Assessing the Impacts of White-nose Syndrome Induced Mortality on the Monitoring of a Bat Community at Fort Drum Military Installation

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Abstract

Since white-nose syndrome (WNS) arrived in the northeastern U.S. in 2006, several affected bat species have exhibited marked population declines (> 90%). For areas such as Fort Drum in northern New York that are subject to regulatory mandates because of the presence of the endangered Indiana bat (*Myotis sodalis*), acoustic monitoring is now likely more effective than traditional capture methodologies. In the summers of 2011 and 2012, I implemented intensive acoustic sampling using Anabat detectors at Fort Drum to develop a summer acoustic monitoring protocol that is both cost efficient and effective at detecting species of high conservation or management interest, such as the Indiana bat and the little brown bat (*Myotis lucifugus*). Habitat analysis of radio telemetry data and occupancy models of acoustic data were congruent in confirming nocturnal spatial use of forested riparian zones by little brown bats. Additionally, occupancy models of passive versus active sampling revealed that passive acoustic sampling is preferable to active sampling for detecting declining species in the post-WNS context. Finally, assessment of detection probabilities at various arrays of acoustic detector layouts in an expected area of use revealed that a grid of detectors covering a wide spatial extent was more effective at detecting Indiana and little brown bats than permanent stations, transects, or double transects. My findings suggest that acoustic monitoring can be affectively implemented for monitoring Indiana and little brown bats even in areas of severe decline. Future efforts should be aimed at determining effective sampling designs for additional declining species.

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Chapter 1: A review of the literature

White-nose syndrome (WNS) is a recently described fungal disease affecting cave-dwelling bats. It was first documented in the United States at Howe's Caverns near Albany, New York in 2006 (Blehert et al. 2009) and has now spread to at least 22 states and 5 Canadian provinces (USFWS 2013). Histologic examination of infected individuals indicates that fungal hyphae invade hair follicles and sebaceous glands without typically causing any inflammation or immune response (Blehert et al. 2009). *Geomyces destructans* (*Gd*), the fungus associated with WNS, is a slow growing, psychrophilic fungus with an optimal growth range of 12.5-15.8°C (Verant et al. 2012), making it prolific in the stable temperatures of cave hibernacula (Gargas et al. 2009). Initially, researchers lacked evidence implicating *Gd* as the true cause of WNS rather than an opportunistic pathogen colonizing individuals (Gargas et al. 2009) during lowered immunity caused by hibernation (Carey et al. 2003). Lack of unusual bat mortality associated with the presence of *Gd* spores on the skin of European bats continued to cast doubt on *Gd* being the cause of WNS (Wibbelt et al. 2010). However, Lorch et al. (2011) finally provided direct evidence that *Gd* is the causative agent of WNS by demonstrating infection and subsequent lesions in healthy little brown bats exposed to pure *Gd* spores. Additionally, Warnecke et al. (2012) provided evidence linking *Gd* to frequent torpor-arousals and associated mortality and implicated *Gd* as a novel pathogen to North American bats.

To date, seven species of bats in eastern North America are known to be clinically affected by WNS. These include: the federally endangered Indiana bat (*Myotis sodalis*), the federally endangered gray bat (*M. grisescens*) the petitioned eastern small-footed bat (*M. leibeii*), the petitioned northern bat (*M. septentrionalis*), the little brown bat (*M. lucifugus*), the big brown bat (*Eptesicus fuscus*), and the tri-colored bat (*Perimyotis subflavus;* USGS 2012). Of these species, the little brown bat, the northern bat, and the tri-colored bat have shown the most drastic declines, i.e., > 90% in overwintering numbers at some hibernacula over a 3-4 year period (Turner et al. 2011). Other species, such as the cave bat *(M. velifer*), and the southeastern bat (*M. austroriparius*) have been documented with *Gd* spores, but have not yet been documented to have clinical WNS (BCI 2012). Finally, two federally endangered species—the Virginia big-eared bat (*Corynorhinus townsendii virginianus*) and the Ozark big-eared bat (*C. t. ingens*)—occur in WNS affected areas but have not yet been confirmed with *Gd* spores or clinical WNS (USFWS 2013).

An estimated 5.7 to 6.7 million bats have died as a result of WNS infection (USFWS 2013). Across the entire United States, 25 of the 45 bat species rely on caves or mines for hibernation and face potential infection as WNS continues to spread (USGS 2012). The little brown bat, previously one of the most common mammals in North America, has experienced a dramatic decline such that it could be at significant risk for regional extirpation in the northeastern United States and southeastern Canada within 20 years (Frick et al. 2010). The United States Fish and Wildlife Service (USFWS) currently is conducting a status review of the little brown bat to determine if federal listing is warranted (Kunz and Reichard 2010). Population declines of such magnitude could lead to the collapse of infected bat species with impacts at the ecosystem scale (Boyles and Willis 2009). As the decline ramifications of once common bat species are as yet unknown, biologists and land managers are also faced with addressing WNS-related population declines of already threatened and endangered bats like the Indiana bat.

The Indiana bat has been listed as federally-endangered since 1967 (USFWS 1967). However, prior to the introduction of WNS, this species was increasing in many areas of its range (USFWS 2011). This was due in part to effective conservation measures introduced through Endangered Species Act (ESA, as amended) agency consultation for the protection of the species such as summer roost and habitat protection, tree cutting restrictions during times when bats may be present on the landscape, and placing cave gates on hibernacula. In the absence of WNS, these programs were seemingly effective, however, with the onset of WNS-associated declines, determining the continued effectiveness of these management actions has become exceedingly difficult. Nonetheless, federal and other public land managers that have Indiana bats present are still tasked to monitor populations to determine if their activities are affecting bats. Meeting this requirement is now challenging at best, particularly in the Northeast where Indiana bat populations have been devastated by WNS impacts for more than five years.

To assess impacts from activities and delineate the use and distribution of the Indiana bat on the installation in association with a nearby hibernaculum (Fenton 1966, Ford et al. 2011), Fort Drum Military Installation (Fort Drum) in northern New York began to monitor Indiana bats across the installation through a combination of acoustic monitoring beginning in 2003. An Indiana bat maternity colony was documented on the installation in 2006. Subsequently, comprehensive mist-netting surveys beginning in 2007 to determine temporal and spatial distribution across the landscape.

Although WNS was first documented in New York in 2006 (Blehert et al. 2009), initial evidence of the impacts were not observed at Fort Drum until 2008 (Dobony et al. 2011). Analysis of acoustic activity patterns on Fort Drum from 2003-2010 showed significant declines in overall summer foraging activity in little brown bats, northern bats, and Indiana bats (Ford et al. 2011). Little brown bats also showed a decline in foraging activity from early to late season in post-WNS years, indicating a probable decrease in reproductive success of surviving individuals and subsequent lack of juvenile recruitment. In addition to acoustic monitoring of foraging activity, a capture-mark-recapture study was initiated in 2009 to monitor the status of a little brown bat maternity colony using an artificial roost structure in the Cantonment (developed) Area of the installation (Dobony et al. 2011). Although there was an absolute decline in numbers approaching 90%, surviving female little brown bats were able to heal from wing damage caused by WNS and were able to reproduce. However, it remains uncertain if females are able to successfully rear pups to volancy or if those neonates survive to be added as recruits going into hibernation. Mist-netting efforts have also revealed drastic declines in capture success of additional populations of little brown, northern, and Indiana bats since the impacts of WNS arrived at Fort Drum (C.A. Dobony, unpublished data).

After WNS impacts began to compound, Fort Drum managers determined that continuing to use standard methods (e.g., mist-nets) to monitor Indiana bats as in previous years would no longer be suitable and desired to develop a more effective and efficient summer monitoring protocol. Primarily, Fort Drum managers were interested in establishing an innovative and cost effective acoustic monitoring program. Previously, traditional monitoring protocols relied on various methods of capture (netting) at roosts, water sources, or along flyways, but not all species or individuals are equally susceptible to these forms of capture on a given night (O'Farrell and Gannon 1999). Previous research suggested that acoustics and mist netting detected similar levels of activity (Kunz and Brock 1975), but the advancement of technology has improved acoustic detectors and call identification software programs such that these techniques may now detect overall greater species richness in less time or spatial extent than capture methodologies.

Acoustic monitoring is a non-invasive sampling technique that has become a routine method for investigating bat ecology, species assemblages and relative abundance (Johnson et al. 2002, Milne et al. 2004, Ford et al. 2011, Johnson et al. 2011). These methods have been shown to detect significantly more species of bats when compared to netting (Murray et al. 1999, O'Farrell and Gannon 1999), as mist nets and harp traps are only able to sample an extremely small portion of the area used by free-flying bats. High flying or alert bats often avoid nets (Murray et al. 1999, O'Farrell and Gannon 1999, MacCarthy et al. 2006), resulting in biased samples and a difficulty in assessing entire species assemblages. Other research has suggested that using capture methodologies in combination with acoustic detectors enhances the likelihood of detecting target bat species and produces greater richness estimates than with either method alone, as some species are differently vulnerable between methods (Patriquin et al. 2003, Flaquer et al. 2007, Robbins et al. 2008). For example, some species such as the northern bat are particularly difficult to record acoustically due to their low intensity call amplitudes (Broders et al. 2004).

In areas severely impacted by WNS, the need for very high levels of sampling effort to detect declining species makes multiple sampling methodologies logistically and fiscally prohibitive. For example, in the summer of 2012, no Indiana bats were captured around known historic maternity areas using mist netting until the thirty-fifth net night at Fort Drum, and northern bats were not captured at all (Coleman et al. 2013). However, both of these species were detected using acoustic detectors within two nights of sampling. As the overall summer foraging activity continues to decline (and presumably overall bat numbers), mist-netting success likely will continue to decline, and therefore will no longer be the most efficacious monitoring technique. The use of acoustic monitoring as a way to track changes in species assemblages during the summer maternity season likely is more effective in WNS-impacted areas, and will likely need to be adopted as the primary method of summer monitoring as bat populations continue to decline.

The term ‘echolocation’ was first provided by Donald Redfield Griffin to describe how bats use echoes of sound to detect objects in their flight paths (Griffin 1958). The echolocation calls of most bats, including the species found in the eastern United States, are ultrasonic, i.e., beyond the upper limit of human hearing of approximately 20 kHz (Fenton 2002). The ability to detect the ultrasonic echolocation calls of bats was first accomplished with a detector developed to study insect noise and vocalizations (Pierce and Griffin 1938). Bat detectors convert the ultrasonic calls of bats into audible signals (Pettersson 2002). Anabat detectors (Titley Electronics, Ballina, New South Wales, Australia) convert ultrasonic sound using frequency-division stored to compact flash-storage Zero-Crossings Analysis Interface Modules (ZCAIM) for subsequent visual or computer analysis. Anabat detectors have been used in numerous studies to evaluate habitat use, foraging activity, and for acoustic identification of species (Betts 1998, O'Farrell and Gannon 1999, Britzke et al. 2002, Britzke 2003, Ford et al. 2006), including the ongoing acoustic surveys that were initiated at Fort Drum in 2003 (Ford et al. 2011). Some researchers have criticized acoustics as a method of identifying bats to the species level, particularly in distinguishing species with similar calls under various habitat conditions (Barclay 1999, Sherwin et al. 2000) such as the myotids of the Northeast. However, substantial research has been conducted recently to distinguish Eastern bat species under various conditions and with different detector types (Britzke and Murray 2000, Ahlén 2002, Ford et al. 2005). The relatively low number of bat species that occur in the Northeast can now be distinguished with high levels of accuracy (Britzke et al. 2002, Britzke et al. 2011).

One of the caveats to using acoustic technology as a method for monitoring bat populations is that individuals within a species cannot be distinguished (Gorresen et al. 2008), consequently making species abundance estimates impossible. However, relative bat activity can be assessed by the number of files (echolocation passes) recorded by a detector (Hayes 1997, Gorresen et al. 2008). Additionally, by using a call library and automated identification software (Britzke et al. 2002, Britzke et al. 2011) presence-absence datasets can be created to estimate site-specific detection probabilities using data from multiple sampling nights (MacKenzie et al. 2002). Finally, unbiased estimates of occupancy can be derived using single-species occupancy models in program PRESENCE (MacKenzie 2005). Using these estimates, the minimum level of effort needed to indicate changes in occupancy relative to management practices, habitat conditions, or seasonal variation can be discerned.

When developing presence-absence studies, understanding the animal-spatial use of the environment is critical to creating the highest possible probability of detection, particularly in systems where capture or presence confirmation may be difficult due to population declines. Home range studies and associated habitat analyses can aid in determining optimal conditions for sampling species. Radio telemetry often is used to classify day-roosts by site and landscape level characteristics (Menzel et al. 2001a, Menzel et al. 2002, Johnson et al. 2009) but has sparingly been used to determine home-range size, habitat use, and foraging behavior of bats (Barclay 1985, Kerth et al. 2001, Menzel et al. 2001b, O'Donnell 2001, Henry et al. 2002, Owen et al. 2003, Menzel et al. 2005) relative to other mammalian species. Using home-range telemetry data to assess summer foraging ecology of bats at Fort Drum will aid in determining whether acoustic monitoring stations are appropriately positioned to capture foraging use on the landscape and may allow for comparisons of home-range size and habitat use with populations that have not been impacted by WNS (Henry et al. 2002).

Natural resources managers at Fort Drum hope to establish an innovative, fiscally and logistically feasible acoustic monitoring protocol that will adequately represent the annual changes in summer foraging activity, use, and distribution of Indiana bats and other species within the landscape. This should help provide a basis for developing mitigation plans for land management activities in areas where Indiana bats still occupy the landscape. This protocol will hopefully also assist in developing plans to help address any new regulatory requirements that may be developed if additional bat species become listed. Such a protocol may be applicable in other landscapes across the country, particularly those that have also been heavily impacted by WNS. To establish guidelines for the best possible acoustic monitoring program at Fort Drum, the objectives of my graduate research were:

1. Conduct foraging telemetry on little brown bats to determine the congruency between nocturnal spatial use patterns of foraging habitat and acoustic monitoring locations.
2. Compare passive and active acoustic sampling designs to determine the more cost efficient and effective method to collect high quality calls and determine presence/absence of bat species.
3. Determine the most efficient and effective passive acoustic landscape sampling design for detecting bat species, with efforts focused at the federally-endangered Indiana bat and the little brown bat.

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Chapter 2: An overview of little brown bat habitat preferences at Fort Drum Military Installation using radio-telemetric home range analysis and acoustic detection

Abstract

Understanding animal-spatial use of the environment is a paramount task of wildlife management. With the dramatic declines of bat populations in the Northeast, assessing the apparent habitat preferences of myotid bats in the Northeast and elsewhere is critical to guide the development of effective monitoring protocols. In the summers of 2010 and 2012, adult female little brown bats (*Myotis lucifugus*) day-roosting in an artificial bat house were captured and radio-tagged at Fort Drum Military Installation in northern New York. I conducted fixed-station simultaneous telemetry to determine nocturnal spatial use of available habitat. Successful location estimates were used to create fixed kernel home range estimates for bats with > 30 locations (n=7) and to perform subsequent habitat analyses. In the summers of 2011 and 2012, I deployed a 52 ha grid of 4 x 4 Anabat acoustic detectors over five, 6-8 day sampling periods in various riparian and non-riparian environments near the same artificial bat house. The mean home range of 143 ha (n = 7, SE = 71.0) completely overlapped the acoustic grid area. Rankings of habitats using Euclidean distances from telemetry data revealed a higher proportional use of forested riparian ($P < 0.0001$) habitats than other habitat type at the landscape scale. Pair-wise comparisons of habitat types indicated that bats were found significantly closer to forested riparian habitats and forests than to open water, development, fields, shrublands, or other riparian habitats ($P < 0.0001$) at the landscape scale. Although overall acoustic detector occupancy estimates were 1.0 in non-riparian habitats versus 0.43 in riparian habitats, naïve occupancy was 0.8 and 0.6 and mean nightly detection probabilities were 0.23 and 0.08 in riparian and non-riparian sites, respectively. Results herein suggest that little brown bats select for forested

riparian and forested habitats during their nocturnal activity at the landscape scale but may be most easily detected acoustically at riparian sites.

Introduction

White-nose syndrome (WNS) is an emerging fungal disease that was first described in the United States in 2006 (Blehert et al. 2009). Currently, 22 states and 5 Canadian provinces have confirmed the presence of the disease (Figure 2.1) and 2 additional states have confirmed the presence of the fungus associated with WNS, *Geomyces destructans* (*Gd*; USFWS 2013). WNS is known to impact 7 species of eastern cave-dwelling bats: the federally endangered Indiana bat (*Myotis sodalis*), the federally endangered gray bat (*M. grisescens*) the petitioned eastern small-footed bat (*M. leibeii*), the petitioned northern bat (*M. septentrionalis*), the big brown bat (*Eptesicus fuscus*), the little brown bat (*M. lucifugus*), and the tri-colored bat (*Perimyotis subflavus;* USGS 2012, USFWS 2013). Of these species, the little brown bat and the tri-colored bat are believed to have suffered the most severe declines (USGS 2012). Since its onset, WNS has caused the death of more than 5.7 to 6.7 million bats (USFWS 2013), decreasing many winter hibernacula colonies 75- 99% (Blehert et al. 2009, Frick et al. 2010, Turner et al. 2011).

The little brown bat, once one of the most common insectivorous bat species in North America (Fenton and Barclay 1980), now faces potential local extirpation in the northeastern United States and southeastern Canada as a result of white-nose syndrome associated mortality (Frick et al. 2010). Little brown bats are known to use a variety of forested and open habitats near bodies of water for foraging (Fenton and Barclay 1980), and also are known to utilize diverse forested and manmade structures as summer roosting sites (Davis and Hitchcock 1965, Fenton and Barclay 1980). Continentally and regionally, available habitat was unlikely to be a limiting factor (Fenton and Barclay 1980, Brooks and Ford 2005) prior to population declines and is probably even less likely with fewer individuals on the landscape following declines. As a species that was once found in colonies of hundreds to thousands of individuals in the summer (Davis and Hitchcock 1965), little brown bats are now rarely observed in the Northeast, as most documented colonies have collapsed. As population declines continue (Frick et al. 2010, Dzal et al. 2011, Ford et al. 2011, Turner et al. 2011), understanding the spatial and temporal use of the landscape by little brown bats will be important for their future conservation. This information will become even more critical for land managers to possess if the current status assessment (Kunz and Reichard 2010) for little brown bats indicates that protection under the Endangered Species Act is warranted.

Home range analysis is frequently used in wildlife studies to assess animals' spatial use of the environment. The term "home range" is defined as the "area traversed by the individual in its normal activities of food gathering, mating, and caring for young" (Burt 1943). Although there are a number of studies, albeit limited relative to other wildlife species, that report on home range estimates of various species of bats in the eastern United States and Canada (Menzel et al. 2001, O'Donnell 2001, Henry et al. 2002, Owen et al. 2003, Menzel et al. 2005a, Broders et al. 2006), there is no research reporting on the foraging home range and associated habitat use of little brown bats in the post-WNS Northeast. In this study, I report on the nocturnal spatial use of adult female little brown bats from a maternity colony from an artificial bat house at Fort Drum Military Installation (Fort Drum; Dobony et al. 2011). This colony has been the focus of a mark-recapture monitoring program that was initiated in 2009 in response to observed WNS-associated declines. Despite overall severe declines of the species in the Northeast and mid-Atlantic (Frick et al. 2010), the colony at Fort Drum, originally estimated at approximately 1500 individuals, has now persisted for several years post-WNS with approximately 150-350 individuals each summer (Dobony et al. 2011).

In addition to the mark-recapture study, there have been multiple ongoing monitoring programs of all bat species at Fort Drum since 2003. Primarily initiated to comply with the Endangered Species Act (ESA 1973, as amended) and determine potential use of the military installation by the federally-endangered Indiana bat, ongoing monitoring efforts have provided important observations of all bat species pre and post-WNS. Analysis of acoustic activity patterns from 2003-2010 showed significant declines in overall summer foraging activity in little brown bats as well as a decline in foraging activity from early to late season in post-WNS years, indicating a probable decrease in reproductive success of surviving individuals and subsequent lack of juvenile recruitment (Ford et al. 2011). In response to severe declines in little brown bats and other WNS-impacted species and traditional monitoring methods becoming less effective, a more intensive acoustic monitoring survey was initiated in summer 2011 to develop an efficient sampling design to track little brown bats and Indiana bats, specifically. Included in this sampling design was a grid of acoustic detection sites that were opportunistically placed in the vicinity of known maternity roosts and presumed foraging areas of little brown bats. The objective of this study was to determine the congruency between nocturnal spatial use of foraging habitat and acoustic monitoring locations for little brown bats by assessing probable foraging home ranges and associated habitat analyses and developing occupancy and detection estimates. Such data will assist Fort Drum managers in choosing optimal detector deployment sites for implementing an effective acoustic sampling protocol for myotid species.

Study Area

Fort Drum Military Installation is a U.S. Army installation of approximately 43,000 ha in Jefferson and Lewis counties in northern New York (44°00'N, 75°49'W; Figure 2.2). The installation lies at the intersection of the St. Lawrence-Great Lakes Lowlands, the foothills of the Adirondack Mountains, and the Tug Hill Plateau ecoregions within the Black River and Indian River drainages. The nearby Niagara Escarpment (10-15 km west of Fort Drum) contains karst formations and caves that Indiana bats are known to use for winter hibernation (Fenton 1966). Approximately 57% of the landscape is made up of forested habitat dominated by northern hardwood types such as sugar maple (*Acer saccharum*), American beech (*Fagus grandifolia*), white ash (*Fraxinus americana*), black cherry (*Prunus serotina*) and white pine (*Pinus strobus*). Early successional understory habitat consists of red maple (*Acer rubrum*), gray birch (*Betula populifolia*), and quaking aspen (*Populus tremuloides*) regeneration. Wetland systems such as wet meadows and beaver (*Castor canadensis*) impacted streams and ponds make up 20% of the landscape. Development is concentrated in a cantonment area, with the remainder of the installation consisting of 18 training areas, an airfield, and a large, centralized main impact zone that are all largely undeveloped.

Methods

i. *Radio telemetry*

In the summer of 2012, I captured little brown bats from an artificial maternity roost using double-stacked mesh mist nets (3 m to 12 m in width; Avinet, Inc., Dryden, NY) from June through July. I placed nets along edges between open field and forests and along forested corridors near the artificial bat house. For each bat that was captured, I recorded age, sex, reproductive condition, mass and right forearm length. I attached 0.34-g radio transmitters (LB

2XT Holohil Systems Ltd., Ontario, Canada) to the intrsacapular region of adult females using Skin Bond (Smith and Nephew, Largo, Florida) or Perma-Type (Perma-Type Company Inc., Plainville, Connecticut) surgical cement. Transmitters to body mass ratios were less than 5% as suggested by Aldridge and Brigham (1988). I released bats near the site of capture and began foraging telemetry the following night to avoid including unusual behaviors as a result of capture.

I used fixed-station telemetry to conduct simultaneous biangulation (Owen et al. 2003, Menzel et al. 2005a) within or adjacent to the expected foraging area (Menzel et al. 2005a) in the Cantonment Area from emergence (approximately 21:00) until bats returned to the bat box, roosted elsewhere, or could no longer be found. I used Wildlife Materials TR4-2000S (Carbondale, Illinois) telemetry receivers and five-element Yagi antennas to estimate the azimuths of foraging bats in 5-10 minute intervals (Menzel et al. 2005a). I monitored signals in synchronization with another observer at an approximately 90° position while surrounding the presumed core foraging area of the bats. I tracked individual bats nightly for the entire duration of the transmitter operability of approximately 21 days or until transmitters were recovered.

I entered coordinates of each fixed telemetry station and all compass readings for Azimuth locations into software LOCATE III (Pacer Computing, Tatamagouche, NS, Canada) to obtain Universal Transverse Mercator (UTM) coordinates of each foraging location (Owen et al. 2003, Menzel et al. 2005a). I calculated home range estimates for bats with ≥ 30 location estimations (Seaman et al. 1999) using the habitat analysis tool in Biotas (Ecological Software Solutions, LLC, Hegymagas, Hungary). I used the fixed kernel density estimator with the least-squares cross validation smoothing factor based on a 95% confidence interval to exclude outliers. In kernel estimation, each point is evaluated and assigned a density value based on the quantity of other points that surround it. This contrasts with minimum convex polygons (Mohr 1947), which are constructed by connecting the outermost location points to form a convex polygon (White and Garrott 1990) and increase indefinitely as the number of locations increases (Jennrich and Turner 1969). Although I only calculated home range estimates for bats with > 30 locations (Seaman et al. 1999), I included 3 additional bats from 2012 and 2 bats that were previously tracked in 2010 in habitat analyses to increase overall sample size. Bats in 2010 were captured, processed, and tracked following the same methods that I conducted herein.

I created a habitat map of the study area using 2006 land cover data provided by the Fort Natural Resources Branch (Fort Drum Environmental Division, Fort Drum, New York) for areas inside the installation boundary and the 2006 National Land Cover Database (Fry et al. 2011) for areas outside the boundary. I reclassified habitat types from both sources into 7 categories: open water, forests, development, fields, shrublands, forested riparian, and all other riparian, i.e. various emergent wetland and wetland meadow complexes. Open water, forests, development, fields, and shrublands were categorized directly from landcover sources. Forested riparian zones were derived by creating a buffer zone that stretched inward approximately 15 meters in each direction from the edge of forested and open water habitats according to the suggested streamside management zone for slopes of less than 10% in New York State (NYDEC 2011). Other riparian zones were derived in a similar fashion at the edge of open water and all other habitat types. I exported home range polygons from Biotas for use in ArcMap 9.3 (Environmental Systems Research Institute, Redlands, California, USA) for area calculations and habitat analysis.

Despite the widespread use of compositional habitat analysis (Aebischer et al. 1993), I chose to use the Euclidean distance approach which analyzes habitat use in a linear fashion and may capture the use of ecotones or edges. The Euclidean distance method evaluates habitat use by comparing the distances from animal locations and random locations to nearest edges of habitat types (Conner and Plowman 2001). Additionally, the Euclidean distance approach requires no explicit error modeling (Conner and Plowman 2001) thereby lessening problems associated with the difficulty in tracking small, volant animals with extreme agility. Furthermore, unequal sampling of individuals is not problematic because individual variation is not assumed to be constant. Finally, Euclidean distance can be adapted to multiple spatial scales. I used Euclidean distance to assess habitat use at the home range scale and at the landscape scale. I defined the landscape scale as a buffered area around the cumulative home range area for which I defined the lateral extent as the greatest distance between all foraging points. At both scales, I used the Distance tool in ArcMap to calculate the Euclidean distance between each location and the edge of the closest representative polygon of each habitat type (Conner and Plowman 2001). I used the Create Random Points tool in ArcMap to create random locations to pair with bat locations within individual bat home ranges and across the entire landscape to represent the two scales, respectively (Figure 2.3). I created a vector of ratios between distances to habitat types from foraging and random locations and used a multivariate analysis of variance (MANOVA) to determine if ratios were different from 1.0, indicating nonrandom habitat use. I then used a paired t-test to determine whether habitats were used in proportion to their availability at each scale and a series of t-tests to determine habitat preference rankings. I defined statistical significance as $\alpha = 0.05$ and used SAS statistical software (SAS Institute 2012, Cary, NC, USA) to perform all statistical analyses.

ii. Acoustic detection

During the summers of 2011 and 2012, I collected acoustic data on the little brown bat as part of a survey program that was initiated to revise monitoring protocols initially established in 2003. Because randomly selected passive acoustic monitoring stations may not successfully detect little brown bats due to WNS declines (Ford et al. 2011), I focused sampling efforts to specifically target the presumed little brown bat foraging area near the known historic maternity area. I deployed a 52 ha grid of 4 x 4 Anabat acoustic detectors over 6-8 day sampling periods in various riparian (n= 11) and non-riparian (n=5) environments near the artificial bat house (Figure 2.4). Sampling occurred during 5 periods: 25 July to 1 August in 2011 and 30 May to 5 June, 20 June to 27 June, 6 July to 13 July, and 23 July to 30 July in 2012.

I collected acoustic data using Anabat II detectors connected to a compact flash-storage Zero-Crossings Analysis Interface Module (ZCAIM), as well as SD1 and SD2 units (Titley Electronics, Ballina, New South Wales, Australia). Before deployment, I calibrated all units using an ultrasonic insect deterring device following the methods of Larson and Hayes (2000). I placed Anabat units in weatherproof boxes with polyvinyl chloride (PVC) tubes attached that contain a small weep hole in the bottom for water drainage according to the methods of O'Farrell (1998; Figure 2.5). Boxes were placed on 1.5 m tripods aligned in a manner that allowed sound to enter the PVC tubes at a 45 degree reflective angle to be received by Anabat transducers perpendicularly (Britzke et al. 2010).

To ensure that more than one Anabat did not collect data on the same bat simultaneously, I placed the detectors in the 4 x 4 design with approximately 200-250 m between each placement site (Figure 2.4). At each sample site, I chose deployment locations and the azimuth of microphone direction to maximize call quality. For example, I targeted sites with uncluttered openings such as canopy gaps, forested trails with open corridors, or open water. I set Anabats to a timer to record data continuously from approximately 1900 to 0700 hours for 6-8 days during each sampling period. I changed batteries and memory cards as needed and downloaded data onto a laptop computer using the CFCread program (Titley Electronics Ballina, New South Wales, Australia).

I used EchoClass (U.S. Army Engineer Research and Development Center, Vicksburg, MS, USA), an automated analysis program currently in final development, to identify bat calls to the species level. Although the ability to identify bat calls to the species level has been criticized (Barclay 1999), research has suggested that good quality calls of eastern North American bats can be identified both qualitatively (O'Farrell et al. 1999) and quantitatively (Britzke et al. 2002, Britzke et al. 2011). To minimize the impact of species identification when accuracy is less than 100%, EchoClass provides a maximum likelihood estimate which allows the user to determine the probable presence or absence of a species with predetermined levels of accuracy (Britzke 2002). In this study, I considered little brown bats to be present at a site if the maximum likelihood values estimate for an individual species' identified call was $\geq$ 90%.

I created a nightly presence-absence detection history from the acoustic data (Gorresen et al. 2008). I considered each night survey independent due to the separation of sites and break in sampling during daylight hours. I used program PRESENCE (version 2.4, Hines and Mackenzie, 2008) to attempt to fit a candidate set of *a priori* models that incorporated broad habitat categorizations as a site covariate and several time parameters as sampling covariates that could affect the probability of detection or occupancy (Table 2.1). I eliminated from the candidate set any models that included illogical parameter estimates or for which the parameter estimates did not converge. For example, I did not include "day" as a sampling covariate in any models that did not also include "day^2", as I did not want to restrict model fit to a straight line. I ranked models using Akaike's Information Criterion (AICc) corrected for small sample size and compared the weight of evidence among candidate models using Akaike weights (Burnham and Anderson 2002). I compared occupancy estimates between habitat types using multi-season models that assumed changing detection probabilities based on site and sampling covariates. I then extracted the 95% confidence set of models (Burnham and Anderson 2002) to recalculate model weights (Weller 2008). Finally, I used PRESENCE to create a simulated dataset of standard errors of occupancy at varying levels of sampling effort at the different habitat types for comparisons to the 95% candidate set of competing models (Figure 2.6).

Results

i. *Radio telemetry*

I tracked and determined fixed kernel density home range estimates for seven adult female little brown bats from June to August 2012. My sample included one pregnant bat, two lactating bats, three post-lactating bats, and one non-reproductive bat; to maintain sample size, bats were not separated by reproductive status in analyses. The mean number of locations used to calculate each bat's home range was 63 (SE = 9.9, range = 33-102). The mean area of little brown bat home ranges was 143.0 ha (n = 7, SE = 71.0; Table 2.2).

Mean distances from bat locations to habitat types (Table 2.3) were not different from random at the home range scale ($F_{df=7,5}$ = 3.88, P = .0775) but were nonrandom at the landscape scale ($F_{df=7,5}$ = 379.37, P < 0.0001). At the landscape scale, bats were found significantly closer to open water (t = -32.00, P < 0.0001), development (t = -5.20, P = 0.0003), forests (t = -29.61, P < 0.0001), shrublands (t = -6.02, P < 0.0001), forested riparian (t = -35.32, P < 0.0001), and other riparian (t = -21.39, P < 0.0001) than expected but further from fields (t = 10.31, P < 0.0001) than expected. A ranking of habitats showed that forested riparian zones and forests were used proportionally most followed by open water habitats (Table 2.4). Pairwise comparisons of the distances between little brown bat foraging locations and habitat types indicated that bats foraged significantly closer to forested riparian zones and forests than to any other habitat type, followed by open water and other riparian zones, but there was not a significant difference between the distance to forested riparian zones or forests (Table 2.4).

ii. Acoustic detection

I collected acoustic data during 5 sampling periods totaling 40 sampling nights at a grid of detectors placed in the vicinity of the little brown maternity colony. I detected all species that occur at Fort Drum at least once during the entire sampling period: big brown bats (*Eptesicus fuscus*), silver-haired bats (*Lasionycteris noctivagans*), hoary bats (*Lasiurus cinereus*), eastern red bats (*L.* borealis), eastern small-footed bats, little brown bats, northern bats, Indiana bats, and tri-colored bats. Little brown bats were detected at 12 of the 16 sites. However, I removed one of the 12 sites where little brown bats were detected due to its close proximity to the artificial bat house because of the multiple instances of severe sound distortion caused by multiple bats being detected simultaneously. Little brown bats were not detected at the remaining 4 sites (Figure 2.4).

My 95% confidence set of models included 3 competing top models according to AIC weights (Table 2.5). The three models present occupancy estimates that varied by habitat and detection probability that varied by habitat and time related covariates (Table 2.6). Simulated standard errors for little brown bat occupancy at Fort Drum suggest that the standard error of occupancy is expected to be approximately 0.12 or 0.03 (Figure 2.6) after 40 nights of sampling at 15 non-riparian or riparian sites, respectively. Models 1 and 3 had large standard errors about the occupancy estimate of riparian habitats and detection probabilities of both riparian and non-riparian habitats. However, standard errors provided by Model 2 were lower for both occupancy and detection probabilities (Table 2.5) than other models. Model 2 suggests higher occupancy of little brown bats at non-riparian sites versus riparian sites but in actuality there was higher naïve occupancy estimates and probabilities of detection at riparian sites than non-riparian sites (Table 2.5, Figure 2.7).

Discussion

Few efforts have been attempted at assessing little brown bat home range and habitat use by telemetric methods, with work to-date restricted to Acadian region boreal forests in eastern Canada (Henry et al. 2002, Broders et al. 2006) and the agricultural Midwest (Bergeson 2012). The little brown bats in my study exhibited larger home range sizes than 90% fixed kernel estimates of pregnant or lactating females in Quebec, Canada (Henry et al. 2002) and 100% minimum convex polygons of males in New Brunswick, Canada (Broders et al. 2006). Conversely, in the Midwest, Bergeson (2012) reported a mean home range size much larger than was found in my study (515 ha versus 143 ha), although 95% fixed kernels were calculated on adult females in both cases. Unlike my study, these studies occurred before the onset of severe WNS in their respective regions. Additionally, bats in my study were pooled together regardless of reproductive status in order to retain sufficient sample size, but there did not appear to be an obvious relationship between reproductive status and home range size. Although these studies may be useful for general comparisons, results are perhaps not representative of the Northeast in general or on Fort Drum in particular. Discrepancies in my home range estimations and other studies may be caused by differences in estimation method, differences between males and females, sample size, habitat quality and spatial arrangement, and the presence or absence of WNS associated declines.

For Euclidean distance habitat analysis, my findings were consistent with the results of Broders et al. (2006) and Bergeson (2012). Specifically, Broders et al. (2006) reported that bats select open water and deciduous forested sites and Bergeson (2012) described that bats selected closed canopy and open hydric sites but avoided open field sites. My analysis revealed a preference of adult female little brown bats for forested riparian zones and forests followed by open water, and other riparian zones at the landscape scale. The high selection for aquatic habitats is unsurprising, as little brown bats are known to have a generalist diet that across a diverse variety of adult aquatic insects (Belwood and Fenton 1976, Edythe and Kunz 1977). It is also well documented that linear landscape features such as riparian zones and corridors are important to bats for foraging habitat, protection from predators, and energetically inexpensive travel between roosting and foraging areas (Verboom and Huitema 1997, Menzel et al. 2005a, Menzel et al. 2005b, Rogers et al. 2006). The suspected foraging area of the bats I observed consisted of a semi-disturbed area within the highly developed Cantonment Area of Fort Drum. This area contains an abundance of corridor habitats such as old logging roads and hiking trails. These corridors may be especially important for pregnant or lactating females that travel back and forth several times throughout the night between open water and riparian foraging habitats and the artificial bat box or other night roosts.

At Fort Drum, acoustic surveys showed that occupancy and detection probability estimates vary by habitat type. The occupancy and detection estimates given by the three models were similar, but Model 2 most closely represents expected standard errors of occupancy given the level of effort in this experiment. As a result, Model 2 is the recommended model from my study, regardless of higher standard errors than expected at riparian sites. Moreover, the simulation shows that the standard error at riparian sites is expected to level off at a sampling effort much less than what was expended in my experiment (i.e., 20 nights). Therefore, it is uncertain if sampling efforts of > 40 nights would have lowered standard errors to more desirable levels. High detection probability estimates at riparian sites were expected, but the discrepancy between occupancy and detection probability estimates at riparian versus non-riparian sites was unforeseen.

It would follow that little brown bats occupy forested habitats due to their low wing loadings and small bodies that facilitate maneuverability (Aldridge and Rautenbach 1987). Therefore, it is unsurprising that forests were equally preferred to forested riparian habitats according to the Euclidean distance habitat analysis and that there was a high occupancy estimate at non-riparian sites. However, occupancy estimates very close to 1 should be cautiously interpreted if obtained when the detection probability is < 0.15 (MacKenzie et al. 2002). Such estimates are based on low amounts of presence data, making it difficult for the model to distinguish between genuine absences and non-detections. Although it is reasonable to expect that little brown bats do occupy forested habitat at Fort Drum, the accuracy of the occupancy estimate at these sites is uncertain. The naïve estimates suggesting higher occupancy in riparian habitats are likely more representative of actual foraging habitat. Schirmacher et al. (2007) found similar acoustic survey results in the New River Gorge area of West Virginia, with little browns more likely to be present in openings in forested habitats than closed forests, particularly when water was present or nearby. Additionally, using acoustics, Brooks and Ford (2005) found ubiquitous habitat use in central Massachusetts in a landscape with abundant watercourses, beaver meadows, bogs and small ponds. Brooks (2011) reported overall declines in *Myotis* species activity since the onset of WNS but with the residual greatest activity levels at forested roads than at other forested sites such as recent clearcuts, streams, and beaver meadows. Despite

WNS, little brown bats continue to utilize habitat in similar patterns acoustically, although overall populations are smaller. In this study, all acoustic detectors that were set in "non-riparian" sites were placed along trails or near canopy gaps where activity was expected. The low probability of detecting little brown bats in forested habitats when they are present, as demonstrated in this study, may inhibit efficient monitoring practices.

Little brown bats are known to forage near bodies of water (Fenton and Barclay 1980) where adult aquatic insect abundance is high (Belwood and Fenton 1976). In fact, Brooks and Ford (2005) sampled predominately at sites associated with water in Massachusetts and avoided closed-canopy forests due to expected low flight activity in these habitat types. Johnson et al. (2008) reported that overall bat activity, for which little browns were the predominant species, was highest at water sources within a rural-urban gradient of the Mid-Atlantic region near Washington, D.C. Studies in the Coastal Plain of South Carolina confirm the importance of riparian and wetland areas as foraging habitats for most species of bats and suggest that monitoring acoustically within forest canopies may underestimate activity levels of many species of bats (Menzel et al. 2005b, Ford et al. 2006). Therefore, the higher probability of detection at acoustic sites dominated by aquatic characteristics in this study is consistent with most acoustic studies.

Although some research has suggested that cluttered habitats do not significantly impact the ability to detect bats acoustically that are adapted to such environments (Patriquin et al. 2003, Menzel et al. 2005b), a threshold may exist whereby acoustic detection is compromised in forested habitats (i.e., call structure is altered). For example, I chose to remove the acoustic detection site that was located directly northwest of the artificial bat roost because preliminary identification analysis indicated that calls contained high frequency, unusual calls that and were difficult to distinguish from other myotid species with similar call structures (O'Farrell et al. 1999, Britzke et al. 2002). Perhaps a similar situation occurred at other forested corridor sites, causing little brown bat calls to go unidentified, even though microphones were placed in my perceived best possible collection circumstances. Although identification accuracy rates of >90% can be achieved for northeastern myotids in open habitats, the impact that clutter has on accuracy rates is not as well understood (Britzke et al. 2002). Alternatively, it is possible that the little brown bats I tracked use forested habitats primarily for travel and were not required to use extensive echolocation for navigation of these familiar corridors, resulting in lower acoustic detection. However, any explanations for differences are wholly speculative, as my model only explains changes in occupancy based on relatively broad habitat classifications, and not on call parameters.

Regardless of the differences between optimum derived occupancy and detection estimates at different habitats, telemetry and acoustic results suggest congruent patterns in bat activity at the landscape scale. My telemetry and acoustic results are consistent with previous research, suggesting that female little brown bats forage in forested riparian habitats and select for these areas over most other habitat types at the landscape scale. Although detectability was low at both habitat types, the probability of detecting little brown bats when they were present more than doubled at riparian sites versus non-riparian sites. Additionally, simulated standard errors suggest that the standard error of occupancy at riparian sites levels off at much less effort than 40 nights. Therefore, sampling of 20 or more nights at riparian sites would likely have produced similar results. Telemetry results further validate the preference of little brown bats for forested riparian habitats, as forested riparian zones, forests, open water, and other riparian zones were selected over development, shrublands, and fields. These data begin to establish use and distribution of the little brown bat around a maternity site, and have clear implications for acoustic detector placement for land managers interested in developing an efficacious acoustic sampling protocol for myotids.

The only other study that has compared telemetry and acoustic sampling found that results were qualitatively similar at the home range scale for determining the importance of pine habitat for evening bats (*Nycticeius humeralis*) in southwestern Georgia (Morris et al. 2011). However, acoustics failed to provide broad scale habitat use resolution, as hardwood forests that were selected for at the landscape scale via telemetry were underrepresented acoustically. In my study, congruency exists between acoustics and telemetry only at the landscape scale. In general, it should not be assumed that acoustic sampling will always provide similar conclusions to telemetry results at any scale, as acoustics are biased towards foraging behavior and may underrepresent habitats that are important for travel or day- and night-roosting. However, in my study, telemetry results successfully validate the use of general habitat types at the landscape scale that were chosen for acoustic sampling in an area where little brown bats were assumed to be present and active.

An important caveat to this study is the lack of a full picture of habitat use. Volant species are very difficult to track accurately using telemetry, and bats in this study were scarcely tracked for their entire foraging durations. Regardless, the results presented here provide a novel perspective for validating optimal conditions for acoustically monitoring little brown bats in a post-WNS environment. Furthermore, acoustics provide an alternative monitoring tool that may deliver similar results to telemetry at some scales, is simpler to implement over wider temporal and spatial scales, and will likely be more successful than other traditional capture methodologies and monitoring techniques as bat population declines continue. Continued monitoring of species like the little brown bat is especially critical in the face of WNS to assess further declines of this and other imperiled species. In areas that have been severely impacted by WNS, little brown bat detection will likely be difficult in any habitat and at any scale. Despite these challenges, the results of my study confirm that summer monitoring can be successfully accomplished by deploying acoustic detectors along riparian habitats for ≥20 nights in areas where little brown bats were historically present and currently still known to be, as confirmed by telemetry. Employing monitoring programs such as these, sooner rather than later, will help managers understand bat spatial and temporal use of their properties and provide them the information they need to address potential new state or federal listings of little brown bats.

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Tables and Figures

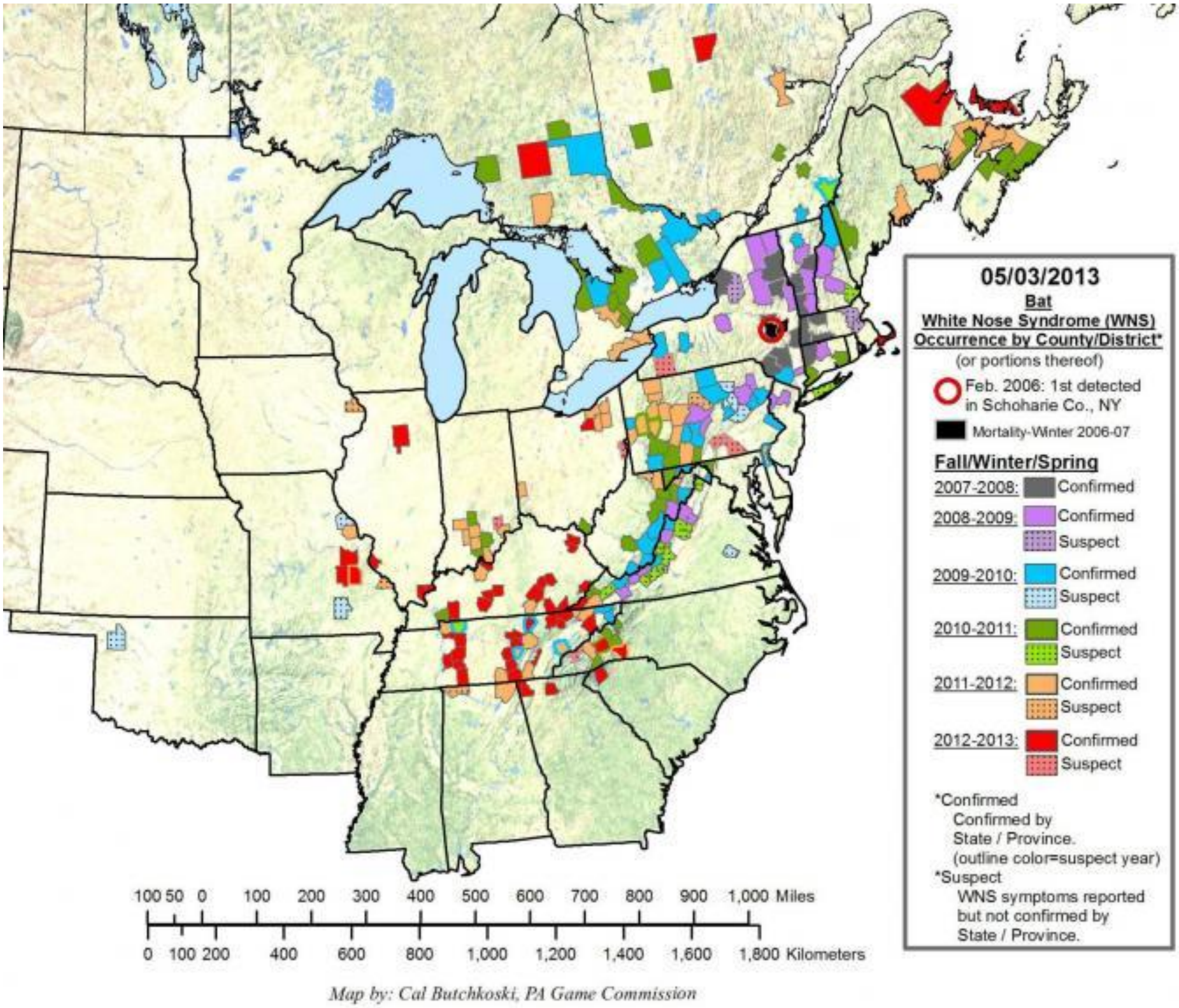

Figure 2.1 Distribution map of white-nose syndrome (WNS) in the eastern United States. Last updated 3 May 2013. Available: http://www.whitenosesyndrome.org/resources/map (USFWS 2013).

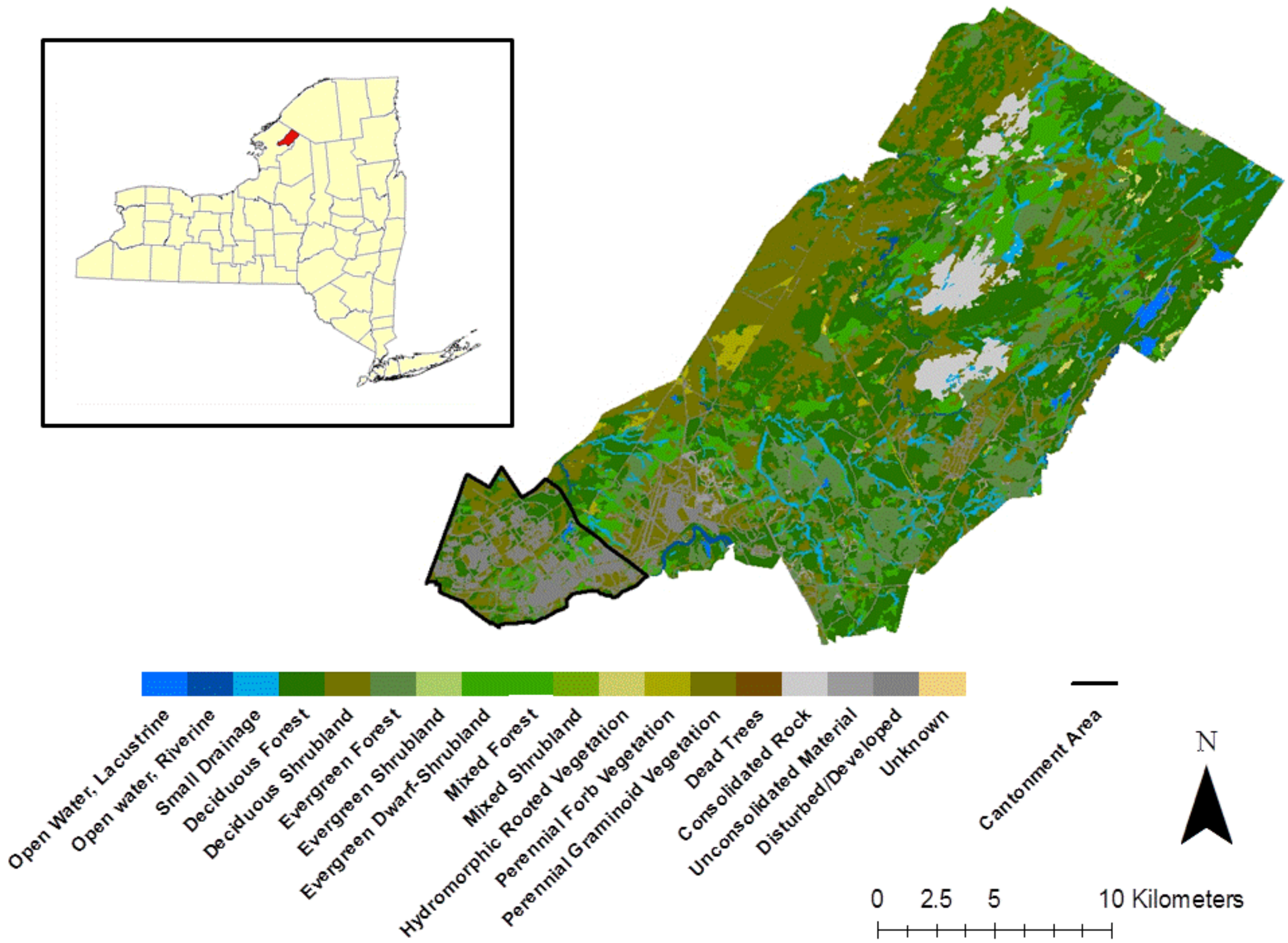

Figure 2.2 Fort Drum Military Installation, Jefferson and Lewis Counties, New York; Cantonment Area site of the 4x4 grid of passive acoustic sampling for little brown bats, summers 2011 and 2012.

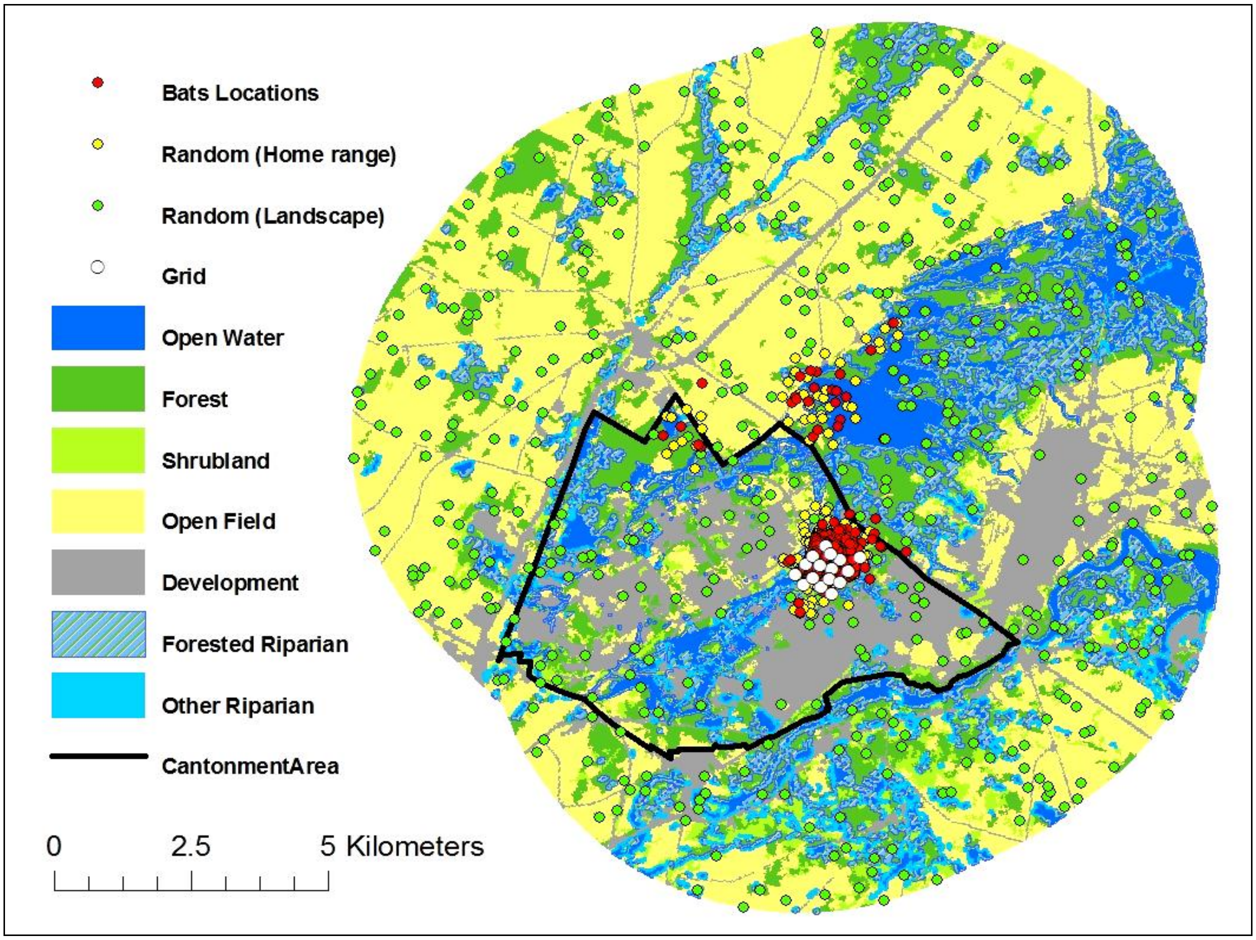

Figure 2.3 Little brown bat (MYLU) foraging locations and associated random locations at the home range and lanscape scales. Home range scale random locations created inside 95% fixed kernel density estimations. Landscape scale random locations created within landscape area defined as a buffered area around the cumulative home range equal to the greatest distance between two foraging locations. Grid points represent acoustic detection sites centralized in the anticipated home range area. Foraging telemetry conducted June-July 2012 at Fort Drum Military Installation, New York.

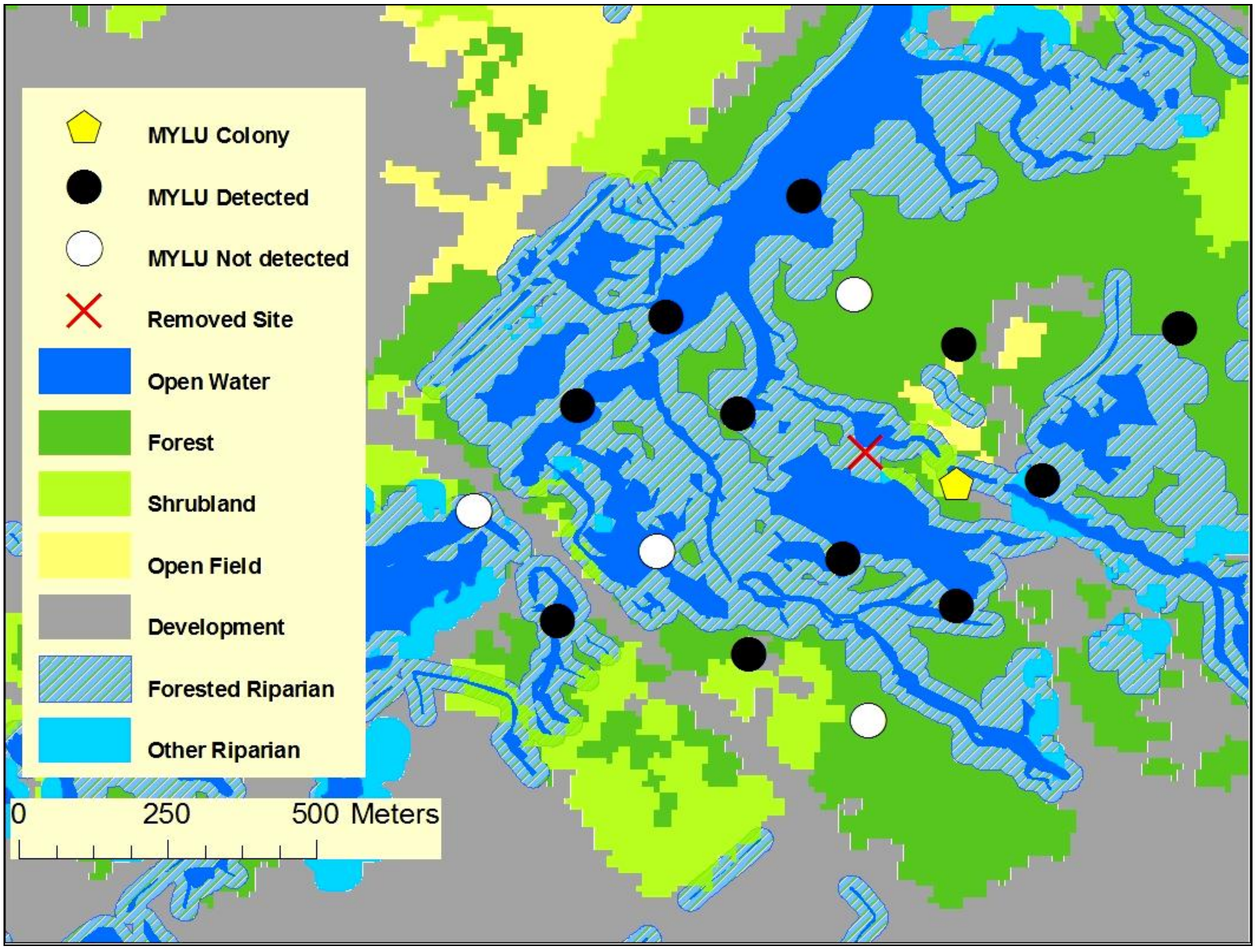

Figure 2.4 Grid of passive acoustic detectors deployed summers 2011 and 2012 near a maternity colony of little brown bats (MYLU) in an artificial bat roost at Fort Drum Military Installation.

Figure 2.5 Passive acoustic sampling setup using Anabat detectors encased in weatherproof boxes mounted on 1.5 meter tripods at a 45° angle with polyvinyl chloride (PVC) tubes attached that contain a small hole in the bottom for water drainage, Fort Drum Military Installation, New York, summers 2011 and 2012.

Table 2.1 Multiple-season occupancy models explaining the influence of habitat and time on occupancy and detection estimates of little brown bats at a grid of acoustic echolocation detectors at Fort Drum Military Installation, New York, summers of 2011 and 2012; Ψ = occupancy, γ = colonization, ε + extinction, p = detection, habitat = riparian versus non-riparian, day = day of the year, day^2 = day of the year squared, year = 2011 versus 2012, day*year = interaction term of day and year, full = full identity, '.' = constant.

Model	*K*	*AICc*	Δ*AICc*	ω*i*
Ψ(habitat), γ(full), ε(full), p (habitat + day + day^2)	14	201.02	0.00	0.4292
Ψ(habitat), γ(full), ε(full), p (habitat + day*year + day^2)	14	201.27	0.25	0.3787
Ψ(habitat), γ(full), ε(full), p (habitat + day + day^2 + year)	15	202.95	1.93	0.1635
Ψ(habitat), γ(.), ε(.), p (habitat)	6	207.43	6.41	0.0174
Null	4	210.65	9.63	0.0035
Ψ(habitat + year), γ(.), ε(.), p (habitat + year)	8	210.97	9.95	0.0030
Ψ(habitat), γ(day*year), ε(year), p (habitat)	8	211.43	10.41	0.0024
Ψ(habitat), γ(full), ε(full), p (habitat + year)	13	213.48	12.46	0.0008
Ψ(habitat), γ(full), ε(full), p (habitat + day*year)	13	213.54	12.52	0.0008
Ψ(habitat + year), γ(full), ε(full), p (habitat + year)	14	215.48	14.46	0.0003
Ψ(habitat + day*year), γ(full), ε(full), p (habitat + year)	14	215.48	14.46	0.0003
Global	24	217.92	16.90	0.0001
Ψ(habitat), γ(full), ε(full), p (habitat)	12	233.31	32.29	0.0000

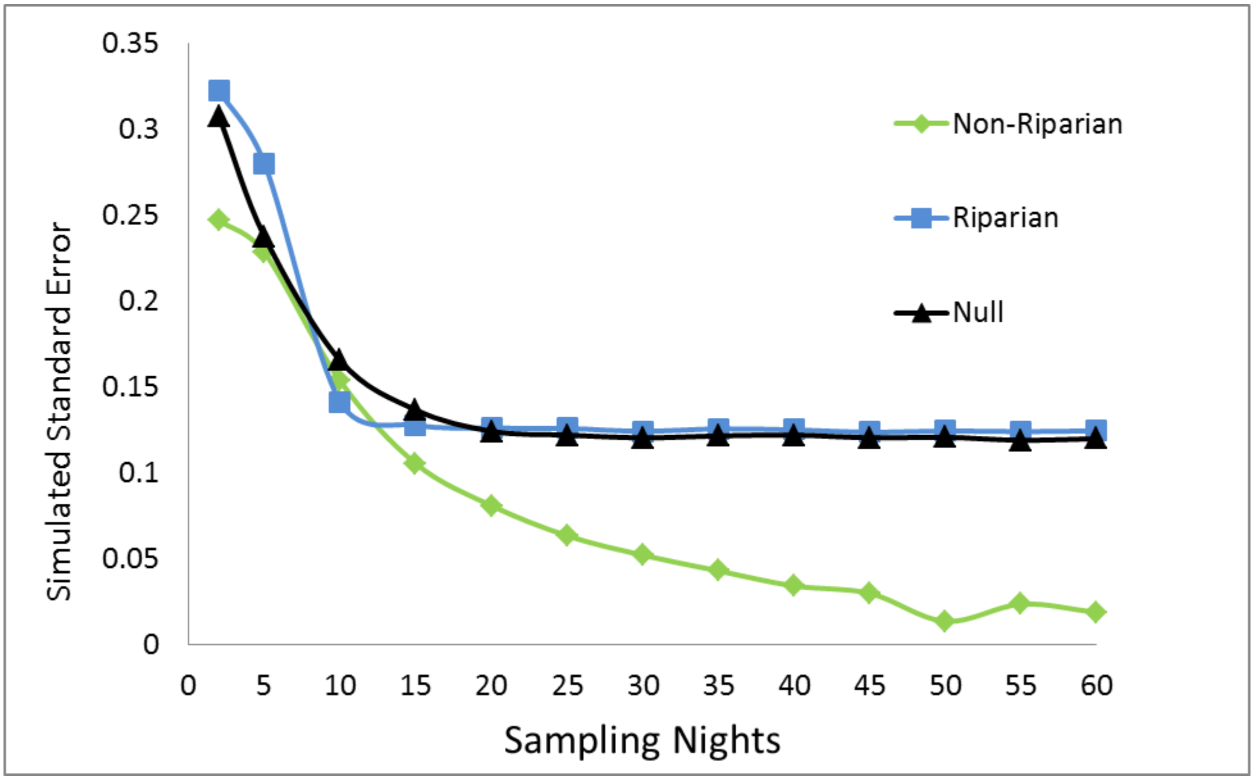

Figure 2.6 Simulated standard errors for little brown bat occupancy over increasing survey effort (sampling nights) at riparian and non-riparian sites and sites not specified to habitat type (null). Data from a grid of acoustic echolocation detectors at Fort Drum Military Installation, New York, summers of 2011 and 2012.

Table 2.2 95% fixed kernel summer home range sizes for adult female little brown bats at Fort Drum Military Installation, June-August 2012; P = Pregnant, L = Lactating, PL = Post-lactating, NR = Non-reproductive.

Bat	Reproductive Status	Locations (#)	Area (ha)
353a	PL	33	59.9
181	L	40	122.4
102	P	44	81.4
265	L	52	63.3
586	PL	70	598.3
354	PL	100	44.6
428	NR	102	31.2
Mean		63	143.0

Table 2.3 Mean (SE) Euclidean distances of bat and random locations to habitat types derived from foraging telemetry conducted at Fort Drum Military Installation, June-August 2012.

Locations	*n*	Open Water	Development	Field	Forest	Shrublands	For. Riparian	Other Riparian
Bats	523	92.4 (6.9)	166.3 (12.2)	233.1 (8.5)	21.4 (3.2)	153.0 (5.9)	38.6 (6.7)	187.4 (6.4)
Random (Home range)	523	112.5 (12.2)	146.7 (11.0)	234.9 (17.5)	23.7 (5.2)	138.5 (6.4)	55.2 (13.7)	185 (10.4)
Random (Landscape)	523	312.3 (13.8)	229.5 (11.0)	143.5 (11.9)	115.1 (7.5)	188.8 (9.4)	276.8 (13.9)	326.8 (13.5)

Table 2.4 Ranking matrix of little brown bat habitat use at the landscape scale derived from foraging telemetry conducted at Fort Drum Military Installation, June-August 2010 and 2012. Numbers are *t*-statistics associated with pairwise comparisons of corrected distances to habitat. Rankings interpreted as relative magnitudes, i.e. larger values associated with higher proportional use.

	Open Water	Development	Field	Forest	Shrublands	Forested Riparian	Other Riparian
Open Water		-7.24***	-19.35***	4.04**	-12.85***	15.57***	-24.74***
Development	7.24***		-7.66***	7.83***	-2.25*	8.80***	2.98*
Field	19.35***	7.66***		30.40***	8.93***	22.68***	14.50***
Forest	-4.04**	-7.83***	-30.40***		-12.66***	1.35 (0.2029)	-14.35***
Shrublands	12.85***	2.25*	-8.93***	12.66***		14.57***	6.48***
Forested Riparian	-15.57***	-8.80***	-22.68***	-1.35 (0.2029)	-14.57***		-36.90***
Other Riparian	24.74***	-2.98*	-14.50***	14.35***	-6.48***	36.90***	

*$P < 0.05$
**$P < 0.01$
***P < 0.0001

Table 2.5 95% confidence set of models and associated mean occupancy (ψ) estimates for little brown bats at a grid of acoustic echolocation detectors at Fort Drum Military Installation, New York, summers 2011 and 2012. Ψ = occupancy, γ = colonization, ε + extinction, p = detection, habitat = riparian versus non-riparian, day = day of the year, day^2 = day of the year squared, year = 2011 versus 2012, day*year = interaction term of day and year, full = full identity.

Model	K	AIC	ΔAIC	ωi	$\bar{\psi}_{Rip.}$ (SE)	$\bar{\psi}_{Nonrip}$. (SE)
1. Ψ(habitat) γ(full) ε(full) p (habitat + day + day^2)	14	201.02	0.00	0.4418	0.6031 (0.5922)	1.0 (0.0)
2. Ψ(habitat) γ(full) ε(full) p (habitat + day*year + day^2)	14	201.27	0.25	0.3899	0.6137 (0.2167)	1.0 (0.0)
3. Ψ(habitat) γ(full) ε(full) p (habitat + day + day^2 + year)	15	202.95	1.93	0.1683	0.6168 (0.5699)	1.0 (0.0)

Table 2.6 Detection probability estimates ($\bar{p}$; standard errors) of little brown bats at riparian and non-riparian sites during 5 sampling events at a grid of acoustic echolocation detectors at Fort Drum Military Installation, New York, summers 2011 and 2012.

Model	Riparian (n = 10)				
	$\bar{p}_{Survey\ 1}$ (SE)	$\bar{p}_{Survey\ 2}$ (SE)	$\bar{p}_{Survey\ 3}$ (SE)	$\bar{p}_{Survey\ 4}$ (SE)	$\bar{p}_{Survey\ 5}$ (SE)
1	0.19 (1.9)	0.29 (3.5)	0.19 (1.5)	0.17 (1.1)	0.19 (1.8)
2	0.19 (0.07)	0.25 (0.10)	0.23 (0.07)	0.22 (0.06)	0.21 (0.08)
3	0.18 (1.8)	0.30 (3.9)	0.18 (1.9)	0.17 (1.5)	0.21 (2.15)
	Non-riparian (n = 5)				
	$\bar{p}_{Survey\ 1}$ (SE)	$\bar{p}_{Survey\ 2}$ (SE)	$\bar{p}_{Survey\ 3}$ (SE)	$\bar{p}_{Survey\ 4}$ (SE)	$\bar{p}_{Survey\ 5}$ (SE)
1	0.04 (0.39)	0.07 (1.1)	0.05 (0.30)	0.04 (0.14)	0.04 (0.38)
2	0.04 (0.03)	0.06 (0.05)	0.05 (0.04)	0.05 (0.03)	0.05 (0.03)
3	0.04 (0.37)	0.07 (1.2)	0.04 (0.40)	0.04 (0.25)	0.05 (0.46)

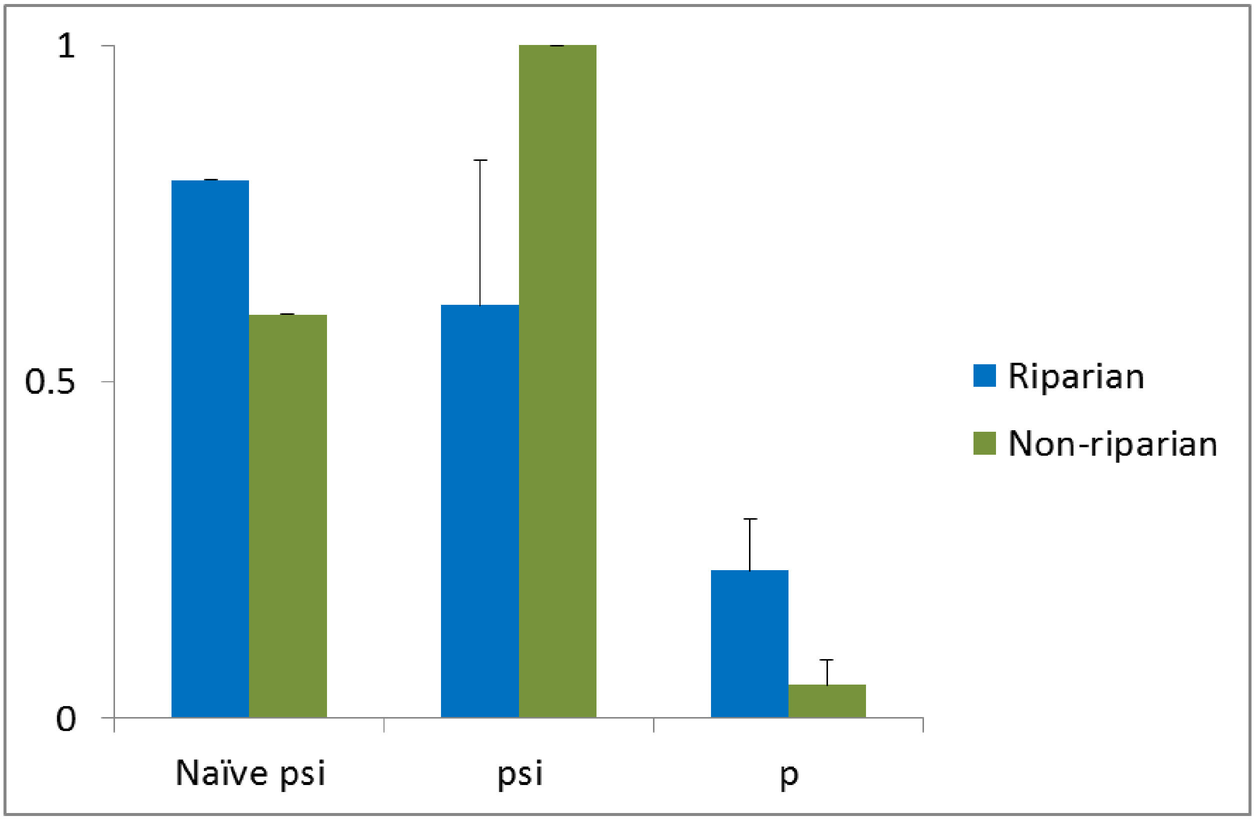

Figure 2.7 Naïve occupancy, mean occupancy (psi) and detection (p) probability estimates of little brown bats at riparian and non-riparian sites at a 4 x 4 grid of acoustic echolocation detectors at Fort Drum Military Installation, New York, summers 2011 and 2012.

Chapter 3: A comparison of passive and active acoustic sampling for monitoring a bat community impacted by white-nose syndrome

Abstract

Since white-nose syndrome (WNS) was documented in the northeastern U.S. in 2006, there has been a marked decline in bat abundance on affected species. Although labor-intensive and costly, summer mist-netting has been the standard technique to monitor bats. However, as bat populations have continued to decline in the Northeast, capture efficiency has decreased to a point that netting is no longer a cost effective methodology for inventorying bats. As a result, acoustical methodologies likely will become the primary means of monitoring declining or low bat populations on local landscapes. Therefore, the objective of this research was to compare active and passive acoustic monitoring for sampling a bat community that has experienced dramatic population declines due to WNS. In the summer of 2011, I used Anabat detectors to repeat passive monitoring at 18 sites previously sampled in 2006, pre-WNS, at Fort Drum Military Installation. I also performed active acoustic sampling at these sites and compared method results between the two types of surveys. In the summer of 2012, I conducted passive and active sampling at 13 additional sites in the Cantonment area of the installation for additional comparison. Single-season occupancy models revealed lower detection probabilities of all species using active sampling versus passive sampling. Additionally, overall detection probabilities declined in WNS-impacted species from 2006 to 2012 regardless of sampling method. A paired t-test revealed that overall recorded foraging activity per hour was greater using active sampling than passive for big brown bats (*Eptesicus fuscus;* t = -2.94, P = 0.006) and eastern red bats (*Lasiurus borealis;* t = -2.41, P = 0.023). There was no significant difference in recorded activity between methods for WNS-impacted species. This result presumably was

because these species have been so reduced in number that their apparency on the landscape is lower.

Introduction

White-nose syndrome (WNS) is a recently described fungal disease of cave-dwelling bats (Blehert et al. 2009). Since 2006, WNS has caused the deaths of an estimated 5.7 to 6.7 million bats (USFWS 2013) and has rapidly spread from northern New York to at least 22 states and 5 Canadian provinces (Figure 3.1). In the context of severe population declines following WNS, the use of acoustic monitoring to track changes in population sizes and species assemblages during the summer maternity season is likely now more effective than the use of mist nets. Individuals and species of bats are not equally vulnerable to capture on a given night (O'Farrell et al. 1999), and an even smaller percentage of bats are now susceptible to physical capture on a given night due to severe declines. As WNS continues to spread and bat populations decline further, biologists will likely need to rely more on acoustic detection, as a matter of necessity, as the primary method of monitoring bat presence on the landscape.

Acoustic monitoring is a non-invasive sampling technique to quantify activity of echolocating bats. These methods have become routine for investigating bat ecology, species assemblages and relative abundance on landscapes and relative to land management (Johnson et al. 2002, Milne et al. 2004, Ford et al. 2011, Johnson et al. 2011). Traditional monitoring protocols typically rely on capture methods (netting) at roosts, water sources, or along flyways. Capture techniques are only able to sample an extremely small portion of the area used by free-flying bats, missing high flying or alert bats (Murray et al. 1999, O'Farrell and Gannon 1999, MacCarthy et al. 2006), thereby resulting in biased samples and incomplete documentation of species assemblages. Previous research suggested that acoustics and mist netting detected similar levels of bat activity (Kunz and Brock 1975). However, improved acoustic detectors and call identification software programs facilitate the identification of greater overall species richness in less time and over greater spatial extent than traditional capture methodologies. For example, O'Farrell and Gannon (1999) detected 86.9 % of the total possible species inventory using frequency-division acoustical detectors versus 63.5% by netting only in the Southwest, and Murray et al. (1999) detected overall greater species richness using Anabats than mist nets in Missouri. Other research has suggested that using capture methodologies in combination with acoustic detectors enhances the likelihood of detecting target bat species and produces greater richness estimates than with either method alone, as some species are differently vulnerable between methods (Patriquin et al. 2003, Flaquer et al. 2007, Robbins et al. 2008). For example, some species such as the northern bat are particularly difficult to record acoustically due to their low intensity call amplitudes (Broders et al. 2004). However, in areas severely impacted by WNS, the need for very high levels of sampling effort to detect declining species makes multiple sampling methodologies logistically and fiscally prohibitive.

Anabat acoustic detectors (Titley Electronics, Ballina, New South Wales, Australia) are a type of frequency-division detector that has been used to evaluate species-specific habitat use and foraging activity (Betts 1998, O'Farrell and Gannon 1999, Britzke et al. 2002, Britzke 2003, Ford et al. 2006). Data are recorded to compact flash-storage Zero-Crossings Analysis Interface Modules (ZCAIM) for subsequent visual or computer analysis. Acoustic monitoring methodologies are categorized as either active or passive sampling. Active sampling refers to surveying where the user is present and able to change the orientation of the microphone relative to bats for optimization of echolocation pass recording (Britzke 2002). Active sampling typically records high quality passes because contact can be maintained between microphone directionality and bats. Passive sampling refers to automatic recording of passes without a user present, with the microphone fixed in a predetermined direction. Passive sampling designs often include multiple sampling stations that are spaced across a wider spatial scale. This method typically records lower quality passes than active sampling due to the fixed directionality of the microphone and climatic influences on the recording system. Both techniques have been widely tested in the field (Johnson et al. 2002, Britzke 2003, Milne et al. 2004, Brooks and Ford 2005, Britzke et al. 2011), but no direct comparisons of active and passive sampling have been made in the context of WNS-impacted landscapes, such as the Northeast.

Nine species of bats have been documented at Fort Drum (Table 3.1). These include the following species impacted by WNS: the big brown bat (*Eptesicus fuscus*), the federally endangered Indiana bat (*Myotis sodalis*), the petitioned eastern small-footed bat (*M. leibeii*), the little brown bat (*M. lucifugus*), the petitioned northern bat (*M. septentrionalis*), and the tri-colored bat (*Perimyotis subflavus*). The silver-haired bat (*Lasionycteris noctivagans*), the hoary bat (*Lasiurus cinereus*), and the eastern red bat (*L. borealis*) occur at Fort Drum but are not believed to be impacted by WNS. As a result of a previous, comprehensive monitoring program at Fort Drum, the basic assemblage of bats, their activity relative to habitat associations, and the impacts of WNS are known (Ford et al. 2011). Analyses of passive acoustic activity patterns showed significant declines in overall summer foraging activity in little brown bats, northern bats, and Indiana bats, as well as a decline in foraging activity from early to late summer season in post-WNS years— indicating a probable decrease in reproductive success of surviving individuals and subsequent lack of juvenile recruitment in little brown bats.

As WNS continues to spread and similar declines as observed at Fort Drum become apparent elsewhere, monitoring of impacted species will become increasingly difficult, yet exceedingly important for management and regulatory purposes. An important step in developing a monitoring protocol for declining bat species in a post-WNS impacted area is determining whether active or passive sampling is more efficient. Tradeoffs exist between high quality pulses and shorter sampling durations of active sampling and the ability to conduct widespread simultaneous sampling with multiple detectors during passive sampling (Britzke 2002). The objective of my study was to compare passive and active acoustic sampling designs to determine the more efficient and effective method to collect high quality calls and determine presence/absence of bat species at Fort Drum.

Study Area

Fort Drum Military Installation is a U.S. Army installation of approximately 43,000 ha in Jefferson and Lewis counties in northern New York (44°00'N, 75°49'W; Figure 3.2). The installation lies at the intersection of the St. Lawrence-Great Lakes Lowlands, the foothills of the Adirondack Mountains, and the Tug Hill Plateau ecoregions within the Black River and Indian River drainages. The nearby Niagara Escarpment (10-15 km west of Fort Drum) contains karst formations and caves that Indiana bats are known to use for winter hibernation (Fenton 1966). Approximately 57% of the landscape is made up of forested habitat dominated by northern hardwood types such as sugar maple (*Acer saccharum*), American beech (*Fagus grandifolia*), white ash (*Fraxinus americana*), black cherry (*Prunus serotina*) and white pine (*Pinus strobus*). Early successional understory habitat consists of red maple (*Acer rubrum*), gray birch (*Betula populifolia*), and quaking aspen (*Populus tremuloides*) regeneration. Wetland systems such as wet meadows and beaver (*Castor canadensis*) impacted streams and ponds make up 20% of the landscape. Development is concentrated in the Cantonment area, with the remainder of the installation consisting of 18 training areas, an airfield, and a large, centralized main impact zone that are all largely undeveloped.

Methods

During the summers of 2011 and 2012, I collected acoustic data at 31 wetland and forested corridor sites across Fort Drum; 18 of these sites were in training areas in July-August 2011 and 13 sites were in the Cantonment Area in June-August 2012. I used Anabat II detectors connected to a compact flash-storage Zero-Crossings Analysis Interface Module, as well as the SD1 and SD2 units (Titley Electronics, Ballina, New South Wales, Australia). I calibrated all units using an ultrasonic insect deterring device following the methods of Larson and Hayes (2000) prior to use in the field.

As part of the long term acoustic survey that was initiated at Fort Drum in 2003, 15-20 sites were sampled annually in pre-WNS years (2003-2007), and have since been revisited on a 5-year basis to document activity trends in the post-WNS context. For example, sites that were originally sampled in 2003 were revisited in 2008, 2004 sites were resampled in 2009, and so forth. In 2011, I was able to resample the 18 sites that were originally sampled in 2006. Due to military training restrictions, sites that were originally sampled in 2007 could not be replicated during 2012, therefore, I chose 13 additional sites in the Cantonment area to compare active and passive sampling.

I conducted passive and active sampling at each deployment site. For passive sampling, I placed Anabat units in weatherproof boxes with polyvinyl chloride (PVC) tubes attached that contain a small weep hole in the bottom for water drainage according to the methods of O'Farrell (1998). Boxes were placed on 1.5 m tripods aligned in a manner that allowed sound to enter the PVC tubes at a 45 degree reflective angle to be received by Anabat transducers perpendicularly

(Britzke et al. 2010). I set Anabats to a timer to record data continuously from approximately 1900 to 0700 hours for 2-3 sampling nights. For active sampling, I hand held Anabat detectors for 30 minutes, pointing and sweeping the microphone towards areas of expected activity (Ford et al. 2005). Multiple sites were sampled in a random order each night beginning at sunset and ending no later than 0200 hours. Each site was sampled using active methods 2-3 times throughout a season, with at least one active survey per site occurring simultaneously to passive sampling. I downloaded data nightly onto a laptop computer using the CFCread program (Titley Electronics, Ballina, New South Wales, Australia).

I used EchoClass (U.S. Army Engineer Research and Development Center, Vicksburg, MS, USA), an automated analysis program currently in final development, to identify bat calls to the species level. Although the ability to identify bat calls to the species level has been criticized (Barclay 1999), research has suggested that good quality calls of eastern North American bats can be identified both qualitatively (O'Farrell et al. 1999) and quantitatively (Britzke et al. 2002, Britzke et al. 2011). To minimize the impact of species identification when accuracy is less than 100%, EchoClass provides a maximum likelihood estimate which allows the user to determine the probable presence or absence of a species with predetermined levels of accuracy (Britzke 2002). In this study, I considered species of bats to be present at a site if the maximum likelihood values estimate for an individual species' identified call was ≥ 90%.

I created nightly presence-absence detection histories from the acoustic data (Gorresen et al. 2008). I considered each nightly survey independent due to the separation of sites and break in sampling during daylight hours. For each species, I attempted to fit single-season, single-species occupancy models to determine constant detection probability and unbiased occupancy estimates (MacKenzie et al. 2002) over each sampling type and year using program PRESENCE (version 2.4, Hines and Mackenzie, 2008). Typically, it is preferred that a candidate set of models are attempted and that models are ranked using Akaike's Information Criterion (AIC) and compared using Akaike weights (Burnham and Anderson 2002). However, because I was only interested in constant, mean detection and occupancy estimates and did not measure for potential site or sampling covariates, I chose to run a single model for each species at each combination of sampling type and year. Rather than running a single, multiple-season model for each species using sampling covariates, I separated models because the same sites were not replicated between years. I ran a total of 5 models on each species: 2006 passive, 2011 passive, 2011 active, 2012 passive, 2012 active.

In addition to determining detection probability and occupancy estimates, I determined the relative foraging activity for each species for both sampling types and years, standardized as passes per hour. I defined a "pass" as an acoustic file that contains 3 or more bat sound pulses. I ran a paired *t*-test to discern differences in recorded relative activity between active and passive sampling. I used SAS statistical software (Version 9.2; SAS Institute 2012, Cary, NC, USA) and defined statistical significance at $\alpha = 0.05$.

Results

I detected 7 of the 9 possible species that occur at Fort Drum during the 2011 and 2012 monitoring seasons. Eastern small-footed bats and tri-colored bats were not detected during my study, and therefore could not be adequately incorporated into the occupancy models. I determined estimates of occupancy and detection probabilities for the remaining 7 species (Table 3.1). Overall passive detection yielded higher detection probabilities than active sampling (Figures 3.3 and 3.4). Additionally, there was a declining trend in detection probabilities in WNS-impacted species across years for both sampling types (Figure 3.3-3.4). Occupancy estimates for the little brown bat, the Indiana bat, and the northern bat were higher using active sampling than passive sampling in both years, but the probability of detecting these species was substantially higher using passive sampling. Additionally, although occupancy estimates were similar between sampling methods for big brown bats and species not impacted by WNS, higher detection probabilities were usually achieved using passive sampling (Table 3.2).

From comparisons on mean hourly activity, I found that sampling method had a significant effect on number of passes recorded for eastern red bats ($t = -2.41$, $P = 0.023$) and big brown bats ($t = -2.94$, $P = 0.006$). Higher relative activity was detected for both eastern red bats and big brown bats using active sampling across both sampling years (Table 3.3). There was not a significant difference in relative activity recorded by passive or active sampling for other species. Observed relative activity was generally lower in WNS-impacted species than non-impacted species using either method of sampling in both years.

Discussion

Although previous research has suggested that active sampling produces higher quality calls, greater total number of calls, and higher species richness than passive sampling (Britzke 2002, Johnson et al. 2002, Milne et al. 2004), the results of my study suggest active sampling provides lower detection probabilities and performs no better in recording relative activity patterns than passive sampling for WNS-impacted myotid bats at Fort Drum. Although passive and active acoustic sampling using Anabat detectors have been widely used, previous work has compared measures of relative activity but not the more critical measures such as detection probabilities. Johnson et al. (2002) in northwestern Georgia and Milne et al. (2004) in Australia report that active sampling was superior to passive sampling in detecting higher species richness, detecting high flying bats, and recording better quality calls, particularly for myotid bats.

However, active and passive methods in these studies were performed simultaneously and for the same duration of time, thereby not allowing passive sampling to display its inherent “duration” advantage over active sampling. Furthermore, Johnson et al. (2002) recorded active acoustics directly to a laptop and passive acoustics to a tape recorder system for comparison. I deployed passive detectors for much longer periods than active detectors in a given night and for several nights in a row. Additionally, active and passive acoustics were both recorded to compact flash storage media in my study. Therefore it is difficult to directly compare previous work to my study due to differences in methodologies.

Although there was no difference in species richness detected by the two sampling methods over the duration of my study, passive sampling consistently yielded higher observed detection probabilities than active sampling for all species, despite potentially higher occupancy estimates for some species using active sampling. MacKenzie et al. (2002) suggest that occupancy estimates very close to 1 should be cautiously interpreted if obtained when the detection probability is < 0.15 or when sampling occurs over fewer than 7 events. Such estimates are based on low amounts of presence data, making it difficult for the model to distinguish between true absences and false non-detections. This is the case herein when active sampling was used for all WNS-impacted myotids, as well as for hoary bats in both years and in 2012 for silver-haired bats. Additionally, MacKenzie and Royle (2005) and Gorresen et al. (2008) suggest that when occurrence of a species is low, survey designs should employ as many sites as possible at the expense of repeated visits to the same sites. In my study, 31 unique sites were sampled over 2 years, yet occupancy estimates for active sampling were not reliable, suggesting that even higher levels of effort are needed to successfully monitor for rare species at my study site using active sampling. Passive sampling produced the highest detection probabilities and, therefore, the most robust estimates of occupancy for the majority of species in my study.

Unfortunately, because of severe WNS-induced population declines (Ford et al. 2011, USFWS 2013), some species (e.g., Indiana bat, little brown bat, and northern bat) are unlikely to be easily detected when present regardless of which method is used. Furthermore species such as northern bats are particularly difficult to record acoustically due to their low intensity call amplitude (Broders et al. 2004). Finally, some species were not detected acoustically at all, such as the eastern small-footed bat and the tri-colored bat. Although these species have been captured previously (Table 3.1; Fort Drum, unpublished data), capture rates have always been low at Fort Drum and it is uncertain whether they have ever been numerous or continue to occupy the landscape, post-WNS. Weller (2008) suggests that data-deficient species that are rare or difficult to detect may require > 50% more sampling effort than common species using acoustic monitoring. Initial equipment costs for active and passive sampling are similar, but the cost per hour of data is substantially higher for active than passive sampling (Table 3.4). For rare species that require rigorous sampling effort, the ability to implement passive monitoring over much longer periods and simultaneously at multiple sites is clearly more cost efficient than intensive active sampling. Although occupancy estimation of rare or declining species requires the most effort, and is hence the most costly (Weller 2008), acoustic monitoring as a tool for estimating occupancy and detection probability likely is still more effective and cheaper than traditional capture methodologies at Fort Drum. For example, Indiana bats were encountered within two nights of passive acoustic sampling whereas no Indiana bats were captured using netting around known historic maternity areas until the thirty-fifth net night in 2012 (Coleman et al. 2013). Furthermore, passive sampling produced overall higher detection probabilities in this study for all species that were successfully detected. Accordingly, I would recommend passive sampling as the preferred method for detecting species acoustically not only in areas that have observed severe post-WNS declines similar to those at Fort Drum, but in areas interested in monitoring for non-impacted species, as well.

It is not surprising that overall higher relative activity was recorded using active sampling for eastern red bats and big brown bats given what is known about their population stability relative to WNS and their ubiquitous foraging habitat use (Shump and Shump 1982, Kurta and Baker 1990). Eastern red bats are not currently known to be impacted by WNS, and big brown bats have not exhibited drastic declines as myotid species either locally or regionally (Ford et al. 2011, USFWS 2013). Moreover, the aforementioned detection probability assessment shows that both species are relatively easy to detect using either method relative to the myotids. My results are consistent with the findings of Weller (2008) that occupancy estimate studies are most effective for common species, as both eastern red bats and big brown bats appear to be relatively common on the landscape and can be easily detected using either sampling technique. Finally, it is somewhat surprising that WNS-impacted myotids did not demonstrate measured effects based on sampling method or year. However, these species are now detected so infrequently at Fort Drum that much greater sampling effort using either acoustic method may be required to determine changes in relative activity or adequately model these species with reliable parameter estimates and low standard errors.

For WNS-impacted species that are currently being considered for regulatory listing at the federal level, i.e., the little brown bat, the northern bat and the eastern small-footed bat (Kunz and Reichard 2010, CBD 2011), the ability to detect a species when present is critical from a regulatory perspective whereby managers can employ monitoring programs and mitigation activities to avoid or minimize potential take (ESA 1973, as amended). Detection is not certain even when a species is present at a site (MacKenzie et al. 2002), and in order to logistically comply with such regulations, a monitoring technique that can detect the most species under the most efficient time and effort conditions is necessary. Although differentiating between similar species acoustically is somewhat controversial (Barclay 1999), managers now have the ability to distinguish among northeastern myotids with high levels of accuracy using maximum likelihood estimates (Britzke et al. 2002, Britzke et al. 2011). In many WNS impacted areas where mist-netting now requires levels of effort for myotid species that are logistically and fiscally unfeasible, acoustic monitoring is likely the more efficient monitoring tool. Unfortunately, as WNS impacts compound, any technique will probably require far more effort (and potentially cost) than was previously required pre-WNS.

For managers hoping to detect high quality calls for species central to regulatory mandates with limited equipment, active sampling may seem like the most preferred sampling method, as call quality is inherently compromised in passive sampling. However, the cost per hour of data collected is substantially lower when passive sampling is used. Furthermore, leaving passive detectors deployed in several locations over longer periods may be crucial to successful monitoring for WNS-impacted species that are now rare on the landscape and, therefore, less likely to be detected in any given active sampling period. For certain species such as the tri-colored bat and the eastern small-footed bat that were not detected at all in 2011 or 2012, passive sampling is probably the better method to potentially produce results if these species do, in fact, still occupy the Fort Drum landscape. For most managers, passive sampling is easier to implement, requires less overall manpower because detectors can be deployed for multiple nights at a time, is ultimately more cost effective over longer durations, and may be more time efficient for more species than active sampling. It is apparent from this study that passive sampling far outweighs active sampling as a monitoring tool in the post-WNS landscape. Further investigations are now being implemented to determine the most cost-effective and efficient passive acoustic sampling design to target focal WNS-impacted species at Fort Drum to replace more costly traditional capture methods.

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Tables and Figures

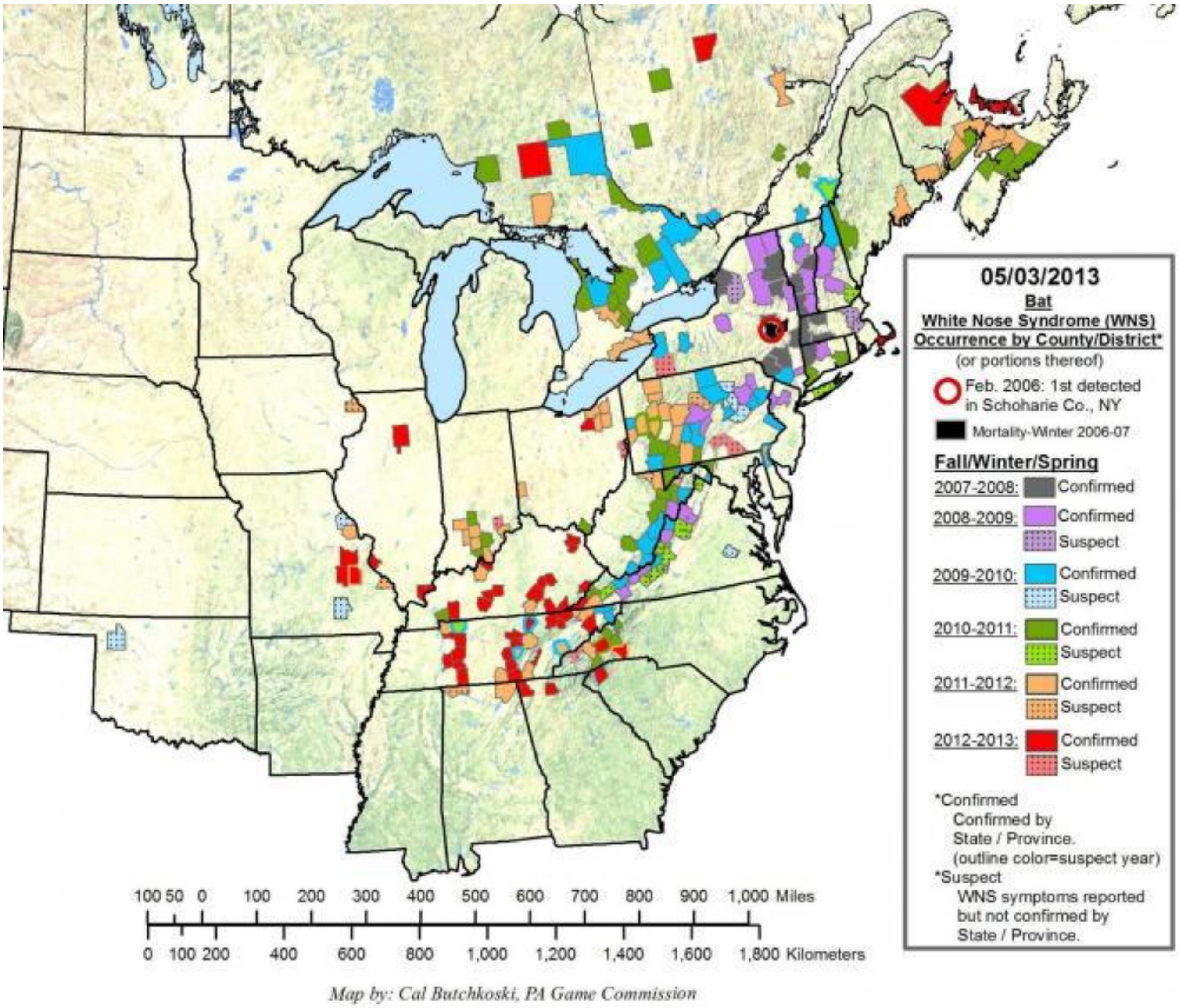

Figure 3.1 Distribution map of white-nose syndrome (WNS) in the eastern United States. Last updated 3 May 2013. Available: http://www.whitenosesyndrome.org/resources/map (USFWS 2013).

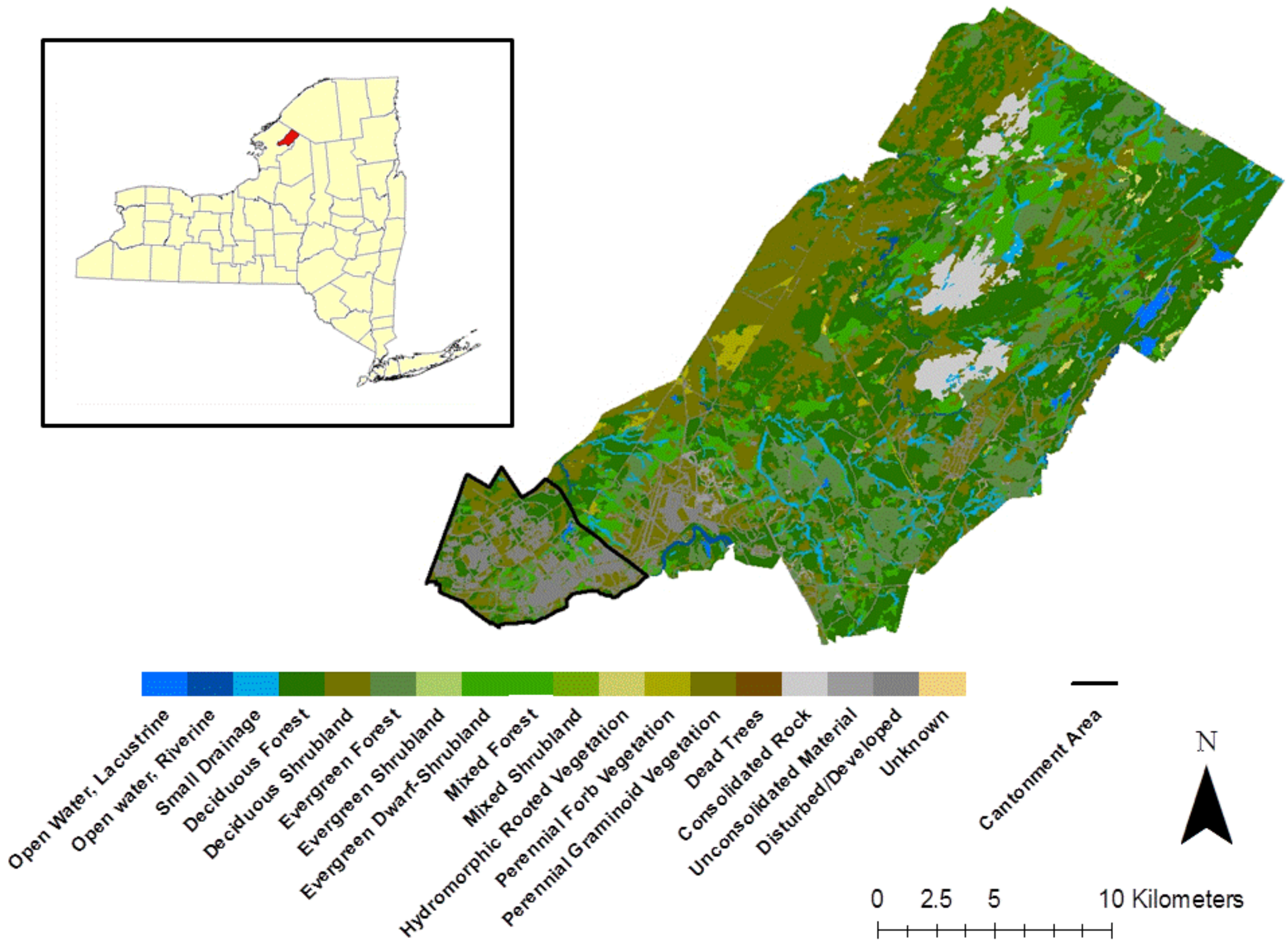

Figure 3.2 Fort Drum Military Installation, Jefferson and Lewis Counties, New York. Passive and Active sampling conducted outside the Cantonment Area in training areas in 2011 and inside the Cantonment Area in 2012.

Table 3.1 Summer mist-netting capture data for bat species occurring at Fort Drum Military Installation, New York 2007-2012; *n* = number of sites (C.A. Dobony, unpublished data).

Species	2007 (*n* = 81)	2008 (*n* = 41)	2009 (*n*=85)	2010 (*n*= 85)	2011 (*n*= 60)*	2012 (*n*= 10)
Silver-haired bat	4	3	4	5	2	0
Eastern red bat	62	14	32	89	72	42
Hoary bat	7	5	3	6	2	7
Big brown bat	574	215	311	486	364	36
Little brown bat	440	104	35	51	14	2
Indiana bat	18	2	0	2	1	1
Northern bat	260	37	5	5	1	0
Eastern small-footed bat	0	0	0	2	0	0
Tri-colored bat	4	0	1	1	0	0
Total	1369	380	391	647	456	88

*30 sites each repeated twice

Table 3.2 Detection probability (p) and occupancy (Ψ) estimates from single season, single species model for bat species at 31 sites of acoustic echolocation detectors at Fort Drum, New York, summers of 2011 and 2012.

Species	Year	Sampling Type	p (SE)	Ψ (SE)
Silver-haired bat	2006	Passive	0.60 (0.13)	0.93 (0.14)
	2011	Active	0.22 (0.18)	0.88 (0.63)
		Passive	0.62 (0.12)	0.86 (0.16)
	2012	Active	0.08 (0.05)	1.0 (0)
		Passive	0.62 (0.15)	0.49 (0.15)
Eastern red bat	2006	Passive	0.83 (0.06)	1 (0)
	2011	Active	0.30 (0.16)	0.78 (0.37)
		Passive	0.64 (0.12)	0.75 (0.15)
	2012	Active	0.50 (0.22)	0.62 (0.26)
		Passive	0.75 (0.09)	0.88 (0.11)
Hoary bat	2006	Passive	1.0 (0)	1.0 (0)
	2011	Active	0.24 (0.07)	1.0 (0)
		Passive	0.93 (0.05)	0.80 (0.10)
	2012	Active	0.12 (0.06)	1.0 (0)
		Passive	0.75 (0.10)	0.63 (0.14)
Big brown bat	2006	Passive	0.97 (0.03)	1.0 (0)
	2011	Active	0.67 (0.10)	0.83 (0.11)
		Passive	0.92 (0.06)	0.74 (0.11)
	2012	Active	0.29 (0.22)	0.94 (0.68)
		Passive	0.60 (0.11)	0.87 (0.13)
Little brown bat	2006	Passive	0.97 (0.03)	1.0 (0)
	2011	Active	0.02 (0.02)	1.0 (0)
		Passive	0.51 (0.13)	0.88 (0.22)
	2012	Active	0.04 (0.04)	1.0 (0)
		Passive	0.26 (0.21)	0.41 (0.32)
Indiana bat	2006	Passive	0.86 (0.06)	1.0 (0)
	2011	Active	0.02 (0.02)	1.0 (0)
		Passive	0.62 (0.17)	0.38 (0.15)
	2012	Active	0.04 (0.04)	1.0 (0)
		Passive	0.08 (0.05)	1.0 (0)
Northern bat	2006	Passive	1.0 (0)	0.89 (0.07)
	2011	Active	-	-
		Passive	0.39 (0.30)	0.10 (0.11)
	2012	Active	0.04 (0.04)	1.0 (0)
		Passive	0.03 (0.03)	1.0 (0)

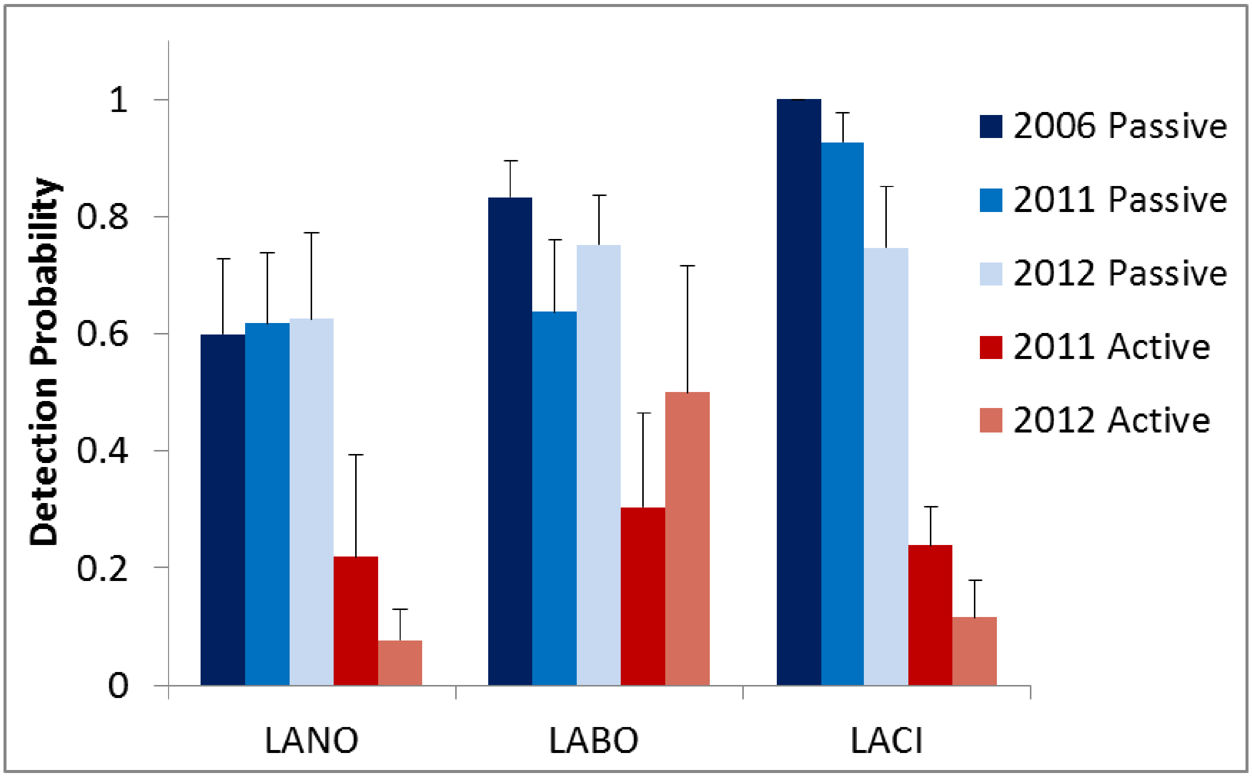

Figure 3.3 Detection probability estimates from single season, single species model for non-WNS-impacted bat species at acoustic echolocation detectors at Fort Drum, New York, 2011 and 2012; Species codes refer to first two letters of the genus and the first two letters of the specific epithet, i.e. LANO = *Lasionycteris noctivagans* (silver-haired), LABO = *Lasiurus borealis* (eastern red), LACI = *L. cinereus* (hoary).

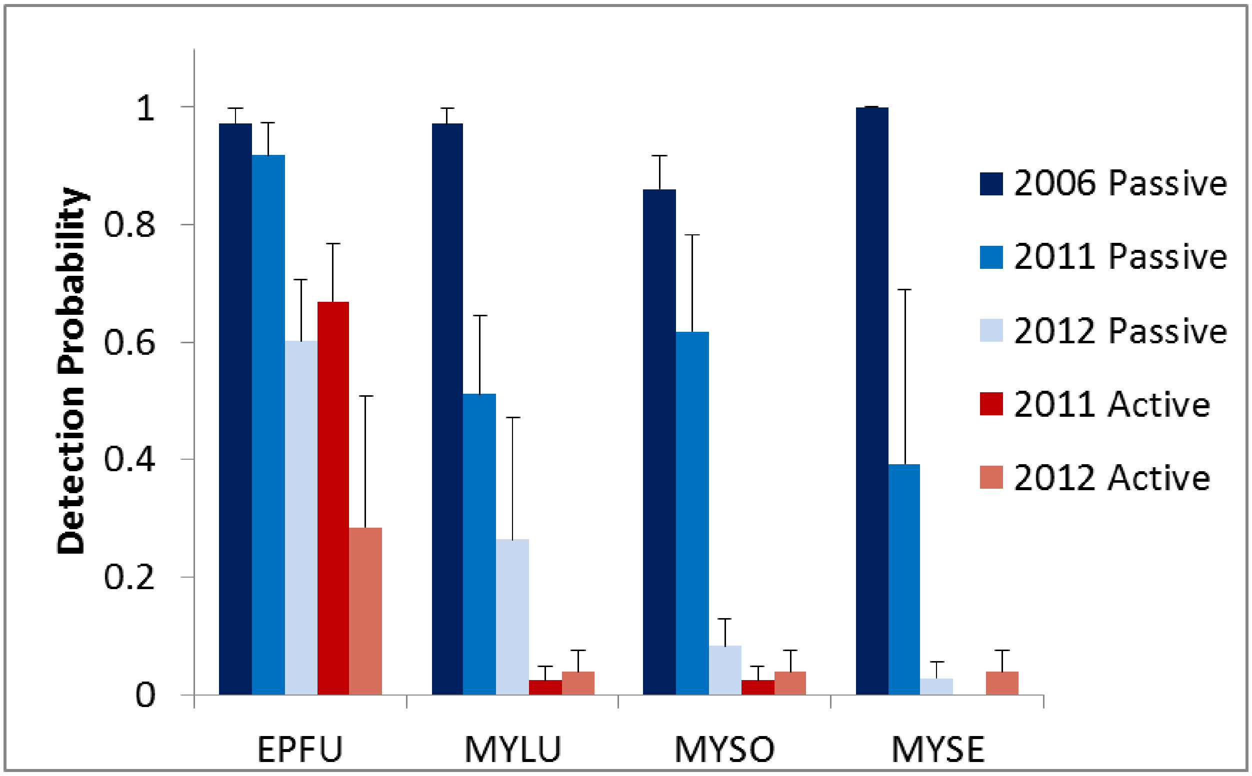

Figure 3.4 Detection probability (*p*) and occupancy (*Ψ*) estimates from single season, single species model for WNS-impacted bat species at acoustic echolocation detectors at Fort Drum, New York, summers of 2011 and 2012; Species codes refer to first two letters of the genus and the first two letters of the specific epithet, i.e. EPFU = *Eptesicus fuscus* (big brown), MYLU = *Myotis lucifugus* (little brown), MYSO = *M. sodalis* (Indiana), MYSE = *M. septentrionalis* (northern).

Table 3.3 Mean (SE) differences of passes per hour recorded using passive and active acoustic sampling of bat species at 31 sites at Fort Drum Military Installation, New York, summers 2011 and 2012.

Species	Mean $_{Passive\text{-}Active}$ (SE)	*t*-statistic	*df*	*P*-value
Silver-haired	-1.59 (1.04)	-1.53	30	0.139
Eastern red	-1.98 (0.82)	-2.41	30	0.023
Hoary	-0.81 (0.54)	-1.50	30	0.145
Big brown	-3.48 (1.17)	-2.94	30	0.007
Little brown	-0.23 (0.30)	-0.78	30	0.443
Indiana	-0.97 (0.14)	-0.71	30	0.482
Northern	-0.24 (0.25)	-0.98	30	0.336

Table 3.4 Estimated cost of 2 nights of summer sampling of bat species at Fort Drum Military Installation, New York.

Type of Sampling	Equipment	Equipment Cost	Gas Cost	Labor Requirement	Labor Cost	Total Cost	Hours of Data Collected	**Cost/Hour of Data**
Passive Acoustic	Anabat Weatherproofing Battery Tripod	$2200 $20 $35 $60	$30	15/hr x 5 hours	$75	$2420	20	**$121**
Active Acoustic	Anabat Batteries	$2200 $10	$30	15/hr x 6 hours	$90	$2330	2	**$1165**
Mist-netting (Contracted)	Mist-nets ($90 ea.) x 5	$450		1125/night	$2250	$2700	10	**$270**

Chapter 4: Effect of passive acoustic sampling array on bat detection in the context of white-nose syndrome-associated declines

Abstract

White-nose syndrome (WNS) has resulted in observed declines in abundance and activity of affected species. As those declines may result in increased regulatory scrutiny, managers are in need of improved data for bat inventory and monitoring efforts. Therefore, I used various sampling arrays of Anabat acoustic detectors to determine the most efficient passive acoustic sampling design for optimizing detection probabilities of multiple bat species. I concentrated sampling efforts in a historic Indiana bat maternity use area and within or near the foraging area of a local little brown bat maternity colony. My sampling protocol included six permanent stations that were deployed for the entire duration of monitoring, and a 4 x 4 grid, four transects, and one double transect of detectors that were deployed for 6-8 nights at a time. I used occupancy modeling in Program PRESENCE to determine which sampling arrays had the best detection probability estimates. Overall, the grid —probably because it contained the most detectors and intercepted the greatest spatial area—produced the highest detection probabilities for most bat species. However, big brown bats (*Eptesicus fuscus*) and species not impacted by WNS were easily detected regardless of sampling array. Indiana bats, little brown bats, and tri-colored bats (*Perimyotis subflavus*) showed declines in detection probabilities at all sampling arrays from 2011 to 2012, potentially indicative of continued WNS-associated declines for these species at Fort Drum.

Introduction

White-nose syndrome (WNS) is fungal disease of cave hibernating bats first documented in North America in 2006 (Blehert et al. 2009). Since its onset, WNS has caused the deaths of an estimated 5.7. to 6.7 million bats (USFWS 2013) and has rapidly spread from its origin of central New York to at least 22 states and 5 Canadian provinces (Figure 4.1). Pre-WNS individual bats and/or bat species were not equally vulnerable to capture on a given night, producing considerable bias associated with netting (O'Farrell et al. 1999). In the context of severe population declines, an even smaller numbers of bats are susceptible to physical capture on a given night. As WNS continues to spread and bat populations decline further, biologists invariably will rely more on acoustic detection, as a matter of necessity, as the primary method of monitoring bat presence on the landscape.

Acoustic monitoring is a non-invasive sampling technique that has become commonplace over the last decade for investigating bat ecology, species assemblages and relative abundance on landscapes and relative to land management (Johnson et al. 2002, Milne et al. 2004, Ford et al. 2011, Johnson et al. 2011b). Use of acoustic has allowed of the detection of greater overall species richness in less time and over greater spatial extent than traditional capture methodologies. For example, O'Farrell and Gannon (1999) detected 86.9 % of the total possible species inventory using frequency-division acoustical detectors versus 63.5% by netting alone in the Southwest, and Murray et al. (1999) detected overall greater species richness using frequency division detectors than mist nets in Missouri. Other research has suggested that using capture methodologies in combination with acoustic detectors enhances the likelihood of detecting target bat species and produces greater richness estimates than with either method alone as some species are differently vulnerable between methods (Patriquin et al. 2003, Flaquer et al. 2007, Robbins et al. 2008). However, the additive benfit of netting is minimized due to the very low detection probability using netting in a post-WNS landscape. In areas severely impacted by WNS, the need for very high levels of sampling effort to detect declining species logistically and probably fiscally prohibits the efficaciousness of using netting in tandem with acoustics.

Anabat acoustic detectors (Titley Electronics Ballina, New South Wales, Australia) are a type of frequency-division detector that have been used widely to evaluate species-specific habitat use and foraging activity (Betts 1998, O'Farrell and Gannon 1999, Britzke et al. 2002, Britzke 2003, Ford et al. 2006). Data are recorded to compact flash-storage Zero-Crossings Analysis Interface Modules (ZCAIM) for subsequent visual or computer analysis. Although Anabats have been field evaluated (Johnson et al. 2002, Britzke 2003, Milne et al. 2004, Brooks and Ford 2005, Britzke et al. 2011), no research has assessed multiple passive sampling arrays for detecting numerous species in the context of WNS-impacted landscapes.

Nine species of bats have been documented at Fort Drum. These include the following species impacted by WNS: the big brown bat (*Eptesicus fuscus*), the federally endangered Indiana bat (*Myotis sodalis*), the petitioned eastern small-footed bat (*M. leibeii*), the little brown bat (*M. lucifugus*), the petitioned northern bat (*M. septentrionalis*), and the tri-colored bat (*Perimyotis subflavus*). The silver-haired bat (*Lasionycteris noctivagans*), the hoary bat (*Lasiurus cinereus*), and the eastern red bat (*L. borealis*) also occur at Fort Drum but are not believed to be impacted by WNS. As a result of a previous, comprehensive monitoring program at Fort Drum, the basic assemblage of bats, their activity relative to habitat associations, and the impacts of WNS are known (Ford et al. 2011). Analyses of passive acoustic activity patterns showed significant declines in overall summer foraging activity in little brown bats, northern bats, and Indiana bats, as well as a decline in foraging activity from early to late summer season in post-WNS years— indicating a probable decrease in reproductive success of surviving individuals and subsequent lack of juvenile recruitment in little brown bats.

As WNS continues to spread, monitoring of impacted species will become more difficult, yet exceedingly important for management and regulatory purposes. It is clear that passive acoustic sampling is preferable to active acoustic sampling or mist netting for detecting WNS-impacted species. As managers begin to incorporate acoustical methods, questions remain about the most effective deployment strategies in terms of duration and configuration of multiple detectors. Accordingly, the objective of my study was to determine the most efficient and effective passive acoustic landscape sampling design for detecting bat species, with efforts focused at the federally-endangered Indiana bat and the little brown bat.

Study Area

Fort Drum Military Installation is a U.S. Army installation of approximately 43,000 ha in Jefferson and Lewis counties in northern New York (44°00’N, 75°49’W; Figure 4.2). The installation lies at the intersection of the St. Lawrence-Great Lakes Lowlands, the foothills of the Adirondack Mountains, and the Tug Hill Plateau ecoregions within the Black River and Indian River drainages. The nearby Niagara Escarpment (10-15 km west of Fort Drum) contains karst formations and caves that Indiana bats are known to use for winter hibernation (Fenton 1966). Approximately 57% of the landscape is made up of forested habitat dominated by northern hardwood types such as sugar maple (*Acer saccharum*), American beech (*Fagus grandifolia*), white ash (*Fraxinus americana*), black cherry (*Prunus serotina*) and white pine (*Pinus strobus*). Early successional understory habitat consists of red maple (*Acer rubrum*), gray birch (*Betula populifolia*), and quaking aspen (*Populus tremuloides*) regeneration. Wetland systems such as wet meadows and beaver (*Castor canadensis*) impacted streams and ponds make up 20% of the landscape. Development is concentrated in the Cantonment Area, with the remainder of the installation consisting of 18 training areas, an airfield, and a large, centralized main impact zone that are all largely undeveloped.

Methods

In the summers of 2011 and 2012, I deployed acoustic bat detectors across Fort Drum in various sampling arrays. I used Anabat II detectors connected to a compact flash-storage Zero-Crossings Analysis Interface Module, as well as the SD1 and SD2 units (Titley Electronics, Ballina, New South Wales, Australia). I calibrated all units using an ultrasonic insect deterring device following the methods of Larson and Hayes (2000) prior to use in the field. I placed Anabat units in weatherproof boxes with polyvinyl chloride (PVC) tubes attached that contain a small weep hole in the bottom for water drainage according to the methods of O'Farrell (1998). Boxes were placed on 1.5 m tripods aligned in a manner that allowed sound to enter the PVC tubes at a 45 degree reflective angle to be received by Anabat transducers perpendicularly (Britzke et al. 2010).

As a result of WNS declines, randomly selected acoustic monitoring stations may not successfully detect Indiana bats and little brown bats due to recent declines (Ford et al. 2011); especially the Indiana bat that previously exhibited a distribution restricted to the Cantonment Area at Fort Drum (Johnson et al. 2011a, C.A. Dobony, unpublished data). Therefore, I focused monitoring efforts on these species by placing arrays of detectors near a little brown bat maternity colony in an artificial bat house (Dobony et al. 2011) and known historic Indiana bat maternity areas (Johnson et al. 2011a). I deployed detectors to record passively at permanent stations that remained in the field for the entire summer season, linear stream transects of 5-10 detectors deployed for 6-8 nights at a time, and a 4 x 4 “grid” of detectors that also were deployed for 6-8 days at a time (Figure 4.3).

To ensure that more than one Anabat did not collect data on the same bat simultaneously, I separated detector sites by 200-250 m. The exception to this was my double transect where Anabats were pointed in opposite directions and data from both units at a site were combined. I chose deployment locations and the azimuth of microphone direction at each site to maximize call quality. For example, I targeted sites with plenty of open space such as canopy gaps, forested trails, or open water. I set Anabats to a timer to record data continuously from approximately 1900 to 0700 hours over 5 sampling periods, once in 2011 for each sampling array and 4 times in 2012, with the exception of the additional Indiana bat transects which were each only sampled twice in 2012. I changed batteries and memory cards as needed and downloaded data to a laptop computer using the CFCread program (Titley Scientific, Ballina, Australia).

I used EchoClass (U.S. Army Engineer Research and Development Center, Vicksburg, MS, USA), an automated analysis program currently in final development, to identify bat calls to the species level. Although the ability to identify bat calls to the species level has been criticized (Barclay 1999), research has suggested that good quality calls of eastern North American bats can be identified both qualitatively (O'Farrell et al. 1999) and quantitatively (Britzke et al. 2002, Britzke et al. 2011). To minimize the impact of species identification when accuracy is less than 100%, EchoClass provides a maximum likelihood estimate which allows the user to determine the probable presence or absence of a species with predetermined levels of accuracy (Britzke 2002). In this study, I considered species of bats to be present at a site if the maximum likelihood values estimate for an individual species’ identified call was ≥ 90%.

I created nightly presence-absence detection histories from the acoustic data for the 9 possible species at Fort Drum (Gorresen et al. 2008). I considered each nightly survey independent due to the separation of sites and break in sampling during daylight hours. Because double transect detectors may have recorded the same bats simultaneously, both Anabats at each site were considered a single unit for computing detection histories—i.e., if a species was detected by either Anabat at a site on a given night it was considered to be present at that site. For each species I attempted to fit a candidate set of 15 models to determine whether sampling habitat, year, or time of season (Table 4.1) impacted estimates of overall occupancy or detection (MacKenzie et al. 2002) using program PRESENCE (version 2.4, Hines and Mackenzie, 2008). I ranked models using Akaike's Information Criterion (AIC) corrected for small sample size and compared the weight of evidence among candidate models using Akaike weights (Burnham and Anderson 2002). Following those analyses and retaining the significant covariates for the top approximating models, I collapsed detection histories so that each entire sampling array was considered a single site for a given night. For example, if a species was detected at any of the 16 grid sites in a given night, it was considered present at the single representative grid site on that night. Because all sites in a particular sampling array were collapsed into a single representative site for the second set of models, I adjusted the covariate for habitat to a continuous variable representing the percent of wet sites in an array. I applied these covariates as a starting point for a single, additional model for each species to determine detection probability estimates among the varying sampling arrays. Included in the covariate set for sampling arrays were permanent, grid, transect, and double transect sites. If the anticipated model failed to converge, I attempted to remove covariates from the previous candidate set in a step-wise fashion until the best possible model was reached to describe detection probabilities based on sampling array. Finally, for WNS-impacted species, I calculated the effort required (in sampling nights) to determine true absence based on detection probabilities derived from occupancy models following the methods of McArdle (Figure 4.5; 1990).

Results

I detected all 9 species that occur at Fort Drum at least once during the 2011 and 2012 monitoring seasons. I removed one permanent site that was included in the grid from detection histories for each species due to its close proximity to the artificial little brown bat house (Figure 4.3) because of high sound distortion caused by multiple bats being detected simultaneously. I was able to fit candidate models for all species to determine impacts of habitat, year, and time of season on occupancy and detection probability estimates (Table 4.2).

Habitat had an effect on occupancy estimates for silver-haired, big brown, and hoary bats, with higher occupancy rates at wet versus dry sites (Table 4.2). Habitat also had an effect on eastern small-footed bat occupancy, but the opposite trend was observed. Additionally, year affected occupancy estimates for hoary bats which increased from 2011 to 2012, whereas tri-colored bats had a decrease in occupancy estimates from 2011 to 2012. Habitat affected the detection probabilities of all 9 species with higher detection probabilities estimates always observed at wet sites versus dry sites. Year had a potential effect on eastern red, little brown, Indiana, eastern small-footed, and tri-colored bats which all exhibited declines in detection probabilities from 2011 to 2012, except for the eastern small-footed bat that was detected at a slightly higher rate in 2012. Finally, time of season had a potential effect on detection probabilities of eastern red, northern, and eastern small-footed bats that all increased in the late, post-volancy season.

For my array-specific detection histories, I created additional models to determine changes in detection probability based on sampling array for 7 species (Table 4.3). I was unable to successfully model northern and eastern small-footed bats due to very low detection data for these species. The percent-wet habitat covariate for the collapsed representative sites of permanent locations and the grid were 60% and 40%, respectively. Transects and the double transect were fully located at wet sites (100%).

Silver-haired bats were widely detected regardless of sampling array, year, or season. However, the grid of detectors produced the highest detection probability estimates (Figure 4.4). Eastern red bats were also widely detected regardless of sampling array. The highest detection probability estimates were recorded at the grid, and an increase in detection probability was recorded at all other sampling arrays from early to late season. Hoary bats were also recorded with the highest detection probability estimates at the grid relative to other sampling arrays and were generally detected with high probabilities regardless of array, year, or season.

Big brown bats were widely detected at all sampling arrays, but were recorded with the highest detection probability at the double transect regardless of year and season (Figure 4.4). Both little brown bats and Indiana bats were detected most at the grid compared to other array types. Additionally, both of these species showed marked decreases in detection probability from 2011 to 2012. Finally, the tri-colored bat that was detected at transects, the grid, and permanent stations in both years was not detected at the double transect in either year. Additionally, this species showed drastic declines in detection probability estimates from 2011 to 2012 at all sampling arrays that it was detected.

Discussion

Using occupancy modeling and detection probability estimates to assess bat species assemblages, relative activity, and habitat use with Anabat acoustic detectors has become an increasingly used analytical approach over the last decade. Yates and Muzika (2006) reported an effect of year on Indiana bat detectability as in my study. However, an increase in Indiana bat detectability was observed over time in the pre-WNS Ozark Mountain region of Missouri, in contrast to my study. Hein et al. (2009) reported an increase in activity of eastern red, little brown, and tri-colored bats with increasing time throughout the season in the South Carolina Coastal Plain. In my study, I observed a slight increase in the detectability of eastern red bats in the late season but did not observe an effect of time of season for little brown or tri-colored bats. Finally, Weller (2008) used occupancy modeling as a monitoring tool for assessing the effectiveness of a multiple-species conservation plan in the Pacific Northwest finding that occupancy modeling to relate species presence to habitat factors is most effective for common species. However, he concluded it was difficult to assess relationships for very rare species unless very high levels of sampling are to be implemented. His findings are consistent with my findings for eastern small-footed and northern bats that I was unable to model due to very low detection data.

Although previous research has focused on occupancy modeling for multiple bat species conservation and management, no prior studies have assessed the differences in multiple species detectability at various passive acoustic sampling arrays in the context of WNS-associated declines. Overall, detection probabilities were the highest in my study for most species at the grid of detectors, regardless of year or time of the season. Each array type was represented by a unique value for percent-wet. Therefore, my simple characterization of habitat did not have an impact on the probability of detecting a species at a particular array type, although it may have influenced the probability of detecting particular species at one array type over another when it was included as a covariate. Indiana, little brown, and tri-colored bats were always detected at the highest probabilities at the grid of detectors despite overall declines from 2011 to 2012. However, species not impacted by WNS and the big brown bat were detected with high probabilities regardless of the sampling array, year, or time of season and should presently be expected to be detected easily under most sampling circumstances at Fort Drum.

Although the grid generally produced the highest detection probabilities, it is important to consider the varying levels of effort that were required for each array. The grid required the highest number of available Anabat units relative to other arrays at 15 units, followed by the double transect requiring 10 units, and the permanent and transect sites that each required five units at a time. For many managers, limited fiscal resources may prohibit the use of greater than five units at once. In such cases, permanent stations or transects may be feasible alternatives. However, MacKenzie and Royle (2005) suggest that the optimal strategy for detecting rare species in a sampling program is to focus on a greater number of sites at the expense of multiple surveys at the same site. Gorresen et al. (2008) confirmed the same optimal strategy on their study of hoary bats in Hawaii, showing that reasonable precision of most parameter estimates were not achieved until 15 sites were sampled.

In my study, permanent sites did not detect little brown bats any better and only detected Indiana bats slightly better than transect sites. Permanent sites were never moved to different locations, whereas transect sites were placed repeatedly along four different streams at different time intervals. Permanent stations were only deployed and removed at the beginning and end of the season and were revisited once approximately every 10 days for memory card and battery changes. Transect deployment and removal required much more effort than permanent stations, but batteries and cards did not need to be exchanged during recording cycles. Additionally, a much greater area of the landscape was sampled using this method. The flexibility in sampling a greater number of sites over a larger area in less time using transects probably makes these the preferable method for high detectability for multiple species on most landscapes. Finally, moving detectors to multiple locations in various habitats offers the potential to assess changes in occupancy at a greater variety of habitat types across the duration of a season than is possible with stationary units.

For focused studies on Indiana bats, benefits of long sampling durations and covering a wide area may be necessary for optimally determining presence or probably absence when there are not enough available detectors to implement a grid. In such cases, deploying detectors for greater than 6-8 days along stream transects of expected foraging use may be a feasible alternative. For example, based on the detection probabilities derived from my study, true absence of Indiana bats can be determined at the grid in 2-3 sampling nights or in 8-12 sampling nights using transects (McArdle 1990). Similar patterns may be observed for determining probably absence of little brown bats. For tri-colored bats, the grid and the transect could both determine probable absence in 2-3 sampling nights relative to detectability of this species in 2011, but a much longer sampling duration would be required based on detection estimates from 2012.

Although longer-duration transects may be adequate for detecting Indiana bats when equipment is limited, the double transect detected Indiana bats second best to grids. Duchamp et al. (2006) suggested that having two detectors at a site increased the probability of detecting bat species in Indiana and Missouri, consistent with my findings for Indiana bats when comparing the double transect to other transects. However, for little brown bats, the double transect was no better than permanent stations or regular transects. This may indicate that although little brown bat are declining at Fort Drum (Ford et al. 2011), their foraging distribution in terms of spatial extent may not be as limited or altered as other WNS-impacted species. For example, tri-colored bats were never detected at the double transect, despite easy detectability of this species at other arrays in 2011. Finally, the big brown bat was detected more at the double transect than at other arrays. Previous research has suggested that this species is probably much less impacted by WNS, and therefore, may be more easily detected than myotid species and tri-colored bat at Fort Drum (Ford et al. 2011). Thus, it is intuitive that transects with double the sampling effort detected this species at a greater rate than other transects.

Perhaps the most important factor that was considered in the design of my study was to focus efforts in known historic areas of the targeted species. As a species that was once found in colonies of hundreds to thousands of individuals in the summer across their range (Davis and Hitchcock 1965) and at Fort Drum (Dobony et al. 2011), little brown bats are now rarely observed in the Northeast, as many documented colonies have collapsed (Frick et al. 2010, Dzal et al. 2011, Turner et al. 2011). Indiana bats have also exhibited population declines regionally and locally as a result of WNS mortality (Ford et al. 2011, USFWS 2013) and were known to have a distribution restricted to the Cantonment Area on the Fort Drum landscape even pre-WNS (Johnson et al. 2011a). Therefore, randomly selected locations are unlikely to be suitable for detecting these species at Fort Drum—and potentially elsewhere—when severe declines have been observed. Indiana bats and little brown bats were detected with high probabilities at all sampling arrays and at much higher detection rates than tri-colored, northern, and eastern small-footed bats, probably because efforts were focused in known historic maternity areas for these species in this study. On landscapes where historic capture information is not available, efforts can be focused at these species by homing in on areas with suitable habitat for these species or other target species.

Although the Indiana bat is the only species at Fort Drum that is presently subject to regulatory mandates of the Endangered Species Act (ESA 1973, as amended), northern, eastern small-footed, and little brown bats are under review by the United States Fish and Wildlife Service to assess their candidacy for federal listing as a result of WNS-associated declines (Kunz and Reichard 2010, CBD 2011). Furthermore, the tri-colored bat is believed to be one of the most severely impact species by WNS (Turner et al. 2011, USGS 2012) and may be subject to status reviews or listing proposals in the near future. Although the dramatic drop in detection probabilities for tri-colored bats from 2011 to 2012 in my study may be due to year to year sampling variation, it is a potential conservation concern that warrants further investigation. If any of these species are listed in the future, the ability to detect them when present will be critical from a regulatory perspective whereby managers can employ monitoring programs and mitigation activities to avoid or minimize potential take (ESA 1973, as amended). Because detection data for eastern small-footed and northern bats was so low in my study, I was unable to adequately fit models to determine the sampling array that produced the highest detection probabilities for these species. Northern bats are not as detectable due to their low intensity call amplitude (Broders et al. 2004). Additionally, although mist-net capture rates for eastern small-footed and tri-colored bats have always been captured at low rates, all three of these species have been scarcely observed since WNS arrived at Fort Drum (C.A. Dobony, unpublished data). Weller (2008) suggested that data-deficient species that are rare or difficult to detect may require > 50% more sampling effort than common species using acoustic monitoring. Future efforts to assess optimal sampling conditions for northern bats, eastern small-footed bats, and tri-colored bats are needed and perhaps should be focused in known historic areas of use to determine if these species still occupy the Fort Drum landscape following WNS declines.

For managers seeking an optimal strategy for determining presence or probable absence of Indiana bats or little brown bats, my study suggests that a grid of detectors in an expected area of use is most effective, possibly due to its inherent ability to survey a wider spatial area than the other methods presented here. The grid of detectors was able to simultaneously detect both of my target species as well as other species in the bat community at Fort Drum. However, in situations where high numbers of Anabats are not available, the other sampling arrays presented here may be viable options for detecting Indiana and little brown bats when placed in areas of anticipated use. For example, if a manager only has five detectors, deploying transects for 8-12 nights in various locations throughout an anticipated area of use may be sufficient. With 10 detectors, efforts could be potentially be lessened to fewer nights of sampling. Furthermore, with 10 detectors, managers may be able to capture a wider spatial and temporal distribution of sampling in less time if multiple transects are implemented simultaneously and repeated throughout the season. A grid using fewer than 15 detectors could also capture detection over a wide spatial area, if detectors are placed at greater than 250 meters apart as in my study.

There are countless options for how detector placement can be implemented to determine presence or probable absence of target species while capturing spatial and temporal use of a landscape. However, deploying detectors across the widest area possible in areas of know previous use, expected use, or suitable habitat is likely more effective than deploying detectors for very long periods at permanent stations or in random locations. Presently, it is unknown whether a single sampling array exists that can adequately detect all WNS-impacted species that occur at Fort Drum or in the wider Northeast region. Sampling efforts for additional WNS-impacted species warrant further investigations, particularly for those species that may be proposed for federal listing in the near future.

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Tables and Figures

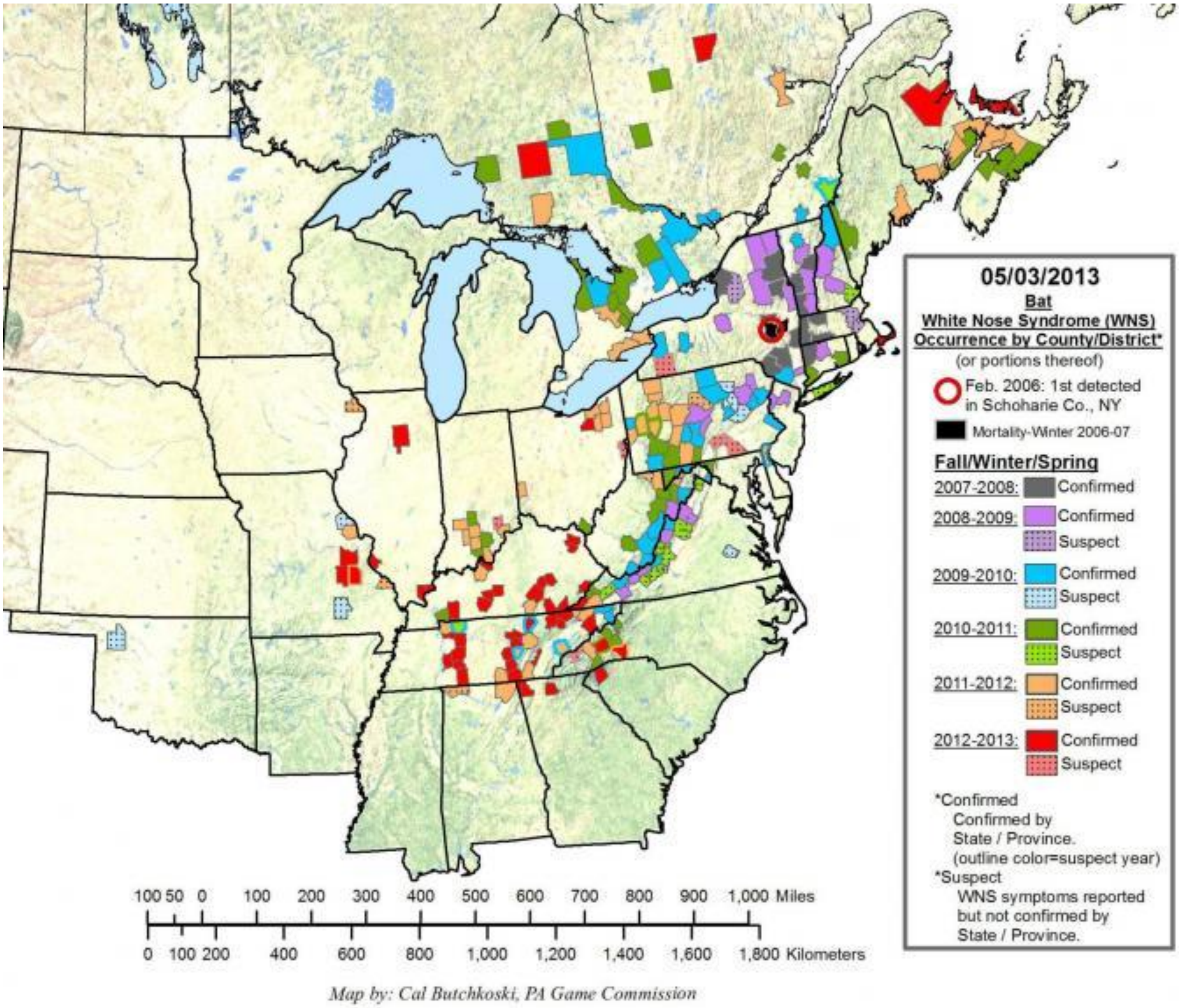

Figure 4.1 Distribution map of white-nose syndrome (WNS) in the eastern United States. Last updated 3 May 2013. Available: http://www.whitenosesyndrome.org/resources/map (USFWS 2013).

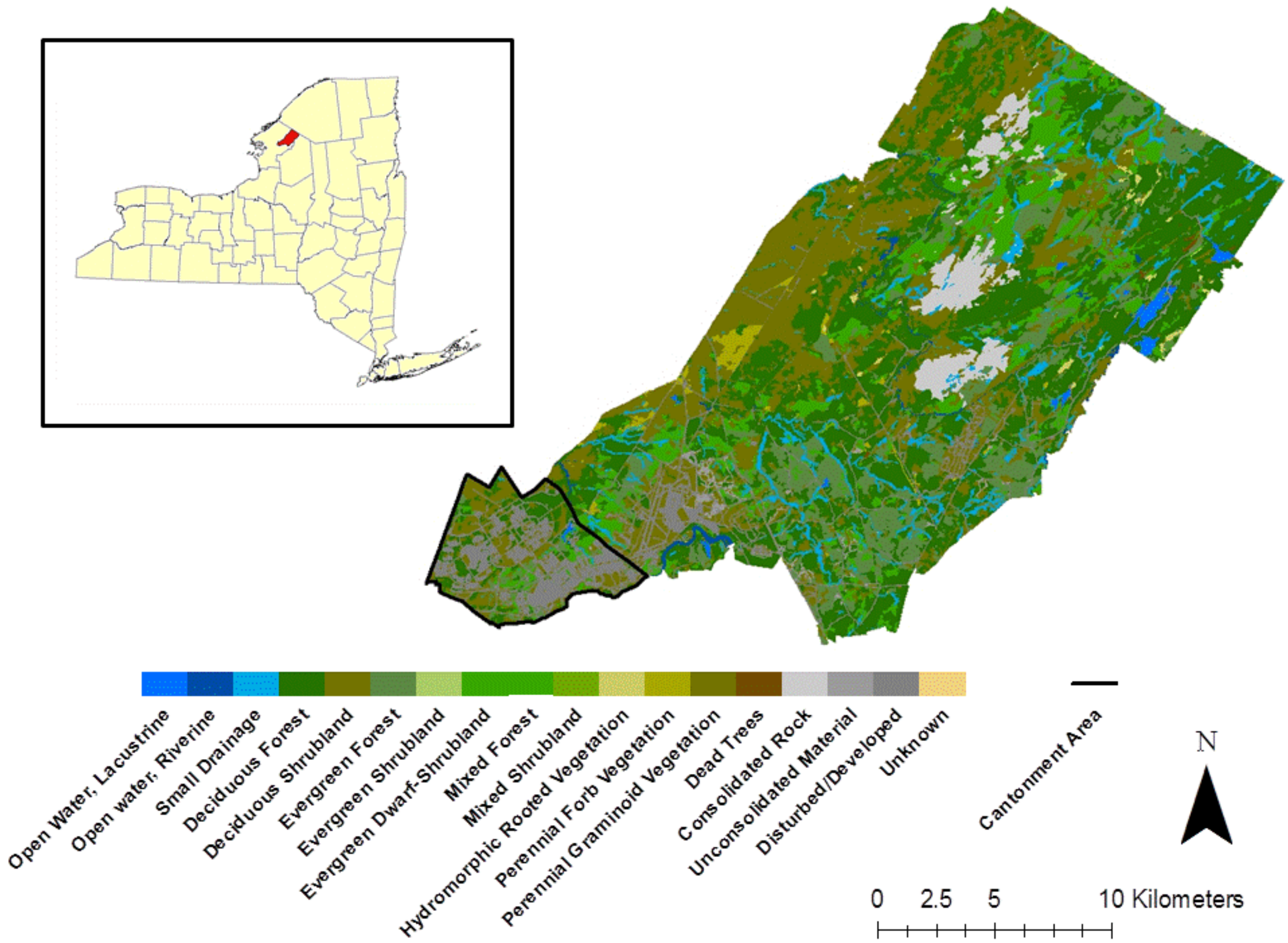

Figure 4.2 Fort Drum Military Installation, Jefferson and Lewis Counties, New York; Cantonment Area site of most passive acoustic sampling for bat species in summers 2011 and 2012.

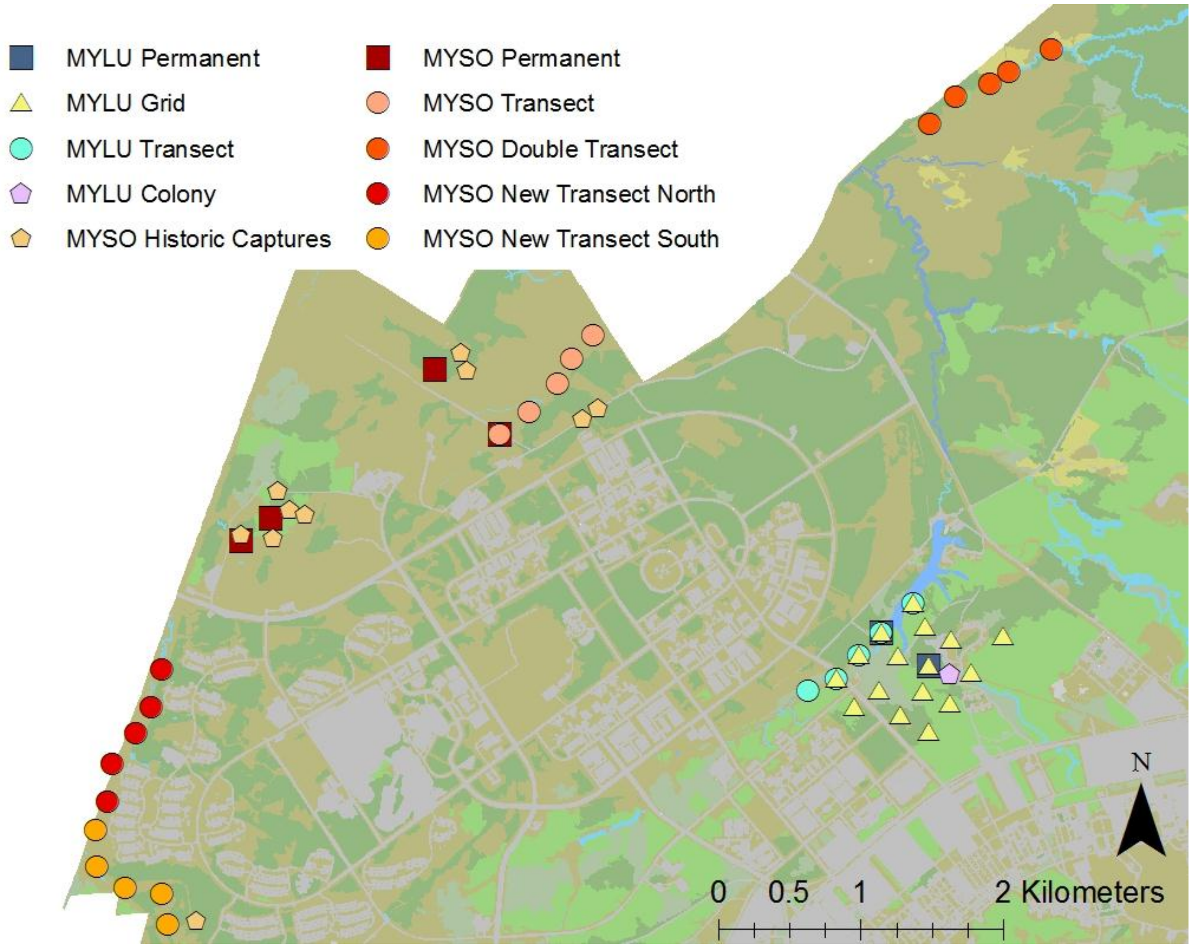

Figure 4.3 Passive acoustic sampling arrays in known maternity use areas of little brown bats (*Myotis lucifugus;* MYLU) and Indiana bats (*M. sodalis;* MYSO) at Fort Drum Military Installation Cantonment Area. Each point represents the site of a one passively deployed Anabat, except MYSO Double Transect which represents 2 Anabats per site. MYSO New Transect North and MYSO New Transect North deployed twice for 6-8 day sampling periods each in summer 2012. All other arrays sampled 4-5 times for 6-8 sampling periods in summers 2011 and 2012.

Table 4.1 Candidate set of models for assessing impacts of habitat, year, or time of season on bat occupancy (*Ψ*) and detection probability (*p*) estimates at Fort Drum Military Installation using passive acoustic sampling; "." = constant, "habitat" = wet vs. dry sites, "year" = 2011 vs. 2012, "season" = 15 May-15 July/pre-volancy of young vs. 15 July-15 Aug/post-volancy of young.

Model	*Ψ*	*p*
1	.	.
2	.	Habitat
3	.	Season
4	.	Year
5	.	Habitat + Season
6	.	Habitat + Year
7	.	Year + Season
8	.	Habitat + Season + Year
9	Habitat	Habitat
10	Season	Season
11	Year	Year
12	Habitat + Season	Habitat + Season
13	Habitat + Year	Habitat + Year
14	Season + Year	Season + Year
15	. Habitat + Season + Year	Habitat + Season + Year

Table 4.2 Top (within 2.00 ΔAICc) models for assessing impacts of habitat, year, or time of season on bat occupancy (*Ψ*) and detection probability (*p*) estimates at Fort Drum Military Installation using passive acoustic sampling at 40 sites; "." = constant, "habitat" = wet vs. dry sites, "year" = 2011 vs. 2012, "season" = 15 May-15 July/pre-volancy of young vs. 15 July-15 Aug/post-volancy of young.

Models	*K*	*AICc*	Δ*AICc*	*ωi*
Silver-haired				
Ψ (habitat), *p* (habitat)	4	1798.21	0.00	0.6480
Eastern red				
Ψ (.), *p* (habitat + year + season)	5	2342.49	0.00	0.4967
Ψ (.), *p* (habitat + season)	4	2343.13	0.64	0.3607
Hoary				
Ψ (habitat), *p* (habitat)	4	1990.24	0.00	0.3756
Ψ (habitat + year), *p* (habitat + year)	6	1990.48	0.24	0.3332
Big brown				
Ψ (habitat), *p* (habitat)	4	1961.30	0.00	0.7888
Little brown				
Ψ (.), *p* (habitat + year)	4	1028.32	0.00	0.6225
Ψ (.), *p* (habitat + year + season)	5	1029.83	1.51	0.2926
Indiana bat				
Ψ (.), *p* (habitat + year)	4	998.16	0.00	0.6853
Northern				
Ψ (.), *p* (season)	3	176.63	0.00	0.2318
Ψ (.), *p* (.)	2	177.26	0.63	0.1692
Ψ (.), *p* (habitat + season)	4	178.00	1.37	0.1169
Ψ (.), *p* (habitat)	3	178.57	1.94	0.0879
Eastern small-footed				
Ψ (.), *p* (year + season)	4	84.92	0.00	0.2717
Ψ (.), *p* (habitat + year + season)	5	86.03	1.11	0.1560
Ψ (habitat), *p* (habitat)	4	86.60	1.68	0.1173
Tri-colored				
Ψ (habitat + year), *p* (habitat + year)	6	191.16	0.00	0.3879
Ψ (.), *p* (habitat + year)	4	191.97	0.81	0.2587

Table 4.3 Occupancy (*Ψ*) models for determining impacts of passive acoustic sampling arrays on the detection probability (*p*) estimates of bat species at 40 sites at Fort Drum Military Installation, summers 2011 and 2012; "." = constant, "array" = permanent, grid, transect, double transect, "habitat" = wet vs. dry sites, "year" = 2011 vs. 2012, "season" = 15 May-15 July/pre-volancy of young vs. 15 July-15 Aug/post-volancy of young.

Bat species	Model
Silver-haired	*Ψ* (habitat), *p* (habitat + array)
Eastern red	*Ψ* (.), *p* (season + array)
Hoary	*Ψ* (.), *p* (array)
Big brown	*Ψ* (.), *p* (array)
Little brown	*Ψ* (.), *p* (habitat + year + array)
Indiana	*Ψ* (.), *p* (habitat + year + array)
Tri-colored	*Ψ* (habitat + year), *p* (year + array)

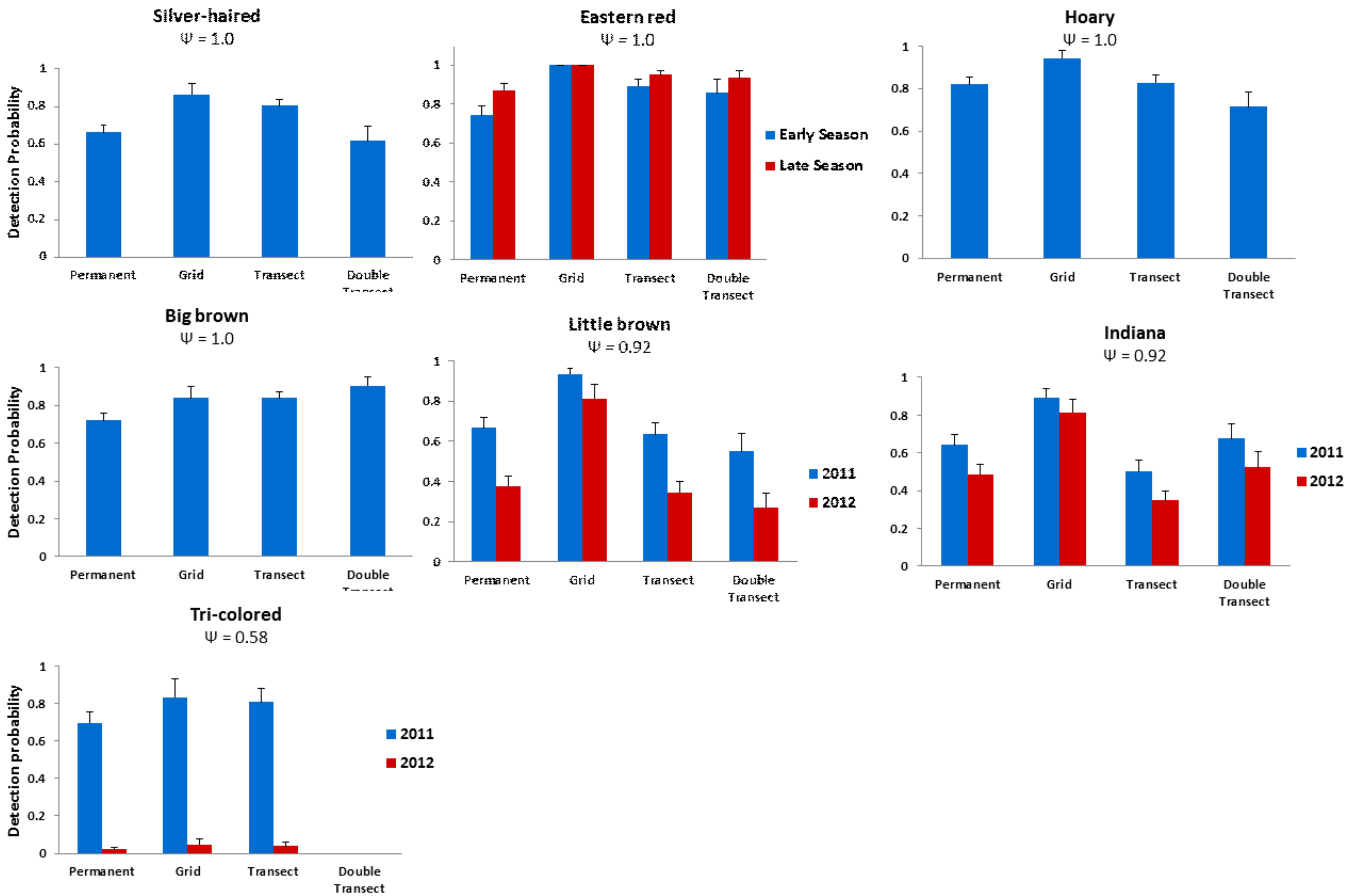

Figure 4.4 Detection probabilities of bats at various passive acoustic sampling arrays at Fort Drum Military Installation, summers 2011 and 2012. Time of season (early/late) and sampling year (2011/2012) included for when important covariates affecting detection probability estimates from single species occupancy models; Ψ = proportion of sites occupied (naïve estimate of occupancy).

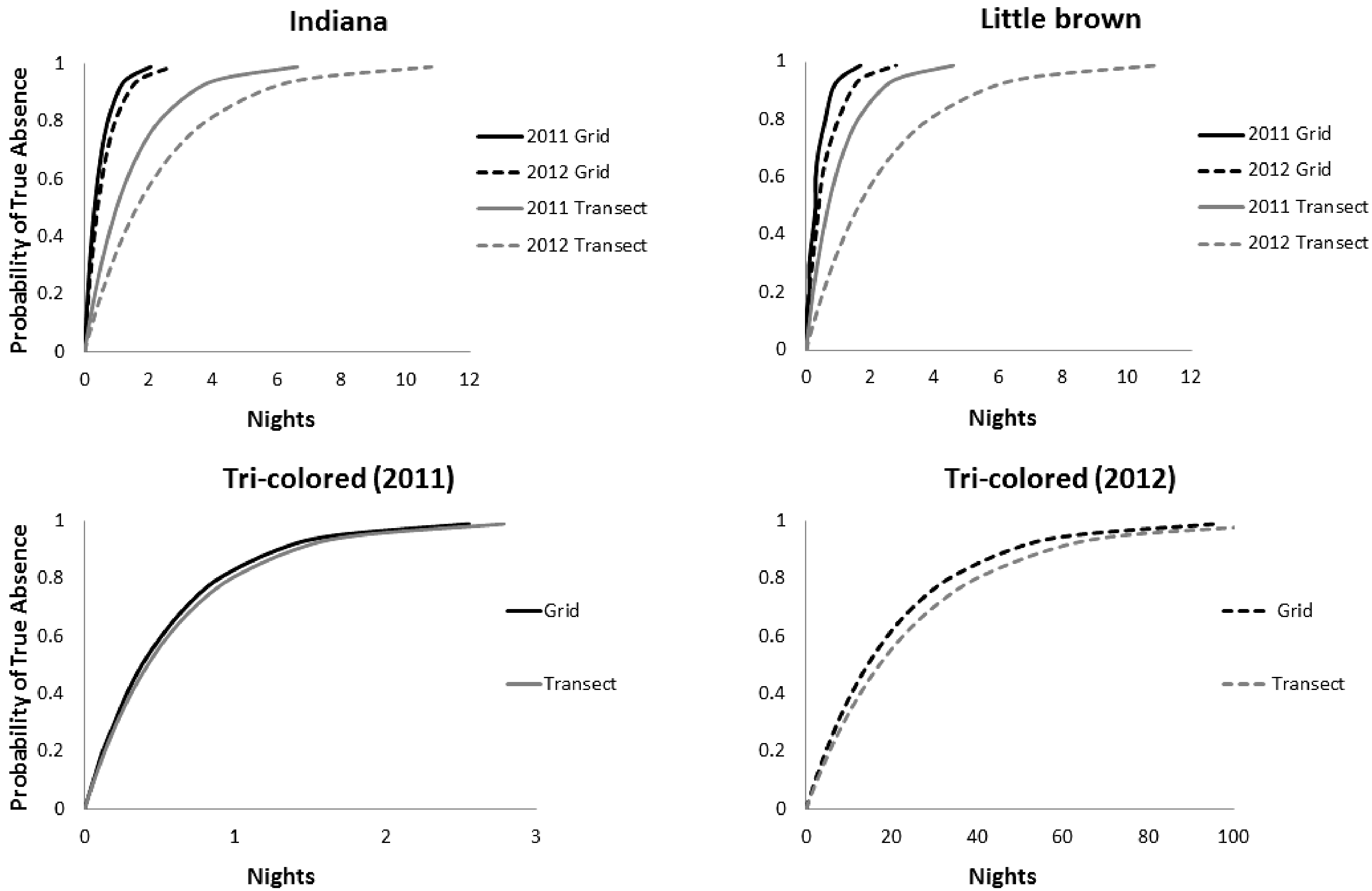

Figure 4.5 Suggested effort of passive acoustic sampling for determining true absence of Indiana (*Myotis sodalis*), little brown (*M. lucifugus*), and tri-colored (*Perimyotis subflavus*) bats using statistical probability according to the methods of McArdle (1990). Detection probabilities derived from single species occupancy models at a grid of 15 Anabat detectors and four transects each of five Anabat detectors at Fort Drum Military Installation, New York, summers 2011 and 2012.

Conclusions and management implications

Since white-nose syndrome (WNS) was documented in the northeastern U.S. in 2006, extreme declines in affected species have been observed (USGS 2012, USFWS 2013). In the face of these declines, managers are challenged with an immediate need to establish innovative summer monitoring protocols in order to comply with regulations of the Endangered Species Act (ESA 1973) for the Indiana bat (*Myotis sodalis*) and other species that may potentially become listed in the near future (Kunz and Reichard 2010, CBD 2011). My graduate research was intended to assist natural resources biologists at Fort Drum Military Installation in northern New York in establishing a summer monitoring program. Such a program was to focus primarily on acoustic methodologies, as traditional mist-netting efforts are now cost prohibitive and ineffective in detecting species of concern on the landscape (C.A. Dobony, unpublished data). I implemented a widespread acoustic sampling effort in order to 1) assess acoustic monitoring locations in congruency with nocturnal spatial use of bats according to radio telemetry, 2) determine whether active or passive sampling is preferable for detecting species of concern and 3) recommend a passive acoustic sampling array for optimizing detectability for determining presence or probable absence of focal species.

I observed considerable congruency between the nocturnal spatial use of female little brown bats (*M. lucifugus*) and sites of acoustic detector deployment that revealed the highest detectability of this species. My telemetry and acoustic results were consistent with previous research, clearly suggesting that female little brown bats forage in forested riparian habitats and select for these areas over most other habitat types at the landscape scale (Broders et al. 2006, Schirmacher et al. 2007, Bergeson 2012). Furthermore, the probability of detecting little brown bats when they were present more than doubled at riparian sites versus non-riparian sites.

Although it should not be assumed that acoustic sampling will provide similar conclusions to telemetry results at all scales, my data suggest that riparian habitat conditions may be optimal for successful detection of little brown bats in northeastern landscapes similar to Fort Drum where severe population declines have made monitoring difficult.

My assessment of passive and active sampling clearly pointed to passive sampling as the more cost efficient and effective method for detecting species of concern in the context of WNS-associated declines. Despite previous research suggesting active sampling as the preferential survey method (Britzke 2002, Johnson et al. 2002, Milne et al. 2004), my study suggests that active sampling provides lower detection probabilities and performs no better in recording relative activity patterns than passive sampling for myotids in the context of WNS declines. Active sampling detected big brown bats (*Eptesicus fuscus*) and eastern red bats (*Lasiurus borealis*) more readily than passive sampling, but both of these species were easily detected with either method. Leaving passive detectors deployed over long periods may be crucial to successful monitoring for WNS-impacted species that are now rare on the landscape and, therefore, less likely to be detected in any given sampling period. Finally, the cost per hour of data collected is substantially lower with passive sampling than for active sampling or contracted mist-netting. For most managers, passive sampling is easier to implement, requires less overall manpower because detectors can be deployed for multiple nights at a time, and is ultimately more cost effective over longer durations for detecting multiple species, particularly those impacted by WNS such as the Indiana bat and the little brown bat.

Although much research has been dedicated to the testing of acoustic techniques for monitoring bats over the last decade, my study was the first to assess the detectability of species at multiple acoustic sampling arrays. I found that a grid of detectors is more efficient in detecting

Indiana and little brown bats than permanent stations, transects, or double transects, probably due to its inherent ability to cover a wider spatial area. However, when enough detectors are not available to conduct a grid of sampling, other sampling arrays may be suitable for monitoring certain species based on my findings. Fewer detectors can be deployed as transects or smaller grids across wide sampling areas to capture temporal and spatial use of the landscape for target species. Moving detectors throughout as much space of an expected area of foraging use is typically preferable to leaving detectors stationary for long durations (MacKenzie and Royle 2005, Gorresen et al. 2008). Therefore, I recommend that managers implement variations of transects or grids even with low numbers of detectors rather than leaving detectors stationary for entire seasons.

Unfortunately, I was not able to draw conclusions on optimal monitoring procedures for detecting all species at Fort Drum. For example, I was unable to draw conclusions on the detectability of eastern small-footed bats (*Myotis leibeii*) with passive versus active sampling and with the assessment of passive sampling arrays. Additionally, I was unable to adequately model tri-colored bat (*Perimyotis subflavus*) detectability using active versus passive sampling and northern bat (*Myotis septentrionalis*) detectability using passive sampling arrays. Although the Indiana bat is currently the only species at Fort Drum central to regulations of the ESA (ESA 1973, as amended), eastern small-footed bats, northern bats, and little brown bats are under status reviews (Kunz and Reichard 2010, CBD 2011). Furthermore, the tri-colored bat has experienced severe declines locally and range wide (Ford et al. 2011, USGS 2012, USFWS 2013) and may be subject to a status review in the near future. My research was primarily focused in areas of expected use by Indiana bats and little brown bats. However, future efforts to assess optimal conditions for northern bats, eastern small-footed bats, and tri-colored bats are needed and should be focused in known historic areas of use or areas of suitable habitat to determine if these species still occupy the Fort Drum landscape following WNS declines.

It is apparent that traditional capture methodologies are no longer logistically or financially feasible as the primary summer monitoring tool for bats at Fort Drum (Coleman et al. 2013; C.A. Dobony, unpublished data) and likely other areas in the Northeast where severe declines of targeted species have been observed. My study suggests that acoustic monitoring can be successfully implemented as a monitoring tool for Indiana and little brown bats when executed in areas of anticipated or historic use. Passive sampling at foresedt riparian sites is likely more effective for detecting these species than active sampling or sampling at other, less frequently utilized habitats. These techniques are also likely to easily collect acoustic information on big brown bats as well as species not impacted by WNS such as the silver-haired bat, the eastern red bat, and the hoary bat. However, it is uncertain from my research whether a definitive sampling protocol exists that can adequately detect all species that occur at Fort Drum or in the wider Northeast region. Further research to define optimal conditions for determining the presence or probable absence of other WNS-impacted species is warranted.

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