggested that bats in highly cluttered environments do not rely on echolocation to detect prey but use other spectral cues. The primary function of echolocation in this environment is to ascertain the position of neighboring structures. The call composition can be dominated by either CF or FM elements depending on the bat species. Vespertilionids that forage in cluttered habitats utilize FM dominated calls. The broadband signals have a short duration (1-3 msec). Species using this technique are often considered gleaners.

Although there are different hunting strategies for each habitat classification, many species of bats demonstrate remarkable variability (Fenton 1995). Several species are capable of using a variety of foraging techniques, including gleaning, aerial hawking, trawling, and perch-hunting. Wing morphology often dictates foraging behavior. Wing loads (weight divided by wing area) and aspect ratios (wing span squared and divided by wing area) relate to speed and maneuverability of bats (Verboom 1997). High wing loads and large aspect ratios indicate that the bat has long, narrow wings and is a fast flyer (e.g. *Lasiurus* spp.). Slow-flying bats, such as *Myotis* spp., with high maneuverability have low wing loads and low aspect ratios.

Bats have a wide range of diets. Some bats are considered generalists while others prey upon specific organisms, such as frogs, spiders, fish, and even other bats

(Schulz 2000, Nowak 1994). The majority of bat species found in the nearctic region belong to the family Vespertilionidae, which primarily consume insects such as agricultural and forest pests (e.g. Anthony and Kunz 1977, Acharya 1999, Agosta and Morton 2003).

Vespertilionids are relatively small in size, ranging from 4 to 50 g (Nowak 1994). These bats are characteristically recognized by a V-shaped uropatagium, which is a membrane that extends the length of the tail. Although these bats are small in size, they are considered K-strategists, meaning they live for long periods and have low fecundity (Findley 1993). Mating typically occurs in autumn, but females delay fertilization until spring. Parturition occurs in late spring to early summer, usually producing 1-2 offspring per female.

Bat roosting sites for bats vary depending on species and time of year. Many species, such as *M. griescens* and *Euderma maculatum*, rely on caves or old mine shafts to provide adequate shelter for nursery colonies and hibernation (Easterla 1973, Tuttle 1976). Others, such as *M. sodalis, M. volans*, and *M. thysanodes*, are known to utilize both rock formations and tree roosts (Cryan et al. 2001, Britzke et al. 2003). In contrast, *Nycticeius humeralis, Lasionycteris noctivagans*, and *Lasiurus* spp. prefer tree roosts in the summer and then will migrate or will remain in the trees or on the ground in a torpid state (Campbell et al. 1996, Robbins 2004). Buildings, bridges, and other man-made structures are often used by *E. fuscus* and *M. yumanensis* for day and night roosts (Adam and Hayes 2000, Arnett and Hayes 2000). Cave bats demonstrate extreme roost fidelity, but tree-roosting bats routinely switch trees relocating on average every 3-5 days (Kunz 1982, Hurst and Lacki 1999, Cryan et al 2001). Bats are strongly influenced by habitat quality, because they require areas with an abundance of insects, a variety of roosting structures, and ample water availability.

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CHAPTER 2

THE EFFECT OF WOODLAND RESTORATION ON BAT ACTIVITY IN A METROPOLITAN LANDSCAPE

INTRODUCTION

Urbanization and human encroachment threaten wildlife through pesticide use (Nowell et al. 1999), habitat fragmentation (Dickman 1987), invasive species introduction (MacDonald et al. 1989), fire suppression (George and Zack 2001), road construction (Forman and Deblinger), and environmental contamination (Kisseberth et al. 1984). Natural habitat remnants in urban environments are particularly at risk from colonization by invasive species (Mooney and Cleland 2001, Borgmann and Rodewald 2005). Shrub species, such as Amur honeysuckle (*Lonicera maackii*) and European buckthorn (*Rhamnus cathartica*), invade Midwestern forest understories and threaten native biodiversity through competition for resources and changes in forest dynamics (Williamson 1996, Mack et al. 2000, Heneghan et al. 2004). Additionally, fire suppression promotes growth of fire-intolerant shrubs and trees, thereby altering plant species composition and structure (Abrams 1992). Failure to manage urban forests with fire or removal of exotic species may lead to increases in tree density, shrub density, and canopy cover resulting in decreased species and forest structural diversity (Anderson and Brown 1986, Albrecht and McCarthy 2006).

Forests are of particular importance to bats (order Chiroptera) in the eastern United States, because several bat species utilize forests for foraging, roosting, and/or rearing young (e.g. Brigham and Barclay 1995, Britzke et al. 2002, Hogberg et al. 2002). Communal bats, such as Indiana bats (*Myotis sodalis*) and northern myotis (*Myotis septentrionalis*), may use snags or standing dead trees during summer for roosting and reproduction (Foster and Kurta 1999). Solitary species, such as red bats (*Lasiurus borealis*), typically roost among foliage in live trees (Hutchinson and Lacki 2000). In winter months, recent research has shown some bat species to continue to roost and forage within forests (Boyles et al. 2005, Hein et al. 2005). Forests also provide bats with edges, which are useful for navigation, foraging, and protection from wind and potential predators (Verboom and Huitema 1997). At the landscape level, woodlands were selected more often by bats than any other landuse types in urban and agriculture regions (Walsh and Harris 1996, Gehrt and Chelsvig 2003).

Interspecific differences in bat use of woodlands probably reflect variation in flight morphology and foraging strategy (Norberg and Rayner 1987). Bats with high wing loading (weight divided by wing area) and large aspect ratios (wing span squared, divided by wing area) typically are strong, fast flyers that forage in more open habitat types. Species with low wing loading and aspect ratios are associated with more cluttered habitats because of increased maneuverability. Due to these morphological differences, bats utilize a variety of forest types, ages, and structures (e.g. Crampton and Barclay 1998, Law et al. 1999, Erickson and West 2003). For example, silver-haired bats

(*Lasionycteris noctivagans*), a large-bodied species, forage more often in clearcuts and avoid interior forests (Patriquin and Barclay 2003). Although there is variability in species-specific responses to habitat structure and types, the overall trend suggests a negative correlation between tree density and general bat activity (Erickson and West 2003, Gehrt and Chelsvig 2003). Areas with high foliage and stem densities (e.g. vegetation clutter) may interfere with echolocation and may increase energetic costs associated with foraging (Aldridge and Rautenbach 1987, Jones 1994, Brigham et al. 1997).

Forest management may benefit bats because it can reduce structural complexity by generating edges and openings and by reducing clutter for foraging bats (Grindal and Brigham 1999). Several habitat use studies have compared bat activity between various forest management methods, such as types of timber harvesting (Krusic 1995, Grindal and Brigham 1999, Zimmerman and Glanz 2000, Elmore et al. 2004). Research has demonstrated differences in bat activity along edges created from clearcuts, in gaps formed from group selection harvests, and in intact, unharvested forests (Grindal and Brigham 1998, Swystun et al. 2001, Menzel et al. 2002, Owen et al. 2004). Collectively, bat activity has been found to be highest along edges or within forest gaps, possibly because of increased foraging or navigation efficiency. However, within stands, Patriquin and Barclay (2003) and Kutt (1995) did not observe statistically significant differences for general bat activity in thinned versus unthinned stands. Conversely, Humes et al. (1999) did not identify a difference between bat activity in old-growth forests and thinned stands; but bat activity was significantly lower in unthinned hemlock (*Tsuga heterophylla*) forests. Habitats in old-growth and thinned forests, which were positively associated with bat activity, also were characterized by lower tree density, increased shrub cover, large diameter trees and snags, and reduced canopy cover. These studies suggest bat activity may be influenced by forest management practices and various forest structures within rural landscapes.

In urban environments, forest management primarily consists of restoration, however little information is available about how bats respond to urban forest management. Urban bat studies demonstrated that bats were positively associated with woodland patches in urban environments, and suggested that bat activity may be affected by microhabitat preferences (Walsh and Harris 1996, Gehrt and Chelsvig 2003, 2004). However, no research has examined relationships between specific vegetation characteristics of forest fragments and bat use or the effects of restoration on bats.

Hourigan et al. (2006) conducted a bat study in an urban tropical environment, and observed no correlation between bat activity and foliage density. The study measured 4 vertical strata in several habitats, but the highest measurement was at 5 m. Bats forage at various heights within a forest, so it is possible their results excluded bats that fly at greater heights. Additionally, the study did not examine effects of urban forest management (Hourigan et al. 2006).

The Lake County Forest Preserve District (LCFPD) was created in 1961 to serve as a means to protect natural areas in the midwestern United States. Historically, the midwestern region was deforested and drained of wetlands to maximize the amount of arable land. Citizens concerned about the loss of natural habitats sponsored referendums that eventually led to creation of a forest preserves system in Lake County, IL (http://www.lcfpd.org). Ultimately, LCFPD became one of the founding members of

Chicago Wilderness, a coalition of public and private organizations that sought to protect, restore, and maintain natural areas at the regional level (Moskovits et al. 2002). One of LCFPD goals was to maintain functional, natural ecosystems in an urban environment. To accomplish this goal, the county purchased land parcels to protect natural areas from development (Moskovits et al. 2004). Forest fragments were set aside and old farmlands were converted to grasslands or savannahs. These forests were at risk due to spread of invasive species, water manipulation, and excessive deer populations. LCFPD implemented the following restoration treatments to restore the structure and function of several forest preserves: prescribed burning, invasive species removal, drain tile disablement, deer population control, reseeding/replanting of native vegetation, snag recruitment, and reduction of canopy cover.

Returning remnant woodlands to European pre-settlement conditions was not always a feasible goal in urban environments. However, LCFPD based management decisions on pre-settlement notes that describe forest conditions, and implemented restoration activities that would return woodlands to a semblance of their condition before European arrival. Historically, forests in Lake County were more open with less structural diversity. Oak (*Quercus* spp.) and hickory (*Carya* spp.) forests were dominant in the western portion of the county and subjected to frequent fires. In the maple forests on the eastern side of the county, fire suppression in the last 100 years has led to an overstock of maples (*Acer* sp.), which has subsequently shaded out some plant species. Restoration practices were used to open canopy cover, reduce tree density, and remove invasive plant species (Ken Klick, Lake County Forest Preserve District, pers. comm.)

For this study, I evaluate relationships between restoration variables and general bat activity. Additionally, I compare microhabitat vegetation variables to relative bat activity. Gehrt and Chelsvig (2003) measured tree density and canopy cover as part of a landscape study to determine if microhabitat or landscape variables were more important to bat activity within the Chicago metropolitan area. However, they did not quantitatively measure other forest characteristics such as vertical structure, shrub density, or tree density of various size classes. To my knowledge, no published work has previously assessed the effects of restoration (e.g. prescribed fire, invasive shrub removal) on bat activity in urban forests.

I had three primary objectives for this study: (1) to determine if relationships exist between restoration efforts and bat use of woodlands at the stand scale, (2) to evaluate urban woodland characteristics associated with general bat activity at the microhabitat level, and (3) to assess species-specific responses, as measured by echolocation activity, to habitat structure at the microhabitat level. I hypothesized that bats would collectively prefer sites that have undergone extensive restoration (e.g. repeated prescribed burns, removal of invasive shrub species, selective thinning of trees, etc.) because of reduction in structural complexity of the forest in Lake County, IL. Larger-bodied species (*L. noctivagans, Eptesicus fuscus, Lasiurus cinereus, and L. borealis*) should utilize habitats with reduced foliage and tree density, and small-bodied (*Myotis* spp. and *Pipistrellus subflavus*) species are expected to have relatively higher activity in association with more dense forest characteristics (Norberg and Rayner 1987).

STUDY AREA AND METHODS

Study Area

I focused my fieldwork within Lake County, IL, which is located within the larger Chicago metropolitan area (Figure 2.1). In 2005, Lake County had a population density of 556/km^2, a total population of 644,000, and a 2.6% annual growth rate (Natural Resources Inventory 2003). Cover composition within the county consisted of urban (47%), agricultural (15%), wetlands (8%), and open spaces (29%; i.e. grasslands, forests, prairies, and savannahs) (Wang and Moskovits 2001).

I qualitatively selected nine Lake County forest preserves scattered across the county with various levels of forest restoration for study sites (Figure 2.1). Figure 2.2 illustrates restoration differences between a site that has undergone multiple restoration efforts and a control site (no restoration). Lake County Forest Preserve District provided current restoration and vegetation classification data, which were used to identify potential study sites (Debbie Maurer, Lake County Forest Preserve District, pers. comm., Wang and Moskovits 2001). Historically, fire influenced plant species composition in several Lake County forest preserves (Figure 2.1; Ethel's Woods, Gander Mountain, Grant Woods, Marl Flats, Sequoit Creek, and Wadsworth Savannah) (Ebinger 1997). Forests subjected to repeated fire exposure were typically dominated by fire-tolerant tree species, such as oaks and hickories. Some forest preserves, specifically those located on the eastern bank of the Des Plaines River (Figure 2.1; MacArthur Woods, Ryerson Conservation Area, and Wright Woods) were less influenced by fire impacts, and therefore were dominated by more shade-tolerant and fire-intolerant maples.

Bat Monitoring

I used acoustic surveys with broadband ultrasonic bat detectors (AnaBat II detectors; Titley Electronics, Ballina, New South Wales, Australia) to monitor bats from mid-June to early September in 2004 and from late May to late August in 2005 (Kunz 1988). Audiocassette recorders (Optimus model, Radio Shack, Fort Worth, Texas, USA) were attached to bat detectors to record bat activity within forest preserves (Krusic et al. 1996, Menzel 2002, Erickson and West 2003). I conducted acoustic surveys at 1 forest preserve each night for a total of 90 nights (5 per site per year). Only 1 forest preserve was monitored each sampling night. Forest preserves were randomly selected without replacement to determine order of sampling and were monitored within 9 days of the other sites to ensure equal participation throughout the study season. Because of the sampling nature of detectors, acoustic surveys were used to provide a relative index of bat abundance (Thomas and West 1989).

I used a stratified random sampling design to adequately represent restoration differences among urban forests. For acoustic sampling, 20 line transects (30 m in length) were randomly located in each forest preserve with 30 m buffers around each line transect (Figure 2.3). Transects were identical for both years unless extenuating circumstances (e.g. flooding, restoration impacts) required transects to be relocated. Detectors (*n*=4) were placed 10 m apart on transects and angled away from forest edges and each other. To ensure an equal rate of detection, detectors were pointed toward vegetation gaps (Barclay 1999).

Detectors were placed in a protective box, mounted ~1.5 m above ground, and angled 45 degrees from the ground. Detectors were calibrated before each field season and checked routinely throughout the summer. Sensitivity was fixed at 8 and the division ratio at 16. Detectors and recorders were controlled via voice-activation and operated for 4 hours, beginning at sunset. Data were removed from analysis if equipment failure or inclement weather occurred during sampling, and the forest preserve was resampled the following night.

Echolocation passes or call sequences were identified using AnaBat V Zero Crossing Analysis Interface Module and AnaBat6 (Titley Electronics, Ballina, New South Wales, Australia) (Gehrt and Chelsvig 2003). For general bat activity, bat passes were defined as 2 or more calls from a single species separated by >1s (White and Gehrt 2001). To identify passes to species, I selected search-phase call sequences (≥4 calls). Passes with < 4 calls or with erratic sequences, such as feeding buzzes, were censured from species identification. Call sequences that met identification requirements were standardized using a filter in AnaLook, which reduced extraneous noise and interference (Gehrt and Chelsvig 2004). Occasionally, the filter was unable to remove all noise, so it was necessary to manually edit echolocation files. By reducing noise, the software program provided more accurate readings of pass parameters.

For each pass, I recorded mean estimates of the following parameters: characteristic slope (slope of the flattest part of the call), maximum, minimum, and mean frequencies, characteristic frequency (kHz) and slope, frequency of the knee (point at which the slope transitions from the steep to flatter portion of a call), total duration of call (ms), time between calls (ms), time from beginning of call to characteristic frequency (ms), and time from start of call until frequency of knee is measured (ms). See White and Gehrt (2001) for more detail.

I used a combination of quantitative and qualitative methods to identify acoustic recordings to species or species groups. To quantitatively identify call sequences, I compared unknown pass parameters to known parameters from reference calls using discriminant function analysis (Gehrt and Chelsvig 2004). Reference calls were previously collected during a landscape study in Chicago, IL (See Gehrt and Chelsvig 2004 for more information). The reference call library consisted of 211 passes from the following bat species: big brown bat (*Eptesicus fuscus*), eastern pipistrelle (*Pipistrellus subflavus*), silver-haired (*L. noctivagans*), red bat (*L. borealis*), little brown bat (*Myotis lucifugus*), Indiana bat (*M. sodalis*), and northern myotis (*M. septentrionalis*). Because of similarities in call structure, *Myotis* spp. were pooled and analyzed as a group (Thomas et al. 1987, Hayes 2000).

When possible, I visually identified bat passes to species, and subsequently compared the qualitative identification to the species prediction determined by the discriminant model. If visual identification was confirmed by quantitative classification, then the pass was considered positively identified to species. Discrepancies resulted in categorizing the pass as unknown. Hoary bat (*Lasiurus cinereus*) and red bat passes were only qualitatively identified because of lack of suitable library passes for comparison or because of high variability associated with the species, respectively. Research protocols were approved by the Institutional Animal Care and Use Committee at Ohio State University (protocol #2004A0001).

Restoration and Microhabitat Characteristics

For each forest preserve (e.g. stand level), the following restoration variables were recorded: woodland area sampled (ha), percent area burned (total area burned/total woodland area), burn frequency, burn intensity, time since initial restoration (in years), and binary (yes or no) variables including invasive shrub/tree removal, snag recruitment, deer population control, and drain tile disablement (Table 2.1). Burn frequency was recorded as the number of prescribed burns that occurred within a sampled forest preserves beginning in 1988, the earliest year restoration activities were documented. LCFPD personnel qualitatively estimated temperature (low, medium, hot) and completeness of prescribed burns (% burned), which I used to develop a categorical scale for burn intensity (1-5 with five being the hottest, most complete burn). A fire index was created by averaging burn frequency, burn intensity, and area burned after each variable was standardized by subtracting the mean and dividing by the standard deviation. The equation for calculating fire index was:

Fire Index= Average (Standardized burn frequency + Standardized burn intensity + Standardized % area burned).

Invasive shrub/tree removal entailed the extraction of invasive plant species via mechanical or chemical means. It also included selective tree removal, which was used to open the forest canopy. Snag recruitment was defined as the creating of snags by girdling a live tree. Girdling included cutting through the tree's cambium layer. Sharp-shooters were used to manage deer, and clay drain tiles were broken to restore a more natural water flow.

In 2005, I measured the following vegetation characteristics for 16 of 20 line transects per forest preserve: tree density (stems/ha), shrub density, canopy cover, and clutter. Trees were identified by species and classified into one of 3 tree size classes determined by diameter at breast height (DBH): small trees (7.7 cm-20 cm DBH), medium trees (20.1 cm-33 cm DBH), and large trees (≥33.1 cm DBH). Heights, DBH, and species were recorded for all trees. The point centered quarter method was used to estimate tree densities (James and Shugart 1970). Shrubs were classified as ≤ 7.6 cm DBH and ≥ 1.5 m height. I counted, identified, and measured shrub diameters and heights along each transect (2 m x 30 m). Canopy cover was estimated using a spherical densitometer at each detector point and then averaged across each transect.

Clutter was measured at various heights to assess vertical forest structure. Measurements were recorded at 0-3 m, 3-6 m, 6-9 m, and 9-12 m heights (Karr 1971). A 12-m pole with twelve 0.25 m subsections was marked to indicate different heights (Tanabe 2002). If foliage or stems touched a subsection, then the subsection was counted and then averaged for each height category.

Statistical Analysis: General Bat Activity

Analyses for general bat activity were performed using the software program R, version 2.2.0, packages nlme, and Rcmdr (R Development Core Team 2005). For the following analyses, I naturally-log transformed bat activity, shrub density, heights for clutter, small tree density, and large tree density to improve skewness and to meet normality assumptions. Canopy cover was arcsine-transformed. Two sets of 11 candidate models were developed to evaluate the effect of restoration techniques on bat activity at the stand level and to assess relationships between vegetation structure and bat activity at the transect level.

To determine the relationship between restoration practices and bat activity, bat activity was linearly regressed on restoration variables (Kutner et al. 2004). Bat activity, the number of echolocation passes per transect, was averaged by forest preserve and between years (n=9). Eleven generalized linear models were created by using no more than 2 restoration variables in each model to avoid overfitting. A null model was created by setting the response variable (bat activity) against a fixed effect (a random number between 1 and 100) and random effect (year). Because snag recruitment and invasive species removal were concurrently performed during restoration treatments, there was high correlation between these 2 variables (post hoc r =0.791, P= 0.011). Therefore, only invasive species removal was included in models. Akaike's Information Criterion (AIC) and model averaging were used to evaluate relative variable importance and obtain multi-model inference (Burnham and Anderson 2002).

Linear mixed-effects models were used to determine the effect of microhabitat variables on general bat activity at the transect level (Venables and Ripley 2002). Sampling year and site (e.g. forest preserve) were modeled as random effects on bat activity, while vegetation characteristics were modeled as fixed effects on bat activity. Modeling year and site as random effects allowed us to account for their variability and to elucidate the underlying relationship between vegetation characteristics and bat activity. General bat activity was obtained by averaging the number of bat passes per line transect and only transects with at least 1 pass detected (0.25) were used in the analysis.

Because shrub density, 0-3 m clutter, and 3-6 m clutter were highly correlated (Pearson correlation: shrub/0-3 m clutter, r= 0.651; shrub/3-6 m clutter, r= 0.595; 0-3 clutter/3-6 m clutter, r= 0.475), a single independent variable was created using principle component analysis (PCA) (McGarigal et al. 2000). Based on scree plot and broken stick criteria, only principle component scores from the first axis, henceforth PC1, were used in model selection. Models were constructed to represent the various layers within forests. For example, PC1 and small tree density were combined into a model to correspond with forest understory. Tree densities were combined in 1 model to determine if excluding PC1 affected bat activity. Forest mid/overstory was signified by canopy cover, clutter at 9-12 m, and large tree density. Because AIC works best with small candidate model sets, I selected 11 models that most appropriately described the overall forest structure. To test the strength of the relationship between the most parsimonious model and general bat activity, I calculated, under the null and reduced models with site as a random effect, an approximation of explained variance by dividing the variance from the selected model by the variance from the null model and subtracting the number from 1.

Statistical Analyses: Interspecific Variation

To determine interspecific variation in the relationship between activity and woodland characteristics, I used CANOCO Version 4.53 and CANODRAW for Windows 4.12 to perform partial canonical correspondence analysis with year and site as covariables (CCA) (ter Braak 1986, ter Braak and Smilauer 1998). This analysis produces ordination axes based on linear combinations of habitat variables, so variation between bat species could be determined directly with vegetation variables. Most importantly, CCA performs well with many problems that are typically associated with bat activity studies, such as skewed species distribution, irregular study designs, highly correlated habitat variables, and unknown factors that may affect species distribution (Palmer 1993). I log-transformed species activity and the following vegetation components as suggested by Palmer (1993): shrub density, small tree density, large tree density, and clutter at all heights. Canopy cover was arcsine-transformed. Transects with missing vegetation data were removed from analysis. Bat species incorporated in analysis were *L. borealis, E. fuscus, Myotis* group, and *L. noctivagans*. *P. subflavus* and *L. cinereus* were excluded from analyses because of lack of data (occurred on < 10 transects). Forest preserves (site) and year were covariables in the analysis (Lepš and Šmilauer 2003).

Scaling for the unimodal method focused on inter-species distances and used biplot scaling. I performed 2 test statistics using Monte-Carlo permutation to determine the significance of the first ordination axis and of the canonical axes together (both $P \leq 0.05$) (ter Braak 2002). I completed 5,000 permutations under the reduced model to test confidence at the 5% significance level. Data were graphically represented in vegetation-species biplots. Vector length and angles referred to the strength of association between species and habitat characteristics (ter Braak 1986).

RESULTS

General Bat Activity

I surveyed bat activity for 90 nights yielding 5,760 detector hours (1 detector per 1 hour). Each forest preserve was monitored for 5 nights each year, and a total of 7, 652 bat passes were recorded.

The number of passes collected from both years (e.g. total number of passes recorded at 1 transect from both years) ranged from 0 to 653 for each transect in a forest preserve. Mean (± SE) number of passes per transect in each forest preserve ranged from 2.45 ± 1.21 to 74.9 ± 29.6 (Figure 2.4). Grant and Ethel's Woods had the highest mean bat activities for both years. These forest preserves had undergone multiple prescribed burns, invasive species removal, and snag recruitment. Sequoit Creek, a control with no restoration, had the lowest mean bat activity for both years.

Descriptive vegetation characteristics for each forest preserve are shown in Figures 2.5-2.8. Mean (± 1 SE) tree densities ranged between 4.7 ± 0.43 and 16.0 ± 0.97 trees per hectare (Figure 2.5), and mean count of shrubs per transect per site varied between 1.0 ± 0.32 and 32.1 ± 3.44 (Figure 2.6). Estimated canopy cover was between 90.9-98.8% (Figure 2.7), and estimates of clutter varied between 0.48 ± 0.11 and 3.3 ± 0.55 mean hits per height class (Figure 2.8).

Model selection criterion provided evidence that bat activity was most parsimoniously explained by the fire index (β=3.03 ± 6.66, Akaike weight = 0.384) (Tables 2.2, 2.3). Moderate support of a positive relationship between invasive species removal (β=8.72 ± 24.51, Akaike weight = 0.260) and bat activity is provided. Because the null model ranked fourth according to Δ AIC_c values, I do not believe bat activity was related to time since initial restoration or subsequent models. Evidence ratios suggest the fire index model was 1.5 times better than the next best model and 4 times more likely than the null model.

Conversely, AIC analysis provided strong evidence that general bat activity was affected by microhabitat variables (Table 2.4). Bat activity was negatively related to PC1 (β=-0.186 ± 0.06) and positively related to small tree density (1.23 ± 0.20, Akaike weight = 0.908). The proportion of variance within a site explained by this model was 10.9%. However, between sites, the most parsimonious model explains 78.4% of the variance, strongly suggesting bat activity differs between sites because of certain vegetation characteristics.

Interspecific Variation

During both years, 5,074 of 7,652 (66%) recorded bat passes were qualitatively identified to species. Quantitative and qualitative species identification matched 4,235 out of 5,074 (83%) passes (Table 2.5). The proportion of correctly identified passes varied between year and site from 63 to 96%. The following bat species were identified from recorded passes: *L. borealis, E. fuscus, P. subflavus, L. cinereus, L. noctivagans,* and the *Myotis* group.

The first 2 axes of the CCA biplot explained 10% of total variation in bat activity, however 89.7% of the variation in bat activity was explained by woodland variables included in analysis (Table 2.6). *L. borealis* had high weighted averages for small and medium tree densities and was inversely related to clutter at 6-9m, indicating red bat activity was positively associated with higher small and medium tree densities but are relatively open 0-9 m above ground (Figure 2.9). The *Myotis* group was associated with increases in canopy cover and clutter at 6-9 m (Figure 2.9). In addition, *Myotis* spp. activity was observed in above average small and medium tree densities. *L. noctivagans* was negatively correlated with canopy cover, clutter at 9-12 m, small tree density, and medium tree density (Figure 2.9), but had a slight positive relationship with clutter at 0-6 m. *E. fuscus* did not show a particular trend with any measured forest structure variable.

DISCUSSION

Restoration efforts have altered the forest structure in urban forest preserves in Lake County, IL. Sites that had undergone prescribed burning and invasive plant removal had reduced shrub densities, canopy cover, and cumulative clutter. The positive relationship between general bat activity and restoration efforts in this study suggests bats may benefit from the resulting alterations to woodland structure, although the sample of forest preserves was limited. Of the management techniques considered in this study, prescribed fire had the strongest association with bat activity. Fire is often used in woodland restoration because it effectively reduces understory density, increases tree mortality and cavity formations, and decreases basal area (Chandler et al. 1983, Peterson and Reich 2001). Snag-roosting bats may benefit from prescribed burns, because fire creates suitable roost trees (Boyles and Aubrey 2006). My findings suggest fire frequency and intensity also may be important for foraging bats. However, I could not differentiate relationships between number and intensity of burns (i.e. multiple fires of low intensity) and bat activity since fire characteristics were combined into an index. Additional research is also needed to determine seasonal affects of prescribed burns on bats (Boyles et al. 2005, Hein et al. 2005).

Previous studies have suggested availability of roosts may be a limiting factor for bats, so increases in roost sites would be beneficial (e.g. Cryan et al. 2001, Erickson and West 2003). Unfortunately, we could not decouple the relative importance of fire and roost availability. I was unable to distinguish between snag recruitment and fire during analysis. Sites that had undergone snag recruitment were also subjected to prescribed fires. Additionally, human-induced snags were newly created, so girdled trees had not fully transformed (e.g. presence of exfoliating bark) into suitable roosts for bats.

Microhabitat characteristics were strongly associated with general bat activity in this study, which is consistent with previous studies (Erickson and West 2003, Gehrt and Chelsvig 2003). Understory vegetation density was the best predictor among microhabitat characteristics for determining bat habitat use. Shrub density and clutter < 6 m height were negatively related with bat activity. This result supports earlier studies that indicated high shrub/vegetation densities were associated with lower bat activity levels (Humes et al. 1999, Loeb and O'Keefe 2006). Flight energy requirements for foraging and navigation are greater in habitats with dense vegetation, so bats have been observed to avoid high-clutter areas (Brigham et al. 1997). However, I observed a positive relationship between bat activity and small tree density, indicating a complex relationship between forest structure and bat use. Perhaps, sites with higher small tree densities provided a compromise between reduced clutter for flight efficiency and protection from predators. Additionally, insect availability could be higher in small tree groups, but I recorded few feeding buzzes during this study. Furthermore, other studies have noted a lack of feeding buzzes within forest interiors, and have shown many species of bats prefer foraging along edges or in forest gaps (e.g. Krusic et al. 1996, Grindal and Brigham 1999, Hogberg et al. 2002).

It is possible the association between small tree density and bat activity may be an ecological artifact related to shrub removal. Competition for resources and potentially allelopathic chemicals were removed with buckthorn removal, which would permit growth of small trees (Ahrens 1999). However, Jung et al. (1999) also noted a positive relationship between *Myotis* spp. and trees > 3m tall and <10 cm DBH. Jung's (1999) study was conducted in pine (*Pinus* sp.) mixed wood habitat and did not mention buckthorn/shrub removal, but it lends support for a positive association between some bats and small tree density.

Combining all bat activity into one general category may mask interspecific differences in habitat use, so it was important to address species-specific responses to microhabitat variables. Specifically, red bat activity was positively associated with small and medium tree densities and canopy cover; and negatively associated with shrubs and clutter at 0-6 m heights. My results agreed with other studies that noted red bats preferred denser forest canopies (Menzel et al. 1998), but disagreed regarding vegetation density (Loeb and O'Keefe 2006, Yates and Muzika 2006). In my study, red bat activity was negatively affected by understory clutter, but Yates and Muzika's (2006)'s study did not select an AIC model with understory clutter as the most parsimonious model for predicting red bat occupancy. Loeb and O'Keefe (2006) also used bat occupancy for measuring bat use in an area and subjectively determined vegetation density around sampling points. Discrepancies may exist between these studies, because of differences in sampling methods and analysis. Their studies related habitat characteristics to bat occupancy (e.g. presence/absence). I chose to use mean number of passes to represent bat activity in an area. By using bat activity, I was able to quantify continuous variation in red bat responses to habitat characteristics. Furthermore, quantitative measurement of surrounding vegetation imparted more information for revealing habitat preferences.

No other previously published work has demonstrated a relationship between red bat activity and tree size classes. Yates and Muzika (2006) developed a model that indicated a negative relationship between basal area and red bat acitivity, but they did not separate size classes in their estimates of live tree densities. Roost studies have shown red bat preferred large trees (>30 cm DBH) and reduced basal area surrounding the roost tree (Hutchinson and Lacki 2000, Mager and Nelson 2001).

The *Myotis* group potentially consisted of northern myotis, *M. lucifugus*, and *M. sodalis* (Hoffmeister 1989). Because *M. sodalis* has not been captured during extensive mist-netting (Stanley Gehrt, Ohio State University, personal communication), or otherwise reported for the Chicago area (Hoffmeister 1989), I doubt that *Myotis* calls were from this species. There is a strong likelihood that *Myotis* recordings could represent either northern myotis or *M. lucifugus*, as both have been reported for the area (Gehrt and Chelsvig 2004), and both were captured during mist-netting in the current study. *Myotis* activity was strongly correlated with increased canopy cover and mid- to upper-levels of clutter. Previous research has consistently indicated that *Myotis* spp. occupy cluttered habitat niches (Jung et al. 1999, Owen et al. 2003, Patriquin and Barclay 2003). They are small-bodied and are more capable of maneuvering in dense habitats than larger bat species (Norberg and Rayner 1987).

In contrast to the *Myotis* group, silver-haired bats were negatively related to canopy cover, clutter 9-12 m, and small and medium tree densities. This is consistent with my prediction and preceding studies that suggest large-bodied species are more often associated with open stands, particularly between the shrub layer and tree-limb canopy (Campbell et al. 1996, Patriquin and Barclay 2003, Owen et al. 2004). *L.*

noctivagans activity had a positive relationship with clutter at 0-6 m heights, but dense vegetation at lower heights may not directly affect larger-bodied bats. *L. noctivagans* may utilize more open mid-stories to commute to other areas or to prey on insects above the shrub layer (Bradshaw 1996).

Big brown bats, the most numerous species recorded in Lake County, was not strongly associated with any measured microhabitat variable in this study. Big brown bats are commonly associated with urban environments, (Barbour and Davis 1969, Kurta and Teramino 1992), possibly due to the fact they exploit man-made structures for roosts and can forage in nearby forests (Geggie and Fenton 1985). *E. fuscus* are positively associated with woodlands in metropolitan landscapes (Gehrt and Chelsvig 2004), and based on morphology, I predicted *E. fuscus* would be associated with more open habitats (Norberg and Rayner 1987). However, this study suggests microhabitat characteristics may not be an important factor in determining *E. fuscus* use, providing additional support that the species is a generalist.

Acoustical Sampling Caveats

Although acoustical sampling allows researchers to remotely monitor bat activity, there are some caveats associated with this sampling method (Gannon et al. 2003). Sampling with echolocation detectors provides a relative index of bat activity within a restricted area, and may not be indicative of bat abundance (Thomas et al. 1987, Gehrt and Chelsvig 2003). Additionally, there is a non-independence factor that is associated with recording bat calls, because detectors are unable to record calls that can be distinguished between individuals or between sexes (O'Farrell 1999). I studied general bat activity and species-specific habitat relationships. I assumed a positive relationship between woodland characteristics and level of bat activity would indicate habitat suitability for bats (Stephens and Krebs 1986). Based on this assumption, I used the number of bat passes as a relative index to determine bat habitat use (Thomas and West 1989). I did not attempt to determine gender differences in habitat preference.

There is inconsistency in the published literature concerning the use of bat echolocation passes to identify species (Thomas et al. 1987, Barclay 1999, Robbins and Britzke 1999, Parsons et al. 2000). I utilized a conservative method to identify bat species, which entailed combining quantitative and qualitative identification methods (Gehrt and Chelsvig 2004). By using more stringent rules for identification, I reduced the quantity of passes used in species-specific analyses, but I was relatively assured accurate species identification.

Nevertheless, caution should be exercised when developing management plans, because one of the implicit assumptions of this project was data censorship or extraneous factors would not affect species-habitat relationships (Hayes 2000). The ability to detect a species varies due to the nature of echolocation calls (e.g. high vs. low frequency), atmospheric conditions, and vegetation density (Fenton et al. 1998, Barclay 1999, Broders et al. 2004). I attempted to minimize variation by standardizing sampling methods, such as adjusting detectors towards vegetation gaps and not sampling in wet conditions. Futhermore, Patriquin et al.(2003) did not find a significant difference in the ability to detect bats that echolocate in the 40 kHz range (e.g. *Myotis* spp.) in open, thinned, or intact forest. Subsequently, vegetation density probably minimally affected this study.

Management Implications

My results suggest restoration of woodland habitat and open forest structures were associated with bat activity in urban forest fragments. Maintaining habitat fragments provides bats with areas for foraging and roosting in relatively inhospitable landscapes. Although fire was the most important management variable associated with bat activity, a combination of restoration methods may be the most appropriate approach for managing bats. Mechanical and chemical removal of invasive shrub species hinders regrowth of species, such as buckthorn (Ahrens 1999). Fire may only top-kill such plant species, leaving a viable root system to regrow (Post and McCloskey 1989). Thinning of trees opens canopy cover and reduces tree density, which is positively associated with general bat activity (Humes et al. 1999). However, species-specific considerations should be taken into account when determining the size class and quantity of trees to remove. Draintile disablement helps recreate forest wetlands, thus providing water sources for bats and other wildlife. Although too many deer per area can lead to habitat degradation, deer management was not a critical factor in determining bat activity (Alverson et al. 1988).

Prescribed burning and invasive species removal effectively altered forest structure, but caution should be exercised when using these management techniques. Intensity, frequency, and timing of fire affects wildlife species in different ways

(Kirkland et al. 1996, Converse et al. 2006). Fire may change successional stages of a forest, thus it may cause shifts in wildlife assemblages (e.g. Schulte and Niemi 1998, Ford et al. 1999). Removal of invasive shrub species may also affect other wildlife species. For example, bats were negatively associated with shrub density in my study, but many bird species select habitats with higher shrub densities for nesting and protection from predators (e.g. Steele 1993). Consequently, land managers should incorporate heterogeneity of habitats and habitat structure into management plans. Results from this study should serve as a guide for managing urban forest fragments in the midwestern U.S. or in habitats similar to the Midwest.

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Site	Woodland Area (ha)	Area Burned* (ha)	Fire Index	Invasive Removal	Draintile Broken	Snag Recruitment	Deer Management
Ethel	38.2	63.0	2.916	y	y	y	n
Gander	38.6	114.1	1.337	n	n	n	n
Grant	42.1	118.0	2.411	y	n	y	n
MacArthur	130.0	124.1	0.574	y	y	y	y
Marl Flats	18.8	0.0	-3.336	n	n	n	n
Ryerson	259.0	115.5	-0.890	n	n	n	y
Sequoit Creek	9.8	0.0	-3.336	n	n	n	n
Wadsworth	46.9	188.0	1.966	y	n	n	n
Wright	93.2	37.4	-1.643	n	n	n	y

*Area burned (ha burned per site) may be greater than total area because of repeated fire application.

Table 2.1. Restoration variables recorded for each monitored study site in Lake County, Illinois in 2005. Symbols y and n represent yes and no, respectively. Restoration techniques include: an index calculated from standardized area burned, standardized fire intensity, and standardized frequency (fire index), invasive plant removal (invasive removal), disabling of agriculture draintile (draintile broken), girdling of trees (snag recruitment), and deer population control (deet management).

Model	K	AIC$_c$	ΔAIC$_c$	ω_i
Passes ~ fire	**3**	**86.402**	**0.000**	**0.384**
Passes ~ removal	**3**	**87.183**	**0.781**	**0.260**
Passes ~ years	**3**	**88.884**	**2.482**	**0.111**
Null	3	89.169	2.767	0.096
Passes ~ deer	3	90.040	3.638	0.062
Passes ~ draintile	3	90.673	4.271	0.045
Passes ~ fire + removal	4	93.211	6.809	0.013
Passes ~ fire + years	4	93.571	7.169	0.011
Passes ~removal + years	4	93.735	7.333	0.010
Passes ~ draintile + removal	4	94.030	7.628	0.008
Global	11	639.904	553.502	0.000

Table 2.2. Akaike's Information Criterion adjusted for small sample size (AIC$_c$) results for linear regression models examining relationships between natural log general bat activity (Passes) and restoration efforts (fire =fire index, removal=eradication of invasive plant species, years=number of years since initial restoration efforts have begun, deer= deer management/harvesting, draintile=removal of agriculture draintiles) with number of parameters in model (K) and Akaike's weight (ω_i). Models are ranked by ascending delta AIC$_c$ values, difference in AIC$_c$ value between model and model with lowest AIC$_c$ value. Models with lower AIC$_c$ values than null model are shown in bold text.

Variable	Beta hat	95% CI	RVI
Fire	3.033	-3.63, 9.69	0.384
Removal	8.715	-15.79, 33.22	0.260
Year	0.252	-0.68, 1.19	0.111
Intercept	15.672	-2.58, 33.93	

Table 2.3. Regression coefficients (Beta hat), 95% confidence intervals (95% CI), and relative variable importance (RVI) from averaging linear regression models with Δ AIC_c less than null model.

Model	K	AIC$_c$	ΔAIC$_c$	ω_i
Passes~PC1+sm	**4**	**737.8577**	**0**	**0.908**
Passes~sm +med +lg	5	743.6764	5.818725	0.050
Passes~ sm +lg +canopy	5	744.0905	6.232825	0.040
Passes~ sm +lg +c6	6	750.2052	12.34749	0.002
Passes~lg +c9 + canopy	7	757.0751	19.21742	0.000
Passes~ PC1	3	760.1793	22.32159	0.000
Passes~PC1+lg	4	761.5719	23.7142	0.000
Global	9	765.3308	27.47313	0.000
Passes~lg +c9+canopy	5	768.2832	30.42553	0.000
Passes~med +lg +c9+ canopy	6	769.5131	31.65539	0.000
Null	2	980.7945	242.9368	0.000

Table 2.4. Models, number of model parameters (K), Akaike's Information Criterion adjusted for small sample size (AIC$_{c)}$, delta AIC$_c$ (difference between model and model with lowest AIC$_c$ value) and Akaike's weight (ω_i). Models were based on linear mixed-effect models comparing natural log general bat activity to microhabitat variables (PC1=principal component scores of shrub density, clutter at 0-3 m and 3-6 m heights, sm=small tree density, med=medium tree density, lg=large tree density, canopy=percent canopy cover, c6= clutter at 6-9 m height, and c9=clutter at 9-12 m height). Models are listed in ascending order by delta AIC$_c$.

Site	*E. fuscus*	*L. cinereus*	*Myotis spp.*	*P. subflavus*	*L. borealis*	*L. noctivagans*	Unknown	Grand Total
				2004				
Ethel	741	1	129	0	17	3	130	1021
Gander	133	0	10	0	10	0	19	172
Grant	413	1	122	1	189	13	134	873
MacArthur	61	0	8	0	7	3	3	82
Marl Flat	92	1	28	0	16	8	19	164
Ryerson	97	0	4	1	1	3	17	123
Seq. Creek	7	1	9	0	0	1	7	25
Wadsworth	73	1	2	1	11	7	20	115
Wright	27	1	38	0	1	1	13	81
				2005				
Ethel	223	1	18	0	59	3	32	336
Gander	26	0	29	2	9	5	16	87
Grant	559	4	152	1	196	26	196	1134
MacArthur	37	0	7	0	0	3	7	54
Marl Flat	13	0	5	0	93	27	51	189
Ryerson	87	0	7	4	92	10	62	262
Seq. Creek	7	2	16	0	2	4	13	44
Wadsworth	56	2	30	0	70	5	96	259
Wright	15	17	5	0	5	7	4	53
Grand Total	2667	32	619	10	778	129	839	5074

Table 2.5. Number of bat echolocation passes (≥ 4 calls) positively identified to species per year in Lake County Forest Preserves, IL. Passes were collected and identified using Anabat and Analook software, respectively.

Axes	1	2
Eigenvalues	0.056	0.039
Species-environment correlations	0.367	0.35
Cumulative % Variance		
species data	5.9	10
species-environment relation	52.6	89.7
Intraset correlations		
Shrub	0.2549	-0.0835
C03	0.4344	0.0876
C36	0.4417	0.022
C69	0.293	0.7619
C912	-0.2762	0.1857
Canopy	-0.613	0.6494
Lg	0.134	-0.1373
Sm	-0.4306	-0.601
Med	-0.2733	-0.3558

Table 2.6. Summary statistics from canonical correspondence analysis (CCA). Vegetation variables include clutter at 0-3 m (C03), 3-6 m (C36), 6-9 m (C69), and 9-12 m (C912) heights, percent canopy cover (Canopy), and large (Lg, >33.1 cm DBH), small (Sm, 7.7-20.0 cm DBH), and medium (Med, 20.1-33.0 cm DBH) tree densities.

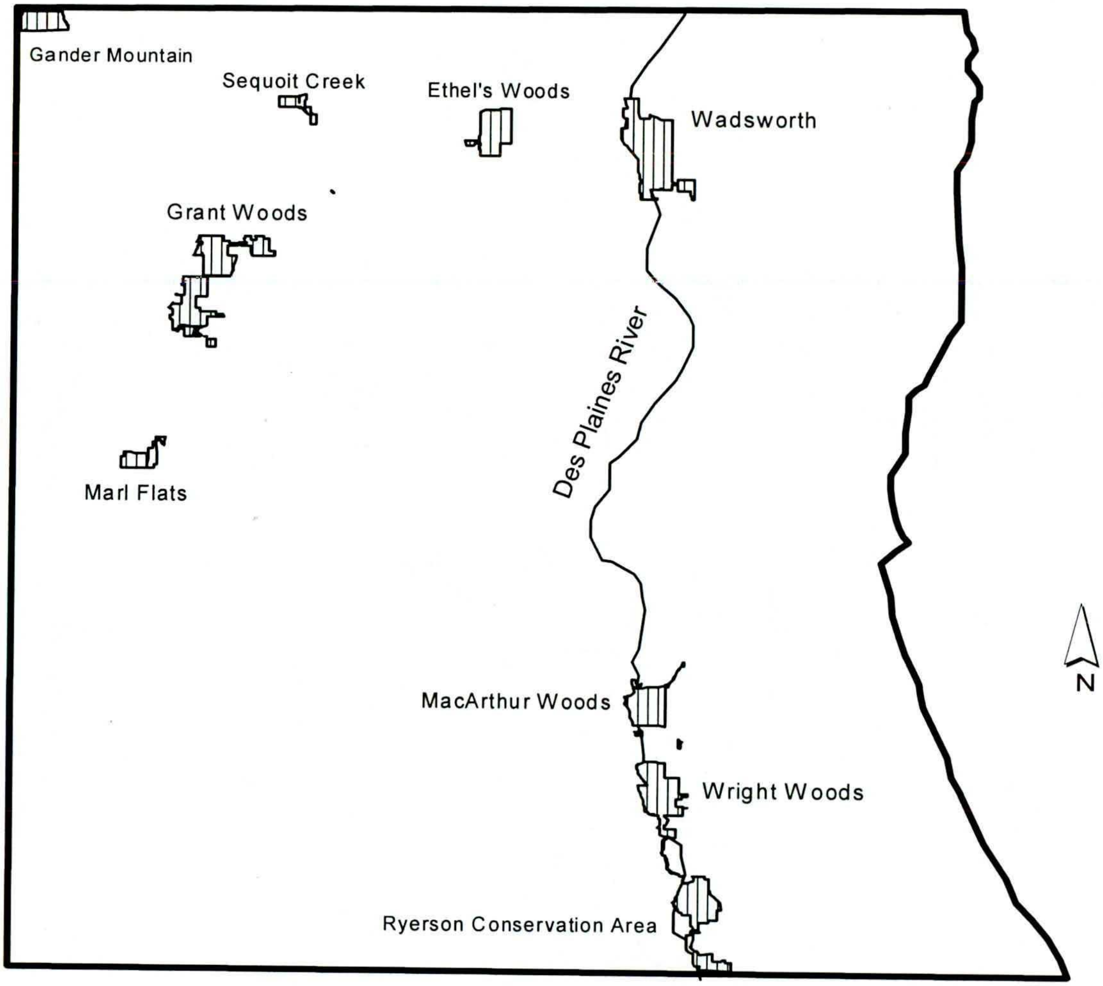

Figure 2.1. Map of forest preserves monitored for bat activity in Lake County, IL (2005).

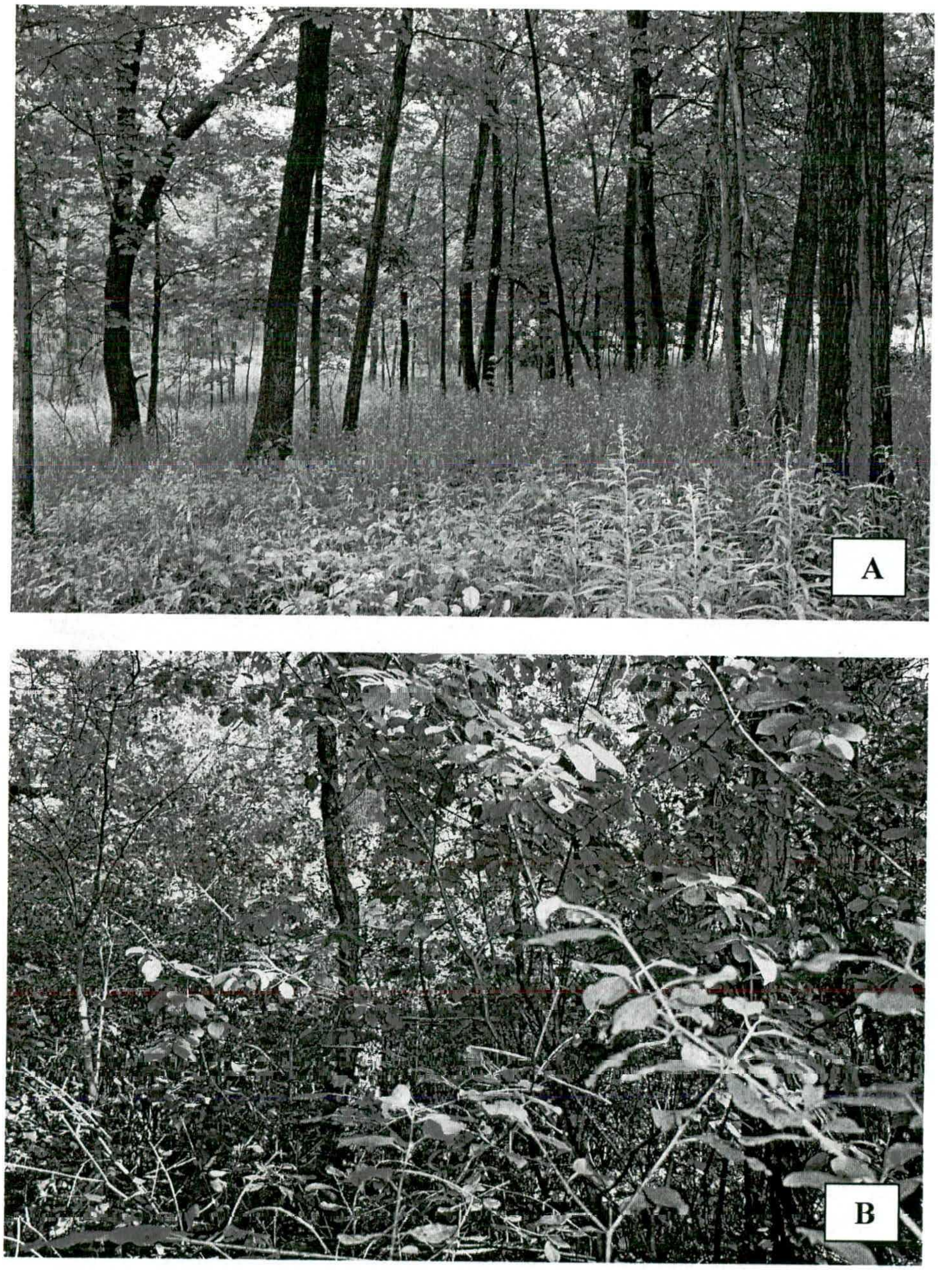

Figure 2.2. Photographs illustrating differences in forest structure in Lake County Forest Preserves, IL. Grant Woods Forest Preserve (A) has been frequently burned, and invasive shrub species have been removed. Marl Flats Forest Preserve (B) has had no restoration treatments.

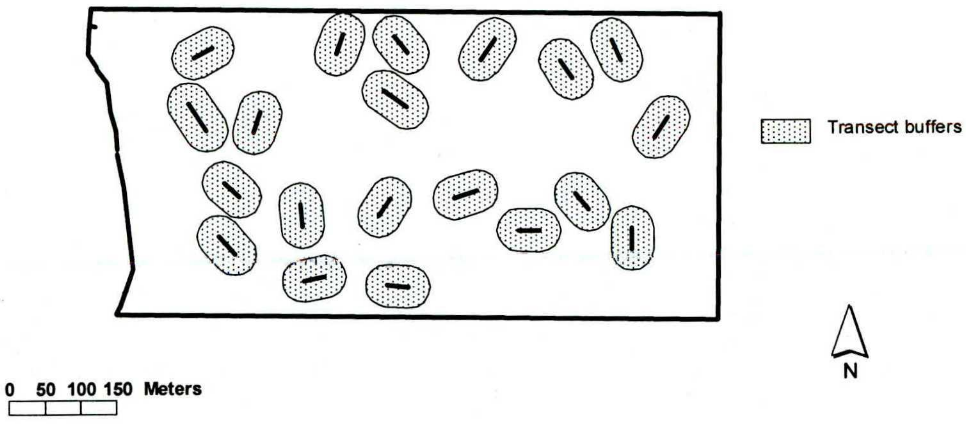

Figure 2.3. Example of study design in one monitored forest preserve. Transects are 30 m in length and have a 30 m buffer. Each transect has 4 echolocation detectors placed 10 m apart.

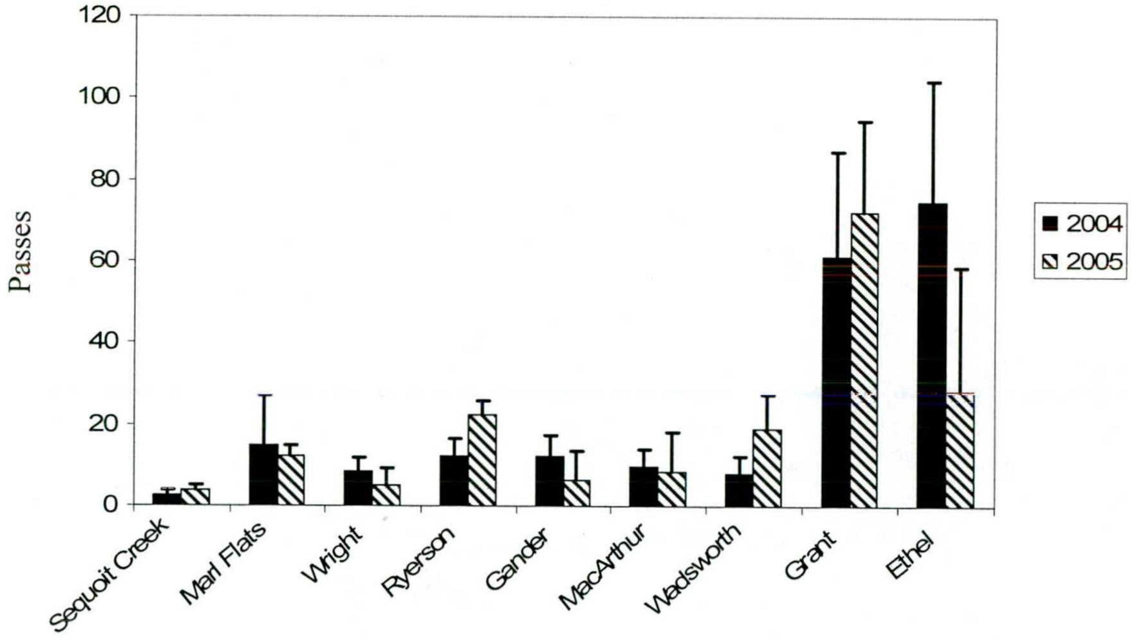

Figure 2.4. Mean (±SE) number of bat passes (mean number of passes/transect/site) per monitored forest preserve in Lake County, IL during 2004 and 2005.

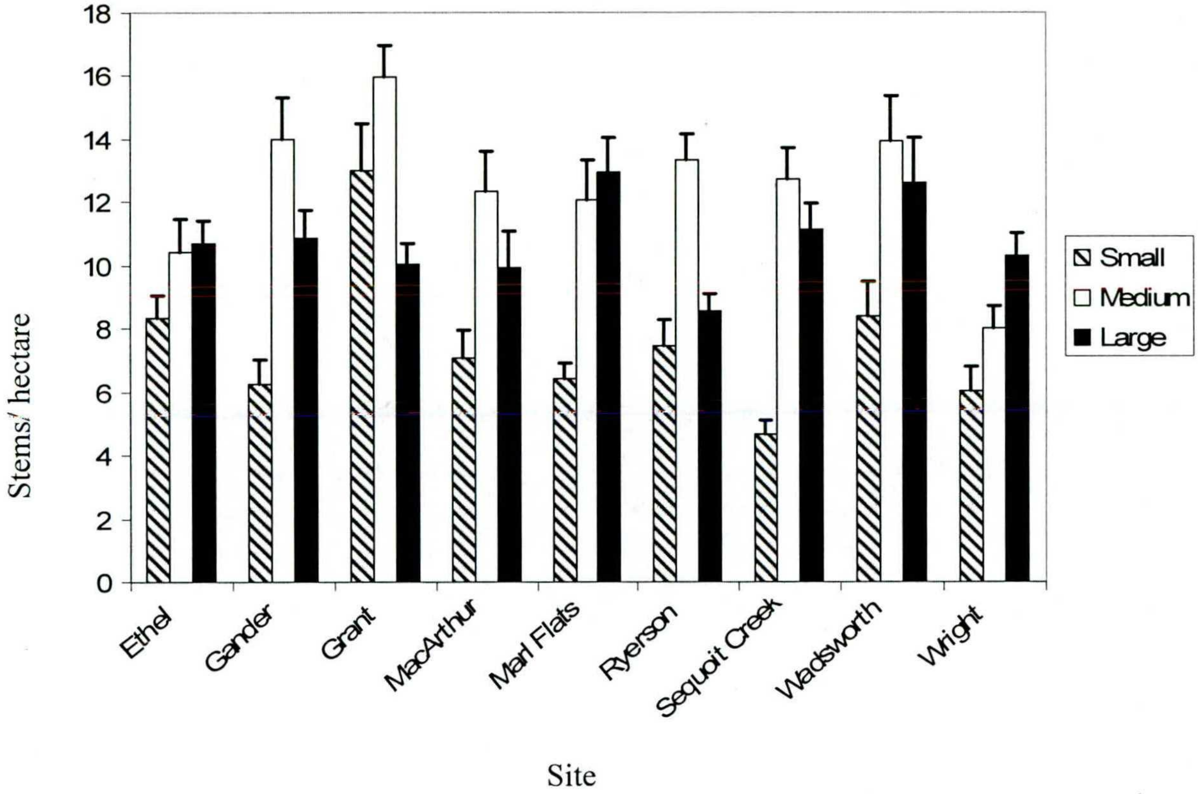

Figure 2.5. Mean tree densities (stems/ha) of various size classes measured in Lake County Forest Preserves, IL in 2005. Small trees have a diameter at breast height between 7.7cm-20cm. Medium and large tree sizes range from 20.1cm-33.0cm and >33.1cm, respectively.

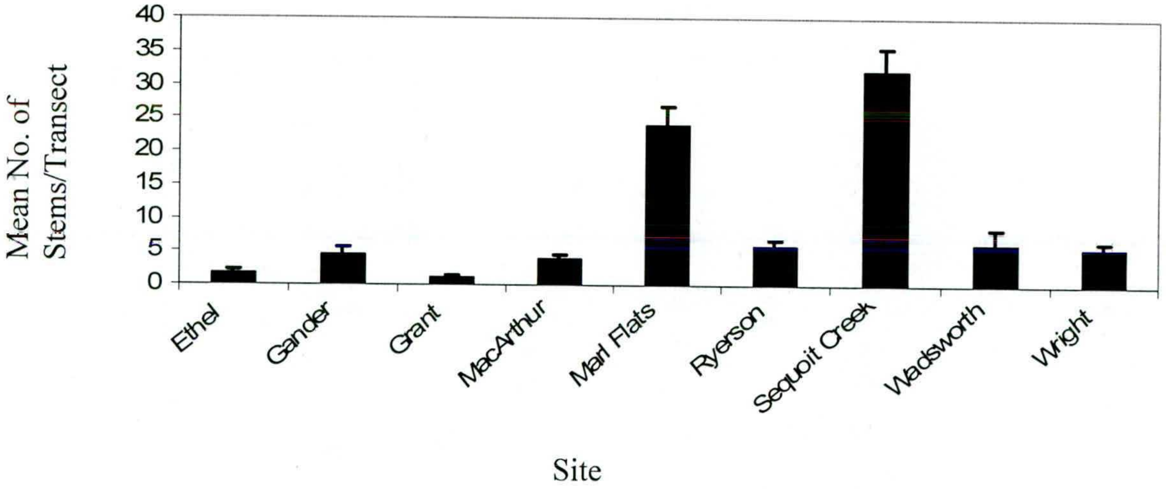

Figure 2.6. Density of shrubs in Lake County, IL forest preserves that have undergone various levels of restoration (2005).

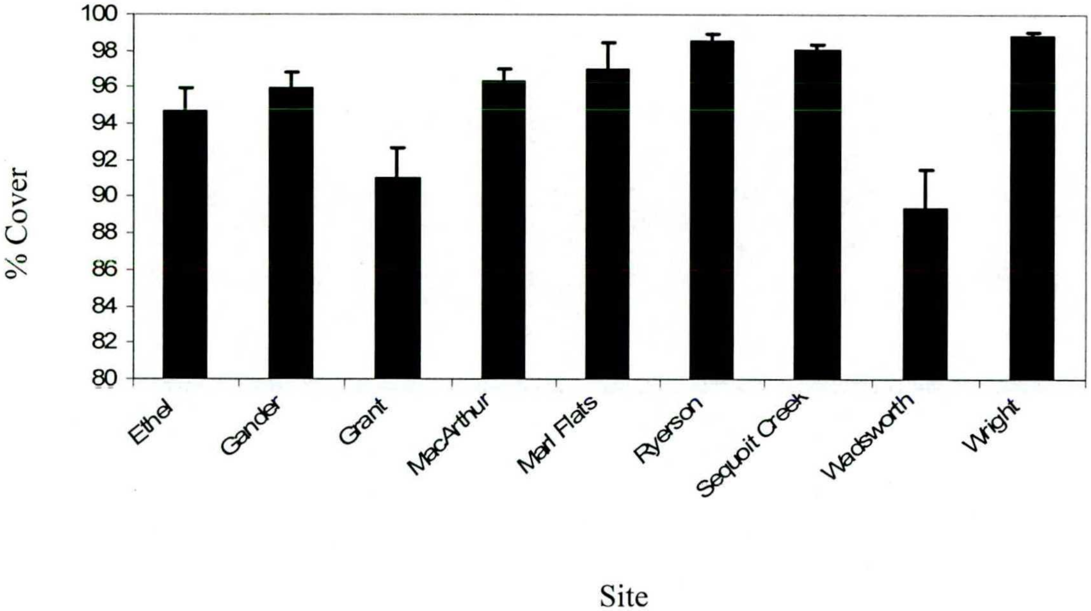

Figure 2.7. Estimated mean canopy cover per site for 9 forest preserves in Lake County, IL (2005).

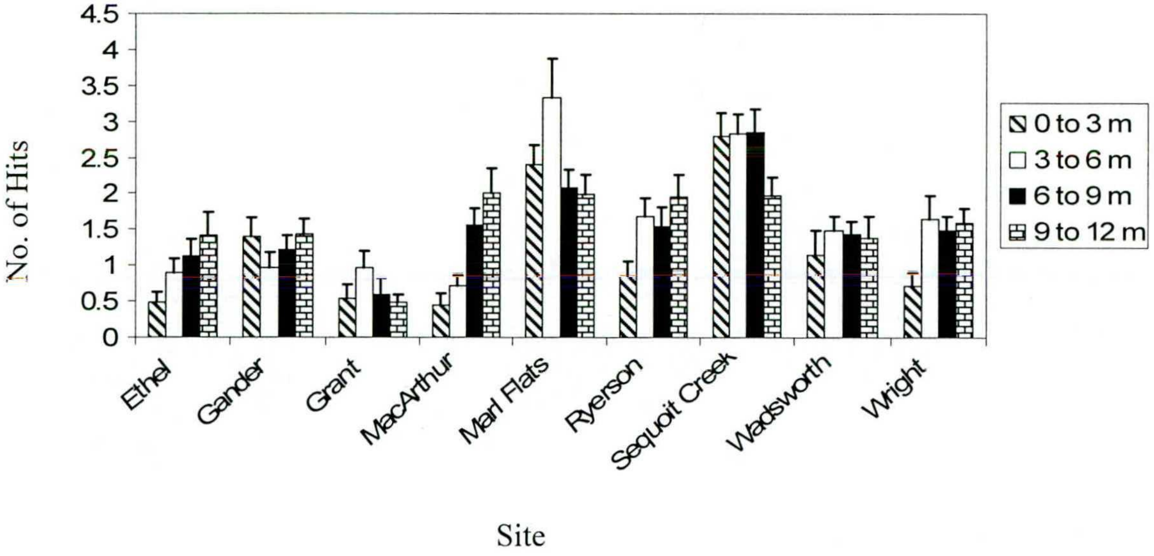

Figure 2.8. Vertical foliage profiles of Lake County, IL forest preserves at 0-3 m, 3-6 m, 6-9 m, and 9-12 m heights (2005).

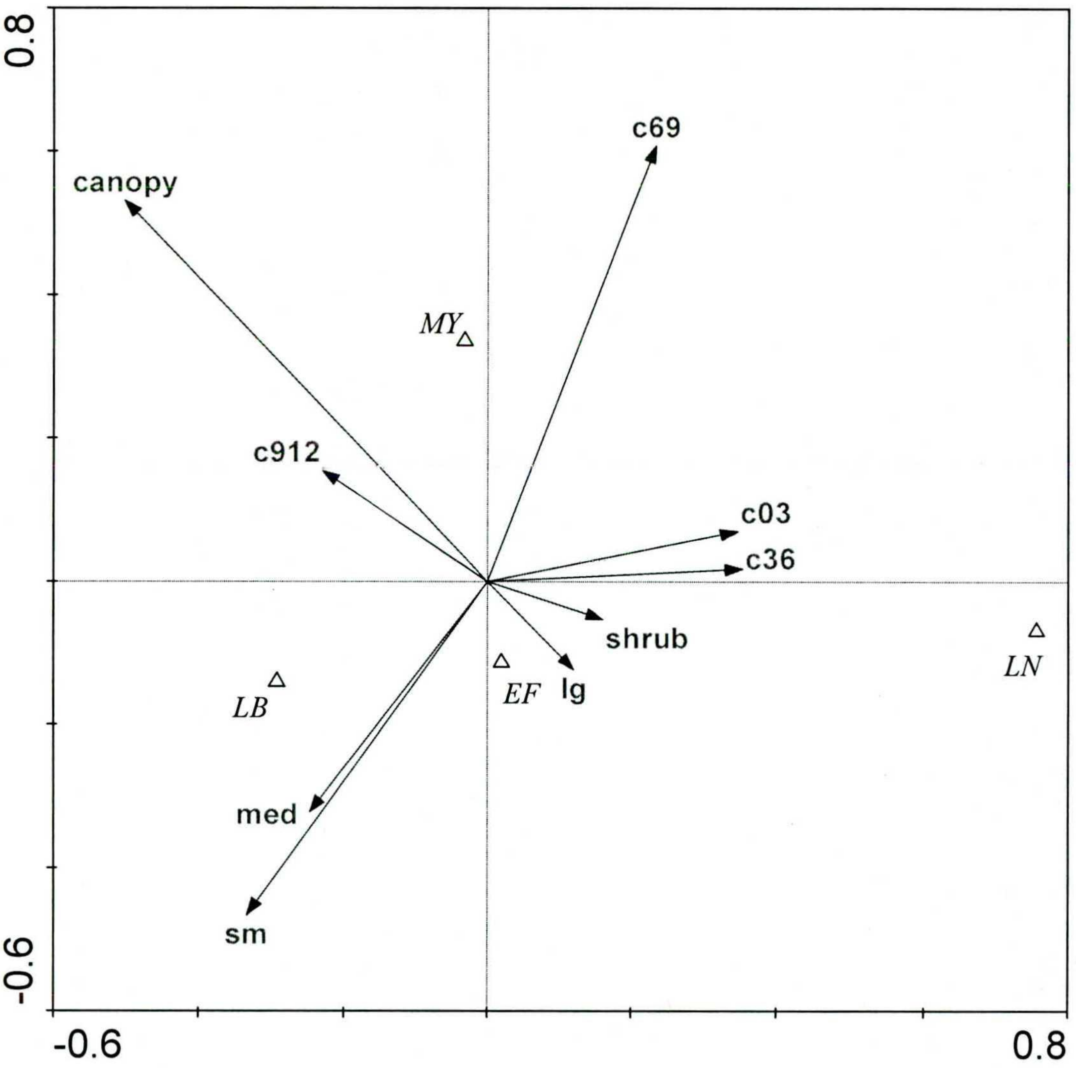

Figure 2.9. Biplot of bat species observed in 2004 and 2005 with associated habitat variables. Bat species are identified by Δ. Environmental variables are represented by arrows. Variables included are clutter at 0-3 m (c03), at 3-6 m (c36), at 6-9 m (c69), and 9-12 m heights (c912), small tree density (sm), medium tree density (med), large tree density (lg), and canopy cover (canopy). Refer to Table 2.6 for intraset correlations that provide more information about the biplot axes.

CHAPTER 3

ROOST SELECTION OF *MYOTIS SEPTENTRIONALIS* AND *LASIURUS BOREALIS* IN MANAGED URBAN FOREST FRAGMENTS

INTRODUCTION

Urbanization has transformed landscapes through reductions of natural habitats and shifts in land use from agriculture (Natural Resource Inventory 2003). Wildlife respond to changes in land use in a variety of ways, that are dependent on scale (e.g. Walsh and Harris 1996, Avila-Flores and Fenton 2005), species sensitivity to urban effects (Crooks et al. 2004), and species' ability to disperse (Barko et al. 2003). Typically, urbanization has been associated with declines in species abundance and richness (e.g. Campbell 1973, Dickman 1987, Kurta and Teramino 1992), however, fragments provide critical habitat for some species (Gilbert 1989, Adkins and Stott 1998). In an agriculture-dominated landscape, bat activity was higher in urban versus agriculture environments because of increased availability of woodland habitat (Gehrt and Chelsvig 2003).

Bats (Order Chiroptera) are highly vagile, thus they are capable of exploiting a mosaic of habitat patches in urban environments. Recent studies have indicated the importance of natural habitat fragments, particularly woodlands, for bats in urban areas

(Everette et al. 2001, Gehrt and Chelsvig 2003, Evelyn et al. 2004). Many species of bats utilize woodlands for roosting, foraging, and/or rearing young (e.g. Brigham and Barclay 1995, Patriquin and Barclay 2003). Subsequently, it has been suggested that the availability of tree day roosts may be a limiting factor for bats (e.g. O'Shea and Vaughn 1999). Roosts provide bats a centralized location for social interaction, a stable environment for raising young, and protection from predators and the elements (Kunz 1982). Bats have narrow requirements for roost selection, which are related to a combination of tree, plot, and landscape characteristics. Several studies have examined how forest characteristics and habitat management methods affect roost selection (e.g. Campbell et al. 1996, Elmore et al. 2004, Miles et al. 2006); however none have investigated roost selection in managed urban forest fragments. Evelyn et al. (2004) found that *Myotis yumanensis*, a species commonly associated with man-made structures, actually used large diameter trees as roosts in open space preserves and residential neighborhoods more often than in buildings. Although the study was conducted in an urban environment, it did not identify the effect of management practices on roost selection. In a conservation area, Boyles et al. (2006) found a positive relationship between prescribed burning and *Nycticeius humeralis* roost selection, possibly because fire led to increased tree mortality and reduced forest understory.

In chapter 2, the effects of different restoration treatments were discussed relative to general bat activity in an urban landscape, and the results indicated that restoration altered forest structure and ultimately influenced bat activity. The objective of this chapter was to determine the characteristics at the tree, plot, and stand scales that are associated with roost selection in managed woodland fragments in an urban landscape.

The northern long-eared bat (*Myotis septentrionalis*) and red bat (*Lasiurus borealis*) were selected as target species because of their roosting ecology. Both forest-dwelling species use trees for roosts but with different traits. Northern myotis roost in tree cavities or underneath exfoliating bark on dead limbs or snags (e.g. standing dead trees)(Sasse and Pekins 1995), and form maternity colonies in the summer (Menzel et al. 2002). In general, northern myotis were found in large diameter, tall trees that had high surrounding canopy cover (Foster and Kurta 1999). Conversely, red bats are solitary and roost among foliage in live trees (Hutchinson and Lacki 2000). Red bats have been found in a variety of roost types (e.g. *Pinus* sp., shrubs, leaf litter) (Mager and Nelson 2001), but are usually associated with larger hardwood trees (Menzel et al. 1998, Hutchinson and Lacki 2000, Elmore et al. 2004). Elmore et al. (2004) observed red bats utilizing roosts in areas with higher canopy cover and increased stem densities.

I hypothesized northern myotis roosts would be primarily located in areas with high snag densities. Based on the assumption restoration processes led to the creation of snags through prescribed burning and girdling of trees, northern myotis will roost more often in restored versus unrestored woodlands. Because of their larger wing size and need for protection from the elements, I predict red bats will select roosts in restored areas that have been thinned but still maintain a dense subcanopy.

STUDY AREA AND METHODS

Study area

This study was conducted within the larger Chicago metropolitan region (Lake County, IL). Lake County was highly populated with an average human population density of 556/km^2 or a total of 644,000 people (Natural Resources Inventory 2003). Land cover was a mosaic of urban, agriculture, and green patches. The county is a member of Chicago Wilderness, which is an organization dedicated to the preservation and restoration of natural habitats in the greater Chicago metropolitan region (See Chapter 2).

I selected 2 forest preserves managed by Lake County Forest Preserve District for this study: Wright Woods and Grant Woods. Both sites contained permanent water sources, fragmented woodlands, and a combination of restored and unrestored habitats. Restoration processes included invasive plant removal, tree thinning, deer population control, prescribed burning, and snag recruitment (See Chapter 2). Both preserves were natural habitat fragments among residential and commercial properties.

Capture and Radiotelemetry

During June 13-August 22, 2005, I captured bats using 6- and 9-m, 3-tier mist nets placed over small wetlands and trails. I identified all captured bats to species and recorded mass (g), forearm length (mm), sex, age, and reproductive condition (e.g. non-lactating, scrotal, pregnant)(Anthony 1988, Racey 1988). Red bat and Northern myotis that weighed ≥ 6 g were affixed with 0.35 g radiotransmitters (LB-2N, Holohil, Ontario, Canada). Mean radio-transmitter load was 4.05% of body mass, range 2.5-5.75% (Aldridge and Brigham 1988). Radiotransmitters were attached between fur-clipped scapulas of the bats using Skin Bond surgical cement (Pfizer, Largo, Florida). Adhesive was allowed to set for 20 min, after which, bats were released at point of capture. No unusual flight behavior was observed for radio-tagged juveniles, so I believed transmitters had a minimum effect on bat activity (Elmore et al. 2004).

I located radio-tagged bats the following day using handheld radio receivers (model R-1000, Communication Specialists, Inc., Orange, CA) with 3-element Yagi antennae (148.000-149.999 MHz, frequency range). Day roosts were identified and flagged, and Universal Transverse Mercator coordinates were recorded for each new roost tree. Bats were tracked daily until either radio-transmitter failure or recovery, approximately 1-11 days post-release. Roosts were occasionally located outside Lake County Forest Preserve District property. I attempted to gain permission to enter private property, however not all requests were granted. In these instances, I approximated the location of radio-tagged bats by encircling the property, and we could estimate roost location in most cases (e.g. private, unrestored woodland vs. lawn shade tree).

For comparative purposes, I located a random tree for each new roost tree. Random trees were selected using a random point generator in ArcView 3.2 and were >30 m apart (Environmental Systems Research Institute, Redlands, California). Points were constrained to woodland habitat in Wright Woods or Grant Woods Forest Preserves. I selected a random tree in the nearest sampled forest preserve for roost trees located on private property. Because of differences in red bat and northern myotis roost preferences, all non-roost trees, including snags, were potential random trees (Menzel et al. 1998, Menzel et al. 2002). I assumed random trees were not used by untagged bats. However, 1 random tree was discovered to be a *Lasionycteris noctivagans*' roost, so another random tree was selected to eliminate potential bias. Research protocols were approved by the Institutional Animal Care and Use Committee at Ohio State University (protocol #2004A0001).

Habitat Sampling

For each roost and random tree, I recorded tree species, height (m), diameter at breast height (DBH in cm), and tree status (live or dead). If the roost or random tree was a snag, I also documented decay class (scale from 1-9; 1=live tree, 9=stump) and percentage of remaining bark (Thomas et al. 1979). Aspect of the bat's position was noted using a compass, and the bat's height in the tree (entrance height) was estimated with a clinometer.

At the stand scale, I noted the type of area that a roost tree was located in, such as restored, unrestored, or residential. Restored areas were defined as having repeated prescribed burns, invasive species removal, or snag recruitment. Trees in manicured lawns were classified as residential.

Plot data consisted of vegetation variables that surrounded the roost tree (≤ 30 m away). Surrounding tree densities were calculated using the point-centered quarter method (Cottam and Curtis 1956), using the roost tree as a center point (Foster and Kurta 1999). The nearest trees from the following size classes were measured: small (7.7 cm-20 cm DBH), medium (20.1 cm-33 cm DBH), large (≥33.1 cm DBH), and snag (>10 cm DBH). Distance to and height of the nearest shrub (≤7.6 cm DBH, ≥1.5 m height) were recorded. If a habitat variable was >30 m from roost tree, then it was considered absent from the plot. Canopy cover was calculated in each of the 4 quadrants and 1 m from the roost tree using a spherical densiometer. The 4 readings were averaged to create 1 measurement of canopy cover per plot.

Analyses

I used an information-theoretic approach to select models with local habitat and roost tree characteristics that best predict bat species occupancy (i.e. presence/absence) (Burnham and Anderson 2002). Logistic regression was used to determine relationships between selected variables and occupancy rates. Because of small sample size, each roost tree was treated as independent, and gender and age classes were pooled for each species. All analyses were calculated in R statistical software, package Rcmdr (R Development Core Team 2005).

Northern Myotis

I developed 6 *a-priori* models comparing roost tree and plot data to occupancy rates. Models were created based on published literature and forest stratification (See Chapter 2, Foster and Kurta 1999, Cryan et al. 2001). Because of high correlation between roost height and DBH, only DBH was used in models (Pearson correlation, r=0.67, p-value < 0.001). The following variables were naturally log-transformed to improve skewness and to meet normality assumptions: shrub, snag, medium, and small tree densities. Canopy cover was cosine transformed. Akaike's Information Criteria (AIC) were used to select the most parsimonious model for explaining bat occupancy.

Red bat

I created 9 *a-priori* models to determine roost tree and plot characteristics that may influence red bat roost selection. Roost height and DBH were highly correlated, so I removed DBH from the models (Pearson correlation: r=0.657, p-value <0.001). The following variables were transformed using a natural log to improve data for statistical assumptions: shrub, small, medium, and large tree densities. Canopy cover was arcsine-transformed. I used Akaike's Information Criterion (AIC) and model averaging to appraise relative variable importance and to obtain multi-model inference (Burnham and Anderson 2002).

RESULTS

I mist-netted 14 nights (66.75 hours) and captured 27 bats of 3 species (Table 1). Capture rates were consistent with other urban bat studies in Illinois (Mager and Nelson 2001, Gehrt and Chelsvig 2003). Overall, capture rate was low and the majority of captures were between July 29 and August 22, 2005.

Northern Myotis

Five northern myotis (2 adult females and 3 adult males) were fitted with radiotransmitters and radio-tracked to 12 roosts (Table 1). Sixty-seven percent of identified roosts were *Ulmus americana* snags, and 33% in cavities of live *Acer saccharum*. On average, bats switched roosts every 1.7 days (range 1-6 days). Northern myotis were located in unrestored (75%) and restored (17%) woodland habitat and in residential areas (8%).

Model selection criteria provided evidence that roost selection was most parsimoniously explained by roost tree characteristics: roost height and decay class (Table 2). The best model (ω_i=0.84) was 11.5 times better than the next top model. Smaller roosts (height and diameter) were more likely selected than was available (odds ratio 0.91; 95% CI 0.75, 1.09). Based on odds ratios, snags with exfoliating bark were 2.4 times more likely to be occupied than live trees (95% CI 1.06, 5.56).

Red Bat

Nine red bats (4 adult females, 2 juvenile females, 1 adult male, and 2 juvenile males) were fitted with radio-transmitters and tracked to 25 roosts (Table 1). Despite an exhaustive search, 3 red bat (2 adult females, 1 juvenile female) were never tracked to a roost and were believed to have left the study area. Identified roosts were in live hardwood species, such as oak (*Quercus* spp., 39%), hickory (Carya spp., 17%), ash (*Fraxinus* spp., 13%), maple (*Acer* spp., 9%), basswood (*Tilia americana*, 9%), and other (11%). Red bats switched roosts approximately every 1.3 days (range 1-2 days). Roosts were detected more often in unrestored (52%) than restored areas (12%). The remaining roosts (36%) were shade trees in residential lawns.

The top 5 models (a combination of tree and plot data) had a delta AIC <2 and were subsequently averaged for multi-model inference (Table 3). However, none of the 6 variables had model-averaged confidence intervals that did not include zero, suggesting that the variables provided little information regarding red bat roost selection (Table 4).

DISCUSSION

Northern myotis roosts were primarily located in Wright Woods/Half-Day Forest Preserves near the Des Plaines River. All except 1 northern myotis roosted within a core area in these forest preserves (Foster and Kurta 1999). Although the area was unrestored, it was subject to periodic flooding, so the presence of invasive shrub species, such as European buckthorn (*Rhamnus cathartica*), were not as abundant as in more upland sites (Ahrens 1999, D. Scott, unpublished data). Foster and Kurta (1999) also observed northern long-eared bats roosting in hydric environments. Riparian conservation may be important for bat conservation in fragmented habitats (Holloway and Barclay 2000, Seidman and Zabel 2001). However, more research is needed to determine the effects of invasive shrub species and restoration processes, such as prescribed burning, on northern myotis roost selection with an emphasis in urban areas.

Results for northern myotis roosts characteristics were consistent with previously published studies that observed northern myotis in cavities and under exfoliating bark in snags (Sasse and Pekins 1995, Foster and Kurta 1999, Menzel et al. 2002). However, this study demonstrated the importance of snags in urban environments (Blewett and Marzluff 2005). Although northern myotis used cavities and crevices in live trees in this study, *M. septentrionalis* were more than twice as likely to roost in snags. All identified snag roosts were *U. americana*, which is inconsistent with previous findings (Sasse and Pekins 1995, Foster and Kurta 1999, Menzel et al. 2002). Snag longevity and decay class of *U. americana* roosts may be important factors in northern myotis roost selection, particularly in urban woodland habitats (Menzel et al. 2002, Thomas et al. 1979).

Northern long-eared bats in this study differed from other snag-roosting bat species by using roosts with smaller heights and diameters, and with higher percent canopy closure (Vonhof 1996, Cryan et al. 2001). Menzel et al. (2002) also observed northern long-eared bats roosting in slightly smaller trees than were available in an industrial forest. Although the roosts selected in this study were slightly smaller than what was available, the mean height (16.9 m) of roost trees were within the mean height range (14.0-24.0 m) of other northern myotis studies (Sasse and Pekins 1995, Foster and Kurta 1999, Menzel et al. 2002). Roost tree mean diameter (33.5) was also within range (29 cm-65 cm), but it was smaller than roost trees in agricultural woodland fragments in Michigan (Foster and Kurta 1999) and in 70-89 mature forests in New England (Sasse and Pekins 1995). Some snags in second-growth forest fragments in urban environments were suitable for northern myotis roosting, so forest fragments are meeting some roosting needs. However, creation of new snags is necessary to replace fallen or advanced-decayed snags, but Blewett and Marzluff (2005) suggest it might not be a concern as long as there are sufficient numbers of live trees in the area. The presence of live trees was highly correlated with snags in forest fragments (Blewett and Marzluff 2005).

Red bats did not appear to select roosts determined by individual roost tree or plot characteristics but by stand characteristics (Elmore et al. 2004). Red bats were significantly more likely to be found in unrestored and residential habitats than in areas where restoration occurred. Within site comparisons between restored and unrestored woodlands were not assessed, however comparisons between restored and unrestored forest preserves showed decreased shrub density, tree density, and overall clutter (e.g. foliage) in restored areas (See Chapter 2). Several studies have suggested bats choose roost sites that moderate thermal and wind conditions (Walsberg 1986, Vonhof 1996). Sites with higher foliage densities may reduce wind velocity and regulate temperatures for red bats (Mager and Nelson 2001). All except 1 residential roost were in close proximity to unrestored woodlands.

Caution should be exercised when interpreting these results. Data were pooled for all gender and age classes and there was a non-independence factor associated with roost variables. Combining gender and age may mask differences in roost selection among the groups (Miller et al. 2003, Elmore et al. 2004). Sample size for northern myotis were too small to differentiate roost selection patterns between age and gender. However, red bat juveniles roosted in unrestored or residential areas (75%) and restored woodlands (25%). All adult red bat roosts were in unrestored or residential habitat. Additional sampling would help elucidate age/gender relationships with roost selection.

Management Implications

Preservation of upland and bottomland forest fragments in urban environments is important for bat conservation (Campbell et al. 1996). Second-growth forests provide essential roost sites for both foliage- and cavity-roosting bat species (Foster and Kurta 1999, Mager and Nelson 2001). Specifically, riparians areas are often associated with suitable roost trees (Campbell et al. 1996).

Invasive shrub removal and selective thinning of trees benefit native herbaceous plants by reducing competition for space, nutrients, and light (Schmit and Wulff 1993, Gould and Gorchov 2000). These restoration activities were associated with increased bat activity (Refer to Chapter 2). Increased canopy cover has been associated with declines in some amphibian species (Skelly et al. 1999), and greater shrub density may restrict animal movement, particularly carnivores (Guevara et al. 2005). However, higher shrub and foliage densities are important for many wildlife species for protection (Bowne et al. 1999), roosting (Hutchinson and Lacki 2000), nesting, and foraging (Steele 1993). Land managers should use caution when removing invasive shrubs if it is the dominant species in the forest understory. Excessive reduction in shrub cover may be detrimental to some wildlife species. Forest structure heterogeneity is recommended for bat conservation.

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Species	n	FA	FJ	MA	MJ	No. roosts	Entrance ht.	Aspect	Tree ht.	DBH
M. septentrionalis	5	2	0	3	1	2.4± 0.51	8.58± 1.2	188.9± 28.6	16.9± 1.63	33.5± 5.15
L. borealis	9	4	2	1	2	5.1± 0.83	12.2± 0.68	171.2± 19.5	21.23 ±1.39	41.44± 4.0
E. fuscus	*	5	0	4	3	*	*	*	*	*

Table 3.1. Summary statistics for captured bats in Lake County Forest Preserves, IL (2005). Variables recorded are: age and gender (FA=female adult, FJ=female juvenile, MA=male adult, MJ=male juvenile), average number of roosts per individual (No. roosts), average height of bat in roost tree (Entrance ht.), aspect, average height of roost tree (Tree ht.), and average diameter at breast height for identified roost trees (DBH).

Model	DF	AICc	ΔAICc	ωi
ms~ ndbh + decay	3	30.758	0.00	0.712
ms~ nsm + med	3	34.068	3.31	0.136
ms~ lg + nmed + ccan	4	36.025	5.27	0.051
ms~ nsm + nsh	3	36.124	5.37	0.048
null	2	37.092	6.33	0.030
ms~ nsnag	2	37.754	6.99	0.021

Table 3.2. Akaike's Information Criterion, adjusted for small sample size (AIC_c), results for logistic regression models examining relationships between *M. septentrionalis* roost occupancy (ms) and vegetative characteristics (ndbh= roost tree's diameter at breast height, decay=decay class of roost tree, nsm= small tree density, med=medium tree density, ccan=percent canopy cover, lg=large tree density, nsh=shrub density, and nsnag=snag density) with degrees of freedom in model (DF) and Akaike's weight (ω_i). The letter n in front of a variable name indicates the variable was transformed using natural log. Models are ranked by ascending delta AIC_c values (difference in AIC_c value between model and model with lowest AIC_c value). Model with highest Akaike's weight is shown in bold text.

Model	DF	AIC_c	ΔAIC_c	ω_i
lb~ nlg +nmed+ acan	4	69.672	0.00	0.22
lb~ nsm + nshr + acan	4	69.774	0.10	0.21
lb~ roost ht + nlg + acan	4	70.278	0.61	0.16
lb~ nmed+ nsm	3	70.313	0.64	0.16
lb~roost ht	2	71.626	1.95	0.08
lb~ nsm +nshr	3	71.996	2.32	0.07
lb~ nlg + nmed + nsm + nshr + acan	6	72.412	2.74	0.06
full	7	74.119	4.45	0.02
null	2	74.856	5.18	0.01

Table 3.3. Akaike's Information Criterion adjusted for small sample size (AIC_c) values for logistic regression that relate *L. borealis* presence/absence in roost selection (lb) and vegetative variables (nlg= large tree density, nmed=medium tree density, nsm=small tree density, nshr=shrub density, acan=percent canopy cover, and roost ht= height of roost tree). Models are ranked via ascending ΔAIC_c values.

Variable	Beta hat	95% CI	RVI
acan	-1.33	-3.98,1.32	0.672
nsm	-0.16	-0.60, 0.27	0.518
nlg	-0.18	-0.74, 0.38	0.463
nmed	-17.53	-72.45, 37.38	0.460
nshr	-0.02	-0.17, 0.12	0.358
roost ht	0.01	-0.02, 0.04	0.269
(Intercept)	-0.71	-5.28, 3.86	

Table 3.4. Vegetative coefficients (Beta hat), 95% confident intervals (CI), and relative variable of importance (RVI) values based on averaging logistic regression models with delta AIC values < 4. Variables (acan= arcsine transformed canopy cover, nsm=small tree density, nlg=large tree density, nmed=medium tree density, nshr=shrub density, and height of roost tree (roost ht) are ranked according to descending relative importance. The letter n in front of variable name indicates that it was natural log transformed.

APPENDIX A

Plot No.	Site	Rest. Level	Roost Species	Live/Dead	Tree Height (m)	DBH (cm)	% bark	Aspect	Roost entrance (m)	Canopy Closure (%)
537R2	Half Day	unrestored	*Ulmus americana*	dead	11.75	22.8	80	260	7.5	77.64
537R1	Half Day	unrestored	*Ulmus americana*	dead	13.6	16	65	160	1.3	88.56
817R3	Half Day	unrestored	*Ulmus americana*	dead	14.25	23.9	60	225	5	66.46
817R2	Half Day	unrestored	*Ulmus americana*	dead	14.25	22.5	15	22	9.66	91.68
817R4	Wright	unrestored	*Ulmus americana*	dead	22.2	42	85	209	8.1	96.36
817R1	Half Day	unrestored	*Ulmus americana*	dead	13.5	17.6	20	296	8.25	69.06
667R1	Residential	other	*Acer saccharum*	live	21.6	65.4	100	316	12	66.2
377R1	Wright	restored	*Acer saccharum*	live	7.2	12.6	100	135	4.6	99.22
377R4	Wright	restored	*Acer saccharum*	live	27.6	47.3	100	326	9.3	85.7
377R3	Wright	unrestored	*Ulmus americana*	dead	18.3	35.5	40	139	11.7	98.96
647R1	Half Day	unrestored	*Ulmus americana*	dead	16.75	33	85	70	8.75	77.64
647R2	Wright	unrestored	*Acer saccharum*	live	21.9	63.4	100	109	16.8	97.92

Appendix A. *Myotis septentrionalis'* roost selection data, which includes the part of the transmitter frequency and roost number (Plot no.), the forest preserve the roost was found in (Site), the level of restoration that the area has undergone (Rest. level), the species of the roost tree (Roost species), if the roost was alive or dead (live/dead), the height of the roost tree (height), the diameter at breast height of the roost tree (DBH), the amount of bark remaining on the roost tree (% bark), the aspect of the bat's location on the roost tree (Aspect), the height of the bat on the roost tree (Roost entrance), and percent canopy cover (Canopy Cover).

APPENDIX B

Plot No.	Site	Rest. Level	Tree Species	Live/dead	Height (m)	DBH (cm)	% bark	Aspect	Roost entrance (m)	Canopy closure (%)
298R1	Residential	unrestored								
298R2	Residential	unrestored								
298R3	Residential	unrestored	*Quercus rubra*	live	23.4	69.9	100	309	9.3	97.14
298R4	Residential	unrestored	*Quercus alba*	live	20.75	52.2	100	49	14.25	98.44
907R1	Wright	unrestored	*Acer saccharum*	live	22.2	14	100	184	12.6	99.74
907R2	Wright	unrestored	*Acer saccharum*	live	34.5	59.6	100	196	15.9	99.22
907R3	Wright	unrestored	*Fraxinus pennslyvanica*	live	27.6	45.3	100	174	16.2	99.22
907R4	Wright	unrestored	*Carya ovata*	live	16.2	32.6	100	223	12.3	97.92
907R5	Wright	unrestored	*Fraxinus nigra*	live	29.7	26.9	100	200	15	98.18
028R2	Grant	restored	*Quercus macrocarpa*	live	12.75	20.6	100	30	9.75	98.18
028R1	Grant	restored	*Carya ovata*	live	21.6	37.3	100	69	15.39	97.4
707R3	Elm Road	unrestored	*Quercus alba*	live	23.7	54.9	100	69	11.7	93.5
707R1	Wright	unrestored	*Fraxinus sp.*	live	29.1	62.5	100	56	16.3	99.74
215R7	Wright	unrestored	*Tilia americana*	live	13.11	16	100	233	9.66	99.22
215R6	priv. prop.	other	*unknown*	live	8.55	11	100	111	5.25	
215R5	priv. prop.	other	*Prunus serotina*	live	22.1	68.6	100	161	9.83	66.2
215R4	priv. prop.	other	*Acer negundo*	live	15.6	52.4	100	322	9.3	85.44
215R2	Residential	other	*Carya cordiformis*	live	7.8	12.1	100	90	5.1	85.7
215R1	Wright	unrestored	*Tilia americana*	live	18.48	19.5	100	186	11.2	99.74
788R6	Grant	partially restored	*Quercus macrocarpa*	live	27	64.5	100	31	15.3	99.22
788R3	Residential	other	*Quercus alba*	live	18.75	47.9	100	240	11.25	66.46
788R5	Residential	other	*Quercus macrocarpa*	live	25.5	59.2	100	320	15.6	94.8
788R7	Residential	other	*Quercus alba*	live	22.5	44.2	100	201	11.4	99.22
788R1	Residential	other	*Quercus alba*	live	22.5	51.1	99	292	14.[illegible]	81.8
788R4	Residential	other	*Carya ovata*	live	24.9	30.8	100	192	12.6	69.06

Appendix B. *Lasiurus borealis*' roost selection data, which includes the part of the transmitter frequency and roost number (Plot no.), the forest preserve the roost was found in (Site), the level of restoration that the area has undergone (Rest. level), the species of the roost tree (Roost species), if the roost was alive or dead (live/dead), the height of the roost tree (height), the diameter at breast height of the roost tree (DBH), the amount of bark remaining on the roost tree (% bark), the aspect of the bat's location on the roost tree (Aspect), the height of the bat on the roost tree (Roost entrance), and percent canopy cover (Canopy Cover).

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Thesis