FORECASTING RECOVERY OPPORTUNITIES FOR THE RED-COCKADED WOODPECKER ON PRIVATE LANDS IN EASTERN NORTH CAROLINA USING A SPATIAL MODEL OF TREE AGE

A Thesis

by

AMANDA MICHELLE DUBE

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Chair of Committee,	Roel R. Lopez
Committee Members,	Russell A. Feagin
	Sorin C. Popescu
	Robert N. Wilkins
Head of Department,	Michael P. Masser

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ABSTRACT

Currently, recovery efforts for the federally endangered red-cockaded woodpecker (RCW; *Picoides borealis*) primarily occur on public lands throughout the Southeast, where proven management practices ensure availability of mature, open pine savannahs able to support populations. Many populations on public lands are approaching carrying capacity, suggesting RCW management on private lands will become increasingly important to achieve recovery goals. Recovery on private lands will involve developing recruitment clusters through management practices that produce sufficient quality and spatial aggregation of trees age 60 or older to provide nesting habitat, and trees age 30 or older to provide foraging habitat, as outlined in the U.S. Fish and Wildlife Service (USFWS) Recovery Plan. In this analysis, relationships between tree age, canopy height, and site index were applied to land cover, LiDAR-derived canopy height, and expected site index data in a geographic information system (GIS) to produce a tree age model for pines on private lands in eastern North Carolina. Modeling provided a means to spatially and temporally identify recovery opportunities over the next 10 to 40 years, predict locations for potential recruitment clusters within the next 10 years, and assess connectivity between potential recruitment clusters.

Depending on predominant species, modeling produced acceptable estimates for tree age and suitability timeframes for 69-95% and 85-92% of surveyed parcels, respectively, compared to expected age and suitability timeframes derived from field-collected diameter at breast height (DBH). Over 90% of existing RCW clusters on

public lands were modeled to contain trees age 40 or older, suggesting age was underestimated in some cases. Results indicate almost 80% of existing pines will remain too young over the next 10 years to support RCW cavity trees. However, over 3,000 potential recruitment cluster sites were identified. These could contribute to increased carrying capacity by providing habitat for potential breeding groups, and create links between existing populations. The prevalence of young pines suggests more opportunities to create RCW recruitment clusters will become available over time with proper habitat and population management. Modeling such as done in this study can serve as a valuable conservation planning tool to guide recovery efforts over space and time.

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1. INTRODUCTION

1.1 Background

The red-cockaded woodpecker (RCW; *Picoides borealis*) is a federally endangered species associated with mature, open pine savannahs of the Southeastern United States. At the time of the 2003 U.S. Fish and Wildlife Service (USFWS) Recovery Plan, approximately 14,000 RCWs existed, accounting for less than 3% of the projected population size preceding European settlement. Population decline has mostly resulted from the harvest of mature pines needed for nesting, and territory abandonment caused by hardwood encroachment (USFWS 2003). Remnant RCW habitat and populations have generally been left in small, highly fragmented, isolated configurations (Ligon et al. 1986, Costa and DeLotelle 2006) following the reduction in range and quality of the 37 million hectares (Frost 1993) of longleaf pine (*Pinus palustris*) that once spanned the Southeast in dominant or mixed stands.

The unique social structure and habitat requirements of the RCW have made recovery challenging, particularly considering the present composition and context of landscapes throughout its range. As cooperative breeders, RCWs live in family groups consisting of a breeding pair and helpers, who assist with nesting, brooding, foraging, and territory defense (Conner et al. 2001, USFWS 2003). Territories include the cavity cluster (i.e., the aggregation of all active and inactive cavity trees used for nesting and brooding) and foraging areas (Walters et al. 1988). This breeding system allows the helper class to contribute to local population stability by filling breeding vacancies;

however, greater spatial isolation of groups and slow rates of natural cavity excavation have reduced the effectiveness of this strategy (USFWS 2003). RCWs prefer longleaf pine for cavity excavation, but will readily utilize other southern pines including loblolly (*Pinus taeda*), shortleaf (*Pinus echinata*), pond (*Pinus serotina*), slash (*Pinus elliottii*), and Virginia (*Pinus virginiana*) pine (Conner et al. 2001). Typically, minimum average age of cavity trees is 60 to 80 years (DeLotelle and Epting 1988, Hooper 1988), but observed cavity tree ages range from 40 to over 450 years (Conner et al. 2001). Likewise, older and larger trees are selected in greater proportion than their availability for foraging (Engstrom and Sanders 1997, Doster and James 1998, Zwicker and Walters 1999). Research suggests foraging habitat suitability is positively correlated with density of large pines, but negatively correlated with increased density of small pines or hardwoods, and height of hardwood midstory (Walters et al. 2002).

Federal and state lands accounted for almost 90% of known active RCW colonies in 2000 (USFWS 2003), largely due to employment of various management practices which produce favorable habitat and promote population stability or growth. Midstory control through prescribed burning and mechanical or chemical treatment, retaining the oldest and largest pine trees, increasing timber rotation, regenerating pine, and thinning pine stands have been essential in creating, improving, and maintaining open, pine-dominated habitat (Conner et al. 1995, Franzreb 1997, Provencher et al. 2001, Walters et al. 2002, USFWS 2003). Such habitat management techniques have been successfully used in conjunction with construction of artificial cavities or cavity inserts, and translocation of RCWs to stabilize or augment populations and even form new groups (DeFazio 1987, Copeyon 1990, Allen 1991, Copeyon et al. 1991, Franzreb 1997).

Although continued management of existing habitat and populations is critical, recovery is not attainable without increasing the number of potential breeding groups (PBG), necessitating the establishment of recruitment clusters (i.e., artificial cavities aggregated in high quality RCW habitat) and formation of new groups (USFWS 2003). Recovery efforts are currently focused on public lands, meaning private lands hold the greatest potential to further contribute to recovery. The Department of Defense (DOD), a major participant in RCW conservation, has begun exploring opportunities to engage private landowners in RCW recovery by providing financial incentive for landowners to create and manage RCW habitat through a Recovery Credit System (RCS). One such effort is currently under consideration in eastern North Carolina, where opportunities for RCW recovery on private lands remain relatively unexplored, but will become increasingly important as experts expect the Coastal North Carolina primary core population, which is distributed among three public properties, to approach maximum size by 2025 (USFWS 2003).

Inclusion of private lands in recovery efforts could provide opportunities to increase the amount of available RCW habitat, which could lead to development of recruitment clusters over broad spatial extents, and increase population size (USFWS 2003), with application of active management practices. Locating existing pine stands with potential to support RCWs is essential in assessing the feasibility and conservation value of pursuing recovery opportunities on private lands in the near future. The spatial and temporal context of recovery opportunities must be considered in evaluating where recovery on private lands is most likely to succeed. This study identifies potential RCW habitat and opportunities to contribute to RCW recovery through the establishment of recruitment clusters on private lands in the next 10 to 40 years.

1.2 RCW habitat modeling

Development of recruitment clusters requires sufficient area and quality of nesting and foraging habitat. While relying solely on field surveys to identify potential RCW habitat is not feasible for large extents (e.g. eastern North Carolina), it is possible to utilize remotely sensed spatial data and knowledge of RCW ecology to detect recovery opportunities where implementation of proven management techniques and strategies can provide adequate quality, quantity, and spatial aggregation of habitat to support RCWs.

Habitat modeling with geographic information system (GIS) and remote sensing technologies has been used extensively by natural resources managers to provide cost-efficient information for broad spatial extents. Until recently, much of this modeling has focused on classifying suitability from two-dimensional data for the species of interest (Vierling et al. 2008). However, because forest structure plays a prominent role in determining RCW habitat suitability, Light Detection and Ranging (LiDAR) lends itself to evaluating habitat. As a form of active remote sensing, LiDAR models surface features with pulse or continuous-wave lasers emitted from a GPS-enabled sensor by measuring the distance between the point where the laser reflects off objects (e.g. trees) and the sensor (Wehr and Lohr 1999). Over forested areas, this provides a 3-D model of canopy structure at spatially explicit locations. Because forest attributes including tree height, crown diameter, tree density, and biomass can be accurately generated or predicted from LiDAR (Wehr and Lohr 1999, Lim et al. 2003), its integration has improved habitat suitability modeling for various endangered bird and mammal species (Davenport et al. 2000, Hinsley et al. 2002, Bradbury et al. 2005, Nelson et al. 2005, Wilsey et al. 2012), effectively reducing dependence on obtaining field-based structural measurements (Vierling et al. 2008).

Several studies have explored the utility of LiDAR in modeling RCW habitat. Ability to evaluate key structural components that define suitable habitat (e.g. diameter at breast height [DBH], basal area by size class, and midstory structure) mainly depends on resolution and extent of available LiDAR and supplementary datasets. Although high density, small footprint LiDAR has been successfully used to evaluate RCW habitat structure, deriving the same metrics over large regions is limited when low density, large footprint LiDAR must be used, such as that collected for the North Carolina Floodplain Mapping Program (NCFMP; Tweddale and Newcomb 2011, Walters et al. 2011). Derivation of canopy height using the NCFMP dataset produced accurate results compared to field measurements on several accounts. Breckheimer (2012) found a strong correlation (R^2 = 0.57, RMSE of 3.65 m) between field and LiDAR canopy height estimates from pine forest plots throughout Camp Lejeune. Sexton et al. (2009) encouraged using the NCFMP dataset to estimate evergreen vegetation structure in North Carolina after finding a strong correlation (R^2 = 0.83, RMSE 4.18 m) between field and LiDAR pine heights in the Duke Forest. Tweddale and Newcomb (2011) and Walters et al. (2011) concluded the NCFMP dataset provided an acceptable mean predominant canopy height model (R^2 = 0.7) useful for evaluating RCW habitat suitability at regional extents.

Canopy height has been incorporated as an indicator of relative tree age or diameter in RCW habitat modeling efforts (Smart 2009, Breckheimer 2012). Tree age or tree size is currently the primary limiting factor inhibiting recovery; availability of older, larger trees is directly correlated to the number of PBGs (Conner et al. 2001, USFWS 2003). Given the physiological attributes of older trees which make them more suitable for excavating cavities (e.g. greater heartwood diameter and greater incidence of heartwood decay [Conner et al. 1995]) and foraging (e.g. increased biomass of arthropods [Hooper 1996]), tree age is an important component of RCW habitat suitability. Although tree height and size do generally increase with tree age, these relationships vary over broad regions, and depend on local site conditions. A different modeling approach, as proposed here, is needed to account for variations in age and height relationships in the process of identifying potential habitat.

1.3 Tree age modeling

Few efforts to directly model tree or stand age using LiDAR are apparent regarding RCW habitat identification. Walters et al. (2011) attempted to predict stand age from LiDAR-derived canopy height of 800 stands at Fort Bragg, North Carolina, but obtained a weak correlation (R^2 = 0.22) between the two variables. Smart (2009) determined the proportion of small-sized trees in stands using LiDAR, and then predicted stand birth year from those proportions ($R^2 = 0.61$). Tweddale and Newcomb (2011) and Walters et al. (2011) suggested using a LiDAR-derived predominant canopy height model (CHM), land cover products, and site index to estimate relative stand age.

Tree height growth depends on local environmental factors (e.g. soil characteristics, climate, etc.) that determine site quality, but it is generally unaffected by density, species composition, or thinning intensity of a stand (Avery and Burkhart 1994). Site index, determined by the relationship between "average total height and age of dominant and codominant trees in well-stocked, even-aged stands," is often used in forestry as a predictor of site productivity or quality (Avery and Burkhart 1994: 279). Several studies have explored using site index, LiDAR, and growth curves to predict tree age for alternative applications. Farid et al. (2006) used small footprint LiDAR metrics to classify cottonwood trees as young, mature, or old. Weber and Boss (2009) incorporated LiDAR-derived canopy height in classification of forest as <= 30 years old, 30-70 years old, or >70 years old, and then applied such classifications to aid conservation decision making. Finally, Stukey (2009) used a field-generated growth curve and expected site index from United States Department of Agriculture (USDA) Natural Resources Conservation Service (NRCS) Soil Survey Geographic (SSURGO) Database soils data to accurately predict (R^2=0.9873) average plot age from LiDAR-estimated height for pine trees. These studies suggest it is possible to model tree age to evaluate RCW habitat suitability.

This study involved modeling current expected pine tree age, using the method recommended by Tweddale and Newcomb (2011) and Walters et al. (2011), to forecast where and when existing pine stands could support recruitment clusters with implementation of necessary habitat management strategies (e.g. prescribed burning, thinning, etc.). Spatial modeling of recovery opportunities is particularly important because spatial distribution of habitat greatly effects RCW populations due to the need for large amounts of contiguous habitat and the relatively short dispersal distances characteristic of the species (Cox and Engstrom 2001). Fragmentation, isolation, and low availability of habitat negatively affect population demographics and impede creation of new PBGs (Conner and Rudolph 1991, Rudolph and Conner 1994, Letcher et al. 1998). Consideration of habitat spatial configuration from a regional perspective is more likely to result in short-term population increases and support long-term population survival (Huxel and Hastings 1999, Cox and Engstrom 2001, USFWS 2003). Region-wide spatial modeling of tree age composition is valuable in spatially and temporally locating recovery opportunities, while also allowing assessment of connectivity between potential recruitment clusters. Understanding where and when recovery opportunities exist, and where potential recruitment clusters are most likely to successfully contribute to population stability or growth, both over time and space, is essential in ensuring recovery efforts on private lands are effective as recovery limits are met on public lands.

1.4 Research objectives

The goal of this study was to identify recovery opportunities for RCW on private lands in eastern North Carolina by locating and quantifying potential RCW habitat that can be managed to become suitable habitat within 10 to 40 years, assuming tree age is currently the most limiting factor inhibiting habitat suitability, and therefore, population growth. This specifically involved:

1. Spatially modeling tree age to determine location of potentially suitable habitat within 10, 20, 30, and 40 years, assuming habitat management practices aimed at achieving RCW habitat suitability will occur,
2. Identifying potential recruitment clusters that could support RCW within 10 years of active management, using USFWS guidelines for nesting and foraging habitat, and
3. Assessing connectivity between potential short-term cluster sites (i.e., potential recruitment clusters within 10 years) to guide the pursuit of conservation efforts across the study area.

2. METHODS

2.1 Study area

Recovery opportunities were identified in 26 eastern North Carolina counties, where private lands account for almost 90% of 4 million hectares (Fig. 1). All but two counties in the study area historically contained RCWs (Jackson 1971, Hooper et al. 1980); 16 counties contained RCWs as of 2002 (USFWS 2002). Within this area, all properties containing the Coastal North Carolina Primary Core (i.e., Croatan National Forest, Holly Shelter Game Lands, and Marine Corps Base Camp Lejeune) population, and all but one property (i.e., Pocosin Lakes National Wildlife Refuge) containing the Northeast North Carolina/Southeast Virginia Essential Support (i.e., Alligator River National Wildlife Refuge, and Dare County Bombing Range) population are expected to reach property goals for number of active RCW clusters by 2025 (USFWS 2003), meaning additional recovery must eventually occur on private lands. Longleaf and loblolly, in addition to pocosins, dominate pine systems associated with this region. The Middle Atlantic Coastal Plain, Southeastern Plains, and Piedmont ecoregions cover approximately 64%, 34%, and 2% of the study area, respectively. Elevation ranges from sea level to 398 feet above sea level (Fig. 2).

2.2 Tree age model development

A 30-m resolution tree age model for longleaf and loblolly pine was created for the study area based on relationships between age, canopy height, and site index, using

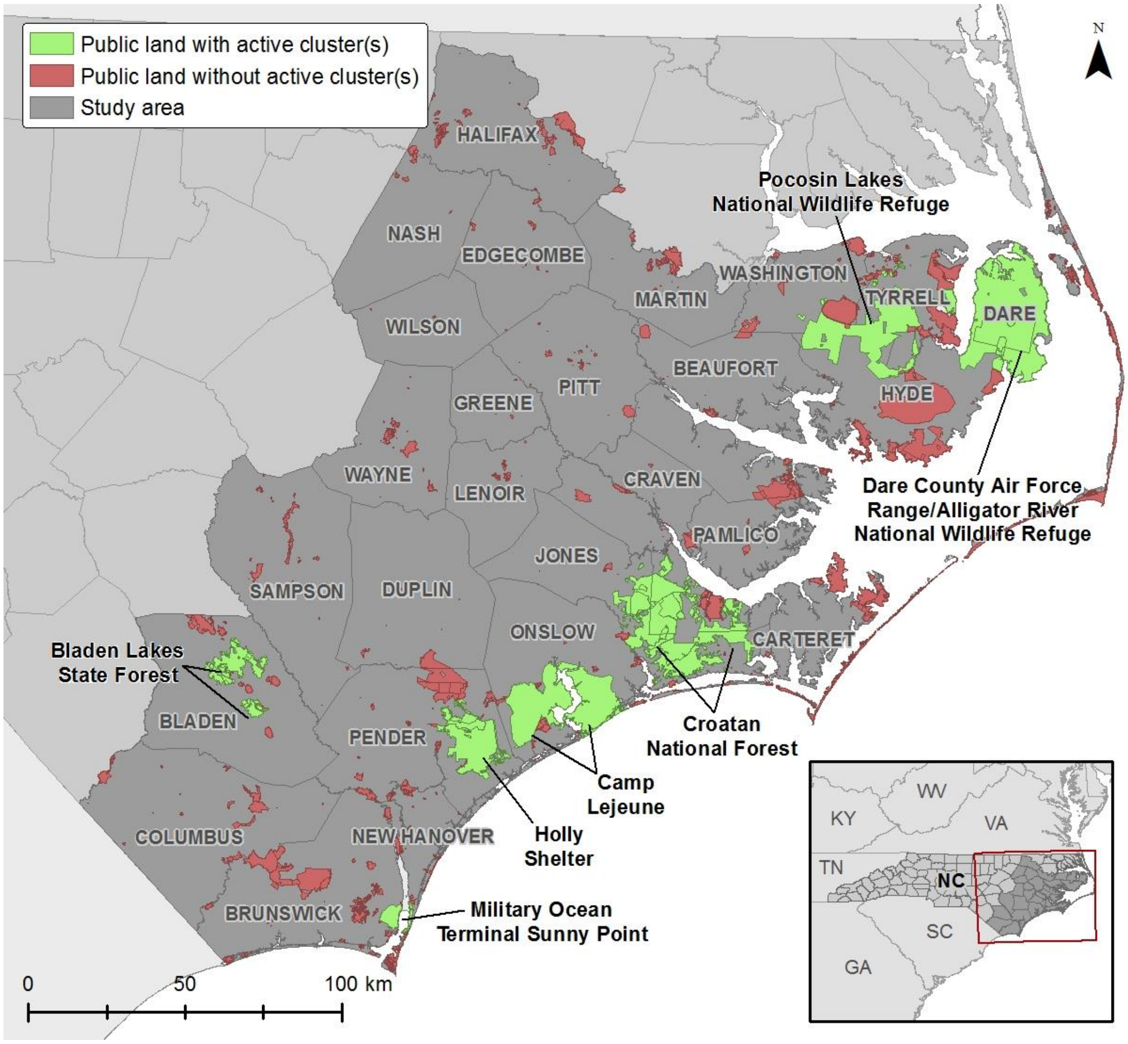

Figure 1. Study area for assessment of potential red-cockaded woodpecker (*Picoides borealis*) habitat on private lands in 26 eastern North Carolina, USA counties, 2012. Public lands with active clusters correspond to those identified in the 2003 USFWS Recovery Plan.

ArcMap (Esri 2012). LANDFIRE Existing Vegetation Type (2006) and SEGAP (2008) land cover datasets were used to locate longleaf and loblolly pines. Classifications of managed pines or evergreen plantations were assumed to represent loblolly pine, the dominant plantation species in this region. Other pine species occasionally used by RCW were not specifically excluded from consideration. Land cover data did not

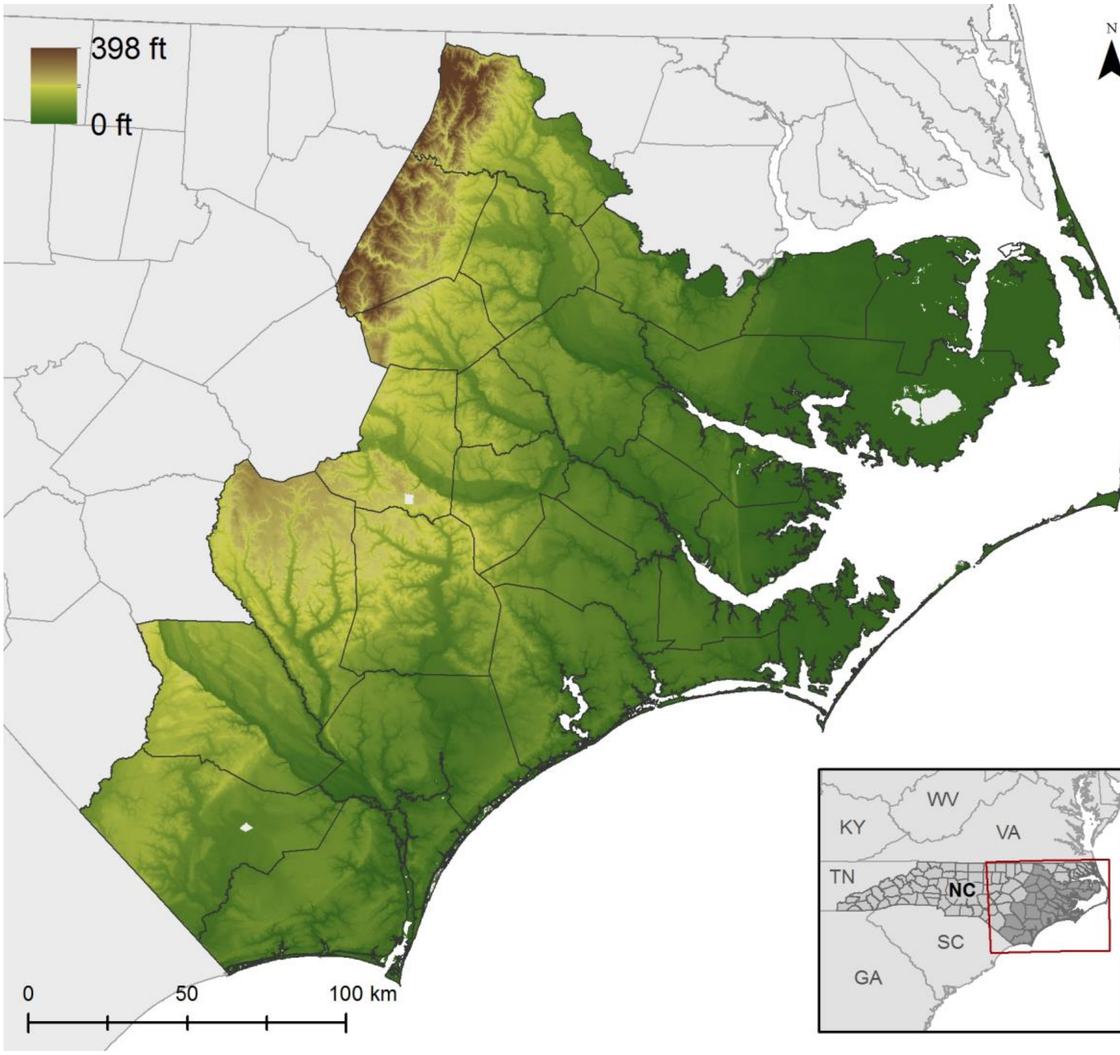

Figure 2. Elevation, in feet, for the study area in eastern North Carolina, USA.

distinguish these other species, which are not major components of the study area, and site index data for these species was limited. Incorporation of the 2012 Cropland Data Layer developed by the National Agricultural Statistics Service eliminated non-forested areas from the model, providing a more current representation of forest distribution than provided by LANDFIRE and SEGAP datasets alone. Modeling relied only on existing land cover products, and did not account for potential changes (e.g., climate change) in the landscape that could alter future pine distribution.

Bare-earth digital elevation models (DEM) and raw discrete-return large footprint LiDAR data acquired from the NCFMP from January to March of 2001 and 2003 were processed in FUSION (McGaughey 2012) to create a seamless DEM and maximum canopy surface model (CSM). A maximum CHM was created by subtracting the DEM from the CSM. Visual inspection of the CHM led to the exclusion of eleven large areas, totaling approximately 126,310 ha, from the model. These generally rectangular-shaped areas contained height values extremely greater than surrounding areas, where height values were expected to remain similar based on comparison with aerial imagery.

The Soil Data Viewer (NRCS 2012*a*) ArcMap extension was used to query and produce an expected site index vector layer by county for longleaf and loblolly pine using USDA NRCS SSURGO soils spatial and tabular data, available from the Web Soil Survey (NRCS 2012*b*). Site index was assigned based on North Carolina Forest Service (2011) estimates by soil name for locations lacking a SSURGO-provided estimate, when possible. For locations where land cover layers identified pine, but expected site index was not available, the average site index value associated with longleaf or loblolly for the entire study area was applied.

Expected height of dominant and co-dominant trees was calculated using formulated growth curves provided by Carmean et al. (1989) for each expected site index value pertaining to the study area at ages 10, 20, 30, 40, 50, 60, and 70. Growth curves are based on original site index curves created by USDA (1929) for longleaf pine, and Schumacher and Coile (1960) and Coile and Schumacher (1964) for loblolly pine. Although many growth curves have been developed for both species, these were assumed to be most consistent with SSURGO site index values because they are provided by the same respective authors, and include plot sites from North Carolina in growth curve development. Recently developed growth curves display a similar growth pattern to these, and mainly focus on improving projected growth at ages too young to be relevant for the timeframes considered in this study. The CHM was reclassified so that height ranges were assigned an age (e.g., a longleaf pine at site index 60 is predicted to be 44 feet tall at age 40, and 53 feet tall at age 50; tree heights between 44 and 53 feet at locations with a site index of 60 were classified as 40 years old). Ages in the model were adjusted to range from 20 to 80 years, in increments of 10 years. This represented a present-day predicted age, accounting for the approximately 10 years since LiDAR collection. Separate models for longleaf and loblolly were made in raster format, and then combined to create a comprehensive model of the study area.

2.3 Model validation data

Field-measured tree age data was not available for model validation. Instead, field-collected maximum DBH was compared to expected DBH derived from maximum modeled age in 48 parcels. Expected relationships between DBH, tree age, and site index were derived from U.S. Forest Service Forest Inventory Analysis Program (FIA) data for longleaf and loblolly pines from coastal plain and piedmont regions of Alabama, Florida, Georgia, Louisiana, Mississippi, North Carolina, South Carolina, and Virginia. Multiple Southeastern states were used to increase sample size and represent various growing conditions.

FIA DBH measurements were summarized by site index and age classes corresponding to those used in the tree age model. Summarizing FIA data made it comparable to the model, and ensured a higher sample size for older tree ages. Expected ranges of DBH were calculated using the average DBH plus or minus one and two standard deviations for each combination of site index and age class. This exact method for model verification was not found in a literature search, however, a similar method of binning plot-level FIA data into age classes has been used to compare against LiDAR-based models of aboveground biomass (Lefsky et al. 2005), and FIA data has been used to develop diameter growth models for stands during the years forest inventories do not occur (Lessard et al. 2001). Although DBH varies based on local conditions, it increases with age, and its growth can be predicted over short periods of time (Avery and Burkhart 1994). Summarizing FIA DBH data in this way provided a substitution for field-measured age to determine whether modeled age values were realistic.

Zonal statistics were used to extract the maximum model value for each parcel (i.e., this identified the cell with the highest age value, and then the highest site index if a tie occurred for age). Comparisons between this extracted value and field-observed DBH were made by:

1. Using modeled site index and age to identify the expected DBH range, according to summarized FIA data, for a surveyed parcel (Fig. 3). If the field-observed DBH fell within this expected range, modeled age was considered acceptable.
2. Classifying expected age (i.e., using field-observed DBH and FIA-derived relationships between DBH and age) and modeled age into suitability timeframes (Fig. 4). Tree ages 20-29, 30-39, 40-49, and 50 or greater were equivalent to suitability timeframes of 40, 30, 20, and 10 years, respectively. This converted tree age into a minimum expected time required for a tree to develop into a potentially suitable cavity tree (e.g., trees age 50 or older are considered potentially suitable cavity trees within 10 years). If both expected and modeled ages converted to the same timeframe, modeled age was considered acceptable.

The model's ability to predict locations of stands old enough (i.e., age 60 or older) to support RCW nesting habitat was also assessed by comparing it to known locations of active and inactive RCW clusters within Camp Lejeune, Croatan National Forest, and Holly Shelter Game Land. To do this, the maximum modeled tree age within 100 m of each cluster was identified.

2.4 Identification of recruitment clusters

The tree age model aided in identification of stands with potential to support recruitment clusters within 10 years (i.e., potential short-term clusters), assuming active habitat management occurs to address midstory and understory structure criteria. These

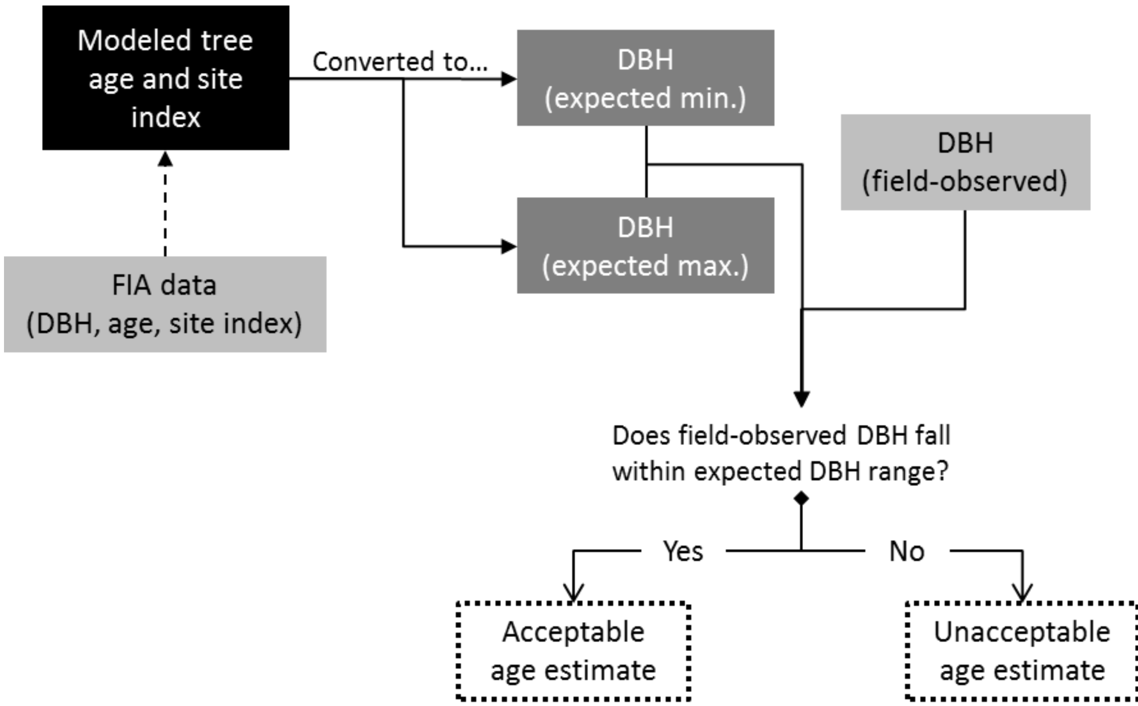

Figure 3. Validation of modeled tree age by comparing expected and observed diameter at breast height (DBH). Relationships between age, site index, and DBH observed in Forest Inventory Analysis (FIA) data were summarized to determine expected DBH values (i.e., minimum and maximum expected DBH within one and two standard deviations of the mean, for specific age and site index classes) for a given location based on modeled age and site index. Modeled age estimates were deemed acceptable for surveyed parcels when field-observed DBH fell within the expected DBH range derived from FIA data. For example, longleaf DBH on a parcel with a modeled age of 70 and site index of 60 was expected to be between 12 and 16 inches. If field-observed DBH was between 12 and 16 inches, this parcel was deemed to contain an acceptable age estimate.

represented potential translocation sites for RCW group establishment in the near future where spatial modeling suggested potential habitat meets tree age and area requirements. The USFWS Recovery Plan (2003) states a cluster, defined as the minimum convex polygon (MCP) surrounding a group's cavity trees and the 61 m forested buffer around the MCP, should be at least 4.05 ha in size. USFWS (2003) recommends cavity trees be at least 60 years of age, and foraging trees be at least 30 years of age.

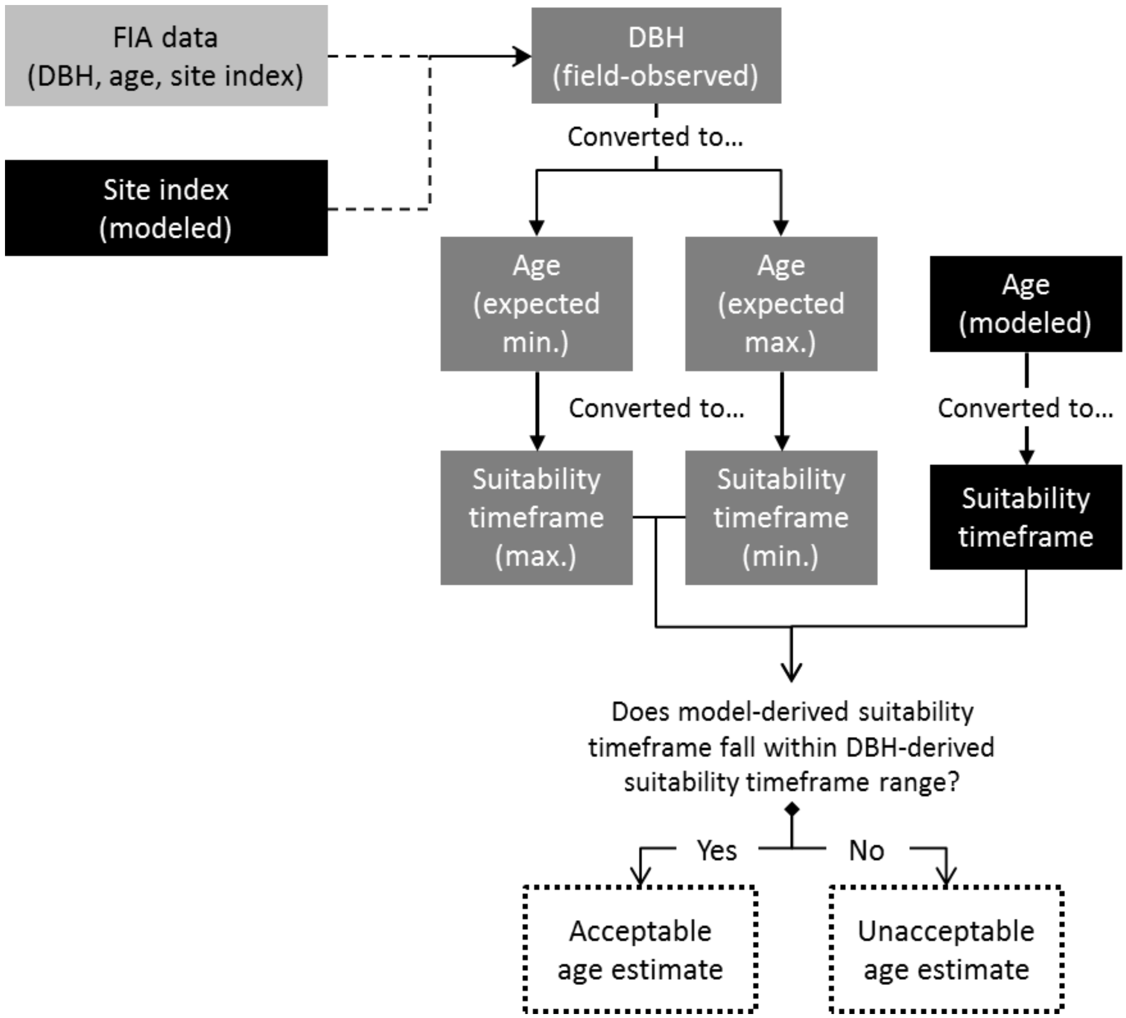

Figure 4. Validation of modeled tree age by converting modeled age and field-observed diameter at breast height (DBH) into suitability timeframes. Relationships between age, site index, and DBH observed in Forest Inventory Analysis (FIA) data were summarized to determine expected age from field-observed DBH. Expected age derived from DBH, as well as modeled age, was converted into suitability timeframes (i.e., expected time required for a tree to develop into a potentially suitable cavity tree, or reach age 60; age 50 was converted to a suitability timeframe of 10 years, age 40 was converted to a timeframe of 20 years, etc.). Modeled age estimates were deemed acceptable for surveyed parcels when suitability timeframes derived from modeled age were between expected minimum and maximum suitability timeframes derived from field-observed DBH. For example, a parcel with a maximum DBH of 18, and modeled site index of 90, was expected to be suitable within 10 to 20 years. If modeled age for this parcel was at least 40, equivalent to a suitability timeframe of 20 years or less, modeled age was deemed acceptable for this parcel.

Potential short-term cluster sites were defined as stands age 50 or older, with a core area of at least 8,442 m^2. The core area was measured after applying an inside buffer of 61 m to stands meeting the age criterion, ensuring stands met the 4.05 ha size requirement regardless of stand shape, assuming cavity trees are positioned to create an adequately-sized MCP. USFWS offers no specific guidelines concerning cluster shape complexity aside from requiring the buffer area. Because this analysis used a minimum core area based on a circle (i.e., the shape with the lowest perimeter to area ratio), stands with a higher perimeter to area ratio may have been excluded even though they have an adequate total area and buffer. The number and connectivity of potential sites could be underestimated in some cases. Research suggests RCW prefer habitat with less total edge (Cox et al. 2001), meaning complexly-shaped stands not identified in this analysis could be less suitable candidate sites for recruitment clusters.

Stands containing at least 24.5 ha of 20 year-old stands within 0.4 km of the cluster center, and at least 49 ha within 0.8 km of the cluster center were considered potential future cluster sites, meeting minimum standards concerning quantity and age of habitat (USFWS 2003). Remaining stands were converted to a point shapefile, representative of the stand centroid. This did not provide a total number of potential clusters, rather, a total number of stands with potential to support at least one cluster, and therefore, one PBG.

2.5 Connectivity assessment

Isolation of RCW groups negatively contributes to population size and genetic flow between populations. Connectivity between potential sites was assessed using a buffer method similar to that employed by Conner and Rudolph (1991) to measure isolation of RCW groups. Although simplistic and sensitive to distance, buffer measures have been shown to reasonably indicate connectivity of a landscape without requiring complex input data (Moilanen and Nieminen 2002). Because precise delineations of stands and clusters were not available, this method was preferable for this study.

The number of potential or active cluster sites within a 4 km radius of each recruitment stand centroid (i.e., potential future cluster site) was summed. This distance accounts for approximately 90% of foray and approximately 60% of dispersal events observed for female juveniles leaving their natal territory on Marine Corps Base Camp Lejeune and in the North Carolina Sandhills (Kesler et al. 2010, Walters et al. 2011). Larger totals were assumed to translate to greater potential connectivity between sites, providing more opportunities for RCW females, which are more likely than males to disperse from the natal territory (Conner et al. 2001), to interact with other groups and find breeding vacancies. Only Euclidean, not functional, distance was used in analysis. Varying habitat gap crossing behavior and dispersal distances by age and sex was not considered, although these have been shown to influence landscape connectivity as perceived by RCW (Moody et al. 2011, Walters et al. 2011). Recent telemetry data suggest there is no significant difference between used and unused paths between territories in regard to forest type (i.e., pine or hardwoods) or habitat quality (Walters et al. 2011). Forest gaps were not considered in connectivity analysis due to the limited ability of medium resolution land cover data to delineate pine stands, and distances between forest patches generally being less than the 500 m maximum forest gap crossing distance observed for dispersing female juveniles (Walters et al. 2011).

3. RESULTS

3.1 Model validation

Compared to field surveys, the model correctly identified species for 20 of 35 (57%) parcels with stands identified primarily as longleaf or loblolly. Of those correctly identified, 18 were longleaf and 2 were loblolly. Of those incorrectly identified, 3 were longleaf and 12 were loblolly. Only correctly identified parcels were used to compare observed and expected DBH using the complete tree age model. Parcels identified as primarily longleaf or loblolly were further validated after dividing the tree age model into separate models for each species (i.e., to ensure the correct species age value was extracted). Twenty-one longleaf and 14 loblolly dominated parcels were compared to species-specific models. Thirteen parcels with mixed stands were compared against all three models.

Over 70% of parcels in each species-model combination contained a field-measured DBH that was within the expected range at two standard deviations, calculated from modeled tree age and site index, and corresponding FIA data (Fig. 5). When comparing DBH at one standard deviation, observed and expected DBH matched for 36-81% of parcels. Modeling produced better results for parcels containing predominantly longleaf, compared to those containing loblolly and mixed stands. In all cases where observed DBH was not within the expected DBH range for longleaf stands, observed DBH was higher than expected. In cases where observed and expected DBH did not agree for loblolly stands, observed DBH was typically higher than expected. Observed

DBH in mixed stands was typically higher than expected when zonal statistics associated loblolly with the maximum age value, and lower than expected when longleaf accounted for the maximum age value.

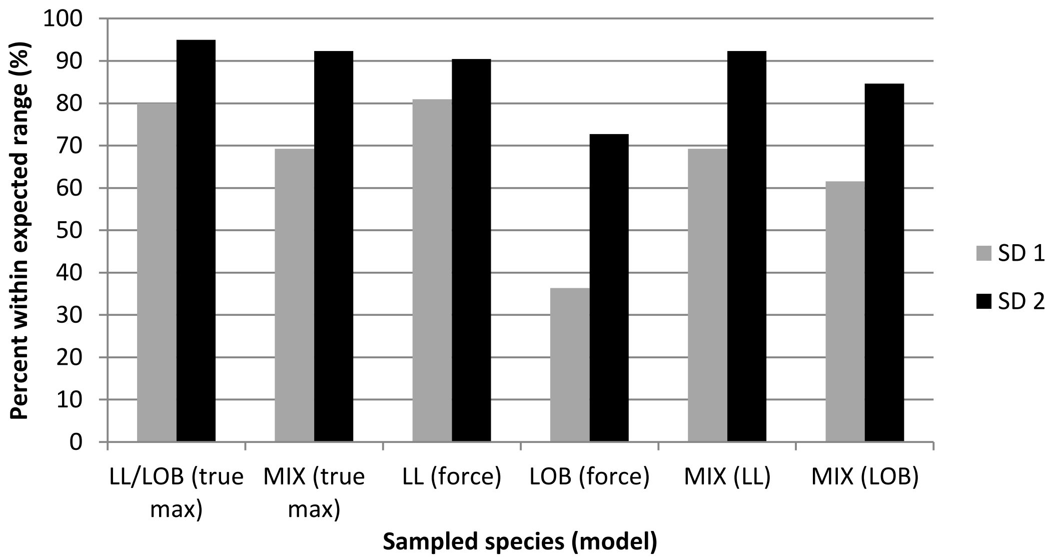

Figure 5. Percent of parcels with acceptable modeled age values for parcels surveyed in eastern North Carolina, USA in 2011. Field-observed diameter at breast height (DBH) was compared against expected DBH range, calculated within one and two standard deviations (SD) of mean DBH according to Forest Inventory Analysis data summarized by site index and age. Modeled age and site index were used to determine applicable DBH range for each parcel, based on relationships between DBH, age, and site index. Comparisons were made for parcels with predominantly longleaf (LL; *Pinus palustris*), loblolly (LOB; *Pinus taeda*), and mixed (MIX) stands using the complete tree age model (true max), longleaf-only model (LL), and loblolly-only model (LOB). Only parcels with correctly identified species were compared to the complete tree age model. Column labels indicate dominant species, followed by the model used for comparison in parentheses.

Predictive results improved when converting observed DBH and modeled age to expected suitability timeframes (Fig. 6). Suitability timeframes predicted from the model produced acceptable results for 45-92% of parcels at one standard deviation, and 82-95% at two standard deviations. In cases where suitability timeframes did not match, modeled suitability was typically later than expected for longleaf and loblolly stands. For mixed stands, modeled suitability was later than expected when using the loblolly model, but sooner than expected when using the other two models.

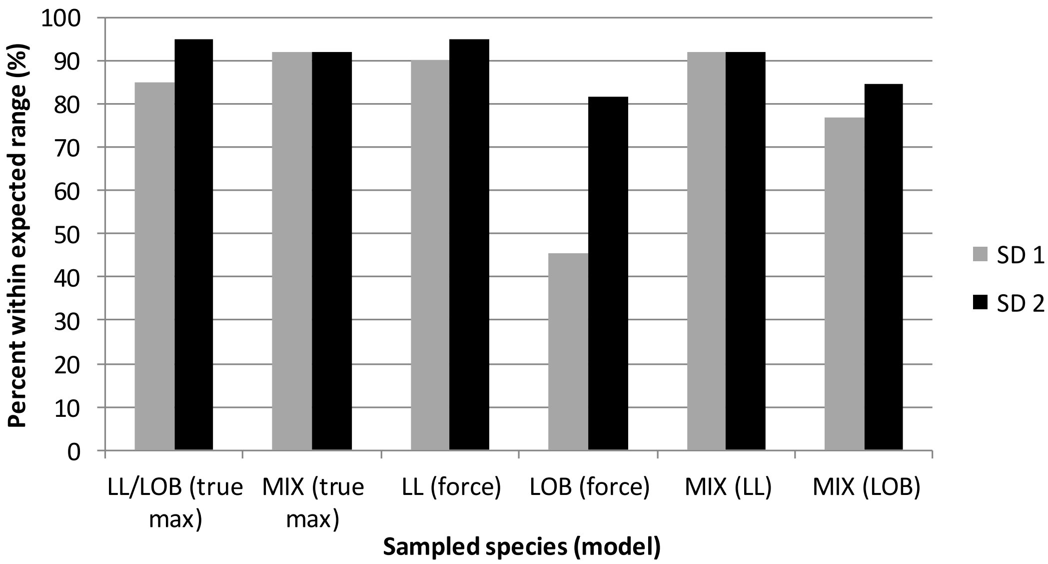

Figure 6. Percent of parcels with acceptable suitability timeframe predictions for parcels surveyed in eastern North Carolina, USA in 2011. Modeled age and expected age (i.e., determined from field-observed diameter at breast height [DBH], and one and two standard deviations [SD] of mean DBH according to Forest Inventory Analysis data summarized by site index and age) were converted to a suitability timeframe so that ages 20-29, 30-39, 40-49, and 50 or greater were equivalent to suitability timeframes of 40, 30, 20, and 10 years, respectively. If both expected and modeled ages converted to the same timeframe, the modeled estimate was considered acceptable. Comparisons were made for parcels with predominantly longleaf (LL; *Pinus palustris*), loblolly (LOB; *Pinus taeda*), and mixed (MIX) stands using the complete tree age model (true max), longleaf-only model (LL), and loblolly-only model (LOB). Only parcels with correctly identified species were compared to the complete tree age model. Column labels indicate dominant species, followed by the model used for comparison in parentheses.

The model identified 98 out of 192 (51%) active and inactive cluster locations as having trees of age 60 or older within a 100 m radius (Fig. 7). Over 90% of clusters were modeled to contain trees age 40 and older.

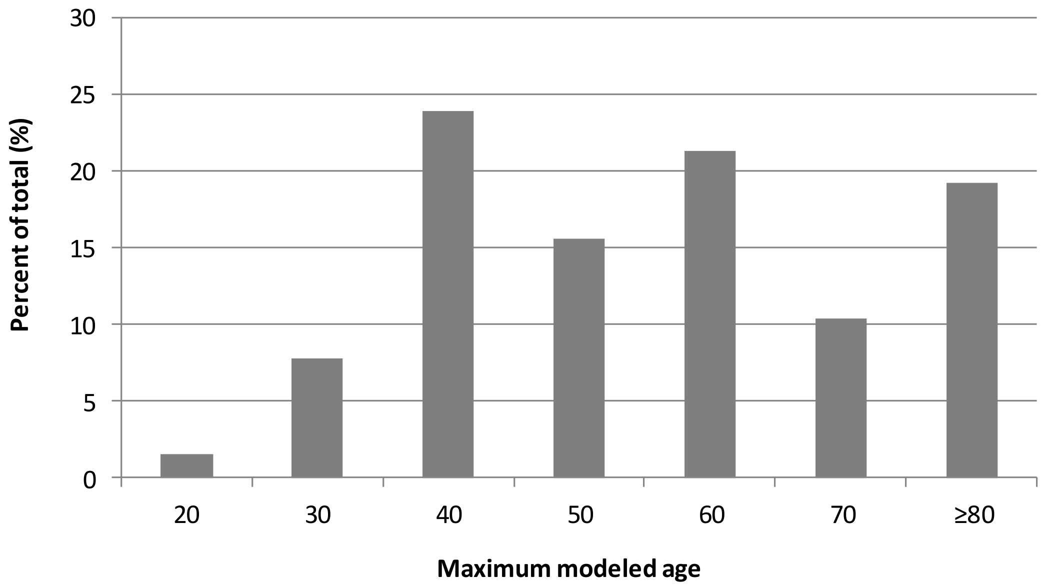

Figure 7. Distribution of modeled maximum tree age near Onslow Bight, North Carolina, USA red-cockaded woodpecker (*Picoides borealis*) clusters detected in 2008. Maximum tree age was extracted within 100 m of all active and inactive clusters.

3.2 Potential RCW habitat

According to tree age modeling, concentrated short-term opportunities for habitat restoration primarily occur to the north of the Croatan National Forest toward Dare County, and between Bladen Lakes State Forest and Holly Shelter Game Land (Fig. 8). Younger stands are noticeably more abundant in this landscape (Fig. 9). Approximately 20% of modeled pines are estimated to be age 50 or older, the minimum age considered suitable within 10 years for cavity trees. Only 4% of pines are predicted to be at least 80 years of age. Approximately 60% of modeled pines are currently old enough to support foraging habitat.

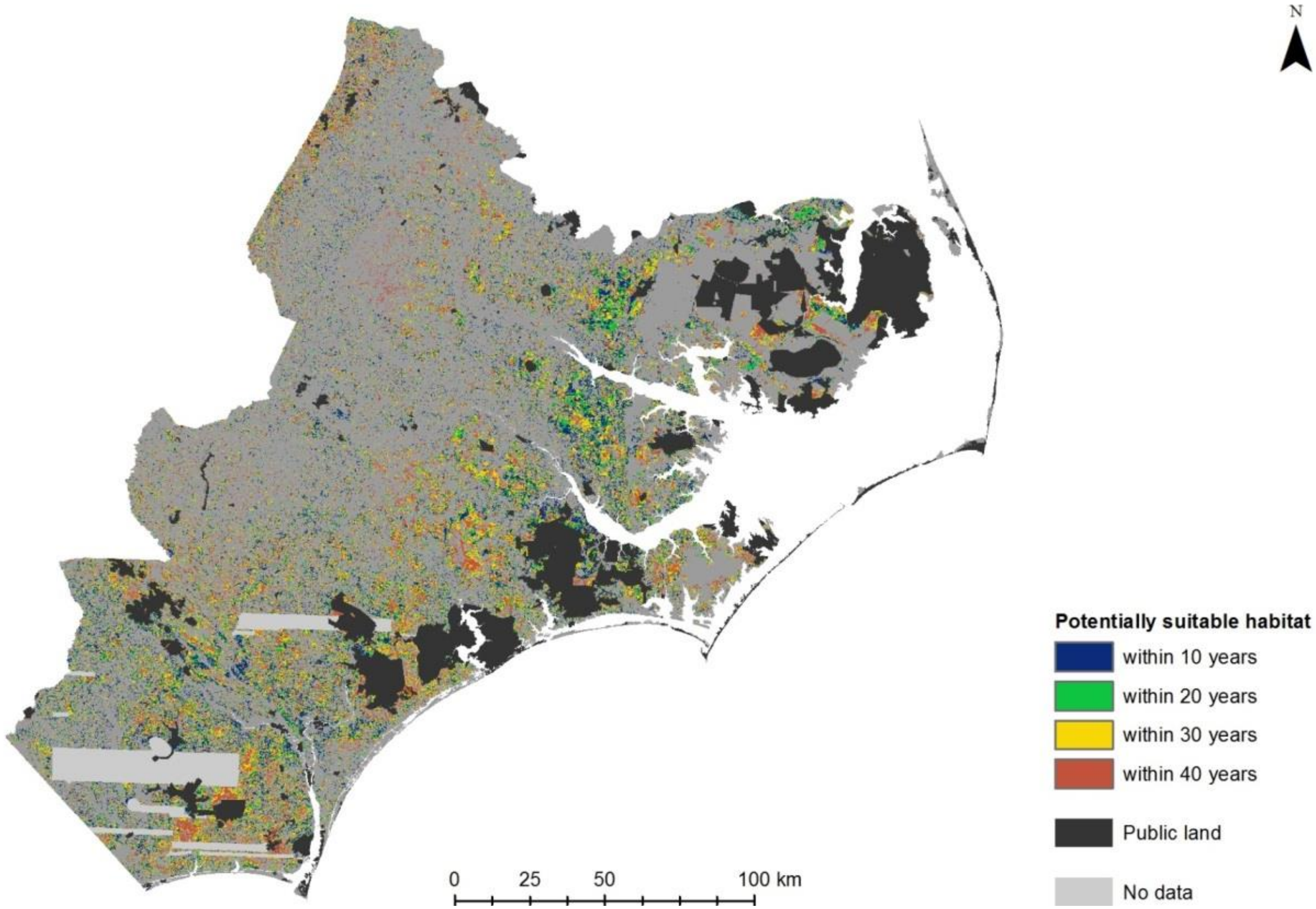

Figure 8. Potentially suitable red-cockaded woodpecker (*Picoides borealis)* habitat on private lands in eastern North Carolina, USA, 2012. Suitability is based on modeled tree age; ages 50 or older, 40, 30, and 20 were considered potentially suitable within 10, 20, 30, and 40 years, respectively.

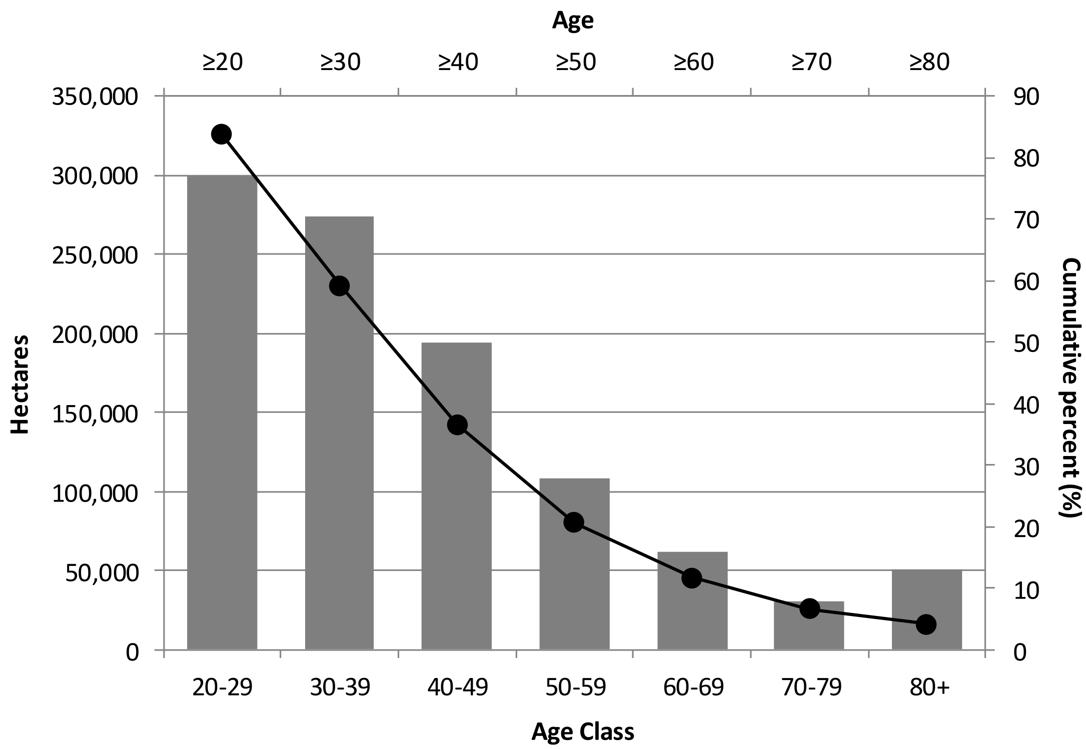

Figure 9. Modeled age distribution of longleaf (*Pinus palustris*) and loblolly (*Pinus taeda*) pine on private lands in eastern North Carolina, USA, 2012. Bars represent estimated area (hectares) of pines by age class. Points represent cumulative percentage of pine area by age, where percent accumulates from older to younger ages (right to left). For example, 21% of pines are estimated as age 50 or older, 12 % are estimated as age 60 or older, etc.

3.3 Potential future cluster sites

Beaufort, Pamlico, and Pender counties had the highest densities of potential clusters. Beaufort, Bladen, and Pender counties contained 34% of all potential sites identified (Fig. 10). Almost 3,450 sites were identified as potentially able to support recruitment clusters within 10 years. Nearly 23% of these were within 20 km (i.e., maximum natal dispersal distance observed for females on Marine Corps Base Camp Lejeune [Walters et al. 2011]) of active clusters in the Onslow Bight, suggesting the populations at these sites could interact with populations on public lands (Fig. 11).

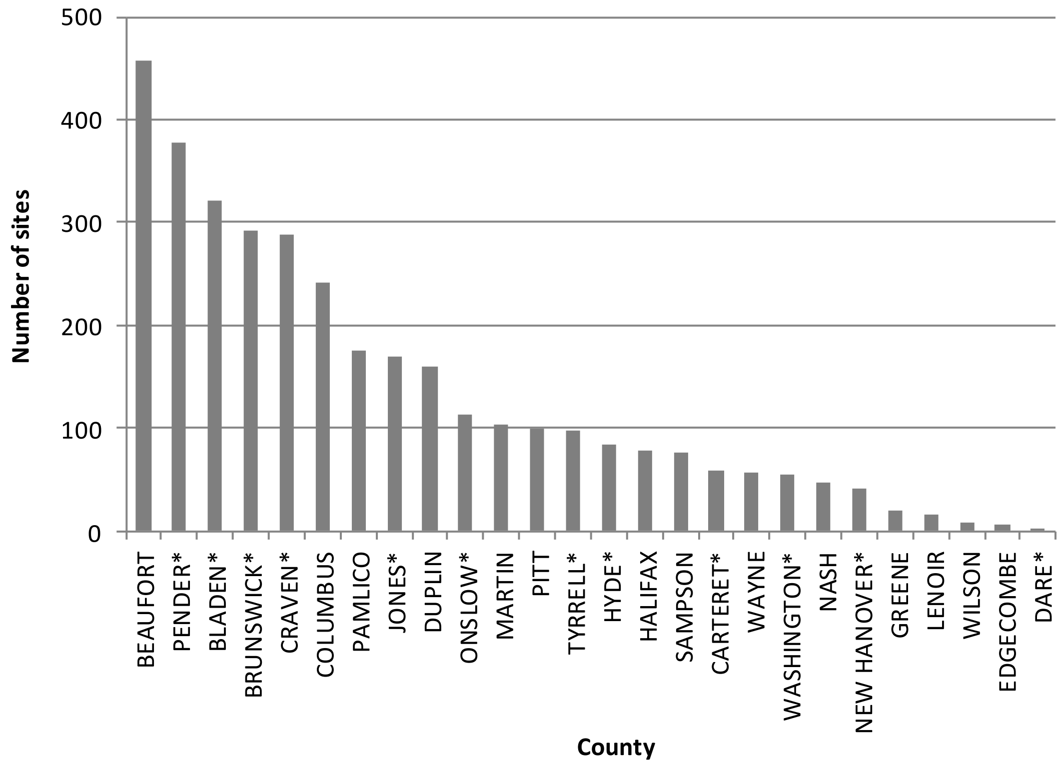

Figure 10. Number of sites with potential to support at least one red-cockaded woodpecker (*Picoides borealis*) recruitment cluster within 10 years in eastern North Carolina, USA, 2012. Counties with at least one active cluster are denoted with an asterisk (*).

Less than 3% of potential sites were not located within 4 km (i.e., centroid to centroid distance) of at least one other potential or existing cluster site. Eighty-two percent had at least 5 other sites within 4 km, and 56% had at least 10 sites within 4 km. Potential cluster sites were primarily connected to other potential cluster sites, and existing sites were primarily connected to other existing sites. On average, each existing cluster was within 4 km of approximately 16 other existing clusters, but only two potential sites, while each potential site was within 4 km of approximately 12 other potential sites, but only one existing cluster.

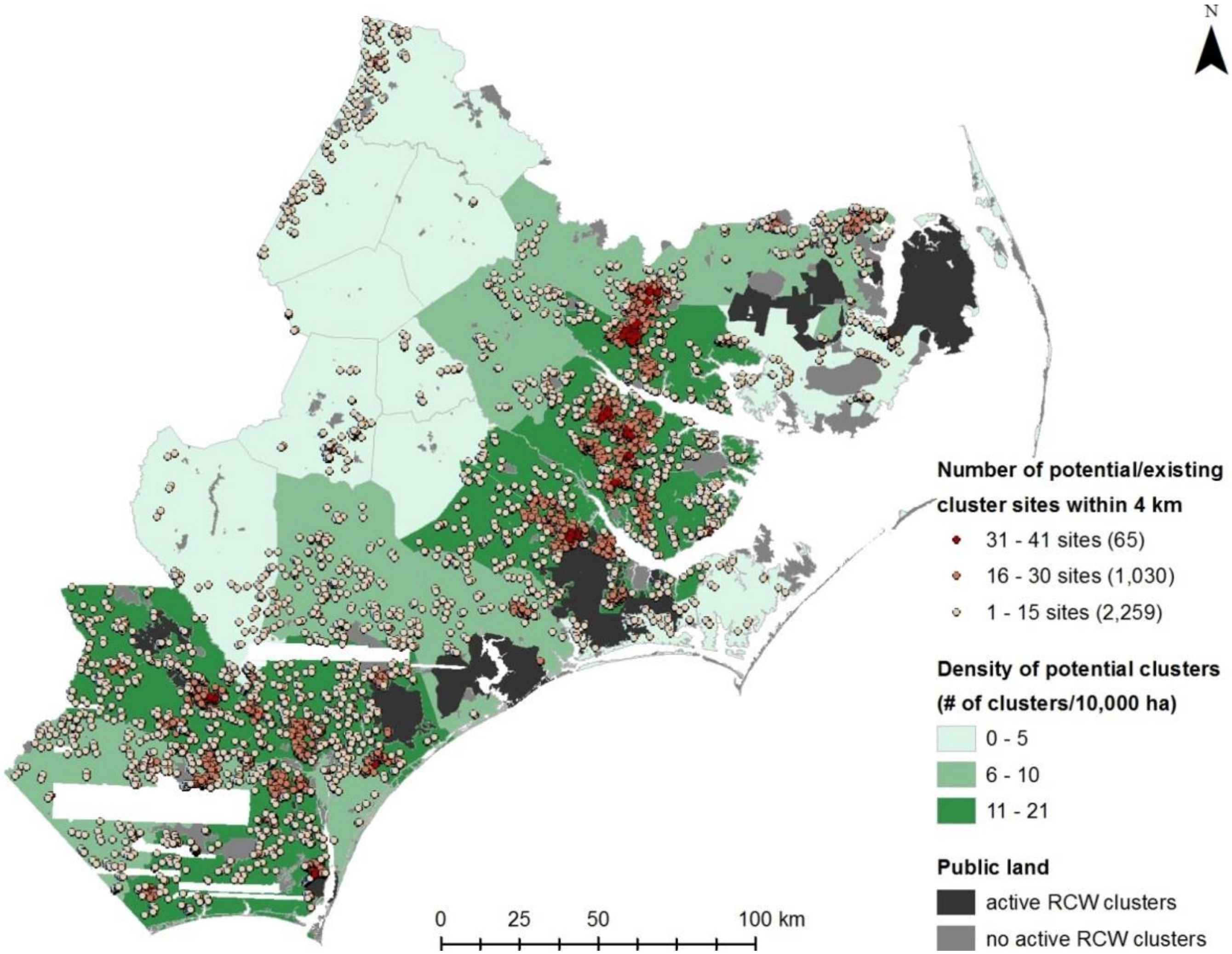

Figure 11. Density and connectivity of potential red-cockaded woodpecker (*Picoides borealis*) recruitment cluster sites in eastern North Carolina, USA, 2012. Potential sites were identified based on USFWS guidelines concerning nesting and foraging habitat.

4. DISCUSSION

4.1 Tree age model performance

Comparison of the tree age model to field surveys suggests the model does provide a reasonable estimate of when stands could become suitable RCW habitat. This assumes the most limiting suitability factor is tree age or tree size, and active management would be employed to improve insufficient midstory and understory habitat in a timely manner. Although the model mostly produced acceptable age and suitability timeframe predictions according to active and inactive cluster locations and field surveys, the spatial and temporal scale of data used to build the model must be considered. This model provides a useful landscape-level guide for further investigation of properties with potential RCW habitat.

In cases where observed DBH was higher than expected, or tree age was predicted to be less than age 60 for existing clusters, it is possible that tree age was underestimated. This is likely due to the underestimation of canopy height characteristic of low sampling density LiDAR available for the study area, or site indexes that were lower than expected (Walters et al. 2011). Tree age could be less than 60 years at some locations where artificial cavities are used in existing clusters; however, data was not available to confirm this. Age is likely overestimated at sites with higher than expected soil site indexes, especially affecting estimations for intensively managed plantations, or sites where stands were harvested between the time of LiDAR data collection and field surveys. Converting modeled age into a suitability timeframe led to a higher percentage of acceptable values, as expected. Although doing so requires a less accurate prediction of age, it nonetheless is useful in assessing potential recovery opportunities over space and time.

Due to field survey efforts being focused on finding existing RCW clusters and currently suitable habitat (i.e., older, larger trees) on private lands, comparisons with surveys do not fully assess to what extent the model could overestimate age and potential suitability. For surveyed parcels, trees were typically 15 inches DBH, and maximum modeled age was mostly 50 or older, meaning limited comparisons were made to stands predicted to be relatively young. Further model validation should include comparison to field-measured tree age across all age and site index classes, and include areas not expected to currently have suitable habitat, to provide a better measure of model accuracy.

Because canopy height, site index, pine species identification, and growth curves were used in conjunction at a large scale to produce the model, improvements or updates for any of these datasets will lead to improvement in the model's ability to predict locations and ages of stands. Particular assumptions and limitations are associated with each dataset used to build and validate the model:

1. LiDAR: Updated, finer-resolution LiDAR would provide a better representation of stand location, improved estimates of tree height and age, and allow an assessment of midstory and understory structure, which are important in determining RCW habitat quality. Improved LiDAR could also be used to estimate tree DBH, providing better opportunities to identify trees suitable for artificial cavity inserts.

2. Land cover: SEGAP and LANDFIRE layers did not always predict the presence of pines for the same locations, nor did they always consistently or correctly differentiate between species (e.g., pine versus hardwood, or longleaf pine versus loblolly pine). Better ability to distinguish between pine species could improve age estimation by ensuring application of the appropriate growth curve and site index. Because a pixel can only represent longleaf or loblolly in the model, misidentification of species leads to inaccurate age estimates, which could explain poorer model performance for loblolly pine. Land cover data modeled predominantly longleaf or loblolly ecosystems; however, RCW have been observed using other pine species.
3. Site index: Expected site index was based on soil characteristics, but was not available for all locations identified as longleaf or loblolly pine. Because site index varies according to local conditions (i.e., past land use, management practices, erosion, etc.; Barry 2011), better age estimates can be made if localized site index is known.
4. Growth curves: These were derived from data associated with natural stands in the Southeast, and developed from similar datasets as those used to determine expected soil site index. These were assumed to apply across the entire landscape for both loblolly and longleaf pines. Similar to localized effects on site index, growth curves can also differ based on site preparation and stand density (Boyer 1980).
5. FIA and field data: Validation in this study required assuming the maximum modeled age corresponded to the maximum recorded DBH, and grouping model and

FIA data into age and site index classes did not significantly impact analysis. These assumptions could influence results for both modeled age values and expected accuracy of age estimates.

Additional data, such as that associated with climate change, would also be useful in modeling habitat. According to various models in the USFS Tree Atlas (Prasad et al. 2007-ongoing), relative abundance of longleaf will remain stable or increase in the southern portion of the study area over the next 100 years. In some scenarios, its spatial distribution is also expected to expand. Loblolly is projected to decrease in relative abundance and distribution. Impacts of disease, insect outbreaks, weather (e.g., hurricanes), and harvest on pine distribution should also be considered. Modeling pine forests at smaller scales, using tools such as the Forest Vegetation Simulator (FVS) would be useful in determining how pine distribution may change, as well as how stand structure is expected to change over time with implementation of certain habitat management practices. Incorporation of these potential changes in modeling could alter present and future recovery strategies and priorities.

4.2 Opportunities for recovery on private lands

Many short-term opportunities to contribute to RCW recovery exist throughout the landscape. Modeling potential RCW habitat and recruitment cluster sites indicates some of the best short-term recovery opportunities exist in the southwestern portions of the study area, and in counties north of the Croatan National Forest; these areas contain some of the highest densities of potential recruitment clusters, and the most highly connected potential cluster sites. Comparison with 2012 aerial imagery shows lands in the southwestern study area are more representative of open, natural pine stands, whereas lands north of the Croatan National Forest mostly include intensively managed pine plantations. For this reason, lands in the southwestern area may be better candidate sites for recovery in the near future.

Modeling confirms older pines are very rare in the study area. Furthermore, many of these older stands do not have adequate amounts of potential nesting or foraging habitat in close proximity to support a cluster. Because older or larger trees are currently considered the most limiting suitability factor, recovery efforts should maximize conversion of existing pines into suitable habitat, creating opportunities to develop habitat sooner, while increasing the amount of available habitat over time. Effective RCW recovery will require practices that produce high quality habitat in adequate quantities and spatial configurations that encourage cavity excavation at rates that can balance or exceed cavity losses (Rudolph et al. 2004). Carrying capacity for RCW will only increase with active management practices that have already proven effective (e.g., forming new groups in abandoned or previously unoccupied territories with artificial cavities, translocating groups into artificial clusters, and ultimately changing silvicultural practices) (Conner et. al 2001), especially on private lands in this region.

Focusing on instances where potential recruitment sites are within 4 km of active sites could be key in developing links to and between major existing populations. Over time, it appears feasible to connect existing populations within the study area (e.g., Holly Shelter Game Land and Bladen Lakes State Forest populations, and Croatan National Forest and Dare County populations), and adjacent to the study area (e.g., Camp Lejeune to Sandhills populations west of the study area), creating more opportunities for natural dispersal of individuals to new groups, thereby increasing genetic flow. Since potential recruitment sites were primarily connected to other potential sites, but rarely within 4 km of existing sites in the Onslow Bight, forming connections between populations will likely require efforts that extend beyond 10 years.

Deciding where to focus recovery efforts must account for current and future availability of nesting and foraging habitat, and how it can contribute to connectivity across multiple spatial and temporal scales. Existing stands of all ages are particularly important in improving connectivity between existing clusters as well as areas where there is great potential to develop many, highly-connected clusters. Present conservation and subsequent conversion of long-term opportunities is equally as important as short-term opportunities. Strategic management of younger stands, which are highly available throughout the study area, can increase connectivity between existing or recruitment clusters, provide foraging habitat, and eventually contribute to nesting habitat.

A more in-depth analysis of this landscape could result in reduced connectivity between potential sites when considering the various dispersal behaviors of individuals, dispersal limits in terms of functional, not Euclidean, distance, and effects of habitat gaps on movement. Establishment of new clusters will require field visits to confirm adequate habitat quality and quantity exist before translocations occur, and careful planning to allocate territories at optimal distances from existing or future clusters. Recruitment clusters must be located a minimum distance away from existing groups to discourage them from inhabiting the recruitment cluster, yet within a maximum distance from existing groups to encourage occupation by dispersing individuals (Conner et al. 2001). Not all recruitment clusters identified in this analysis should be developed; site-specific data is needed to ultimately determine suitability. However, modeling tree age and potential cluster sites is useful in guiding where further investigation should occur, and assessing where recruitment clusters have the greatest potential to effectively contribute to recovery.

4.3 Feasibility of recovery on private lands

Recovery efforts will not only be driven by habitat attributes, but also by which private landowners are willing to participate, and the costs associated with converting existing pines into RCW habitat. DOD is interested in pursuing a RCS, which allows Federal agencies to implement a system where credits (i.e., typically, an explicit amount and quality of habitat able to contribute to recovery of a threatened or endangered species) can be created through recovery efforts off-site (i.e., on private lands), stored, and then redeemed to offset adverse actions on-site (USFWS 2008). The RCS is intended to contribute to recovery of a listed species, and grant Federal agencies more flexibility to manage their lands toward agency-specific missions and goals, while continuing to meet their responsibilities for threatened and endangered species management (USFWS 2008). A properly designed RCW RCS that encourages landowners to engage in strategic, long-term recovery efforts would provide a way to collectively increase carrying capacity for RCW over the study area, as populations approach carrying capacity on public lands.

DOD has already utilized RCSs to offset adverse effects of military training on at-risk species. In the case of the endangered golden-cheeked warbler (GCW) (*Setophaga chrysoparia*) on Fort Hood, goals for habitat conservation were met, landowners were satisfied with the process of participating in the RCS, and efforts resulted in increased training flexibility on Fort Hood (Robertson and Rinker 2010). Similarly, the pilot RCS developed for at-risk populations of the gopher tortoise (*Gopherus polyphemus*) in Alabama and Georgia has resulted in less expensive pre-compliance credits, allowed for monitoring funding in perpetuity, and offered better compensation to landowners for management activities (Gartner and Dolan 2011). Although a relatively young conservation program, success with other species suggests a RCS would be beneficial to RCW, DOD, and participating landowners.

Financial incentives to encourage landowner participation can be especially important in areas such as eastern North Carolina, where the forest products industry accounts for a major portion of the economy. In order to encourage conversion of presently forested or agricultural lands to RCW habitat under conservation contracts (i.e., an RCS), costs associated with changes in management regimes must be offset, so that endangered species management is transformed from a liability into an asset (Wilkins et al. 2008, Gartner and Dolan 2011). Economic analysis suggests the financial incentives required to convert longleaf and loblolly currently managed for maximum timber revenue to RCW management are feasible, especially if contracts allow intermediate income-producing practices to continue (e.g., pine straw harvest), and delay or eliminate complete forest harvest to reduce required payment amounts (Glenn 2012). Changes in forest management that result in longer timber rotations would likewise directly benefit RCW.

Relatively low costs associated with management conversion are especially appealing considering 50% of surveyed landowners in 18 eastern North Carolina counties—all of which overlap the study area—are willing to apply for conservation contracts (Rodriguez et al. 2010). On average, respondents were willing to enter over 160 acres (i.e., areas large enough to support an RCW cluster) into such agreements, and reported forestry or agriculture as the primary land use. Short-term contracts (e.g., up to 10 years) were most popular among landowners, who likely prefered these contracts to minimize land use constraints and limit deed restrictions (Rodriguez et al. 2010). Rodriguez et al. concluded that the percent of respondents willing to participate are encouraging, and may increase over time as younger generations inherit land. This suggests habitat restoration opportunities may increase over time, further contributing to connectivity and availability of habitat within the study area.

Given landowner preference for short-term contracts, careful program design and provision of attractive incentives are needed to bolster participation in long-term contracts. Surveyed forest landowners were more willing to participate in long-term gopher tortoise conservation contracts when able to receive increased compensation, and retain ability to make land management decisions (Sorice et al. 2013). Reverse auctions used in the market-based GCW RCS resulted in landowners increasing contract lengths from 10 to 25 years and decreasing bid amounts with consecutive bid rounds, as they realized longer contracts at lower prices were more likely to be accepted (Wolfe et al. 2012). These trends should be considered when formulating a RCW RCS to encourage long-term participation.

As indicated with spatial modeling of current pine forest conditions, continued landowner participation in recovery will be needed for decades to provide a substantial contribution to RCW recovery. Successful development of the RCS will require consideration of both the species', and landowners' needs, and adequate landowner participation in sufficient spatial extents and contexts (Sorice et al. 2013). This includes identifying existing and potential habitat, defining suitable habitat, and accounting for the importance of context within the landscape so that conservation units, and therefore recovery credits, truly contribute to species recovery (Wilkins et al. 2008). Spatial modeling of potential habitat, such as that done in this study, can aid in development and implementation of an RCS. As spatial data becomes available to identify landowners willing to participate and financial compensation required for recovery, it can be used in conjunction with modeled habitat data to locate lands most likely to enroll and succeed in an RCW RCS.

5. CONCLUSIONS

Opportunities to increase carrying capacity for RCW and contribute to recovery over the next 10 to 40 years exist on private lands in eastern North Carolina; however, only one-fifth of existing pines have potential to produce suitable RCW cavity trees within 10 years. Southwestern portions of the study area between Bladen Lakes State Forest and Holly Shelter Game Land, and areas north of the Croatan National Forest contain some of the best opportunities to develop recruitment clusters within 10 years, assuming appropriate management practices are implemented. More recovery opportunities will become available over time, but will require active management of existing pines, and translocation of RCWs to create new groups. Prevalence of young pine stands suggests long-term recovery efforts will be particularly essential in increasing habitat quantity and connectivity throughout the landscape.

Further investigation will be needed to assess habitat quality, quantity, and connectivity at smaller scales. In addition to habitat attributes, levels of landowner participation and costs associated with habitat management will determine the degree to which private lands contribute to RCW recovery. Strategic planning will be required across multiple spatial and temporal scales to maximize success of recovery efforts and establish RCW populations throughout the landscape that can eventually persist without intensive management. Spatially forecasting recovery opportunities over time provides a unique conservation planning tool that can help guide and assess recovery efforts in this landscape.

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