

Distribution of Northern Long-eared Bat Summer Habitat on the Monongahela National Forest, West Virginia

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Abstract: Species distribution models enable resource managers to avoid and mitigate impacts to, or enhance habitat of, target species at the landscape level. Persistent declines of northern long-eared bats (*Myotis septentrionalis*) due to white-nose syndrome have made acquisition of contemporary data difficult. Therefore, use of legacy data may be necessary for creation of species distribution models. We used historical roost and capture records, both individually and in combination, to assess the distribution and availability of northern long-eared bat habitat across the 670,000-ha Monongahela National Forest (MNF), West Virginia, USA. We created random forest presence/pseudo-absence models to examine influences of various biotic and abiotic predictors on both roosting and foraging presence locations of northern long-eared bats. Predicted northern long-eared bat habitat was abundant (43.1% of the MNF) and widely dispersed. Generally, all models suggested that northern long-eared bat habitat was characterized by interior forests containing linear edge features. We observed only 3.4% spatial overlap of habitat based on complete model agreement, but 38.5% of all habitat areas resulted from agreement between capture-only and combination models. Our models provide important assessments of habitat availability necessary for addressing state and federal conservation requirements on the MNF and adjacent eastern West Virginia mountains.

Key words: *Myotis septentrionalis*, random forest, species distribution model

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Pseudogymnoascus destructans (Pd), the fungal pathogen that causes white-nose syndrome (WNS), has prompted extreme population declines of several bat species, including the federally threatened northern long-eared bat (*Myotis septentrionalis*; Cheng et al. 2021). Declines of northern long-eared bats in the inland Northeast through the central Appalachian Mountains of West Virginia have been severe and threaten the species with extinction (Cheng et al. 2021, Johnson et al. 2021). The collapse of northern long-eared bat populations occurred quickly within the High Appalachian Plateau portion of the central Appalachian Mountains, where hibernation periods are long and bats rely on WNS-infected caves in karst areas as hibernacula (Johnson et al. 2013, Ford et al. 2016a, Austin et al. 2018). After first being federally listed as threatened (USFWS 2015), the northern long-eared bat was proposed for up-listing to endangered after a 2022 status reassessment (USFWS 2022).

Although well examined on the Fernow Experimental Forest (FEF) within the Monongahela National Forest (MNF) and the MeadWestvaco Wildlife and Ecosystem Research Forest (Keyser

and Ford 2005) adjacent to the MNF, the distribution and habitat relationships of the previously common northern long-eared bat has not been well-investigated on the broader MNF (Menzel et al. 2002, Silvis et al. 2012, Ford et al. 2016a, Ford et al. 2016b). The northern long-eared bat often is described as a forest generalist (Silvis et al. 2016), typically selecting day-roosts based on the availability of suitable roosts and forest stand conditions (Ford et al. 2006, Perry and Thill 2007, Silvis et al. 2012). Northern long-eared bats are most often associated with contiguous canopy, deciduous forest stands where bats often select decaying black locust (*Robinia pseudoacacia*) and sassafras (*Sassafras albidum*), along with more shade tolerant and often suppressed maples (*Acer* spp.) as day-roosts (Ford et al. 2006, Gorman et al. 2022). On the FEF, northern long-eared bats select small black locust trees and snags located in forest canopy gaps created by prescribed fire (Johnson et al. 2009) or large canopy dominant black locusts receiving full sunlight (Ford et al. 2006). Throughout the northern long-eared bat's distribution, roosts also have been associated with ridges and high side-slope positions (Lacki and Schwierjohann 2001, Brack et al. 2002, Jung et al.

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2004, Johnson et al. 2009, Johnson et al. 2012) and/or forest stands disturbed by frequent wildfire and windthrow or that experienced large-scale disturbance in the previous decades (Silvis et al. 2012). Specifically, the dynamic processes that occur at ecotones or during earlier forest successional patterns appear to promote both the availability and quality of northern long-eared bat day roosts (Johnson et al. 2009, Silvis et al. 2012, De La Cruz et al. 2022). For foraging, northern long-eared bats select harvested and intact stands in both deciduous and coniferous forests (Owen et al. 2003, Broders et al. 2006). Foraging areas, while often proximal to roosting sites (Broders et al. 2006), are typically nearer to forested streams and road corridors than are roosts (Henderson and Broders 2008). Such areas are typically characterized by lower canopy forests having higher structural complexity (Henderson and Broders 2008). Foraging habitat selected by the species also includes high-elevation pools having little canopy cover (Johnson et al. 2010).

Use of pooled data derived from previous research to construct species distribution models (SDM) for rare, threatened, or endangered bat species has been limited (Pauli et al. 2015a, Ford et al. 2016b). Landscape-level distribution and habitat relationship data for northern long-eared bats may assist land managers in conserving the species in eastern North America where it is in severe decline (Ford et al. 2016b). At the scale of the MNF, such information can inform decisions about land management practices (e.g., timber harvesting, prescribed burning, trail and road construction, natural gas production) that may negatively impact remaining populations

of northern long-eared bats through roost tree removal (Silvis et al. 2015) or the alteration of foraging habitat (Loeb and O'Keefe 2014). Although occupancy modeling derived from repeated survey designs are foundational methods for assessing both the detection and occupancy probabilities relative to habitat conditions (MacKenzie et al. 2002), such approaches are difficult for declining bat species (Ford et al. 2016b). In contrast, other types of species distribution models, often created using machine learning tools (e.g., random forests [RF], maximum entropy [MAXENT]), are compatible with presence-only data (De La Cruz and Ward 2016). These techniques allow for the robust analysis of species distribution and habitat associations, based on historical documentation and limited continued presence identifications, of species in severe decline (De La Cruz and Ward 2016, Mi et al. 2017). Our objectives were to: 1) build SDMs from historical presence data (i.e., roost locations, capture locations, and combination) for northern long-eared bats using a random forest framework and assess model performance; 2) describe northern long-eared bat habitat associations on the MNF; 3) assess inter-model (i.e., roost, capture, and combination models) agreement; and 4) quantify the availability of northern long-eared bat habitat (i.e., foraging and roosting) across the MNF.

Study Area

We estimated the availability of northern long-eared bat habitat within the proclamation boundary of the greater MNF (Figure 1). This boundary comprises 670,000-ha of Grant, Greenbrier,

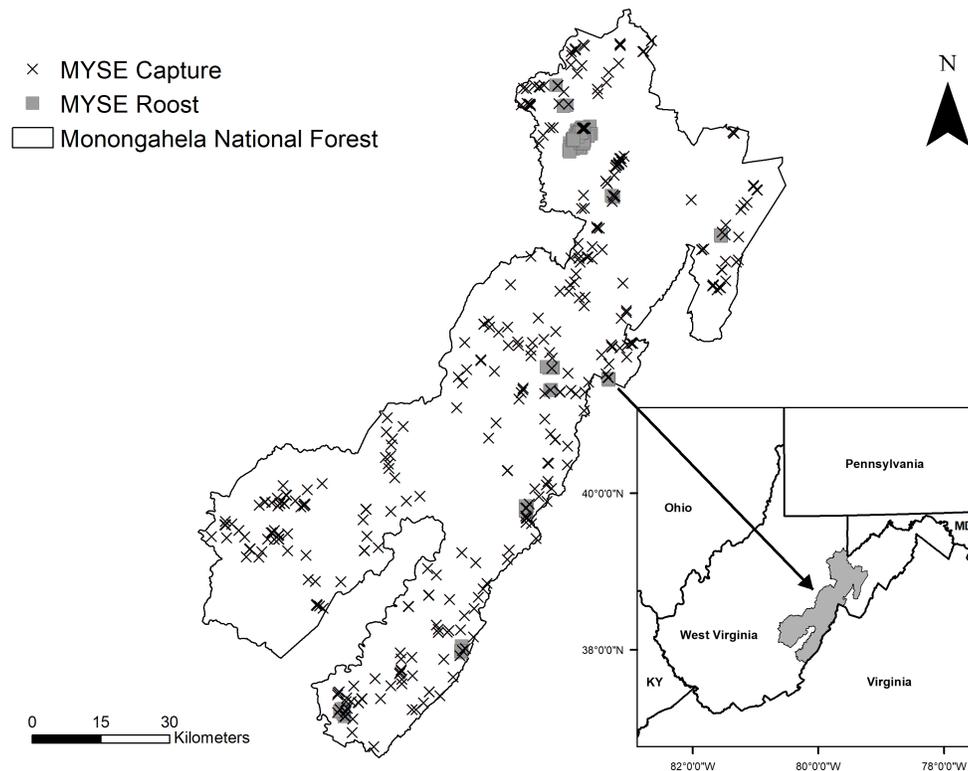


Figure 1. Approximate northern long-eared bat (MYSE) capture and roost location in the proclamation boundary of the Monongahela National Forest, West Virginia, 1997–2019.

Nicholas, Pendleton, Pocahontas, Preston, Randolph, Tucker, and Webster counties, West Virginia (USFS 2011). The MNF lies in the Allegheny Mountain and Upland Section and Ridge and Valley sub-physiographic provinces of the central Appalachian Region of West Virginia (Johnson et al. 2021). Depending on elevation, aspect, and soil types, the Allegheny Mountain and Upland Section is composed of northern evergreen (i.e., red spruce; *Picea rubens*) forests, northern hardwood forests, mixed mesophytic forests, and oak (*Quercus* spp.)-hickory (*Carya* spp.) forests whereas the more xeric Ridge and Valley portion is dominated by oak-hickory and mixed hardwood-pine (*Pinus* spp.) forests (Johnson et al. 2021). Regional topography is comprised of steep, deeply incised mountains ranging 275–1480 m in elevation (USFS 2011, Johnson et al. 2021). The MNF receives approximately 150 cm of precipitation annually in the Allegheny Mountain and Upland Section and as little as 70 cm in the Ridge and Valley east of the rain shadow created by the highest elevations of the Allegheny Front. The MNF experiences a mean annual temperature of 9 C (USFS 2011).

Methods

Historical Capture and Roost Records

We collated historical bat presence data collected from 15 April to 15 November 1999–2013 on the FEF in Tucker County, West Virginia, and 22 July to 3 August 1997–2019 throughout the wider MNF. Following Ford et al. (2016b), we retained capture sites as unique presence locations only when sites were >30 m apart using the BiodiversityR package in R (Kindt and Coe 2005, R Core Team 2013). From the FEF, we included 11 northern long-eared bat capture locations and 131 roost tree locations collected by academic institutions and state and federal agencies (see Ford et al. 2016b). From mist-netting surveys conducted by the U.S. Forest Service and its partners at >400 locations (≥ 30 m apart) across the MNF (USFWS 2006, Johnson et al. 2021), we identified 360 additional capture locations along with 44 northern long-eared bat roost locations on the broader MNF. In total, we compiled $n = 371$ capture locations and $n = 175$ roost locations.

Habitat

Following the framework of De La Cruz and Ward (2016), we examined the effect of land use/land cover using the West Virginia Land Use Land Cover raster dataset (Strager 2020a), and corresponding habitat diversity (Shannon diversity; H), evenness, and richness (Zuckerberg et al. 2016), on species distributions. Specifically, the 5-m resolution land use/land cover raster data detailed 25 cover types: water, river floodplains, small stream riparian

forests, roads, impervious surfaces, mixed development, uncategorized barren, mine barren, oil and gas development barren, other forests, red spruce forests, northern hardwood forests, mixed mesophytic forests, dry-mesic oak forests, dry oak-pine forests, pine-oak-emergent rock forests, dry calcareous forests, montane red oak (*Quercus rubra*) forests, low vegetation, hay/pasture, cultivated crop, mine grass, shale barrens, emergent wetlands, and forested wetlands. These cover types were described in WVDNR (2015), then were spectrally updated (Maxwell et al. 2019) to create the West Virginia Land Use Land Cover dataset. We incorporated measures of forest fragmentation using the Forest Fragmentation of West Virginia raster dataset, classifying areas as patch, edge, perforated, core (<100 ha), core (100–200 ha), or core (>200 ha; Strager 2020b). We also included canopy height (EROSC 2019), solar radiation (watt-hours per square meter), average annual temperature (PRISM Climate Group 2019), elevation (Gesch et al. 2002), aspect (i.e., cosine and sine transformations), and slope degree in models examining the distribution of northern long-eared bat presence records. Terrain measures i.e., solar radiation, aspect, slope, were calculated from digital elevation models using the Spatial Analyst toolbox in ArcGIS (Version 10.3, Environmental Systems Research Institute, Redlands, California). Additionally, we examined distance from forest edge (0) into forest interior (–) and from forest edge into (+) non-forest cover (White et al. 2017). We standardized all raster datasets to a 30-m resolution. We calculated focal means of categorical variables using a 500-m moving window analysis (Thalke et al. 2018), which approximates the day roost to foraging site dispersal distance of northern long-eared bats (Owen et al. 2003, Timpone et al. 2010, Badin 2014, Pauli et al. 2015a).

Statistical Analysis

We created presence/pseudo-absence random forest (RF) SDM models (Mi et al. 2017) using the *sdm* package in R (Naimi and Araújo 2016). The RF models create several classification and regression tree (CART) models from bootstrap sampling of training data whereby each new model contributes to the prediction of the dependent variable (Evans et al. 2011). The RF models are non-parametric and therefore not subject to distributional assumptions, do not require transformation (e.g., scaling), can incorporate a variety of predictor variables (i.e., continuous, categorical, and ordinal), do not assume spatial independence, and are considered robust to noise even when a very large number of independent variables are used (Evans et al. 2011).

The number of pseudo-absence points equaled the number of known presence locations (Barbet-Massin et al. 2012), relative to

location type (i.e., roost, capture, combination). To mitigate against excessive false pseudo-absences and clustering, we required that all random pseudo-absence points be ≥ 1000 m from any presence point and any other pseudo-absence point (Olivier and Wotherpoon 2006). Prior to initial modeling, we removed correlated variables using pairwise correlation (threshold = $|0.8|$). We evaluated the predictive performance of our models using k-fold cross-validation (i.e., 5 folds and 10 replications; $n = 50$ models) of area under the curve (AUC) of the receiver operating characteristic and Cohen's Kappa (κ) statistic (Evans et al. 2011). We interpreted AUC values of 0.70–0.80 as fair, 0.80–0.90 as good, and 0.90–1.00 as excellent, and similarly viewed κ values of 0.21–0.40 as fair, 0.41–0.60 as moderate, 0.61–0.80 as good, and 0.81–1.00 as excellent (Koma et al. 2022). We also determined relative variable importance (RVI) by measure of AUC improvements to model performance, assessing model performance during inclusion of each variable and comparing to exclusion of predictors during cross-validation (Naimi and Araújo 2016). Our final models contained only those variables scoring greater than or equal to initial mean RVI (Evans 2011, Naimi and Araújo 2016). Finally, we examined the relation of probable habitat suitability to predictor variables using response curves (Evans et al. 2011).

We created binary (i.e., suitable vs unsuitable) SDMs using predictive model averaging and thresholding based on constraining predicted prevalence to be equal to observed prevalence for all models (Freeman and Moisen 2008). We then combined the three binary rasters created from individual roost, capture, and combination models using spatial addition. Final concordance maps reflect habitat suitability as predicted by one or more models (i.e., capture, roost, capture and roost, combination, capture and combination, roost and combination, or all models).

Results

Our capture-only SDM for northern long-eared bats displayed moderate to good predictive power (AUC = 0.81; $\kappa = 0.53$) and contained distance to forest edge, mixed mesophytic and “other” forests, woodland roads, slope, and low vegetation. The most important variable to the model was slope (RVI = 22.8; Table 1). Response curves suggested that foraging habitat was associated with flat to moderately steep hillsides (0–20 degrees; Figure 2) comprised of interior (>500 m; RVI = 6.4) mixed mesophytic ($\geq 10\%$; RVI = 9.0) and “other” forests ($\geq 10\%$; RVI = 4.7). These areas are further described as containing moderate amounts of woodland roads (1–4%; RVI = 14.8) and little to no low vegetation (RVI = 7.1; Figure 2).

The SDM we created using roost-only data demonstrated good

Table 1. Final mean area under the curve (AUC) relative importance of select variables for northern long-eared bat random forest model replicates ($n = 50$) derived from roosts locations, capture locations, and the combination of roost and capture locations on the Monongahela National Forests, West Virginia, 1997–2019.

Variable	Capture model	Roost model	Combination model
Canopy height	–	1.8	10.7
Distance to forest edge	6.4	1.7	14.6
Dry oak-pine forests	–	3.2	4.6
Mixed mesophytic forests	9.0	6.6	–
Montane red oak forests	–	0.5	–
Northern hardwood forests	–	2.2	–
Non-forest areas	–	3.6	–
Other forests	4.7	–	7.2
Richness	–	1.5	–
Roads	14.8	1.1	28.1
Slope	22.8	–	10.4
Red spruce forests	–	1.5	5.1
Small stream riparian forests	–	1.8	–
Mean annual temperature	–	5.1	–
Low vegetation	7.1	–	–

to excellent predictive ability (AUC = 0.94; $\kappa = 0.80$) and, similar to our capture-only SDM, contained distance to forest edge and roads. In addition to these predictors, the model also contained canopy height, dry-oak pine forests, mixed mesophytic forests, montane red oak forests, northern hardwood forests, non-forest areas, landscape richness, red spruce forests, small streams riparian forests, and mean annual temperature. The most influential variable to the model was presence of mixed mesophytic forests (RVI = 6.6; Table 1), with roosting habitat located in areas containing $\geq 20\%$ of the cover type (Figure 3). Based on our roost-point SDM, northern long-eared bat habitat was positively associated with dry oak-pine ($\geq 10\%$; RVI = 3.2), montane red oak ($\geq 5\%$; RVI = 0.5), and northern hardwood forests ($\geq 10\%$; RVI = 2.2). Roosting habitat contained few roads ($\leq 1.5\%$; RVI = 1.1) and was characterized by moderate temperatures (8–10 C; RVI = 5.1) and tall trees (>20 m; RVI = 1.8). Northern long-eared bat habitat appeared somewhat homogenous (≤ 8 cover types; RVI = 1.5), containing little high-elevation red spruce forests ($<15\%$; RVI = 1.5), small stream riparian forests ($<5\%$; RVI = 1.8), and sparse non-forest cover ($\leq 10\%$; RVI = 3.6; Figure 3).

The combination model (i.e., capture and roost records) displayed moderate to good predictive power (AUC = 0.84; $\kappa = 0.58$), and, similar to the roost- and capture-only SDMs, contained distance to forest edge and roads. Additionally, this SDM included canopy height, dry oak-pine, “other” forests, roads, slope, and red spruce forests. Roads (RVI = 28.1; Table 1) was the most important

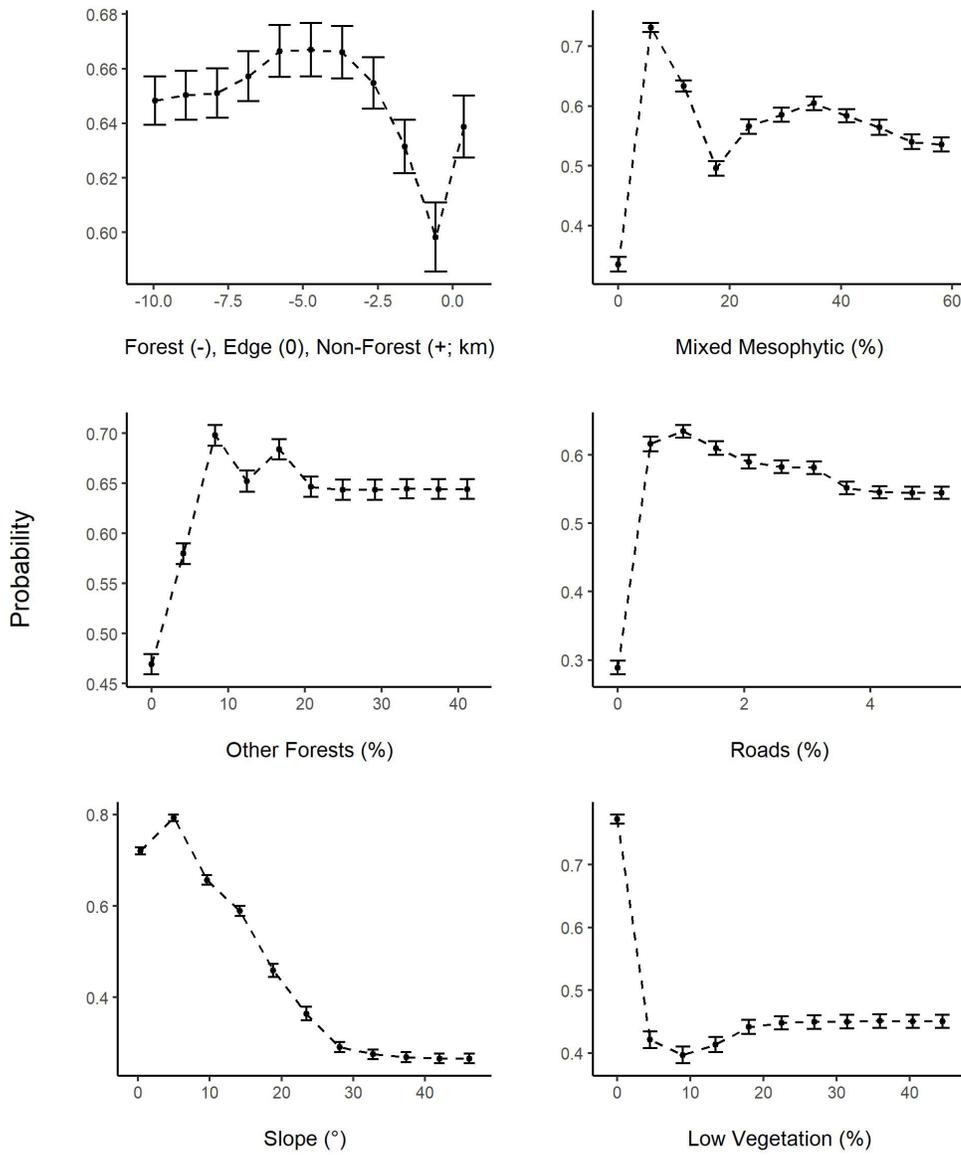


Figure 2. Predictor response curves (mean point estimates and 95% confidence interval) for $n = 50$ random forest model replicates predicting northern long-eared bat habitat based on capture locations on the Monongahela National Forest, West Virginia, 1997–2019.

Table 2. Total contribution of models and model agreement regarding northern long-eared bat habitat on the Monongahela National Forest, West Virginia, 1997–2019.

Model	Total hectares	Percent total	Percent suitable
Unsuitable	379,925	56.9	–
Capture model	76,750	11.5	26.7
Roost model	24,725	3.7	8.6
Capture/roost models	2475	0.4	0.9
Combination model	40,275	6.0	14.0
Capture/combination models	110,600	16.6	38.5
Roost/combination models	10,025	1.5	3.5
All models	22,550	3.4	7.8

variable to the combination model, with habitat suitability highest in areas comprised of 1–4% of these flight corridors (Figure 4). Furthermore, northern long-eared bat habitat appeared positively associated with interior (500–8000 m; RVI = 14.6) dry oak-pine ($\geq 10\%$; RVI = 4.6) and “other” forests ($\geq 5\%$; RVI = 7.2) on flat to moderately steep hillsides (0–20°; RVI = 10.4) containing taller forest canopies (≥ 20 m; RVI = 10.7). Finally, habitat suitability was negatively associated with red spruce forests ($\geq 25\%$; RVI = 5.1; Figure 4).

Based on summary totals of binary thresholding of capture-only ($P \geq 0.45$), roost-only ($P \geq 0.47$), and combination ($P \geq 0.46$)

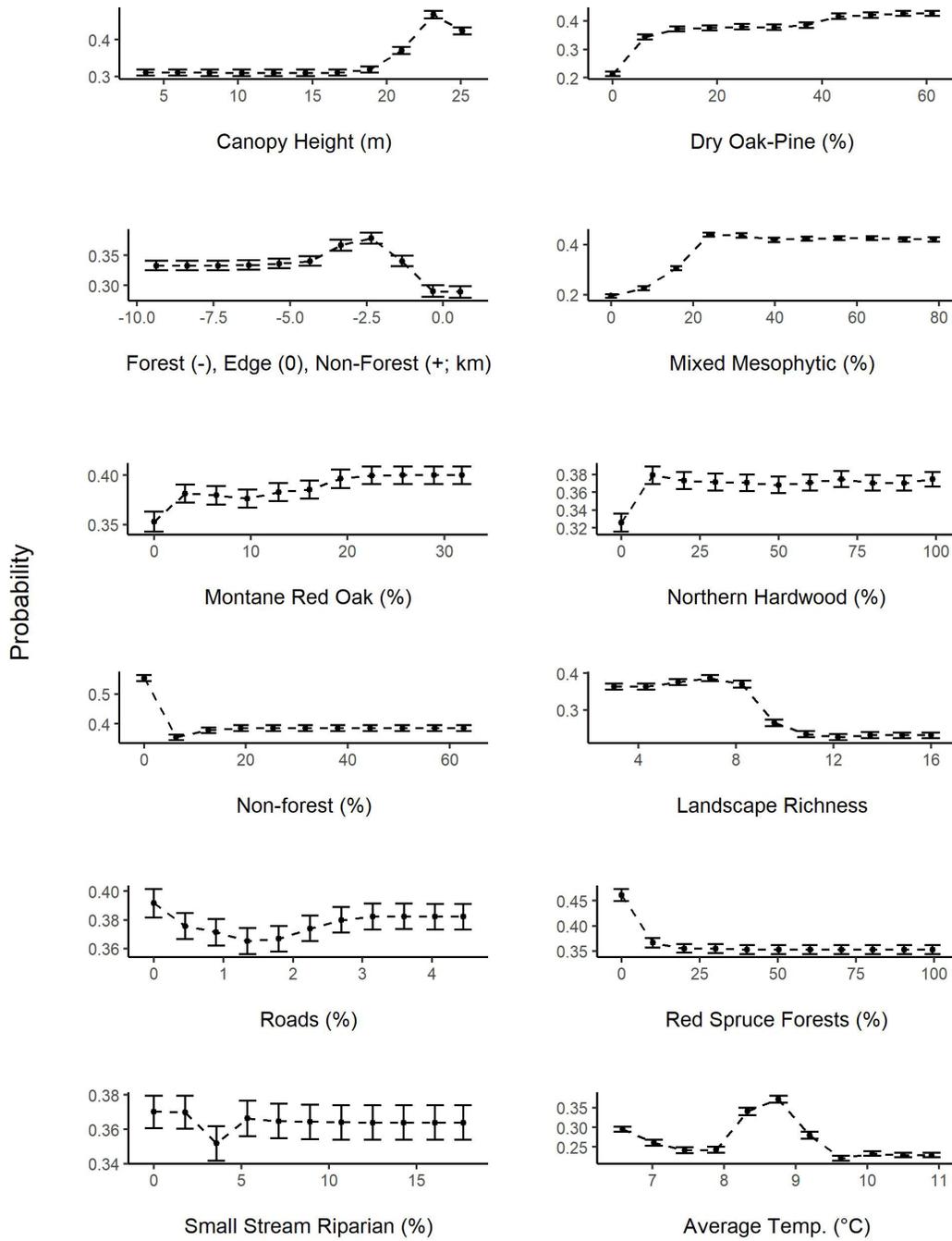


Figure 3. Predictor response curves (mean point estimates and 95% confidence interval) for $n = 50$ random forest model replicates predicting northern long-eared bat habitat based on roost locations on the Monongahela National Forest, West Virginia, 1997–2019.

models, northern long-eared bat habitat on the MNF was widespread and abundant (287,400 ha, 43.1% of the MNF; Figure 5). The additive grouping of the capture-only and combination models identified 38.5% (110,600 ha) of all potential northern long-eared bat habitat on the MNF. Independently, the capture-only

and combination models identified an additional 76,750 (26.7%) and 40,275 (14.0%) ha of potential habitat, respectively. All other models and additive agreement of models described <10% of identified habitat, with complete model agreement accounting for 7.8% (22,500 ha; Table 2).

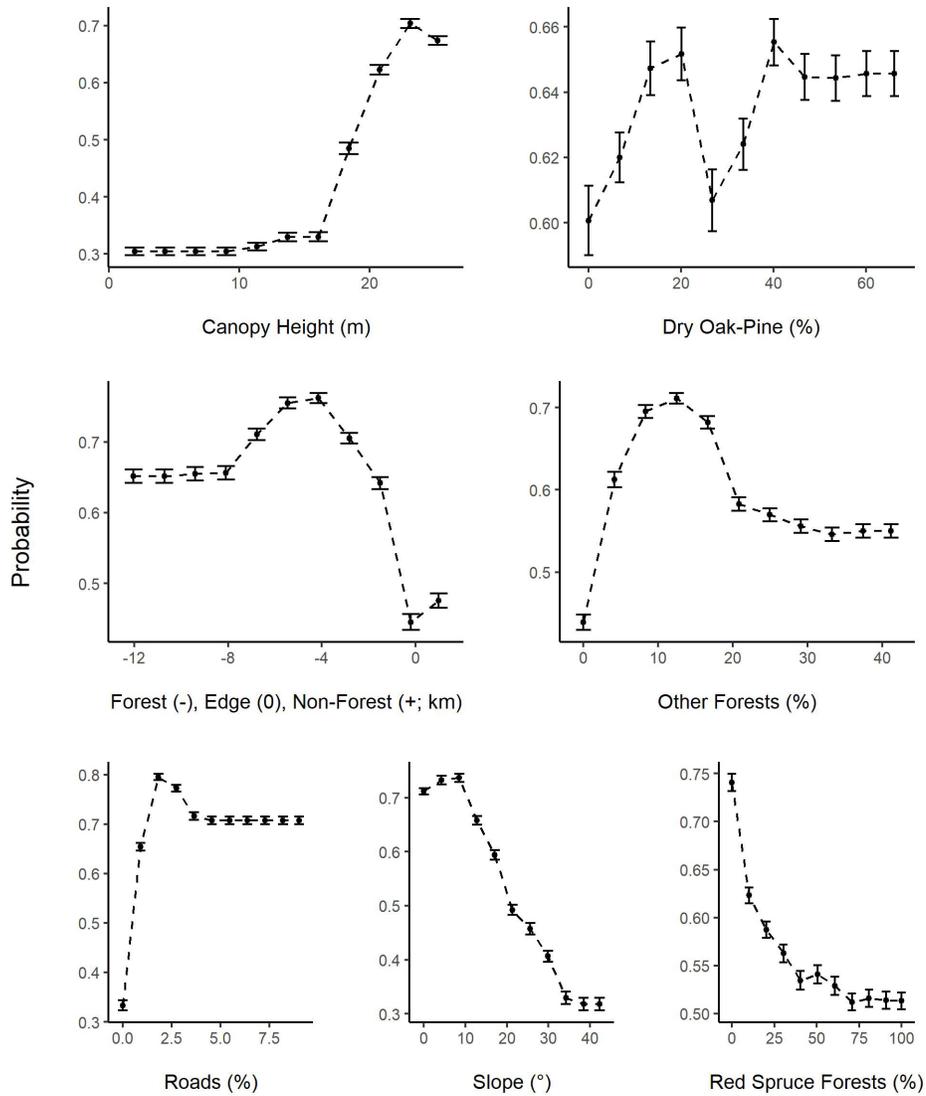


Figure 4. Predictor response curves (mean point estimates and 95% confidence interval) for $n = 50$ random forest model replicates predicting northern long-eared bat habitat based on both capture and roost locations on the Monongahela National Forest, West Virginia, 1997–2019.

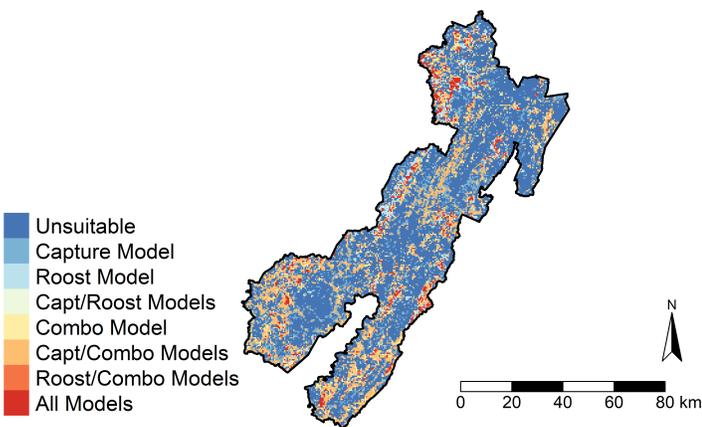


Figure 5. Northern long-eared bat habitat suitability concordance map on the Monongahela National Forest, West Virginia, 1997–2019.

Discussion

Quantitative bat research in the central Appalachian Mountains of West Virginia often has been limited to focal areas rather than large landscapes (De La Cruz and Ward 2016, Johnson et al. 2021). Prior to our work, few data existed capable of assisting U.S. Forest Service efforts in meeting regulatory requirements and/or supporting population-level recovery (Ford et al. 2016b). Our SDMs suggest that northern long-eared bat habitat is not a limiting ecological factor on the MNF. Moreover, widespread regional northern long-eared bat colony collapse has resulted in recruitment declines and/or maternity colony abandonment (Kalen et al. 2022), potentially decoupling land management impacts from the species.

Northern long-eared bats can require relatively large areas of

contiguous forest for both roosting and foraging (Sasse and Perkins 1996, Carter and Feldhamer 2005, Broders et al. 2006, Perry and Thill 2007). Our results support this expectation, as all models suggested that northern long-eared bat habitat suitability increased progressively into the forest interior and was negatively associated with increasing proportions of low vegetation and non-forest cover types, based on capture-only and roost-only models, respectively. Our roost-only model indicates that roosting habitat suitability was highest in areas of moderate annual temperatures (i.e., 8–10 C), and both roost-only and combination models indicated the importance of mature forests (>20 m tall) to the species. In general, northern long-eared bats use suppressed and shaded roosts that receive little direct sunlight in areas of contiguous forest cover (Lacki and Schwierjohann 2001, Menzel et al. 2002, Johnson et al. 2012, Silvis et al. 2012). However, local landscape heterogeneity may benefit northern long-eared bat populations in some regions (Gorman et al. 2022), and cover types selected may vary regionally (e.g., Kaminski et al. 2020), supporting the idea that conservation efforts for rare, threatened, and endangered bats should be based upon information collected at the local or regional level (De La Cruz and Ward 2016).

Our analyses and previous work indicate that northern long-eared bat habitat was related to increasing proportions of dry-mesic oak (roost-only model; Badin 2014), dry oak-pine (i.e., roost-only and combination models; Perry and Thill 2007), mixed mesophytic (i.e., capture-only and roost-only models; Lacki and Schwierjohann 2001), montane red oak (i.e., roost-only model; O’Keefe 2009), northern hardwood forest (roost-only model; Broders et al. 2006), “other” forests (i.e., capture-only and combination models; Perry et al. 2008), and forest roads that intersect these cover types (i.e., all three models; Owen et al. 2003). These results support the idea of northern long-eared bats as a forest generalist (Silvis et al. 2016). Still, we observed a negative relation in our roost-only and combination models between habitat suitability and red spruce forests, which contain a suite of tree species rarely used as roosts throughout their distribution (Silvis et al. 2016). Furthermore, our roost-only model also indicates that roosting habitat suitability decreased with high landscape richness, further supporting the supposition that northern long-eared bats are forest obligates (Ford et al. 2005, Gorman et al. 2022). Nonetheless, northern long-eared bats select forests frequently altered by small-scale disturbances or longer-term successional/stand development dynamics that create more roosts and enhance the quality (i.e., increased solar exposure, cavity creation, exfoliating bark) of existing roosts available for use within the interior forest matrix (Johnson et al. 2009, Ford et al. 2016a, Ford et al. 2016b, Divoll et al. 2022). This suggests the need for forest interior heterogeneity as northern long-eared

bat roosting and foraging habitat (Owen et al. 2003, Gorman et al. 2022). Finally, northern long-eared bat habitat appears related to moderately steep slopes (i.e., capture-only and combination models; De La Cruz et al. 2018) and lower proportions of small stream riparian forests (i.e., roost-only model; Ford et al. 2005).

Colonies of post-WNS reproductively successful northern long-eared bats have recently been discovered in many parts of their range (De La Cruz et al. 2018, Jordan 2020, Deeley et al. 2021, Arant et al. 2022, De La Cruz et al. 2022, Gorman et al. 2022). As the effects of WNS cause additional colony failures (Cheng et al. 2021), particularly in populations associated with traditional karst hibernacula (i.e., MNF), determining how remnant populations persist and successfully reproduce will be necessary to conserve the species. Nonetheless, recent findings suggest that the use of unique, non-traditional hibernacula (i.e., basements, coal adits) may allow this species to avoid long-term exposure to WNS in colder portions of its range, allowing it to avoid contact with WNS-vectoring bat species such as the little brown bat (*Myotis lucifugus*; Dowling and O’Dell 2018, Lituma et al. 2021). Furthermore, our ongoing acoustic sampling has documented widespread detection of northern long-eared bats throughout the MNF, suggesting either the presence of numerous non-reproductive individuals or possibly the continued presence of reproductive populations. We believe this highlights a need to continue to identify, conserve, and enhance northern long-eared bat habitat on the MNF.

Because many historical bat records were collected in conjunction with regulatory clearance, the distribution of sampling sites was often located at or near highly productive foraging areas (e.g., streams, wetlands) or easily accessible sites (e.g., flatter topography and access roads), potentially raising the concern that sampling bias may have promoted clustering of presence records and affected our results. However, research suggests that northern long-eared bats actively select such resources as commuting and foraging habitat in areas of contiguous forests (Henderson and Broders 2008, Johnson et al. 2010), supporting use of these data and inclusion of covariates (i.e., roads) for modeling. Although our models do not directly address spatial autocorrelation of point data, which may lead to model overfitting (Hammond et al. 2016), we likely reduced overfitting by performing initial removal of landscape variables using a mean RVI threshold (Evans et al. 2011). Furthermore, although capture locations (i.e., nets) were collapsed to a relatively small minimum distance (Ford et al. 2016b, Johnson et al. 2021), mean distance between sites was 1188 m, which incorporates fair spatial distribution across the MNF. Unsurprisingly, unlike capture locations, roost locations were far more clustered (\bar{x} = 176 m). However, maternity colonies of northern long-eared bats switch between nearby roosts frequently, often due to exposure and thermoregulatory

requirements (Patriquin et al. 2016), suggesting the potential utility of including areas of high resource density when modeling roosting habitat.

Presence data types (i.e., roost and capture points) often represent very different aspects of bat ecology, and species distribution models made using these data separately and in combination often produce variable results (Pauli et al. 2015b, Ford et al. 2016b). Specifically, our capture (i.e., nocturnal foraging selection) and roost (i.e., diurnal roost selection) models often contained differing covariates, and when variables were shared, importance measures were dissimilar. For example, our roost model included predictors that reflect diurnal roosting habitat requirements (canopy height, dry oak-pine forests, montane red oak forests, northern hardwood forests, and mean annual temperature), variables absent from the capture model. This illustrates both the need to assess habitat selection specific to ecological need and to address these habitat requirements individually when devising species conservation and management plans.

Our results contribute to understanding conservation priorities related to habitat requirements of the northern long-eared bat on the MNF and presumably much of the eastern mountains of West Virginia. This work can assist managers in establishing future monitoring sites (i.e., capture and acoustic sampling) and candidate areas for conservation planning with respect to forest management goals on the MNF (USFS 2011). By combining individual model habitat determinations, our concordance model represents a conservative assessment of habitat availability necessary for addressing state and federal conservation requirements relative to the species. Because northern long-eared bats continue to decline from WNS, collection of robust contemporaneous presence-data is unlikely, so historical records represent the best available data for species distributional modeling (Ford et al. 2016b). Future research on the MNF and elsewhere focusing on collection of post-WNS northern long-eared bat presence to assess current occupancy and detection probabilities and to validate final concordance distribution maps is warranted.

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