

ABSTRACT

YIRKA, LIANI MARIE. Demographic rates and Prioritization of Habitat for Conservation for Painted Buntings in North Carolina. (Under the direction of Dr. Jaime A. Collazo).

Populations of Painted Buntings (*Passerina ciris*) have been declining across their range since the 1970's, especially in the southeastern United States, which experienced a steep initial decline and no indication of improvement. In an effort to identify and understand the cause of this decline, recent studies have highlighted wintering and breeding distributions, and one study estimated demographic rates for a population of Painted Buntings in South Carolina. However, more information about demographic rates, specifically juvenile survival and recruitment rates, and understanding how these might be exacerbated by habitat degradation and loss, will aid in a more comprehensive conservation approach for the Painted Bunting. This study works to identify demographic rates for Painted Buntings at two sites in North Carolina, which life stages are most vulnerable to threats, and identify which habitat most promotes the persistence of this species in the state.

In order to gain insight to demographic rates for Painted Bunting in North Carolina, we estimated age and sex-specific survival for two sites, Bald Head Island (BHI) and Hammocks Beach State Park (HBSP) using capture-recapture data collected between 2007 and 2015. Using program MARK, we employed the Cormack-Jolly-Seber and Robust Design models to estimate rates. Juveniles had lower apparent annual survival rates than adults (BHI: 0.28 ± 0.04 (HY) vs. 0.66 ± 0.02 (AHY); HBSP: 0.31 ± 0.15 (HY) vs. 0.66 ± 0.07 (AHY)). Data from Bald Head Island provided sex-specific apparent annual survival estimates of 0.63 ± 0.03 (males) and 0.71 ± 0.03 (females). The

population of Bald Head Island averages 120 juveniles and 329 adults annually and a mean average growth rate (λ) of 1.16 indicating the population of Painted Buntings on Bald Head Island is growing on average. The recruitment rates also indicate this while also strongly suggesting that there is connectivity between the population on Bald Head Island and other coastal populations.

To determine if habitat affected occupancy for Painted Buntings in North Carolina, we used survey data from 2008-2009 and a single-season, single-species occupancy model and found that occupancy rates increased with increasing amounts of maritime forest. This relationship was consistent at a sampling spatial scale of 200m radii as well as at an average natal dispersal unit of 6km which estimates $\leq 90\%$ of juvenile passerines return within this area after their first winter. Using this habitat-occupancy relationship and projections of maritime forest loss generated using a habitat threat-risk assessment tool, we evaluated the probability that at least one Painted Bunting would persist in 5 areas of interest (AOI) across North Carolina. Persistence probability ranged from 0.82 to 0.98. Numerical ranking of AOIs by persistence probability is the first level of decision criteria for area selection, but we also enumerate potential trade-offs associated with a more complex decision scenarios involving selection of paired AOIs. These scenarios highlighted the importance of benchmark occupancy and habitat condition, and the timing of actions to foster persistence.

While our estimates of demographic rates are were obtained in North Carolina, they expand existing knowledge of Painted Bunting vital rates and allow for more informed conservation management decisions throughout its range. Occupancy and habitat threat-risk assessments represent steps forward in the estimation of persistence to

guide area-selection. Existing multi-season models, coupled with measures of uncertainty for projected land cover changes, represent a path forward to model persistence in a non-stationary environment and a stronger foundation for decisions.

© Copyright 2017 Liani Marie Yirka

All Rights Reserved

Demographic rates and Prioritization of Habitat for Conservation for Painted Buntings in
North Carolina

by
Liani Marie Yirka

A thesis submitted to the Graduate Faculty of
North Carolina State University
in partial fulfillment of the
requirements for the degree of
Master of Science

Zoology

Raleigh, North Carolina

2017

APPROVED BY:

Dr. Jaime A. Collazo
Committee Chair

Dr. Nick M. Haddad

Dr. J. Krishna Pacifici

Dr. Brian J. O'Shea
Technical Consultant

DEDICATION

To my ever-constant rock, Adrian, for your quiet, steady, soul-deep love, patience and encouragement. To the Paloma, you are the reason for all I do. To Percy and Jaime, you are my pillars, my inspiration and my compass.

BIOGRAPHY

Liani was born in south Florida and grew up in Apex, North Carolina with her family. Her love of nature and passion for science began early in life, was nurtured along the way by some wonderful science teachers, and matured through her participation in the junior curator program at the North Carolina Museum of Natural Sciences. Liani attended NC State University for her undergraduate degree where she received a B.S in Zoology in 2009. During this time she gained experiences working as a research technician for Dr. Nick Haddad's Saint Francis Satyr project on Fort Bragg, for Dr. Paul Hamilton and the Biodiversity Group's herpetofauna biodiversity inventory project in western Ecuador, and as an environmental educator for both the NC Museum of Natural Sciences and Wake County's Environmental Services division.

While in graduate school, Liani has continued to work full-time for the NC Museum of Natural Sciences as the head of accessibility and inclusion. Through this position and her work, Liani strives to make science accessible to audiences who might normally face barriers to engage in science and museums using innovative assistive technology. In her spare time, Liani tutors students in science, math, Spanish and English at all levels, reads tirelessly, and enjoys being outdoors whether birding, herping, hiking, or camping with her husband and friends.

Upon the completion of her master's degree, Liani hopes to continue to find avenues to communicate science content to a large variety of audience and aspires to be a role model for youth interested in science careers. Her drive to succeed and to learn will pave the way.

ACKNOWLEDGMENTS

There are many people to sincerely thank for their contributions to this project. I am very thankful to Dr. Jaime Collazo and my committee, Drs. Nick Haddad, Krishna Pacicifi, and Brian O'Shea for their time, dedication and guidance through this entire process. We are thankful to Drs. Rua Mordecai and Clint Moore for generously sharing survey data collected in 2008 and 2009. The contributions and support from John Gerwin and Dr. Jamie Rotenberg are greatly appreciated. Special thanks go Dr. David T. Cobb and the North Carolina Wildlife Resources Commission for providing the financial support to conduct the project. Additional generous support from the Cape Fear Audubon Society is appreciated.

The countless hours of field assistance and support from Sam Jolly and Julia Jacobs ensured this project would be successful; thank you! Additional thanks to all those who helped with this project over the years: Stephanie Bohr, Martha Fisk, Debbie Huston, Ginger Deason, Tyler Michels, Edward Landi, Sarah McGrath and Brett Williams. This project would not have happened without help on Bald Head Island from the Bald Head Island Conservancy, the Giacci family, the Grantmyer family, the Quanstrom family, the Coryell family, the Prak family, and the Douglas family.

To all of my NCMNS colleagues and friends, thank you for your endless support over the years. Thank you to the individuals in the Applied Ecology department at NC State University for their guidance as well as logistical and administrative support, Steve Williams, Curtis Belyea, Ruby Valetton, Susan Marschalk, Brooke Clemmons, Freha Legoas, and Drs. Tom Kwak, Craig Layman, and Rob Dunn.

Finally I am grateful to my family and friends for the support in all I strive to accomplish. To my husband, Adrian, thank you for helping me follow my dreams. To my parents, Percy and Jaime, and my siblings, Paloma and Jaime, I hope to make you proud. To Abuela, Jose y Nelida, gracias por todo tu cariño y apoyo en todo lo que yo aspiro a hacer. To Dave, thanks for believing in me. To Debbie, you know I can't replace you. And finally, to Amariyls, ¡Yo voy a ti!

TABLE OF CONTENTS

LIST OF TABLES	viii
LIST OF FIGURES	ix
CHAPTER 1: CONCEPTUAL OVERVIEW	1
LITERATURE CITED	4
CHAPTER 2: ANNUAL SURVIVAL, POPULATION SIZE AND RECRUITMENT RATES OF PAINTED BUNTINGS AT BALD HEAD ISLAND AND HAMMOCKS BEACH STATE PARK, NORTH CAROLINA, 2007-2015	6
ABSTRACT	6
INTRODUCTION	7
METHODS	9
STUDY SITES	9
SAMPLING METHODS	10
DATA ANALYSIS	12
RESULTS	16
DISCUSSION	17
LITERATURE CITED	22
TABLES AND FIGURES	26
CHAPTER 3: PRIORITIZING AREAS FOR SPECIES PERSISTENCE USING OCCUPANCY AND THREAT-RISK ANALYSES	33
ABSTRACT	33
INTRODUCTION	34

METHODS	37
STUDY SITES	37
AVIAN SURVEYS	38
HABITAT AND RISK ASSESSMENT	39
DATA ANALYSIS	41
OCCUPANCY	41
RISK ASSESSMENT	42
AREA PRIORITIZATION	43
RESULTS	44
DISCUSSION	46
LITERATURE CITED	50
TABLES AND FIGURES	55
APPENDICES	65
APPENDIX 1	66
APPENDIX 2	72
APPENDIX 3	73

LIST OF TABLES

CHAPTER 2

Table 1. Model selection table for Cormack-Jolly-Seber models used to estimate apparent, age-specific annual survival of Painted Buntings at Hammocks Beach State Park (top table) and Bald Head Island (bottom table), North Carolina 2007-2015. Models are ranked by their respective AICc values.26

Table 2. Model selection table for Cormack-Jolly-Seber models used to estimate apparent, sex-specific annual survival of Painted Buntings at Bald Head Island, North Carolina 2007-2015. Models are ranked by their respective AICc values. Competing models ($\Delta AICc \leq 2$) are indicated with an asterisk.....27

Table 3. Model selection table for robust design models used to estimate apparent, age-specific annual survival of Painted Buntings at Bald Head Island, North Carolina 2007-2015. Secondary sampling occasions (3) occurred in late May/mid-June, mid-July, and mid-August. Models are ranked by their respective AICc values. Competing models ($\Delta AICc \leq 2$) are indicated with an asterisk.28

CHAPTER 3

Table 1. Model Selection tables for assessment of Painted Bunting occupancy as a function of 12 land cover classes found in coastal North Carolina. Occupancy was estimated at the 200 m (survey station; top table) and 6 km (natal dispersal unit; bottom table) based on surveys of 133 stations collected in 2008-2009. Effort means that survey stations were sampled 6 times, or just 3 time in one of two years.55

Table 2. Summary statistics (mean \pm SE) for percent loss of maritime forest and severity per 12 digit Hydrologic Unit (HUC) in 5 areas of conservation interest (AOI) for Painted Buntings in southeastern North Carolina.56

LIST OF FIGURES

CHAPTER 2

- Figure 1. Map of coastal North Carolina depicting the location of Hammocks Beach State Park and Bald Head Island. The inset depicts southeastern United States and the range of Painted Buntings along the coast of Florida and North Carolina, and coast and inland South Carolina and Georgia.29
- Figure 2. Estimates of apparent, age-specific annual survival estimates of Painted Buntings from 2007 to 2015 at Bald Head Island, North Carolina. Average estimates for the study period and parameter estimates \pm SE are reported.30
- Figure 3. Estimates of age-specific population size of Painted Buntings from 2007 to 2015 at Bald Head Island, North Carolina. Parameter estimates \pm SE are reported. Estimates were derived from the robust design model with highest support (AICc Weights) in the data (no-movement).31
- Figure 4. Estimates of recruitment rates of Painted Buntings from 2007 to 2015 at Bald Head Island, North Carolina. Estimates were derived from the robust design model with highest support (AICc Weights) in the data (no-movement). Recruitments are classified by origin: *in-situ* or immigrants. Precision levels (95% CIs) for annual average by origin is reported in the text.32

CHAPTER 3

- Figure 1. Map of the range of eastern Painted Buntings (inset), highlighting the range of the species in southeastern North Carolina. The map depicts surveys station locations (dots) within natal dispersal units (6 km, black circles), and areas of conservation interest (squares). Areas of conservation interest (AOI) were labeled based on distinctive communities or federal lands along the coast. AOI-1 = Sunset Beach, AOI-2 = Bald Head Island/Wilmington, AOI-3 = Topsail/Hammocks Beach State Park, AOI-4 = Camp Lejeune, and AOI-5 = Croatan National Forest.57
- Figure 2. Stepwise process to estimate persistence probability using approach outlined by Araujo and Williams (2000). Objective of each step is provided. Subscripts are: i =species, a =area of interest, t =time.58
- Figure 3. Occupancy probability (\pm SE) of Painted Buntings as a function of percent maritime forest in natal dispersal units (ndu; 6 km) in southeastern North Carolina. The functional relationship was developed using survey data collected at 133 stations in 2008-2009.59
- Figure 4. Occupancy probability of Painted Buntings by natal dispersal unit (6 km; top panel) by regions and area of conservation interest (AOI; bottom panel) in southeastern

coastal North Carolina. Units were grouped using AOI labels. SB/W/BHI= Sunset Beach/Bald Head Island/Wilmington, TS/HBPS = Topsail/Hammocks Beach State Park, LEJ/CNF = Camp Lejeune, and Croatan National Forest.60

Figure 5. Projected percent loss of maritime forest in 2020 and 2050 relative to the baseline levels in 2000 (top panel). Percent loss is expressed on a per hectare basis (bottom panel). SB = Sunset Beach, W/BHI= Bald Head Island/Wilmington, TS = Topsail/Hammocks Beach State Park, LEJ = Camp Lejeune, and CNF = Croatan National Forest.61

Figure 6. Projected change in occupancy probability of Painted Buntings from 2008-09 to 2050 in southeastern coastal North Carolina. Estimates of occupancy assume the species-habitat relationship developed for Painted Buntings using 2008-2009 survey data holds through 2050.62

Figure 7. Average severity (\pm SE) that loss of maritime forest of a specified magnitude occurs in 2050 at five areas of conservation interest for Painted Bunting in southeastern North Carolina (top panel). Probability (95% CIs) that an area of conservation interest (AOI) will remain occupied (persist) in 2050. Magnitude of loss of habitat is depicted in figure 5. SB = Sunset Beach, W/BHI= Bald Head Island/Wilmington, TS = Topsail/Hammocks Beach State Park, LEJ = Camp Lejeune, and CNF = Croatan National Forest.....63

Figure 8. Risk rating matrix of projected changes in 2050 due to sea level rise in 5 areas of conservation interest for Painted Buntings in southeastern North Carolina. Rating is based on occurrence (e.g., proportion of HUCs within each AOI where terrestrial land cover changes were detected) and severity (e.g., probability of change equal or less than a specified value (x) of terrestrial land cover change). Terrestrial land cover change caused by sea level rise is the loss of all land cover types (including marshes) to open water since 2000. SB = Sunset Beach, W/BHI= Bald Head Island/Wilmington, TS = Topsail/Hammocks Beach State Park, LEJ = Camp Lejeune, and CNF = Croatan National Forest.64

CHAPTER 1

CONCEPTUAL OVERVIEW

The Painted Bunting (*Passerina ciris*) is a small, colorful, songbird that breeds in the southeastern and south central United States and winters in the Central America and the Caribbean (Shipley et al. 2013, Lowther et al. 2015). Like many other Neotropical migrants, populations have declined since the 1970s (Sauer et al. 2014). The causes of these declines are myriad, but among them are loss and degradation of habitat across their breeding and wintering range (Robbins et al. 1989). It is not known whether other drivers (e.g., global climate patterns) might be contributing to these declines, factors that impinge on survival and fecundity of other species like Black-throated Blue Warblers (*Dendroica caerulescens*; Sillett et al. 2000). Although a series of studies have been conducted over the past 30 year, the dearth of information on Painted Bunting demographic rates and primary habitats still precludes ascertaining the status of the species (Lowther et al. 2015).

Donovan et al. (2002) put forth several recommendations with regards to research priorities that still hold for Painted Buntings. These were: 1) identify high-quality habitat during all phases of their life cycle, 2) understand the scale at which populations interact, including winter-breeding linkages to identify the part of the annual cycle most limiting to the population, and 3) monitor population sizes and responses to conservation actions (Nichols and Williams 2006). We review these with respect to the eastern population, the focus of this study. Available information indicates that primary breeding and wintering habitat is poorly documented, and linkages between both stages in their annual cycle are

now being elucidated with the use of geolocators (Cotina et al. 2013). There are several studies on nesting success and distribution (Sykes Jr. and Holzman 2005, Springborn and Meyers 2005), but none have indexed habitat and reproductive output, survival, or immigration or *in-situ* recruitment rates. These studies, however, have suggested that coastal habitats, such as maritime forest, are optimal for Painted Buntings (Hamel et al. 1982, Hamel 1992). Using long-term data, DeSante et al. (2015) found that apparent annual adult survival for a population in South Carolina was 0.61 ± 0.05 (95% CI: 0.49-0.72). They suggested that attention should also be given to the estimation of first-year survival and recruitment of young birds (linked to wintering habitat quality) because low values may be important contributors to the southeastern population decline. Recent information from population surveys (2007-2009), albeit unpublished, suggested that the size of the eastern population was 1.2 M (95% CIs: 1-1.4 M) individuals (R. Mordecai, USFWS, unpublished data; Delany et al. 2013). These data suggested that 68% of the estimated population occurs in inland habitats of Georgia and South Carolina.

The northern extent of the eastern population of Painted Buntings is located in North Carolina with an estimated population in the state of 154, 600 individuals (95% CIs: 59,000-251,000; R. Mordecai, USFWS unpublished survey data). Individuals are restricted to a narrow corridor of coastal habitat, a geographic extent subject to continued human encroachment. For example, maritime forests along the coast of North Carolina have declined by 36% over the last 23 years (Jones et al. 2013). This trend raises the possibility that the advantages gained by high population numbers, individuals purportedly responding independently to other environmental factors, lose that advantage

because all are exposed to loss of habitat (sensu Myers et al. 1987). It is for this reason that Painted Buntings are designated as a species of conservation concern in North Carolina (North Carolina Wildlife Resources Commission 2015).

There is much work to be done if the underlying causes of Painted Bunting decline are to be understood, and hopefully, curbed (Donovan et al. 2002). This thesis explores two research priorities essential to advance the conservation of Painted Bunting in North Carolina and regionally. First, we estimate demographic rates pertinent to the assessment of the status of the species. Using capture-recapture data collected in 2007-2015, we estimated age and sex-specific apparent survival rates, recruitment rates and size of the population at Bald Head Island and Hammocks Beach State Park, North Carolina. Second, we combined occupancy analyses and a habitat threat-risk assessment tool to estimate the probability of an area becoming unoccupied (extinct) as a function of projected habitat loss. This area-selection process prioritizes habitat for conservation on the basis of persistence probability in North Carolina, linking demographic attributes and habitat conservation.

As this research, which constitutes my thesis, reflects the work of many (see Acknowledgements section), I use the collective “we” throughout the thesis.

LITERATURE CITED

- Contina, A., E. S. Bridge, N. E. Seavy, J. M. Duckles, and J. F. Kelly. 2013. Using geologgers to investigate bimodal isotope patterns in Painted Buntings (*Passerina ciris*). *The Auk* 130:265–272.
- Delany, M. F., B. Pranty, and R. A. Kiltie. 2013. Painted Bunting Abundance and Habitat Use in Florida. *Southwestern Naturalist* 12:61–72.
- DeSante, D. F., D. R. Kaschube, and J. F. Saracco. 2015. Vital Rates of North American Landbirds. www.VitalRatesOfNorthAmericanLandbirds.org: The Institute for Bird Populations.
- Donovan, T. M., C. J. Beardmore, D. N. Bonter, J. D. Brawn, R. J. Cooper, J. A. Fitzgerald, R. Ford, S. A. Gauthreaux, T. Luke, P. D. Vickery, and T. B. Wigley. 2002. Priority research needs for the conservation of Neotropical migrant landbirds. *J. Field Ornithology* 73:329-339.
- Hamel, P. B., H. E. LeGrand Jr., M. R. Lennartz and S.A. Gauthreaux. 1982. Bird habitat relationships on southeastern forest lands. Asheville, NC: U.S. For. Serv.
- Hamel, P. B. 1992. Land manager's guide to the birds of the south. The nature Conservancy, Chapel Hill, NC and U.S. Forest Service, Southern Region, Atlanta, GA.
- Jones, G., A. Snider, and S. Luo. 2013. Changes in the Extent of North Carolina Barrier Island Maritime Forests 1988-2011: An Assessment of Past Efforts at Protection. *Journal of Forestry* 111:186–193.
- Lowther, P E., S. M. Lanyon, and C. W. Thompson, 2015. Painted Bunting (*Passerina ciris*). *The Birds of North America*. The America Ornithologist's Union. Philadelphia, PA.
- Myers, J. P., R. I. G. Morrison, P. Z. Antas, B. A. Harrington, T. E Lovejoy, M. Sallaberry, S. E. Senner and A. Tarak. 1987. Conservation strategy for migratory species. *American Scientist* 75: 18-26.
- Nichols, J. D., and K. B. Williams. 2006. Monitoring for Conservation. *Trends in Ecology and Evolution* 21(12):668-673.
- North Carolina Wildlife Resources Commission. 2015. North Carolina Wildlife Action Plan (SWAP). Raleigh, NC.

- Robbins, C. S., D.K. Dawson, and B.A. Dowell. 1989. Habitat Area Requirements of Breeding Forest Birds of the Middle Atlantic States. The Wildlife Society, Wildlife Monographs no. 103.
- Sauer, J. R., J. E. Hines, J. E. Fallon, K. L. Pardieck, D. Ziolkowski Jr, and W. A. Link. 2014. The North American Breeding Bird Survey, Results and Analysis 1966-2013. Laurel, MD.
- Shipley, J. R., A. Contina, N. Batbayar, E. S. Bridge, A. T. Peterson, and J. F. Kelly. 2013. Niche conservatism and disjunct populations. *The Auk* 130:476–486.
- Sillett, T. S., R. T. Holmes, and T. W. Sherry. 2000. Impacts of a global climate cycle on population dynamics of a migratory bird. *Science* 288:2040-2041.
- Springborn, E. G., and J. M. Meyers. 2005. Home range and survival of breeding Painted Buntings on Sapelo Island , Georgia. *Wildlife Society Bulletin* 33:1432–1439.
- Sykes Jr., P. W., and S. Holzman. 2005. Current range of the eastern population of Painted Bunting (*Passerina ciris*). *North American Birds* 59:4–17.

CHAPTER 2

ANNUAL SURVIVAL, POPULATION SIZE AND RECRUITMENT RATES OF PAINTED BUNTINGS AT BALD HEAD ISLAND AND HAMMOCKS BEACH STAT PARK, NORTH CAROLINA, 2007-2015

ABSTRACT

Painted Buntings (*Passerina ciris*) in southeastern United States have been declining since the 1970's. Low juvenile survival and recruitment rates are believed to contribute to declines, but estimates of either parameter are not available. Reliable estimates of annual adult survival are only available for a population in South Carolina. We used capture-recapture data collected from 2007-2015 at Bald Head Island (BHI) and Hammocks Beach State Park (HBSP), North Carolina to estimate age- and sex-specific apparent annual survival rates, and population size and recruitment rates. Estimates were obtained using Cormack-Jolly-Seber and Robust Design models in program MARK. Juveniles had lower survival rates than adults (BHI: 0.28 ± 0.04 (HY) vs. 0.66 ± 0.02 (AHY); HBSP: 0.31 ± 0.15 (HY) vs. 0.66 ± 0.07 (AHY)). At BHI, sex-specific survival was 0.63 ± 0.03 (males) and 0.71 ± 0.03 (females), the population averaged 120 juveniles and 329 adults annually, and the mean observed growth rate (λ) was 1.16. Estimated recruitment rates averaged 35 (95% CIs: 25-57) individuals (*in-situ*) and 157 (95% CIs: 116-213) immigrants annually. Recruitment rates at BHI yielded evidence of successful reproduction and connectivity with other coastal populations. Lower male survival may affect local fecundity, but also draws attention to potential mortality induced by pet trade. Although findings come from North Carolinian populations, adult survival estimates

were consistent with previously reported estimates in coastal habitats (~60%). We augment knowledge about Painted Bunting vital rates, particularly juvenile survival and recruitment rates, laying a stronger foundation to help guide effective conservation actions.

INTRODUCTION

Painted Buntings (*Passerina ciris*) are small Neotropical migrants that breed across the southern United States (Shipley et al. 2013). There are two disjunct, allopatric breeding populations (Thompson 1991a, Shipley et al. 2013). The eastern population occurs from northern Florida, Georgia, South Carolina and North Carolina, and the western population in parts of Kansas, Texas, Oklahoma, Arkansas, and Louisiana (Herr et al. 2011). These populations are divided by a 500 km gap with no evidence of gene flow between them (Herr et al. 2011). Studies by Thompson (1991b) implied that the two populations even differ in molt and migration patterns, with the western population beginning migration to their wintering grounds earlier than the eastern population (mid-to-late-July) and molting along the migration route. In contrast, individuals from the eastern population do not migrate until they have completed molting (ca. late-August).

Numbers of Painted Buntings have declined since the 1970's, but more consistently for the eastern population (Meyers 2011, Sauer et al. 2014). Declining trends have triggered studies aimed at identifying factors accounting for declines (e.g., Sykes Jr. and Holzman 2005, Springborn and Meyers 2005, Delany et al. 2013). Yet, there is still scant information on key demographic rates (Lowther et al. 2015), and particularly for the eastern population (DeSante et al. 2015). The dearth of information for the eastern

population prompted the U.S. Fish and Wildlife Service to conduct region-wide surveys between 2007 and 2009 (R. Mordecai, USFWS, unpublished survey data). These surveys yielded a regional estimate of 1.2 million individuals, with an estimated 68% occurring in inland habitats of South Carolina and Georgia. Also recently, analyses of long-term, capture-recapture data from coastal South Carolina yielded an apparent annual adult survival rate of 0.61 ± 0.05 (95% CI: 0.49-0.72; DeSante et al. 2015). The population at this location is declining ($\lambda = 0.87$), a contrast with population maintenance or growth observed in five western locations where lambda ranged from 0.99 to 1.01 (DeSante et al. 2015).

North Carolina harbors the northern extent of the eastern population (Sykes Jr. et al. 2007), and the estimated population in the State was 154,600 individuals (95% CIs: 59,000 - 251,000; R. Mordecai, USFWS unpublished survey data). These estimates provided, for the first time, a demographic parameter with which to quantitatively appraise the status of the species. Notwithstanding, the conservation status of the species continues to be of concern in North Carolina for two reasons. First, demographic analyses to date suggest that low first-year survival or low subsequent recruitment of young birds are important contributors of the southeastern population decline (DeSante et al. 2015). Second, the species distribution in North Carolina is restricted to a narrow band of coastal habitats (1,547 km²) as compared to the much wider band in South Carolina and Georgia (8,125 km²; Hobbs and Meyers 1999, Meyers 2011). The restricted distribution of coastal habitats led Sykes Jr. and Holzman (2005) to suggest that the vulnerability of the species does not stem from population size constraints, but from the risk of losing prime

habitat for survival and reproduction. In North Carolina, this is exemplified by the 36% loss of maritime forest in the last 23 years (Jones et al. 2013), considered high quality habitat for Painted Buntings (Hamel et al. 1992).

In this study, we used capture-recapture data of Painted Buntings collected from 2007 to 2015 at Bald Head Island and Hammocks Beach State Park in North Carolina to estimate demographic rates germane to the assessment of the status of the species (DeSante et al. 2015). Specifically, we report age-specific apparent annual survival rates for both locations, and sex-specific survival, recruitment rates and population size for the population occurring at Bald Head Island. We hypothesized that age-specific survival would favor adults (higher) because juveniles are expected to lack experience in foraging, predator avoidance and dispersal (Gardali et al. 2003, Maness and Anderson 2013, Cox et al. 2014). We also hypothesized that adult annual survival rates from Bald Head Island and Hammocks Beach State Park would be similar (i.e., within 95% CIs). Our rationale was that both populations shared many environmental conditions (spatial correlation), including high quality habitat (e.g., maritime forest). Parameters were estimated using Cormack-Jolly-Seber and Robust Design models in program MARK (Pollock 1982, Pollock et al. 1990, Kendall et al. 1997, 2015, Cooch and White 2015). We discuss the conservation implications of our results for Painted Buntings and other Neotropical migrants.

METHODS

STUDY SITES

We trapped and released Painted Buntings at Bald Head Island and Hammocks Beach State Park (figure 1). Bald Head Island is located at the extreme southeastern coastal point of North Carolina at the mouth of the Cape Fear River in southeastern North Carolina (Lat: 33.86°, Long: -77.99°). Approximately 620 ha of the island is upland habitat with various successional stages of maritime forest/shrub, dune/grassland, tidal marsh and urban development (Oosting 1954, Bourdeau and Oosting 1959, Lopazanski et al. 1988). A 191 acre reserve, Bald Head Woods, is located at the center of the island as part of the Smith Island Complex. The reserve consists of old growth live oak (*Quercus virginiana*) and laurel oak (*Quercus laurifolia*) as canopy species and a mixture of understory shrubs such as American holly (*Ilex opaca*) and yaupon (*Ilex vomitoria*) (North Carolina Coastal Reserve 2015). Our sampling took place at the extreme portion of the island covering an area of approximately 840 ha.

Hammocks Beach State Park (HBSP) is located between Camp Lejeune (USMC) and Croatan National Forest (USFWS) (Lat: 34.67°, Long: -77.13°; Figure 1). The park totals over 400 ha and is composed of: a) a 13 ha mainland gateway, b) two islands designated as nature preserves and with no public access, Huggins and Jones Islands, and c) Bear Island. Sampling took place between the mainland gateway area and Bear Island covering an area of approximately 2000 ha. Vegetation is comprised of shrub-scrub thickets and maritime forest in addition to the large swaths of salt marsh in between the mainland and Bear Island. Bear Island is predominantly covered with large sand dunes

and beach habitat that has overtaken much of the maritime forest on the island in recent decades (North Carolina Department of Parks and Recreation 2016). In shrub maritime and maritime forest thickets the primary vegetation consists of wax myrtle (*Myrica cerifera*), Eastern red cedar (*Juniperus virginiana*), yaupon (*Ilex vomitoria*), and live oak (*Quercus virginiana*) among others while bordered by salt marsh habitat dominated by two species of cordgrass (*Spartina alterniflora* and *Spartina patens*) (Dickerson 1978).

SAMPLING METHODS

We collected capture-recapture data between 15 May and 30 August from 2007 to 2015. There were 3-4 sampling occasions per year, lasting approximately 3-5 days each. Sampling occasions were in late-May/mid-June, mid-July, mid-August, with occasions spaced by a minimum of 14 days. Painted Buntings were trapped at 4-7 sampling stations at Bald Head Island and 2 sampling stations at Hammocks Beach State Park. Each site was visited in the morning at 6:00-10:30 or 6:30-11:00, and in the afternoon at 14:00-18:30 or 14:30-19:00. Timing was based on sunrise and sunset hours during the season, and peak Painted Bunting activity. We alternated morning and afternoon sampling sessions to avoid systematic errors (e.g., recapture probability).

Painted Buntings were trapped using two wire cages with feeders containing a tube feeder filled with white millet seed (Sykes Jr. 2006). Birds were not extracted immediately, but after they showed no signs of stress, either caused by entering the cage or the presence of other species. Females, particularly those captured in the morning, were evaluated for eggs in the oviduct, and further manipulation was minimal to avoid undue stress if eggs were present or forming. At the completion of the 4.5 hour sessions,

trap feeders were removed and replaced with regular feeders. The latter ensured that birds would come and go to the same spot between trapping occasions, and would be available for capture in the next sampling occasion. All sampling occasions were carried out by a minimum of two individuals, an experienced bander and a data recorder. Captured individuals were uniquely marked with 3 colored plastic bands (either Darvic or Acetal; Avinet 2015) and one USGS band from the Bird Banding Laboratory (United States Geological Survey 2015). Before releasing, we weighted (gr), and aged and sexed birds (if discernible).

DATA ANALYSIS

We treated Bald Head Island and Hammocks Beach State Park as our primary experimental units, thus, data from all stations within each location were pooled for analyses. We captured 1,162 individuals at Bald Head Island and 248 at Hammocks Beach State Park over the 9-years study period. We screened data from Bald Head Island, excluding records that did not contain information on age or sex. This process yielded a data set with 729 encounter histories for sex-specific analyses (age ≥ 2 years), a dataset (N=1,107) where birds were classified as juveniles (HY or hatch year), subadults (SY or second year) and adults (ASY or after second year), and a third one (N=1,112) where birds were grouped as either juveniles (HY) and adults (AHY=SY+ASY). Encounter histories (N = 158) from Hammocks Beach State Park were classified as either juveniles (HY) or adults (AHY). Data were too sparse to group by sex. We did not include body mass at time of capture as covariate because it would likely change in a matter of days (Guglielmo et al. 2005), likely influenced by feeders.

We used Cormack-Jolly-Seber (CJS) models to estimate sex- and age-specific survival rates for Hammocks Beach State Park and Bald Head Island using program MARK (White and Burnham 1999, White 2015). These models yield estimates of apparent (or local) survival (ϕ) and capture probability (p). We note that age-specific estimates reflect appropriate parameterization within MARK to account for the transition in survival probability from juvenile to age ≥ 2 . Cormack-Jolly-Seber models assume that: 1) every marked animal present in the population at year (i) has the same probability of recapture (p); 2) every marked animal in the population immediately after year (i) has the same probability of surviving to year ($i+1$); 3) marks are not lost or missed; and 4) all samples are instantaneous, relative to the interval between occasion (i) and ($i + 1$). Our sampling design led us to believe that assumptions 1, 2, and 4 were met, and tag loss (assumption 3) was negligible (e.g., Bald Head Island 1 recorded loss in >1100 individuals).

We further analyzed data from Bald Head Island using a Robust Design modeling framework. We did so because we were also interested in estimating population size and recruitment rates, which were not possible with the CJS approach (Pollock 1982, Kendall et al. 1995, 1997). For this analysis, we used that largest data set ($N=1,112$), categorizing Painted Buntings as juveniles and adults. In this study, years were designated as primary sampling periods, a time interval when the population is open or subject to change (immigration, emigration, deaths, and births). Within-year sampling periods were designated as secondary sampling occasions, time intervals within which population closure is assumed (Pollock 1982, Kendall et al. 1995). Secondary sampling occasions

were late-May/mid-June, mid-July and mid-August. We tested for evidence of temporary emigration using two competing models as these can be parameterized for both adults and juveniles: Markovian temporary emigration and no movement models. In the Markovian temporary emigration model the probability of transitioning between the available and unavailable states between primary periods depends on the availability of the bird in the previous primary period. This model has two different parameters: gamma prime (γ') and gamma double-prime (γ''). The γ' parameter represents the probability of remaining unavailable during a sampling period i , given that the bird was not available during the previous sampling period $i-1$ and that it survives to period i . The γ'' parameter represents the probability of being unavailable for capture during a sampling period i , given that the bird was available for capture during the previous sampling period $i-1$ and that it survives to period i (Kendall et al. 1997). The second parameterization, the no movement model, assumes that unavailable birds remain unavailable and available birds remain available over all sampling periods. To parameterize the Markovian temporary emigration model, we modeled γ' and γ'' separately, and for the no movement model, we set $\gamma'=0$ and $\gamma''=1$.

We did not use the random or classic temporary emigration to parameterize a model for juveniles because there is no γ' parameter (i.e., prob. of juvenile returning from temporary emigration). This is because they become adults as soon as they temporarily emigrate. To have a random movement model for juveniles, one would need several primary periods where the birds remain as a juvenile (within season), and this is not how we structured the data for our objectives. In the random movement model, the probability

of transitioning between the available and unavailable states between primary periods does not depend on the availability of the bird in the previous period. Gamma parameters are set as $\gamma' = \gamma''$. We, however, parameterized a model that assessed random movement for adults and Markov for juveniles to include the possibility of an adult random movement component in the data. Assumptions for robust design include those for CJS models, but also benefits from sampling that begins when the population has arrived and ends before it departs. We believe we sampled within these seasonal bounds, scheduling sampling occasions between late May and mid-August.

We assessed a candidate set of 5-6 age-specific competing models for Hammocks Beach State Park and Bald Head Island. We modeled survival and capture-recapture probability as constant (.) and year-specific (Yr) for both locations, and the interaction between age and year for Bald Head Island data. We modeled sex-specific survival for Bald Head Island using 5 competing models, with a model including age (SY, ASY) as an additive covariate. We assessed 8 additional models using the robust design with the Huggins closed captures data type option in program MARK (Cooch and White 2015; Kendall et al. 2015). We started with a Markovian and no-movement model that featured age-specific constant survival, age-specific constant gamma and constant capture-recapture probability. We then assessed no-movement and Markov models with year-specific age-specific survival (Age*Yr), age-specific constant gamma, and constant capture-recapture probability. We modified this model to parameterize gamma as random movements for adults and Markov for juveniles as noted above. Finally, the best supported model up to this point (i.e., no-movement) was parameterized in three

additional ways to further explore sources of variation in the data. First, a model with age-specific survival, year-specific age gammas, and constant capture-recapture probability. Second, a model with constant age-specific survival, age-specific gamma, and year-specific capture-recapture probability, and a third one with the same parameterization except that it featured year-specific age survival. A global (saturated) Markovian model was not included because it was overparameterized.

Recruitment rates were estimated following Kendall et al. (1997). The estimate of *in-situ* recruits as surviving juveniles is given by $\hat{B}_i^{(1)'} = \hat{N}_i^{(0)} \hat{\phi}_i^{(0)}$; the estimate of immigrant recruits by subtracting surviving adults and young from abundance is given by

$\hat{B}_i^{(1)''} = \hat{N}_{i+1}^{(1)} - \hat{N}_i^{(1)} \hat{\phi}_i^{(1)} - \hat{N}_i^{(0)} \hat{\phi}_i^{(0)}$; where juveniles (age 0) and adults (age 1). $N_i^{(a)}$ = abundance for age a at period i ; ϕ_i = survival rate (i to $i+1$) for age a , $B_i^{(1)'}$ = new recruits via *in situ* reproduction entering adult population between i and $i+1$, and present at $i+1$, $B_i^{(1)''}$ = new recruits via immigration entering adult pop between i and $i+1$, and present at $i+1$. We used annual estimates of adults to obtain an average estimate of exponential growth (r_{obs}) to assess the status of the species (no growth, increasing or decreasing). The estimate of r_{obs} is obtained by regressing log-transformed population estimates by years, which we express as lambda (λ , Caughley 1977).

We used Akaike's Information Criterion (AIC) to select the most parsimonious model. Models were ranked by corrected AIC (AIC_c), where the model with the lowest AIC_c was the model with the most support in the data. The difference in AIC_c units between the best supported model and any other model (ΔAIC_c) was used to calculate model weights (AIC_c weight), which indicate the relative likelihood of the model given

the (Burnham and Anderson 2002). Models with $\Delta AIC_c \leq 2$ were considered as models with the highest support. We considered an effect (i.e., covariate $\hat{\beta}$ coefficient) to be strongly supported if the 95% confidence interval did not overlap zero; weak otherwise. Parameter estimates \pm SE are reported.

RESULTS

Variation in the data for age-specific annual survival rates at Hammocks Beach State Park was explained by a model that featured age (HY, AHY) and year-specific recapture probability (AICc weight = 0.99; Table 1). Juvenile apparent annual survival was 0.31 ± 0.15 , and 0.66 ± 0.07 for adults. Highest support in the data also favored a similar model structure for estimates at Bald Head Island (AICc weight = 0.69, Table 1). Juvenile survival was estimated at 0.28 ± 0.04 and adults at 0.66 ± 0.02 . A competing model ($\Delta AIC_c \leq 2$) split survival rates into three age classes (HY, SY, ASY; AICc weight = 0.31). Juvenile survival was estimated at 0.28 ± 0.04 , sub-adult at 0.63 ± 0.05 , and adults at 0.66 ± 0.02 .

Variation in sex-specific annual survival rates was best explained by a model that featured sex and age (SY, ASY), and year-specific recapture probability (AICc weight = 0.61; Table 2). A competing model ($\Delta AIC_c \leq 2$) explained 30% of the variation in the data and featured only sex and year-specific recapture terms. Estimates of apparent survival for the top model were 0.63 ± 0.03 for males and 0.71 ± 0.03 for females. Although age was featured in the top model, its influence on sex-specific survival was

weak ($\hat{\beta} = 0.29 \pm 0.15$). Estimates from the competing model were 0.60 ± 0.03 (males) and 0.69 ± 0.03 (females).

Robust design analyses indicated that variation in Bald Head Island data was best explained by two models that split their support evenly (Table 3). These featured constant and year-specific age-specific survival (HY, AHY), age-specific gammas (no movement), and year-specific capture-recapture probability. Estimates of the constant age-specific model were 0.29 ± 0.04 (juveniles) and 0.62 ± 0.02 (adults). Year-specific estimates are shown in Figure 2. Population size estimates from the top model (no movement) averaged 120 juveniles and 329 adults annually, and annual estimates of adults suggested that the population was on an increasing (growth) path ($r_{\text{obs}} = 0.15$ or $\text{Lambda} = 1.16$). Annual fluctuations in population numbers are shown in figure 3. Annually and on average, 35 (95% CIs: 25-57) individuals were recruited *in-situ*, and 157 (95% CIs: 116-213) as immigrants at Bald Head Island. Annual variation in recruitment by source (*in-situ* or immigrants) is shown in figure 4.

DISCUSSION

We report the first estimates of juvenile survival rates for Painted Buntings, and of population size and recruitment rates for Bald Head Island. These findings addressed two of three research priorities required to better understand the demographic basis for declines of Painted Buntings in southeastern United States (DeSante et al. 2015). As expected, adults had higher survival estimates than juveniles. Lower juvenile survival has been attributed to lack of experience in behaviors such as foraging and predator avoidance (Gardali et al. 2003, Maness and Anderson 2013, Cox et al. 2014), and to

mortality risks during their first-year migration as they encounter novel habitats of varying quality (Sillett and Holmes 2002, Klaassen et al. 2014). Estimates of juvenile survival are rare (McKim-Louder et al. 2013), triggering theoretical work that suggests that juvenile survival for neotropical migrants needs to be at least 50% of adult rates to achieve population maintenance (Donovan et al. 1995, Faaborg et al. 2010, Ricklefs 1973). Our findings suggested that the growing population at BHI approximated that survival threshold (0.29 juveniles; 0.66 adults). Admittedly, our juvenile survival rates could be slightly lower because juveniles were captured in cages, often accompanied by parents, likely a few days after fledging. Reporting survival estimation contexts (e.g., time of season, time since fledging) is important to derive appropriate survival rates required for population maintenance (McKim-Louder et al. 2013).

The average population size at Bald Head Island was 449 individuals (HY+AHY). The observed average annual growth rate was 16% ($\lambda = 1.16$). This is in contrast to the negative growth rate of 13% ($\lambda = 0.87$) reported by DeSante et al. (2015) for a population in coastal South Carolina. The contrast in growth rates could be viewed as an indication that juvenile survival at BHI was indeed close to that required for maintenance (50% of adults). But it is also possible that Painted Buntings at BHI could have offset lower juvenile survival rates through the production of multiple broods per season (Farnsworth and Simons 2001, Podolsky et al. 2007). Certainly, the demographic vigor of the Bald Head Island population was affirmed by recruitment rates. We found that, on average, 35 (range: 15-84) local recruits were added annually to the population, providing evidence of successful reproductive activity. We note that the average number of immigrant

recruits (157) fluctuated markedly (range: 0 – 478). These estimates indicated that there were periodic influxes (colonization) of individuals, and that the magnitude of the influx likely reflected variable reproductive success outside Bald Head Island. Identifying populations interconnected with Bald Head Island was outside the scope of this work, but identified as a research priority by DeSante et al. (2015). We list two sites harboring populations that could interact with Bald Head Island, namely, Fort Fisher, 14 km to the north, and Sunset Beach, 49 km away to the south, both in North Carolina.

As hypothesized, estimates of age-specific survival rates between Bald Head Island and Hammocks Beach State Park were similar (95% CIs overlapped). Two factors may contribute to similarities between locations. First, maritime forests were the dominant vegetation type within sampled areas (13% Hammocks Beach State Park; 55% Bald Head Island) and this habitat is considered optimal for Painted Buntings (occurrence, reproduction; Hamel et al. 1982, Hamel 1992). In concert with this assessment, we found that the variability in occupancy probability was best explained by a model that featured maritime forest in coastal North Carolina (Chapter 3). Second, both populations occur in relative close proximity of each other (116 km), thus, similarities may reflect the influence of other factors associated with spatial correlation (e.g., similar suite of habitats).

Differences between estimates of second year (subadults) and >2 year old (adults) survival were negligible, but not between sex-specific survival rates, where apparent male survival was lower than females. These parameters influence how age of first breeding and sex-specific survival are parameterized in demographic models, important

as it may affect estimates of fecundity and life-time reproductive output (Noon and Sauer 2001, Bosman et al. 2013). Estimates of second year (SY) survival rates are of interest because male Painted Buntings start breeding at age 2, but in some populations it might be as early as age 1, similar to most females (Lowther et al. 2015). Thus, low male survival could be a constraint on fecundity if the predominant mating system is monogamy rather than a mix of monogamous and polygynous mating reported for some populations (Lowther et al. 2015). It would be of conservation value to investigate whether differences in sex-specific survival are commonplace throughout the species range or simply the case at Bald Head Island. If sex-specific survival differences are prevalent, one could test the hypothesis that sites with higher male mortality are also the sites with higher proportion of polygynous mating. We advance two mechanisms that may explain sex-specific differences. One is that male conspicuousness (i.e., distinctive feather coloration) increases their vulnerability to predation during periods of higher exposure (e.g., calling, mating). The other is that males are selectively trapped for pet trade (Sykes Jr. 2006, Sykes Jr. 2007), and removals might effectively be considered an additive source of mortality. Any evidence supporting linkages between male survival and trapping should prompt the formulation of management actions to curb its impact, including studies to generate estimates of allowable take as done for the Lazuli Bunting (*Passerina amoena*, Johnson et al. 2012).

Although vital rates reported in this study were limited to North Carolina, it contributes towards more robust inferences about eastern Painted Bunting status at a broader geographic scale in two ways. First, it provided confirmatory evidence that

apparent adult annual survival in coastal habitats hovers around 60% (DeSante et al. 2015, Lowther et al. 2015). Our point estimates fluctuated between 0.63 and 0.66. Second, it highlights a case study that links coastal habitats, like the one in Bald Head Island (e.g., low human density, maritime forests, marshes), to non-zero recruitment rates. The contrast between estimated lambda values at Bald Head Island (>1.0) and Dill Conservatory, South Carolina (<1.0 , DeSante et al. 2015) likely reflect heterogeneity in habitat quality in coastal habitats. These results underscore the importance of broadening geographic sampling efforts to generate demographic rates, which needs to include inland habitats to achieve a comprehensive quantitative assessment of the status and persistence of eastern Painted Buntings. We concur with Sykes Jr. and Holzman (2005) that restricted distribution of coastal habitats and their high rate of loss could undermine, in terms of persistence, the advantages conferred by the estimated 32% of the eastern population (401,660 individuals) occurring in those habitats (sensu Myers et al. 1987, R. Mordecai, USFWS, unpublished survey data). Therefore, we stress the importance of identifying habitat features for management associated with successful reproduction and survival in breeding and wintering habitats (Martin 1992, DeSante et al. 2015, Lowther et al. 2015), and of integrating demographic rates and habitat threat-risk assessment approaches to guide habitat conservation prioritization (Araújo and Williams 2000, Chapter 3).

LITERATURE CITED

- Araújo, M. B. and P. H. Williams. 2000. Selecting areas for species persistence using occurrence data. *Biological Conservation* 96: 331-345.
- Avinet. 2015. Avinet- Leg Bands. <https://www.avinet.com/en/bands>.
- Bosman, D. S., H. J. P. Vercrujse, E. W. M. Stienen, M. Vincx, and L. Lens. 2013. Age of First Breeding Interacts with Pre- and Post-Recruitment Experience in Shaping Breeding Phenology in a Long-Lived Gull. *PLoS ONE* 8(12): e82093. doi:10.1371/journal.pone.0082093
- Bourdeau, P. F., and H. J. Oosting. 1959. The Maritime Live Oak Forest in North Carolina. *North* 40:148–152.
- Burleigh, T. D. 1935. The Breeding Range of the Painted Bunting in South Carolina. *The Auk* 52:52–53.
- Burnham, K. P., and D. R. Anderson. 2002. Model selection and multimodel inference : a practical information-theoretic approach. 2nd edition. Springer US, New York.
- Caughley, G. 1977. Analysis of Vertebrate Populations. New York Wiley & Sons. 234 p.
- Cooch, E., and G. White. 2015. Program MARK “ A Gentle Introduction .” 14th edition. Ithaca, NY; Ft. Collins, CO.
- Cox, W. A., F. R. Thompson, A. S. Cox, and J. Faaborg. 2014. Post-fledging survival in passerine birds and the value of post-fledging studies to conservation. *The Journal of Wildlife Management* 78:183–193.
- Delany, M. F., B. Pranty, and R. a Kiltie. 2013. Painted Bunting Abundance and Habitat Use in Florida. *Southwestern Naturalist* 12:61–72.
- DeSante, D. F., D. R. Kaschube, and J. F. Saracco. 2015. Vital Rates of North American Landbirds. www.VitalRatesOfNorthAmericanLandbirds.org: The Institute for Bird Populations.
- Dickerson, T. J. 1978. Floristic Survey and Plant Community Descriptions of Bear Island (Hammocks Beach State Park), Onslow County, North Carolina. NC State University.
- Donovan, T. M., F. R. Thompson, J. Faaborg,, and J. R. Probst. 1995. Reproductive success of migratory birds in habitat sources and sinks. *Conservation Biology* 9:1380-1395.
- Dunn, E. H. 2004. Relationships Among Body Mass, Fat, Wing Length, Age, and Sex for

170 Species of Birds Banded At Powdermill Nature Reserve. *The Wilson Bulletin* 116:189–190.

- Faaborg, J., R. T. Holmes, A. D. Anders, K. L. Bildstein, and K. M. Dugger. 2010. Recent advances in understanding migration systems of New World land birds. *Ecological Monographs* 80:3-48.
- Farnsworth, G. L. and T. R. Simons. 2001. How many baskets? Clutch sizes that maximize annual fecundity of multiple-brooded birds. *Auk* 118:973–982.
- Froehlich, D. 2003. Ageing North American landbirds by molt limits and plumage criteria: A photographic companion to the identification guide to North American birds, Part 1.
- Gardali, T., D. C. Barton, J. D. White, and G. R. Geupel. 2003. Juvenile and adult survivorship of swainson's thrush (*Catharus ustulatus*) in Coastal California: Annual estimates using capture-recapture analyses. *Auk* 120:1188–1194.
- United States Geological Survey. 2015. The North American Bird Banding Program. <https://www.pwrc.usgs.gov/bbl/>.
- Guglielmo, C.G., D.J. Cerasale, and C. Eldermire. 2005. A field validation of plasma metabolite profiling to assess refueling performance of migratory birds. *Physiological and Biochemical Zoology* 78:116-125.
- Hamel, P. B., H. E. LeGrand Jr., M. R. Lennartz and S.A. Gauthreaux. 1982. Bird habitat relationships on southeastern forest lands. Asheville, NC: U.S. For. Serv.
- Hamel, P. B. 1992. Land manager's guide to the birds of the south. The nature Conservancy, Chapel Hill, NC and U.S. Forest Service, Southern Region, Atlanta, GA.
- Herr, C. a., P. W. Sykes, and J. Klicka. 2011. Phylogeography of a vanishing North American songbird: The Painted Bunting (*Passerina ciris*). *Conservation Genetics* 12:1395–1410.
- Hobbs, M., and J. M. Meyers. 1999. A Bird Without Equal- Will It Survive? Georgia Department of Natural Resources Wildlife Resources Division.
- Johnson, F. A., M. A. H. Walters and G. S. Boomer. 2012. Allowable levels of take for the trade in Nearctic songbirds. *Ecological Applications* no. 22 (4):1114-1130.
- Jones, G., A. Snider, and S. Luo. 2013. Changes in the Extent of North Carolina Barrier Island Maritime Forests 1988-2011: An Assessment of Past Efforts at Protection. *Journal of Forestry* 111:186–193.

- Kendall, W. L., K. H. Pollock, and C. Brownie. 1995. A Likelihood-Based Approach to Capture-Recapture Estimation of Demographic Parameters under the Robust Design. *Biometrics* 51:293–308.
- Kendall, W. L., J. D. Nichols, and J. E. Hines. 1997. Estimating Temporary Emigration Using Capture-Recapture Data with Pollock’s Robust Design. *Ecological Society of America* 78:563–578.
- Kendall, W., and C. Fish. 2015. Chapter 15 The “robust design.” Pages 15-1-15–50 Program MARK: ‘A Gentle Introduction.
- Klaassen, R. H. G., M. Hake, R. Strandberg, B. J. Koks, C. Trierweiler, K-M. Exo, F. Bairlein, and T. Alerstam. 2014. When and where does mortality occur in migratory birds? Direct evidence from long-term satellite tracking of raptors *Journal of Animal Ecology* 83:176–184.
- Lopazanski, M. J., J. P. Evans, and R. E. Shaw. 1988. An Assessment of Maritime Forest Resources on the North Carolina Coast 27706.
- Lowther, P E., S. M. Lanyon, and C. W. Thompson, 2015. Painted Bunting (*Passerina ciris*). *The Birds of North America*. The America Ornithologist’s Union. Philadelphia, PA.
- Maness, T. J., and D. J. Anderson. 2013. Predictors of Juvenile Survival in Birds. *Ornithological Monographs* 2013:1–55.
- Martin, T. E. 1992. Breeding productivity considerations: what are the appropriate habitat features for management? Pp 455-473 in *Ecology and Conservation of Neotropical migrant landbirds* (J. M. Hagan, III, and D. W. Johnston, eds.). Smithsonian Institution Press, Washington, DC.
- McKim-Louder, M. I., J. P. Hoover, T. J. Benson, and W. M. Schelsky. 2013. Juvenile survival in a Neotropical Migratory Songbird is lower than expected. *PLoS ONE* 8(2): e56059. doi:10.1371/journal.pone0056059.
- Meyers, J. M. 2011. Population Densities of Painted Buntings in the Southeastern United States. *Southeastern Naturalist* 10:345–356.
- Myers, J. P., R. I. G. Morrison, P. Z. Antas, B. A. Harrington, T. E Lovejoy, M. Sallaberry, S. E. Senner and A. Tarak. 1987. Conservation strategy for migratory species. *American Scientist* 75: 18-26.
- Noon, B. R., and J. R. Sauer. 2001. Population Models for Passerine Birds: Structure, Parameterization, and Analysis. *Page Wildlife 2001: Populations* 2.
- North Carolina Coastal Reserve. 2015. Bald Head Woods.

<http://www.nccoastalreserve.net/web/crp/bald-head-woods>.

North Carolina Department of Parks and Recreation. 2016. Hammocks Beach State Park-Ecology. <http://www.ncparks.gov/hammocks-beach-state-park/ecology>.

Oosting, H. J. 1954. Ecological Processes and Vegetation of the Maritime Strand in the Southeastern United States. *Botanical Review* 20:226–262.

Podolsky, A. L., T. R. Simons, and J. A. Collazo. 2007. Modeling population growth of the ovenbird in the southern Appalachians. *Auk* 124:1359-1372.

Pollock, K. H. 1982. A Capture-Recapture Design Robust to Unequal Probability of Capture. *Journal of Wildlife Management* 46:752–757.

Pollock, K. H., J. D. Nichols, C. Brownie, and J. E. Hines. 1990. Statistical Inference for Capture-Recapture Experiments. *Wildlife Monographs* 107:1–97.

Ricklefs, R. E. 1973. Fecundity, mortality, and avian demography. In Farner D. S., editor. *Breeding biology of birds*. National Academy of Science: Washington, D.C. 366-435 p.

Sauer, J. R., J. E. Hines, J. E. Fallon, K. L. Pardieck, D. . Ziolkowski Jr, and W. A. Link. 2014. *The North American Breeding Bird Survey, Results and Analysis 1966-2013*. Laurel, MD.

Shipley, J. R., A. Contina, N. Batbayar, E. S. Bridge, A. T. Peterson, and J. F. Kelly. 2013. Niche conservatism and disjunct populations. *The Auk* 130:476–486.

Sillett, T. S., and R. T. Holmes. 2002. Variation in survivorship of a migratory songbird its annual cycle throughout. *Ecology* 71:296–308.

Springborn, E. G., and J. M. Meyers. 2005. Home range and survival of breeding Painted Buntings on Sapelo Island , Georgia. *Wildlife Society Bulletin* 33:1432–1439.

Sykes Jr., P. W., and S. Holzman. 2005. Current range of the eastern population of Painted Bunting (*Passerina ciris*). *North American Birds* 59:4–17.

Sykes Jr., P. W. 2006. An Efficient Method of Capturing Painted Buntings and Other Small Granivorous Passerines. *North American Bird Bander* 31:110–114.

Sykes Jr., P. W., S. Holzman, and E. E. Iñigo-Elias. 2007. Current range of the eastern population of Painted Bunting (*Passerina ciris*)— Part II : Winter range. *North American Birds* 61:378–406.

Thompson, C. W. 1991a. Is the Painted Bunting actually two species? Problems determining species limits between allopatric populations. *The Condor* 93:987–1000.

Thompson, C. W. 1991b. The sequence of molts and plumages in Painted Buntings and implications for theories of delayed plumage maturation. *The Condor* 93:209–235.

White, G. C., and K. P. Burnham. 1999. Program MARK: survival estimation from populations of marked animals. *Bird Study* 46:S120–S139.

White, G.C. 2015. Markhelp. Program, Colorado State University, Fort Collins.

TABLES AND FIGURES

Table 1. Model selection table for Cormack-Jolly-Seber models used to estimate apparent, age-specific annual survival of Painted Buntings at Hammocks Beach State Park (top table) and Bald Head Island (bottom table), North Carolina 2007-2015. Models are ranked by their respective AICc values.

Hammock Beach State Park Models	AICc	Delta AICc	AICc Weights	Num. Par	Deviance
{Phi(Age) p(Yr)}	249.05	0.00	0.99	11.00	61.84
{Phi(Age) p(.)}	260.75	11.70	0.00	3.00	91.13
{Phi(Age) p(Age*Yr)}	260.92	11.87	0.00	18.00	56.70
{Phi(Age*Yr) p(Age*Yr)}	265.32	16.27	0.00	26.00	39.53
{Phi(Age*Yr) p(.)}	276.49	27.43	0.00	17.00	74.80

Bald Head Island Models	AICc	Delta AICc	AICc Weights	Num. Par	Deviance
{Phi(Juv vs ASY) p(Yr)}	1927.14	0.00	0.69	10.00	303.23
{Phi(Age) p(Yr)}	1928.77	1.64	0.31	11.00	302.83
{Phi(Age*Yr) p(Yr)}	1942.98	15.85	0.00	31.00	275.47
{Phi(Age*Yr) p(Age)}	1976.56	49.42	0.00	27.00	317.48
{Phi(Age) p(Age)}	1995.63	68.49	0.00	6.00	379.84
{Phi(Age) p(.)}	1995.97	68.83	0.00	4.00	384.22
{Phi(.) p(.)}	2046.01	118.88	0.00	2.00	438.29

Table 2. Model selection table for Cormack-Jolly-Seber models used to estimate apparent, sex-specific annual survival of Painted Buntings at Bald Head Island, North Carolina 2007-2015. Models are ranked by their respective AICc values. Competing models ($\Delta\text{AICc} \leq 2$) are indicated with an asterisk.

Model	AICc	Delta AICc	AICc Weights	Num. Par	Deviance
*{Phi(Sex+AGE) p(Yr)}	1527.04	0.00	0.61	11.00	1504.72
*{Phi(Sex) p(Yr)}	1528.46	1.42	0.30	10.00	1508.19
{Phi(Yr) p(Yr)}	1531.77	4.73	0.06	15.00	1501.17
{Phi(Sex*Yr) p(Sex*Yr)}	1533.23	6.18	0.03	30.00	1470.87
{Phi(Sex) p(Sex)}	1578.10	51.05	0.00	4.00	1570.05
{Phi(.) p(.)}	1584.61	57.57	0.00	2.00	1580.60

Table 3. Model selection table for robust design models used to estimate apparent, age-specific annual survival of Painted Buntings at Bald Head Island, North Carolina 2007-2015. Secondary sampling occasions (3) occurred in late May/mid-June, mid-July, and mid-August. Models are ranked by their respective AICc values. Competing models ($\Delta AICc \leq 2$) are indicated with an asterisk.

Model	AICc	Delta AICc	AICc Weights	Num. Par	Deviance
*{S(Age*Yr) Gamma"(g=0) Gamma'(g=1) p& c (Yr)}no movemenYr	6633.48	0.00	0.50	61.00	8427.68
*{S(Age) Gamma"(g=0) Gamma'(g=1) p& c (Yr)}no movemenYr	6633.51	0.03	0.50	49.00	8453.32
{S(Age*Yr) Gamma"(g=0) Gamma'(g=1) p (.), c(.)no movemenYr	7097.93	464.45	0.00	35.00	8947.16
{S(Age*Yr) Gamma"()=Gamma'() p (.), c(.)} Random AdulYrs; Markov Juv}}	7101.59	468.11	0.00	38.00	8944.56
{S(Age*Yr) Gamma"(Age) Gamma'(Age) p (.), c(.)}Markov	7103.42	469.93	0.00	39.00	8944.29
{S(Age) Gamma"(Age*Yr=0) Gamma'(Age*Yr=1) p (.), c (.)} no movemenYr}	7117.90	484.42	0.00	20.00	8998.12
{S(Age) Gamma" (Age=0) Gamma' (Age=1), p(.), c(.) (no movemenYr)}	7117.90	484.42	0.00	20.00	8998.13
{S(Age) Gamma"(Age*Yr) Gamma'(Age*Yr) p (.), c (.)} Markov}	7145.69	512.21	0.00	41.00	8982.37

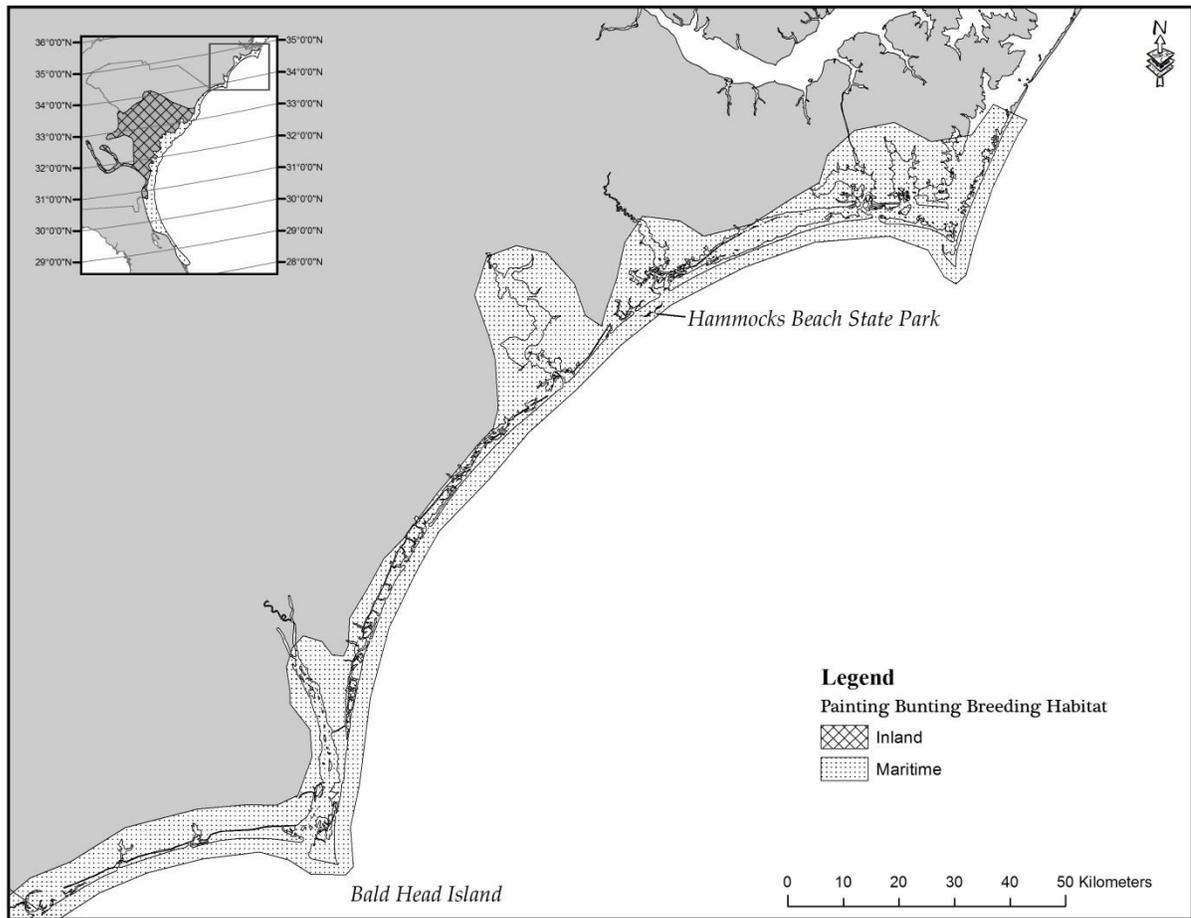


Figure 1. Map of coastal North Carolina depicting the location of Hammocks Beach State Park and Bald Head Island. The inset depicts southeastern United States and the range of Painted Buntings along the coast of Florida and North Carolina, and coast and inland South Carolina and Georgia.

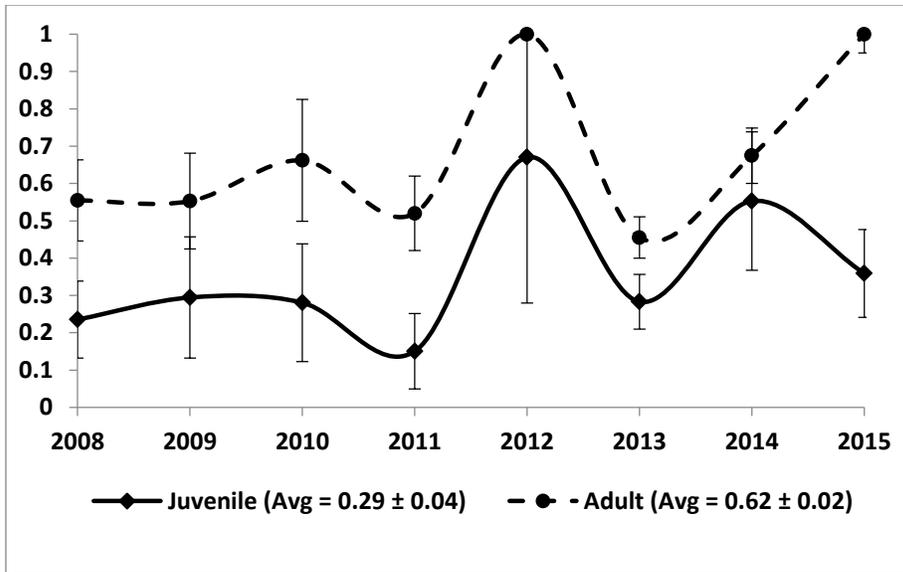


Figure 2. Estimates of apparent, age-specific annual survival estimates of Painted Buntings from 2007 to 2015 at Bald Head Island, North Carolina. Average estimates for the study period and parameter estimates \pm SE are reported.

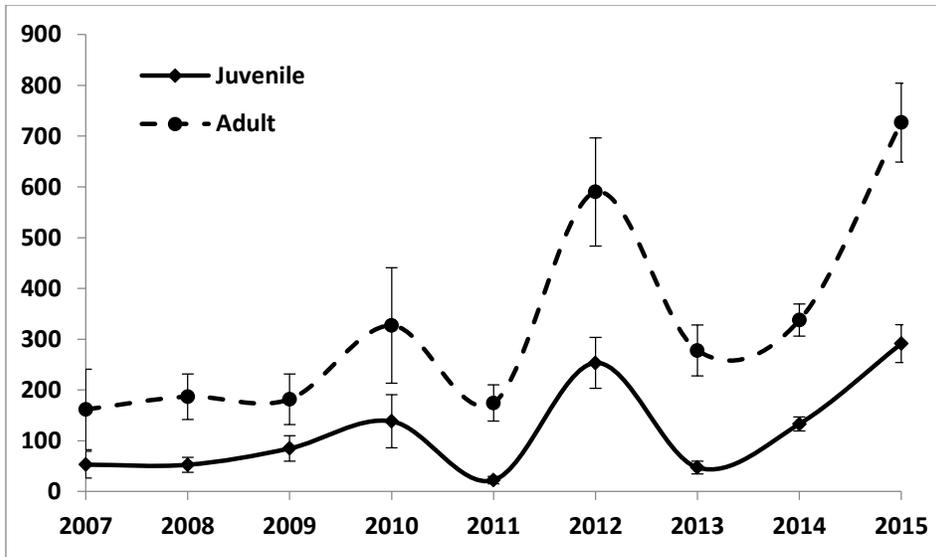


Figure 3. Estimates of age-specific population size of Painted Buntings from 2007 to 2015 at Bald Head Island, North Carolina. Parameter estimates \pm SE are reported. Estimates were derived from the robust design model with highest support (AICc Weights) in the data (no-movement).

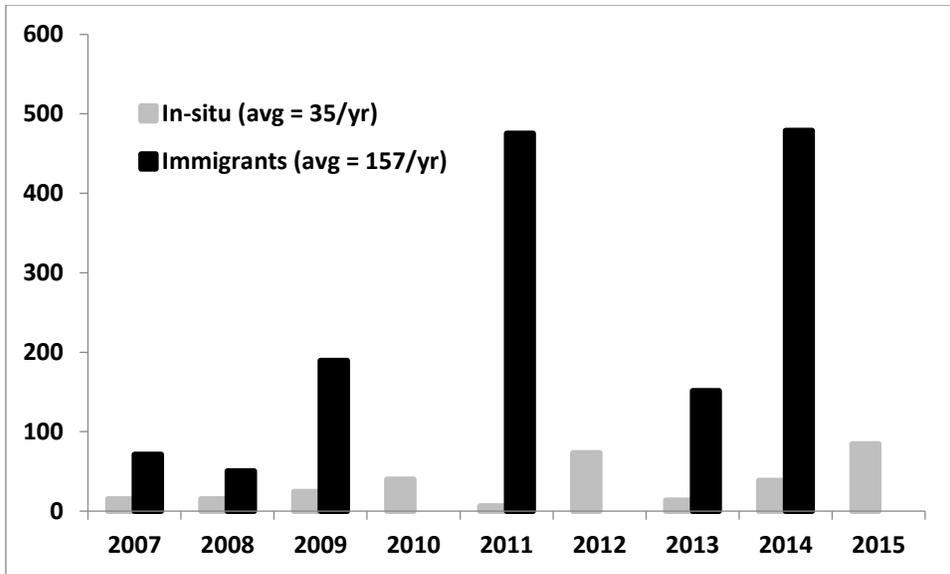


Figure 4. Estimates of recruitment rates of Painted Buntings from 2007 to 2015 at Bald Head Island, North Carolina. Estimates were derived from the robust design model with highest support (AICc Weights) in the data (no-movement). Recruitment is classified by origin: *in-situ* or immigrants. Precision levels (95% CIs) for annual average by origin is reported in the text.

CHAPTER 3

PRIORITIZING AREAS FOR SPECIES PERSISTENCE USING OCCUPANCY AND HABITAT THREAT-RISK ANALYSIS.

ABSTRACT

Effective reserves protect species by increasing their probability of persistence. However, very few area selection approaches utilize this metric, largely because scant data preclude its estimation. Instead proxies of persistence are derived from incidence functions and metapopulation theory (e.g., structural connectivity, patch size), and more recently, minimizing purported adverse effects of threats (e.g., urban growth). Here we estimated persistence probability for 5 areas of conservation interest (AOI) using occupancy analyses and a habitat threat-risk assessment tool for Painted Buntings (*Passerina ciris*) in North Carolina. We used occupancy analyses to identify the habitat, among 12 classes, that explained most of the variation in 133 sampling units that reflected natal dispersal. We used the tool to estimate the added risk posed by the projected loss of habitat identified by the occupancy analysis. Persistence probability was $1 - (\text{probability of an AOI going extinct (unoccupied)} * \text{added risk})$. Painted Bunting occupancy increased with increasing amount of maritime forests. Persistence probability ranged from 0.82 to 0.98. AOIs could be ranked by persistence as a straightforward criteria informing area selection, but potential trade-offs associated with a more complex decision scenarios highlighted the importance of benchmark habitat threats and occupancy levels, and the timing of conservation actions given the magnitude of projected changes. Our joint analyses approach represents an advance in the estimation of persistence to guide area-

selection. Existing multi-season single and community occupancy models, coupled with measures of uncertainty for projected land cover change, represent a path forward to model persistence in a non-stationary environment.

INTRODUCTION

Areas set aside for conservation are often selected to ensure the persistence of a species or multiple elements of biological communities (Araújo and Williams 2000, Araújo et al. 2002, Kareiva and Marvier 2015, Lawler et al. 2003, Pimm and Lawton 1998, Rodrigues et al. 2000). These areas are designed to curb the adverse effects on persistence caused by human-induced habitat loss and degradation, already occurring on an estimated 39-50% of earth's terrestrial surface (Vitousek 1997, Watling et al. 2011). Some species are adept at exploiting human-modified habitats, a trait that might expand conservation opportunities (Radeloff et al. 2015). However, the rate and extent at which landscape changes are occurring underscore the importance of developing robust area-selection approaches for conservation (Araújo and Williams 2000, Moilanen et al. 2009, Arponen 2012).

Extensive effort has been devoted to area selection methodology since the 1980's (Faith and Walker 1996, Frankel and Soulé 1981, Margules et al. 1988, Moilanen et al. 2005, Moilanen et al. 2009). The process is complex because it combines value-based objectives with ecological processes and objectives (Arponen 2012). It is also made difficult because data to understand biological processes and responses to threats or management actions are scant. For this reason, conservation design and reserve selection processes have relied on principles that purportedly lead to higher persistence. Among

the most prominent are connectivity, patch size, and representativeness, which together create a spatial and temporal framework conducive to persistence via metapopulation dynamics (Hanski 1998). Recently, area selection algorithms have incorporated threats that might directly or indirectly affect the ecological integrity of conservation areas. A threat is defined as any physical, chemical or biological entity that can induce an adverse response (SETAC 1997). Lawler et al. (2003) illustrate the application of area selection algorithms by integrating two factors, namely, representativeness and levels of impervious/urban surfaces (human density) surrounding areas of conservation interest as an indicator of vulnerability that could undermine habitat quality. This approach acknowledges that a suite of other extrinsic factors (e.g., human-induced changes) might influence persistence. It does not, however, yield explicit estimates of persistence (Araújo and Williams 2000, Araújo et al. 2002, Moilanen et al. 2009).

Araújo and Williams (2000) advanced an approach to estimate persistence that bridges processes that characterize persistence with factors (threats) that impinge those processes. Specifically, it expresses changes in persistence in terms of the likelihood that an area of interest will remain occupied, given a projected magnitude of a threat and the likelihood it will occur. This framework is suited for multiple taxa, including North American Neotropical migratory avian species. The status of many species in this group is threatened by loss of habitat and degradation (De Wan 2009, Robbins et al. 1989, Sauer et al. 2014). In southeastern United States this conservation challenge is exemplified by the Painted Bunting (*Passerina ciris*), a species considered at risk of becoming threatened in North Carolina (North Carolina Wildlife Resources Commission

2015). Eastern Painted Buntings occur in northern Florida, Georgia, South Carolina, and southern North Carolina (Herr et al. 2011). Its regional population is estimated at approximately 1.2 million individuals (R. Mordecai, US Fish and Wildlife Service, unpublished 2007-2009 survey data). Sixty-eight percent (68%) of the population occurs in inland habitats in South Carolina and Georgia, but the remaining in coastal habitats. In North Carolina, for example, the estimated population is 154,600 (95% CIs: 59,000 - 251,000), but restricted to a narrow coastal strip from Croatan National Forest (northern boundary) to the border with South Carolina (Sykes Jr. and Holzman 2005). Habitat conservation gains impetus because the species vulnerability is believed to be largely driven by habitat loss, not constraints imposed by population numbers (Sykes Jr. and Holzman 2005). In other words, the advantage conferred by numbers in most Neotropical migrants, expressed as P_i^N (where P=probability of extinction, N = population size), is undercut by loss of suitable coastal habitat (sensu Myers et al. 1987).

In this work, we evaluated persistence in five areas of conservation interest that harbor breeding populations of Painted Buntings in southeastern North Carolina. Our parameterization of Araújo and Williams's area-selection prioritization framework is unique because it used input data from occupancy analyses (MacKenzie et al. 2006) and a habitat threat-risk decision support tool (Williams et al. 2015). Occupancy is defined as the probability of detecting at least one individual of Painted Bunting in an area of conservation interest, provided that it is available to be detected (MacKenzie et al. 2006). We used occupancy analyses to determine the habitat type accounting for most of the variation in the data, and as such, establish a functional relationship between occupancy

and habitat availability (threat-response curve). The habitat threat-risk decision tool was used to determine if areas of interest were exposed to threats (e.g., habitat loss) and the severity (magnitude) of those threats. The latter is expressed as a risk or probability that a change of a specified severity will occur. A risk is defined as a state of uncertainty for which there exists an undesirable outcome (Conroy and Peterson 2013). Estimates of risks are based on projections of habitat change (transitions) from 2000 to 2050 (Costanza et al. 2015a, 2015b). We prioritize (rank) areas in terms of persistence, discuss the conservation implications of results, and outline recommendations to improve the utility of the approach advanced by Araújo and Williams (2000) in a decision context.

METHODS

STUDY AREA

This study focused on the breeding range of the Painted Bunting in North Carolina, and specifically on five areas of conservation interest (AOI) within this range (Figure 1). The Painted Bunting breeds along the southern coast of North Carolina in a narrow strip of coastal habitat that ranges north to Morehead City, NC and south to Sunset Beach, NC spanning about 1,547 km². This area is characterized by Middle Atlantic Coastal Plain and the predominant habitat is the Carolina Flatwoods or Tidewater with some Carolina Barrier Islands and coastal marshes (Griffith 2002, North Carolina Wildlife Resources Commission 2015). Approximately 5.1% of this range is covered by maritime forest (77 km²), characterized by southern live oak (*Quercus virginiana*), sand laurel oak (*Quercus hemisphaerica*), and loblolly pine (*Pinus taeda*, North Carolina Wildlife Resources Commission 2015). The majority of the Painted

Bunting's breeding range, however, includes fresh and brackish water (47.4%), developed land (10.9%) and fresh or brackish tidal marsh (9.7%). This tidal marsh habitat is dominated by salt-tolerant vegetation including black needlerush (*Juncus roemerianus*), and saltgrass (*Distichlis spicata*), while edged by bald cypress (*Taxodium distichum*), Atlantic white cedar (*Chamaecyparis thyoides*), and yaupon (*Ilex vomitoria*).

Areas of conservation interest (AOI) were selected with the assistance of Painted Bunting experts from the North Carolina Museum of Sciences (Figure 1). Areas were singled out because they harbor breeding populations and could be considered potential candidates in a State-wide habitat conservation strategy. There are two of these areas in the southern extent of the range in North Carolina: Sunset Beach area (AOI 1) and the Bald Head Island/Wilmington (AOI 2). The remaining three AOIs are located near the northern extent of the Painted Bunting's breeding range in the State: Topsail Beach/Hammocks Beach State Park (AOI 3), Camp Lejeune (AOI 4) and the Croatan National Forest (AOI 5). All areas of interest contained multiple survey stations used by the USFWS to survey Painted Bunting in 2008-2009 (see methods, Figure 1).

AVIAN SURVEYS

We obtained raw data (presence/non-presence) collected at 133 stations during surveys conducted in North Carolina in 2008 and 2009. The sampling design was developed by the U.S. Fish and Wildlife Service (USFWS) Atlantic Coast Joint Venture, Laurel, Maryland. Delany (2011), Delany et al. (2013), and R. Mordecai (USFWS, South Atlantic Landscape Conservation Cooperatives, pers. comm.) provided a detailed account of the sampling protocol, and here we summarized its salient features. First, the eastern

range of the species was overlaid with grid blocks (0.05 degree, 27 km²). Blocks containing > 30% unsuitable habitat (e.g., metropolitan areas) were excluded from the pool. Then, following a pre-specified allocation rule defined by the range of the species per State, blocks were randomly selected and survey stations established within a block if there was at least 1% of suitable habitat within 200 m radius of the station center and were accessible by vehicle. Blocks were excluded from consideration if the canopy of habitat was closed, contained impervious surfaces, agricultural fields without shrubs, open water, and mowed lawn with no trees or shrubs.

Painted Buntings were surveyed for 5 minutes at each station, recording visual and auditory detections within 75 m radius from the station center (Ralph et al. 1993). Surveys were conducted from 01 May to 15 June from sunrise to 1000 hrs. Each station was visited 3 times each year. We characterized vegetation cover for each station using a 200 m radius, or the distance used to determine whether a station was included or not in the survey (Delany 2011). We also characterized vegetation within a 6 km radius of each station. This is the estimated natal dispersal distance, or the $\leq 90\%$ probability that juveniles would return alive at $t+1$, and if so, return to within 6 km of its place of birth (Sutherland et al. 2000). Although there are no empirical data to confirm this estimate for juveniles, Fisk (1974) indicated that Painted Buntings returned to 1-8 km from the station at which originally banded. We note that natal dispersal units (6 km radius) are the primary sampling unit in this work. We created 21 such units, each maximizing the number of stations contained within them, provided the distance between the nearest two stations would be $\cong 6$ km (Figure 1). This process yielded natal dispersal units with 3-12

stations each. We characterized vegetation (i.e., % within specified radius) using 12 cover classes used by USFWS during region-wide surveys and reported by Hamel et al. (1982) and Hamel (1992). We used land cover thematic data from Earnhardt (2012). Landcover types included maritime forests, maritime scrub-shrub, pasture/crops, oak-hickory, bay-swamp pocosin, long-leaf pine, oak-gum-cypress, salt marsh, open pine/hardwood, managed pine, and high and low density residential.

HABITAT THREAT AND RISK ASSESSMENT

We used a spatially-explicit decision support tool (DST) to evaluate threat and risks to areas of conservation interest (AOI) in North Carolina (Williams et al. 2015). The DST contains information on 22 threats (Appendix 1). These data are created at multiple, and often, disparate spatial scales. Thus, to represent multi-source data on the same spatial scale (standardized) and facilitate its interpretation, the tool conducts assessments using single spatial units, based on sub-watersheds (12 digit Hydrologic Unit Code or HUC12). Data from each threat or a combination of threats is projected over the 1,720 sub-watersheds. Some threat data are static, that is, consist of a unique value that may be updated periodically (e.g., number of dams), but others (e.g., loss of habitat) can be projected into the future (dynamic) for preselected time steps (e.g., Constanza 2015a, b).

In this study, we selected two threats that were relevant to the Painted Bunting (coastal habitats), and allowed us to project into the future. These were estimated loss of specified habitats (e.g., suitable) and loss of terrestrial land cover based on sea level rise projections. Projections were made based on habitat transition models developed by Constanza et al. (2015a,b) and SLAMM (Clough 2008, Rubino 2009). Land cover

transitions in the former models are influenced by the effects of urban growth, wildfires and fire suppression, forest harvest and thinning, and changes to forests due to vegetation succession. SLAMM simulates transforming coastal environments by accounting for nearshore geomorphological processes such as accretion, erosion, and marsh migration dynamics due to long-term seas level rise (Park et al. 1986). Terrestrial land cover change is the loss (ha) of all land cover types (including marshes) to open water since 2000.

We used output from the DST tool to conduct the two components of a threat-risk analysis (EPA 1998). The first is a threat or exposure characterization analysis, summarizing the number of threats, their occurrence (proportion within an area of interest), and their intensity or severity relative to an area of interest. Depending on how an AOI polygon is created, the DST also yields an estimate of proximity (spatial) of threats. These data are essential to assess the ecological effects of a given threat, the second component of a threat-risk analysis. Evaluating ecological effects, however, requires a threat-response curve or functional relationship between a threat and the response of the resource in question.

We note that severity of a threat (e.g., percent loss of habitat) is expressed as a risk or probability obtained from a cumulative distribution function (CDF) made up of the 1720 HUCs state-wide. Results are interpreted as the probability of obtaining a value equal or less than a specified value (x) for a given threat. We deviate from this procedure if a resource of interest has a restricted, but known distribution (e.g., coastal obligate habitats). In such instances, the number of HUCs with zero or near zero chance of registering a threat (e.g., loss of coastal habitat) is large, skewing the CDF markedly.

Thus, we rescaled the CDF to have its point of origin at the lowest, positive value of change (>0) for a given threat. Our rescaled expression of severity risk is interpreted as the probability of obtaining a positive value (e.g., loss of habitat) equal to or less than a specified value (x) for a given threat (Appendix 2).

DATA ANALYSIS

OCCUPANCY

We used a single-season, single-species occupancy approach to estimate the probability that a sampling unit (see below) was occupied by a Painted Bunting, conditioned on habitat type (covariate) and the availability of Painted Buntings to be detected (MacKenzie et al. 2002). We combined survey data from 2008 and 2009 (3 visits/station/year) after determining that year has a weak influence on occupancy and detection probability (Appendix 3). No single model with year effects [e.g., Ψ (YR) or p (YR)] accounted for most of the variation in the data. The top model featured year effects on detection, but its influence was weak ($\text{Beta} = 0.54 \pm 0.30$). Combined data had 6 columns of encounter histories, but we note that some stations (53/133) were not sampled both years. We account for this difference in sampling effort by modeling detection probability (p) by whether stations were surveyed 3 or 6 times.

We modeled data at two spatial scales. The first was at 200 m as we reasoned that this scale might pick preferences at local scales (e.g., foraging, breeding; Johnson 1980). We ran 12 competing models testing for constant effects in occupancy (\cdot) and the influence of habitat covariates on occupancy. As indicated above, detection was

conditioned on effort [p (effort)]. We also modeled data at the scale of natal dispersal units. We created 15 competing models that had the same structure as described above, but vegetation covariates were expressed as a function of a sampling unit of 6 km radius. Our final expression of occupancy grouped AOIs based on the spatial patterns of natal dispersal units occupancy estimates. We designated three regions that encapsulated these patterns, and labeled them as the southern region (Bald Head Island/Wilmington area/Sunset Beach), intermediate region (Top Sail/Hammocks Beach State Park region), and northern region (Camp Lejeune/Croatan National Forest). This covariate acknowledged that there could be contagion or spatial correlation influencing persistence (Araújo et al. 2002).

RISK ASSESSMENT

We evaluated 5 areas of conservation interest (AOI) along North Carolina's coast (Figure 1). These areas contained 2-3 natal dispersal units each, and ranged in size from 46 to 79 ha. We focused the habitat threat-risk assessment on the habitat type (land cover) that accounted for most of the variation in data in the aforementioned occupancy models. We estimated the loss of habitat in years 2020 and 2050 relative to our baseline of 2000. We note that these projections do not report estimates of model uncertainty, that is, that owing to our inability to know the behavior of the system accurately (Uusitalo et al. 2015). We do, however, report sampling error associated with average severity value per AOI, and how it reflects itself on estimates of persistence (95% CIs).

AREA PRIORITIZATION

We followed steps outlined by Araújo and Williams (2000) to estimate persistence per area of interest (Figure 2). First, we used occupancy models to establish the functional relationship between occupancy and habitat (threat-response curve). The utility of results is predicated on at least two assumptions. First, species occupancy is a function of habitat quality. The reader is cautioned about the possibility that some habitats could misrepresent their true value as in the case of ecological traps or population sinks (Battin 2004, Kareiva and Marvier 2015). Second, projected consequences or change in occupancy is predicated on a non-changing threat-response function between the time the relationship was established (2008-2009 survey data) and the future time step. This is a niche approach (Araújo and William 2000), thus, the reader is also advised to use results carefully to gain insights and guide next steps in analyses or decision processes.

The next step consisted of expressing local occupancy probability as the risk of local extinction. In this case, extinction refers to the probability that our sampling unit (natal dispersal unit) no longer harbors at least 1 Painted Bunting. We use the term “local” to indicate that we only make inferences about the occupancy “state” of the sampling unit, not the Painted Bunting population occurring throughout the coastal zone of North Carolina. We first estimate the added risk posed by a threat. That is, how likely will a threat (e.g, loss of habitat) in an area of interest (a) change by a specified amount or severity x at time i , given that Painted Buntings are vulnerable to such a threat ($\text{Threat}_{a,t}$, Vulnerability_i). For this estimate, we used the average severity value per AOI. Then, we

estimated the extinction risk for Painted Buntings (i) at an area of interest (a) at time t as the product of the projected AOI extinction (based on 1-occupancy) and the risk of the threat occurring (Added Risk*(1-occupancy $_{i,a,t}$)). This extinction risk is then expressed as a probability of persistence by subtraction (1-extinction risk $_{i,a,t}$). We also ranked areas of interest by projected loss of terrestrial landcover based on sea level rise projections. In this case, we only conducted an exposure analysis, characterizing threats due to sea level rise as a function of occurrence and severity. The threat-response curve to such changes is unknown for Painted Buntings. We used this threat characterization to illustrate how complementary information could help inform prioritizing habitat for conservation.

RESULTS

Maritime forest consistently accounted for the habitat variation in occupancy in Painted Buntings, regardless of the spatial scale of the analyses (Table 1). At the survey station scale (200 m), the top model featured maritime forest (AIC $_{wt} = 0.92$). The influence of maritime forest on occupancy was strong ($Beta = 1.45 \pm 0.64$). At the natal dispersal scale (6 km), the top model also featured maritime forests as best cover type accounting for variation in the data (AIC $_{wt} = 0.98$). The influence of maritime forest on occupancy was strong ($Beta = 0.34 \pm 0.07$). The functional relationship between percent maritime forest (natal dispersal scale) and occupancy is depicted in Figure 3. Estimates of occupancy by natal dispersal unit varied geographically (Figure 4). Thus, our final estimate of occupancy was derived at the area of conservation interest scale (AOI), but included a covariate (regions) to account for the geographic aggregation of natal dispersal

units (Occupancy = AOI-Maritime+Regions; Figure 4). The influence of maritime forest remained positive within these regions ($Beta = 0.17 \pm 0.12$).

Projections indicated that maritime forest will be lost in most of the areas of interest considered in this study (Figure 5 top panel). Losses varied by time step (2020 or 2050), but we focus on 2050 to simplify presentation and interpretation of results. In 2050, maritime forest losses varied between 5 and 27%. Only the Bald Head Island/Wilmington area of interest (AOI-2) exhibited low rates of habitat reduction. Average loss of habitat is summarized per HUC within AOIs on a per hectare scale (Table 2), and totalized per AOI (Figure 5 bottom panel). Highest losses were projected for the Sunset Beach area of interest (AOI-1; Figure 5).

The projected response of maritime forest loss on occupancy probability was negative, except for Bald Head Island/Wilmington area (AOI-2, Figure 6). The probability that a projected loss would occur was highest in the vicinity of Sunset Beach AOI (1), Camp Lejeune AOI (4) and Croatan National Forest AOI (5; Figure 7 top panel). The persistence probability was highest and least variable for the Bald Head Island/Wilmington (0.98; AOI-2) and Sunset Beach areas (0.93; AOI-1; Figure 7 bottom panel). Conversely, it was lowest and most variable for the Camp Lejeune (0.82; AOI-4) and Top Sail/Hammocks Beach State Park (0.84; AOI 3). Threat characterization of sea level rise, reflecting occurrence and severity, suggested that AOI-4 (Camp Lejeune) and AOI-5 (Croatan National Forest) would be at highest risk of losing land cover (all types) to sea level rise. Of the remaining AOIs, AOI-1 (Sun Set Beach) and AOI-2 (Bald Head Island/Wilmington) had the lowest rating (Figure 8).

DISCUSSION

We combined occupancy and habitat threat-risk assessments to prioritize (rank) areas of conservation interest for Painted Buntings based on persistence probability. Prioritization revolved around maritime forests, the habitat type that accounted for most of the variation in occupancy regardless of the scale considered for analysis. The support in the data for this relationship was nearly unequivocal ($AIC_{wt} > 0.90$) despite considering 11 other types of land cover (habitats) in competing models. Our finding does not mean that Painted Buntings will not use other habitat types, but confirmed that maritime forests are optimal habitat as suggested by Hamel et al. (1982) and Hamel (1992). Further support for the importance of maritime forest comes from harboring the highest densities of Painted Buntings in southeastern United States (R. Mordecai, USFWS, unpublished survey data).

Maritime forests have declined 36% in the last 23 years in North Carolina (Jones et al. 2013). Thus, prioritization criteria are valuable to inform decisions that might help curb losses of this type of habitat, valuable on its own merits, but also because it has implications for other species such as the Painted Bunting. Projected losses of maritime forest in 2020 and 2050 were highest at Sunset Beach AOI, followed by Camp Lejeune and the Croatan National Forest AOIs, all $\geq 20\%$. Not surprisingly, decreases in projected occupancy varied accordingly, with the highest drop ($\sim 15\%$) for Sunset Beach AOI. Conversely, the lowest loss of maritime forest was projected for the Bald Head island/Wilmington AOI. Projected losses to 2050 and variation in severity yielded highest uncertainty in persistence for Camp Lejeune AOI. In contrast, projected losses for the

Bald Head island/Wilmington AOI were the lowest, yielding the highest persistence probability projected by 2050. Indeed, occupancy values based on present (2008-09) and projected persistence (2050) were very close.

A straightforward prioritization of areas of interest would be to rank AOIs from highest to lowest persistence probability. However, habitat conservation scenarios will invariably be more complex, and we would like to illustrate ways in which various types of data used in our assessment informs the prioritization process. Let's suppose that the area-selection challenge requires selecting two areas, and let's assume further that Topsail/Hammocks Beach State Park and Sun Set Beach are the focal areas. Arguably, the persistence of Topsail/Hammocks Beach State Park AOI would benefit from also prioritizing the Camp Lejeune AOI, its nearest neighbor. Similarly, we could argue that Sun Set Beach AOI would benefit from being paired up with Bald Head Island/Wilmington AOI. The rationale is that pairings would facilitate interchanges between populations and minimize risks of extinction (Hanski 1998), a possibility inferred from immigration rates at Bald Head Island (Chapter 2). Our assessment suggests that focusing on Topsail/Hammocks Beach State Park AOI would carry a higher risk of failure (i.e., AOI extinction) for several reasons. First, occupancy levels at present (2008-09) were relatively low (<0.4), and although projected habitat losses were low relative to other AOIs, such losses would only undermine persistence. Second, the Camp Lejeune AOI is projected to undergo the most severe habitat losses of any AOI, strongly undercutting its value as a neighboring area due to its low and uncertain persistence. Admittedly, the Sunset Beach AOI will also undergo a substantial amount of projected

habitat loss, but its baseline (2008-09) occupancy level was much higher (0.9). An option for decision makers is to implement conservation actions to pro-actively curb loss of maritime forest in either set of AOIs; however as noted above, the trade-off in terms of costs and benefits would likely favor Sunset Beach AOI as it is more likely that it would yield higher persistence probability. Benefits are strengthened by the Bald Head Island/Wilmington AOI, the AOI that had the highest baseline occupancy and persistence, in part due to the low probability of projected habitat loss by 2050.

In the context of the aforementioned hypothetical scenarios, exploring other data sets, even in the absence of threat-response curves, might prove informative. For example, by 2050 sea level rise projections places Camp Lejeune AOIs at high risk of losing terrestrial land cover. Sunset Beach and Bald Head Island/Wilmington AOIs were at a much lower risk. Additional information could relax the reliance of area-selection decisions on one habitat, adding robustness to the area-selection process. This option is useful in situations where the threat-response curve is fraught with uncertainty or modeling persistence should involve a combination of multiple habitats simultaneously. In the case of Painted Buntings, for example, it is possible to prioritize conservation areas by jointly evaluating the effects of threats on maritime forest (breeding habitat) and salt marshes (foraging habitat).

We underscore that our sampling unit was defined on the basis of estimated natal dispersal area, and our final expression of occupancy grouped data by regions, a proxy of contagion among units. Thus, our inferences reflect factors that influence the state (occupied or not) of areas of interest over time (Araújo et al. 2002). Stronger inferences

about persistence for any taxonomic group, as Araujo and Williams (2000) suggested, would be obtained using dynamic analytical frameworks. This capability is already available for occupancy in the form of multi-season, single or multiple species models (MacKenzie et al. 2006, Dorazio et al. 2010, Pacifici et al. 2014), with a number of extensions to broaden their use (e.g., correlated detections, Hines et al. 2014). Adopting a dynamic framework relaxes constraints imposed by niche-based models (Araujo and Williams 2000), focusing instead on the rate parameters that influence occupancy (colonization, extinction, Yackulic et al. 2015). Johnston et al. (2015) suggested that the value of areas of interest might be misrepresented because occupancy might not always be a good reflection of abundance. However, we point at multi-season, multi-state occupancy models as a means to obtain reliable proxies of abundance as well as estimates of occupancy (Nichols et al. 2007). We also stress that the utility of projected persistence estimates will be intimately related to measures of accuracy and uncertainty in land cover changes. These estimates are seldom reported but may have non-trivial consequences when estimating occupancy, colonization and extinction rates in landscapes characterized by transient dynamics (Veran et al. 2012).

LITERATURE CITED

- Araújo, M. B. and P. H. Williams. 2000. Selecting areas for species persistence using occurrence data. *Biological Conservation* 96:331-345.
- Araújo, M. B., P. H. Williams, and R. J. Fuller. 2002. Dynamics of extinction and the selection of nature reserves. *Proceedings of the Royal Society of London Series B-Biological Sciences* 269:1971–1980.
- Arponen, A. 2012. Prioritizing species for conservation planning. *Biodiversity and Conservation* 21:875–893.
- Battin, J. 2004. When good animals love bad habitats: Ecological traps and the conservation of animal populations. *Conservation Biology* 18:1482–1491.
- Clough, J.S. 2008. SLAMM 5.0.1. Technical documentation and executable program downloadable from <http://www.warrenpinnacle.com/prof/SLAMM/index.html>.
- Conroy, M. J. and J. T. Peterson. 2013. Decision making in natural resource management: a structured, adaptive approach. Wiley-Blackwell, John Wiley and Sons, Ltd., UK. 456 pp.
- Costanza, J. K., R. C. Abt, A. J. McKerrow and J. A. Collazo. 2015a. Linking state-and-transition simulation and timber supply models for forest biomass production scenarios. *AIMS Environmental Science* 2(2):180-202. DOI: 10.3934/environsci.2015.2.180
- Costanza, J. K., A. J. Terando, A. J. McKerrow, J. A. Collazo. 2015b. Modeling climate change, urbanization, and fire effects on *Pinus palustris* ecosystems of the southeastern U.S. *Journal of Environmental Management* 151:186-199.
- Delany, M. F. 2011. Eastern Painted Bunting Monitoring. Florida Wildlife Resources Institute. Final Report (NG07-101), Gainesville, FL. 42 pp.
- Delany, M. F., B. Pranty, and R. A. Kiltie. 2013. Painted Bunting Abundance and Habitat Use in Florida. *Southeastern Naturalist* 12:61–72.
- DeWan, A.A., P. J. Sullivan, A. J. Lembo, C. R. Smith, J. C. Maerz, J. P. Lassoie, and M. E. Richmond. 2009. Using occupancy models of forest breeding birds to prioritize conservation planning. *Biological Conservation* 142: 982-991.
- Dorazio, R.M., M. Kéry, J.A. Royle, and M. Plattner. 2010. Models for inference in dynamic metacommunity systems. *Ecology* 91: 2466-2475.

- Earnhardt, T. S. 2012. Vegetative Dynamic Modeling for the SAMBI Designing Sustainable Landscapes Project. <http://www.basic.ncsu.edu/dsl/downloads/DSL-SAMBI_Landscape_Change_Modeling.pdf> Accessed February 10, 2014.
- Environmental Protection Agency. 1998. Guidelines for Ecological Risk Assessment. Federal Register 63(93):26846-26924)
- Faith, D. P., and P. A. Walker. 1996. Integrating conservation and development: incorporating vulnerability into biodiversity-assessment of areas. *Biodiversity and Conservation* 5:417–429.
- Fisk, E. J. 1974. Wintering populations of Painted Buntings in southern Florida. *Bird-Banding* 45:353-359.
- Frankel, O. H., and M. E. Soulé. 1981. *Conservation and Evolution*. Cambridge University Press, New York, NY.
- Griffith, G. E., J. M. Omernik, J. A. Comstock, M. P. Schafale, W. H. McNab, D. R. Lenat, T. F. MacPherson, J. B. Glover, and V. Shelburne. 2002. Ecoregions of North Carolina and South Carolina, (color poster with map, descriptive text, summary tables, and photographs): Reston, Virginia.
- Hamel, P. B., H. E. LeGrand Jr., M. R. Lennartz and S.A. Gauthreaux. 1982. Bird habitat relationships on southeastern forest lands. Asheville, NC: U.S. For. Serv.
- Hamel, P. B. 1992. Land manager's guide to the birds of the south. The nature Conservancy, Chapel Hill, NC and U.S. Forest Service, Southern Region, Atlanta, GA.
- Hanski, I. 1998. Metapopulation dynamics. *Nature* 396: 41-49.
- Herr, C. A., P. W. Sykes, and J. Klicka. 2011. Phylogeography of a vanishing North American songbird: The Painted Bunting (*Passerina ciris*). *Conservation Genetics* 12:1395–1410.
- Hines, J. E., J. D. Nichols, and J. A. Collazo. 2014. Multiseason occupancy models for correlated replicate surveys. *Methods in Ecology and Evolution* 5:583–591.
- Johnson, D. H. 1980. The comparison of usage and availability measurements for evaluating resource preference. *Ecology* 61: 65–71.
- Jones, G., A. Snider, and S. Luo. 2013. Changes in the Extent of North Carolina Barrier Island Maritime Forests 1988-2011: An Assessment of Past Efforts at Protection. *Journal of Forestry* 111:186–193.

- Kareiva, P. M., and M. Marvier. 2015. *Conservation Science: Balancing the Needs of People and Nature*. 2nd edition. Roberts and Company Publishers, Inc., Greenwood Village, CO.
- Lawler, J. J., D. White, and L. L. Master. 2003. Integrating Representation and Vulnerability : Two Approaches for Prioritizing Areas for Conservation. *Ecological Applications* 13:1762–1772.
- MacKenzie, D. I., J. D. Nichols, G. B. Lachman, S. Droege, A. Royle, and C. A. Langtimm. 2002. Estimating site occupancy rates when detection probabilities are less than one. *Ecology* 83: 2248-2255.
- MacKenzie, D. I., J. D. Nichols, J. A. Royle, K. H. Pollock, L. L. Bailey, and J. E. Hines. 2006. *Occupancy estimation and modeling: inferring patterns and dynamics of species occurrence*. Academic Press. 324 pp.
- Margules, C.R, A.O Nicholls, and R.L Pressey. 1988. Selecting networks of reserves to maximize biological diversity. *Biological Conservation*. 43, 63-76.
- Moilanen, A., A. M. A. Franco, R. I. Early, R. Fox, B. Wintle, and C. D. Thomas. 2005. Prioritizing multiple-use landscapes for conservation: methods for large multi-species planning problems. *Proceedings of the Royal Society B-Biological Sciences* 272:1885–1891.
- Moilanen, A. K., A. Wilson, and H. Possingham. 2009. *Spatial conservation prioritization: quantitative methods and computational tools*. Oxford Press University, Oxford, UK. 304 pp.
- North Carolina Wildlife Resources Commission. 2015. *North Carolina Wildlife Action Plan (SWAP)*. Raleigh, NC.
- Pacifici, K., E. F. Zipkin, J. A. Collazo, J. I. Irizarry, and A. DeWan. 2014. Guidelines for a priori grouping of species in hierarchical community models. *Ecology and Evolution* 4: 877–888.
- Park, R. A., T. V. Armentano, and C. L. Cloonan. 1986. Predicting the Effects of Sea Level Rise on Coastal Wetlands. Pages 129-152 in J.G. Titus, ed. *Effects of Changes in Stratospheric Ozone and Global Climate, Vol. 4: Sea Level Rise*. US Environmental Protection Agency, Washington, DC.
- Pimm, S. L., and J. H. Lawton. 1998. Planning for Biodiversity. *Science* 279:2068–2069.
- Radeloff, V. C., J. W. Williams, B. L. Bateman, K. D. Burke, S. K. Carter, E. S. Childress, K. J. Cromwell, C. Gratton, A. O. Hasley, B. M. Kraemer, A. W. Latzka, E. Marin-Spiotta, C. D. Meine, S. E. Munoz, T. M. Neeson, A. M.

- Pidgeon, A. R. Rissman, R. J. Rivera, L. M. Szymanski, and J. Usinowicz. 2015. The Rise of Novelty in Ecosystems. *Ecological Applications* 25:2051–2068.
- Ralph, C. J., G. R. Geupel, P. Pyle, T. E. Martin, and D. F. DeSante. 1993. Handbook of field methods for monitoring landbirds. Gen. Tech. Rep. PSW-GTR-144. Albany, CA: Pacific Southwest Research Station, Forest Service, U.S Dept. Agricul.; 41 pp.
- Robbins, C. S.; D.K. Dawson; and B.A. Dowell. 1989. Habitat Area Requirements of Breeding Forest Birds of the Middle Atlantic States. The Wildlife Society, Wildlife Monographs no. 103.
- Rodrigues, A. S. L., R. D. Gregory, and K. J. Gaston. 2000. Robustness of reserve selection procedures under temporal species turnover. *Proceedings of the Royal Society B-Biological Sciences* 267:49–55.
- Rubino, M. J. 2009. Sea Level Rise Modeling for the SAMBI Designing Sustainable Landscapes Project. Technical documentation and data downloadable from <http://www.basic.ncsu.edu/dsl/slr.html>. Accessed May 12, 2014.
- Sauer, J. R., J. E. Hines, J. E. Fallon, K. L. Pardieck, D. . Ziolkowski Jr, and W. A. Link. 2014. The North American Breeding Bird Survey, Results and Analysis 1966-2013. Laurel, MD.
- SETAC, Society of Environmental Toxicology and Chemistry. 1997. Technical issue paper: Ecological risk assessment. Pensacola, FL, USA: SETAC. 4 p.
- Sutherland, G. D., A. S. Harestand, K. Price, and K. P. Lertzman. 2000. Scaling of natal dispersal distances in terrestrial birds and mammals. *Conservation Ecology* 4: 44.
- Sykes Jr., P. W., and S. Holzman. 2005. Current range of the eastern population of Painted Bunting (*Passerina ciris*). *North American Birds* 59:4–17.
- United States Geological Survey National Gap Analysis Program (USGS). 2008. Provisional Southeast GAP Regional Land Cover 2001. Biodiversity and Spatial Information Center, Cooperative Fish and Wildlife Research Unit, NC State University (<http://www.basic.ncsu.edu/segap/> last accessed 2016).
- Uusitalo, L., A. Lehtikoinen, I. Hello, and K. Myrberg. 2015. An overview of methods to evaluate uncertainty of deterministic models in decision support. *Environmental Modelling and Software*. 63:24-31.
- Veran, S., K. J. Kleiner, R. Choquet, J. A. Collazo, and J. D. Nichols. 2012. Modeling habitat dynamics accounting for possible misclassification. *Landscape Ecology* 27:943-956.

- Vitousek, P. M., H. A. Mooney, J. Lubchenco, and J. M. Melilo. 1997. Human domination of earth's ecosystems. *Science* 277:494–499.
- Watling J.I., A.J. Nowakowski, M.A. Donnelly and J.L. Orrock. 2011. Meta-analysis reveals the importance of matrix composition for animals in fragmented habitat. *Global Ecology and Biogeography* 20: 209-217.
- Williams, S. G., J. A. Collazo, and A. J. McKerrow. 2015. Statewide Terrestrial Habitat Threats and Risks Projections: A Decision Support Tool for Strategic Wildlife Habitat Conservation in North Carolina. Final Report to the North Carolina Wildlife Resources Commission. 34 pp.
- Yackulic, C.B., J.D. Nichols, J. Reid, and, R. Der. 2015. To predict the niche, model colonization and extinction. *Ecology* 96: 16–23.

TABLES AND FIGURES

Table 1. Model Selection tables for assessment of Painted Bunting occupancy as a function of 12 land cover classes found in coastal North Carolina. Occupancy was estimated at the 200 m (survey station; top table) and 6 km (natal dispersal unit; bottom table) based on surveys of 133 stations collected in 2008-2009. Effort means that survey stations were sampled 6 times, or just 3 time in one of two years.

Model	AIC	deltaAIC	AIC wgt	no.Par.	-2*LogLike
psi(Maritime),p(Effort)	601.96	0.00	0.92	4.00	593.96
psi(Shrub/Scrub),p(Effort)	607.07	5.11	0.07	4.00	599.07
psi(Oak-Hickory),p(Effort)	611.86	9.90	0.01	4.00	603.86
psi(.),p(Effort)	614.67	12.71	0.00	3.00	608.67
psi(Sandhills LongLeaf),p(Effort)	615.17	13.21	0.00	4.00	607.17
psi(Crop-Pasture),p(Effort)	615.31	13.35	0.00	4.00	607.31
psi(LowIntensityDeveloped),p(Effort)	616.32	14.36	0.00	4.00	608.32
psi(Bay-Swamp-Pocosin),p(Effort)	616.90	14.94	0.00	4.00	608.90
psi(.),p(.)	617.28	15.32	0.00	2.00	613.28
psi(Managed Pine),p(Effort)	618.55	16.59	0.00	4.00	610.55
psi(Oak-Gum-Cypress),p(Effort)	619.78	17.82	0.00	4.00	611.78
psi(.),p(t)	622.05	20.09	0.00	7.00	608.05

Model	AIC	deltaAIC	AIC wgt	no.Par.	-2*LogLike
psi(Maritime),p(Effort)	589.67	0.00	0.98	4.00	581.67
psi(UrbanLowDensity),p(Effort)	597.24	7.57	0.02	5.00	587.24
psi(Oak-Hickory),p(Effort)	605.58	15.91	0.00	4.00	597.58
psi(Managed Pine),p(Effort)	612.63	22.96	0.00	4.00	604.63
psi(Shrub/Scrub),p(Effort)	613.81	24.14	0.00	4.00	605.81
psi(Sandhills LongLeaf),p(Effort)	615.21	25.54	0.00	4.00	607.21
psi(.),p(Effort)	615.64	25.97	0.00	5.00	605.64
psi(BaySwampPocosin),p(Effort)	615.90	26.23	0.00	4.00	607.90
psi(.),p(.)	617.28	27.61	0.00	2.00	613.28
psi(Pasture/Crop),p(Effort)	617.93	28.26	0.00	4.00	609.93
psi(.),p(Effort)	618.10	28.43	0.00	3.00	612.10
psi(UrbanHighDensity),p(Effort)	618.12	28.45	0.00	4.00	610.12
psi(Salt Marsh),p(Effort)	619.65	29.98	0.00	4.00	611.65
psi(Oak-Gum-Cypress),p(Effort)	619.78	30.11	0.00	4.00	611.78
psi(.),p(t)	622.05	32.38	0.00	7.00	608.05

Table 2. Summary statistics (mean \pm SE) for percent loss of maritime forest and severity per 12 digit Hydrologic Unit (HUC) in 5 areas of conservation interest (AOI) for Painted Buntings in southeastern North Carolina.

AOI	HUC/AOI	Mean Sev	Std Err	Mean(%chg)	Std Err
SB	11	0.34	0.12	2.41	0.88
W/BHI	6	0.25	0.12	0.26	1.45
TS/HBSP	9	0.23	0.07	0.84	0.41
LEJ	10	0.33	0.11	2.17	0.94
CNF	9	0.35	0.11	2.22	0.98

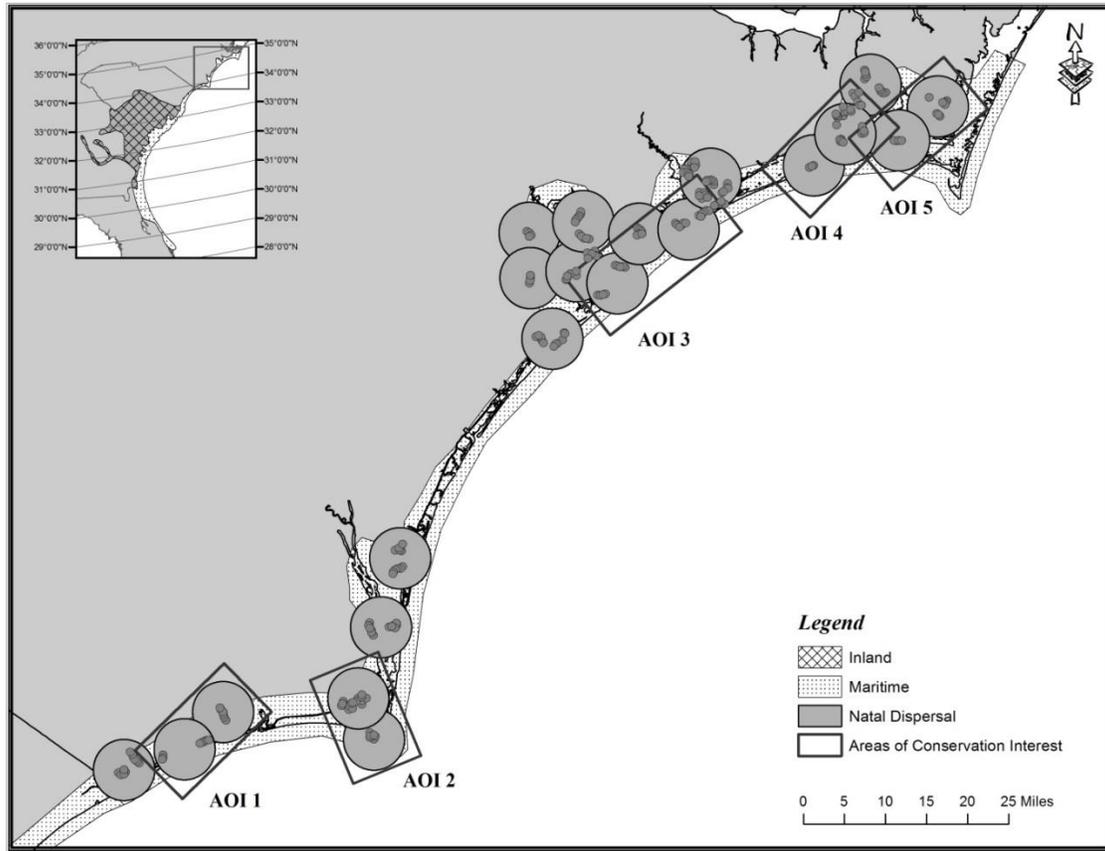


Figure 1. Map of the range of eastern Painted Buntings (inset), highlighting the range of the species in southeastern North Carolina. The map depicts surveys station locations (dots) within natal dispersal units (6 km, black circles), and areas of conservation interest (squares). Areas of conservation interest (AOI) were labeled based on distinctive communities or federal lands. AOI-1 = Sunset Beach, AOI-2 = Bald Head Island/Wilmington, AOI-3 = Topsail/Hammocks Beach State Park, AOI-4 = Camp Lejeune, and AOI-5 = Croatan National Forest.

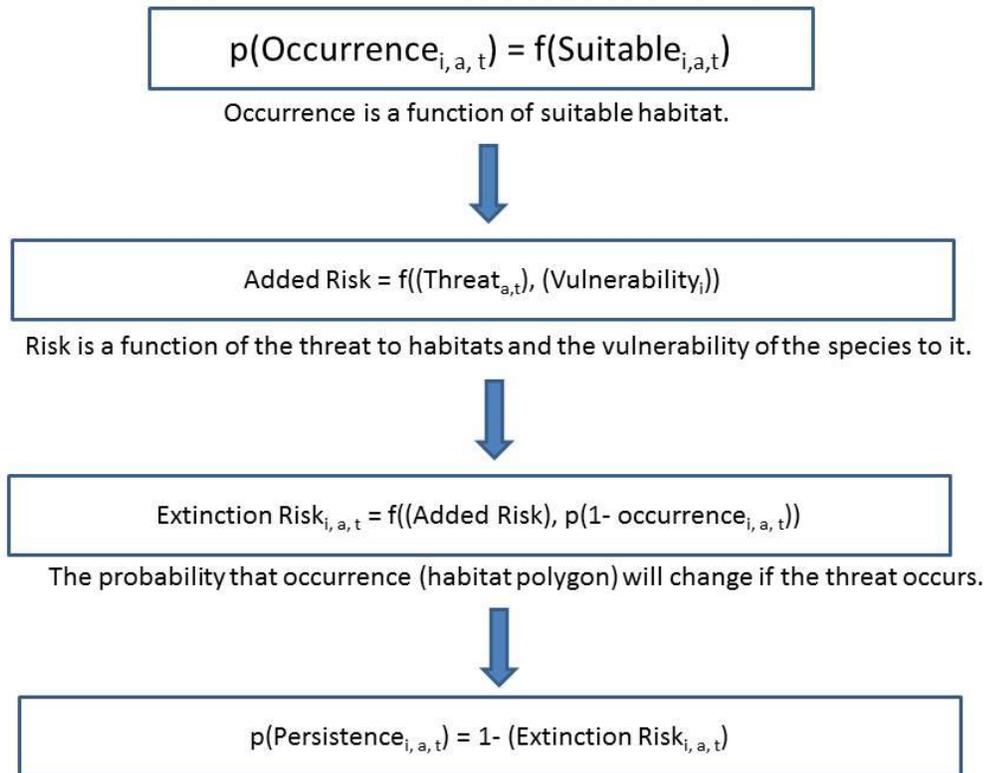


Figure 2. Stepwise process to estimate persistence probability using approach outlined by Araújo and Williams (2000). The objective of each step is described under each rectangle. Subscripts are: i=species, a=area of interest, t=time.

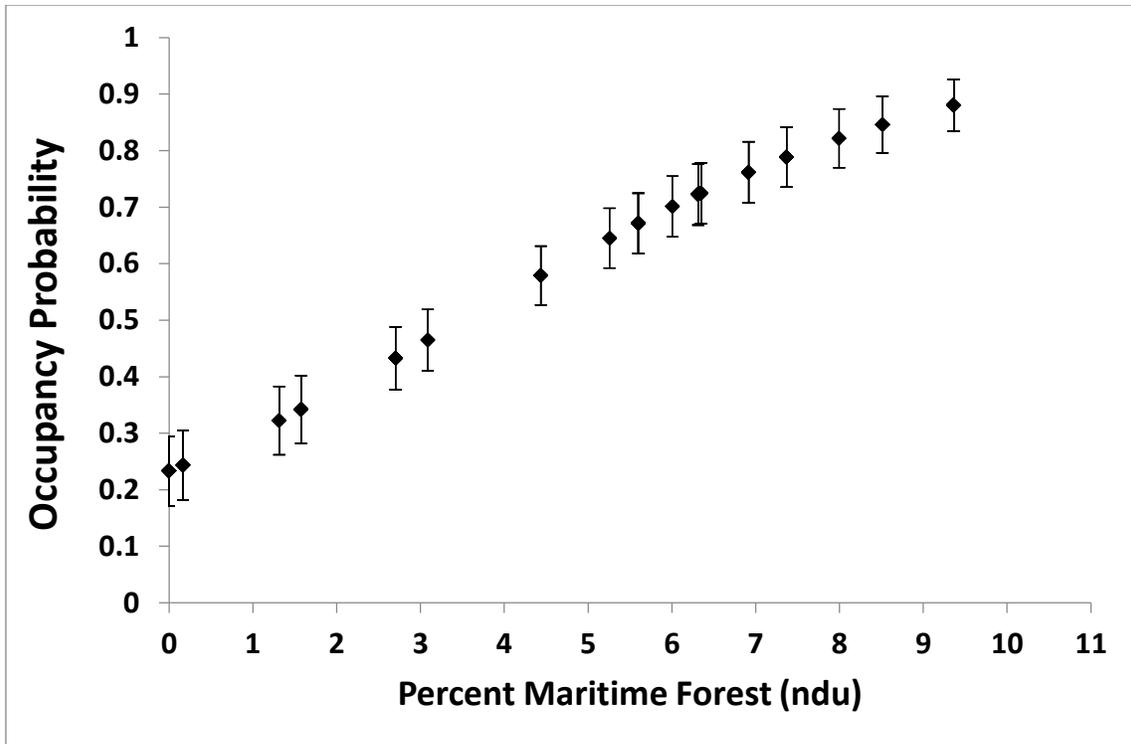


Figure 3. Occupancy probability (\pm SE) of Painted Buntings as a function of percent maritime forest in natal dispersal units (6 km radius) in southeastern North Carolina. The functional relationship was developed using survey data collected at 133 stations in 2008-2009.

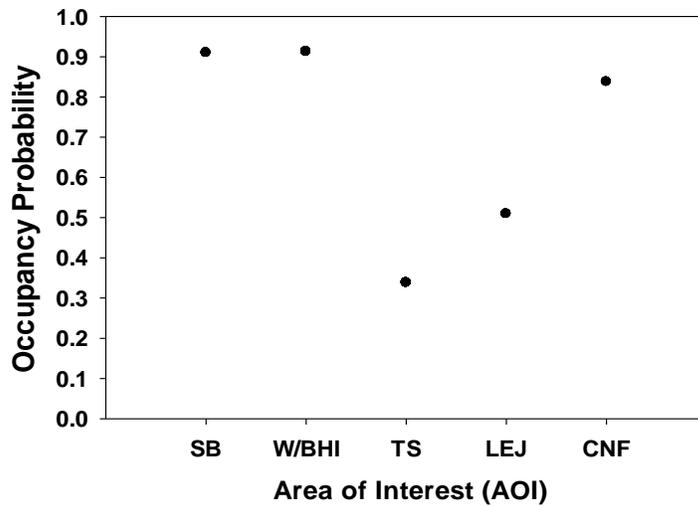
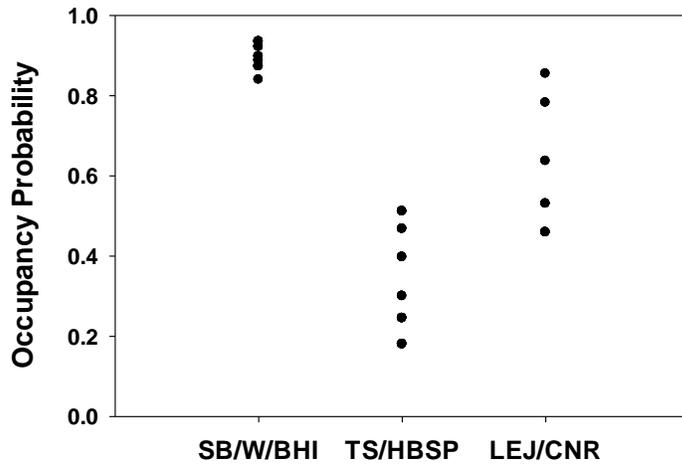


Figure 4. Occupancy probability of Painted Buntings by natal dispersal unit (6 km; top panel) by regions and area of conservation interest (AOI; bottom panel) in southeastern coastal North Carolina. Units were grouped using AOI labels. SB/W/BHI= Sunset Beach/Bald Head Island/Wilmington, TS/HBPS = Topsail/Hammocks Beach State Park, LEJ/CNF = Camp Lejeune, and Croatan National Forest.

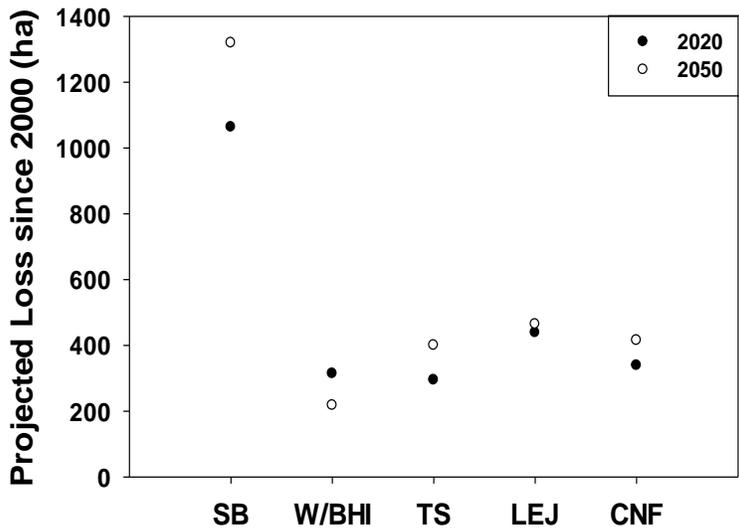
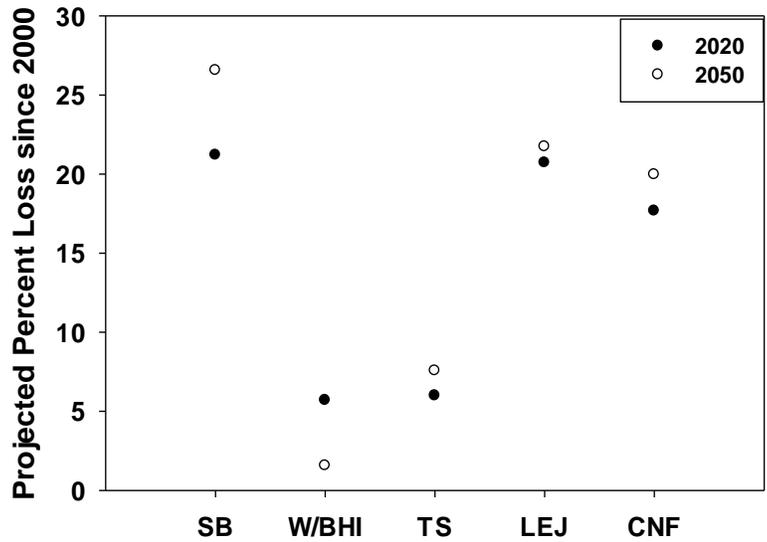


Figure 5. Projected percent loss of maritime forest in 2020 and 2050 relative to the baseline levels in 2000 (top panel). Percent loss is expressed on a per hectare basis (bottom panel). SB = Sunset Beach, W/BHI= Bald Head Island/Wilmington, TS = Topsail/Hammocks Beach State Park, LEJ = Camp Lejeune, and CNF = Croatan National Forest.

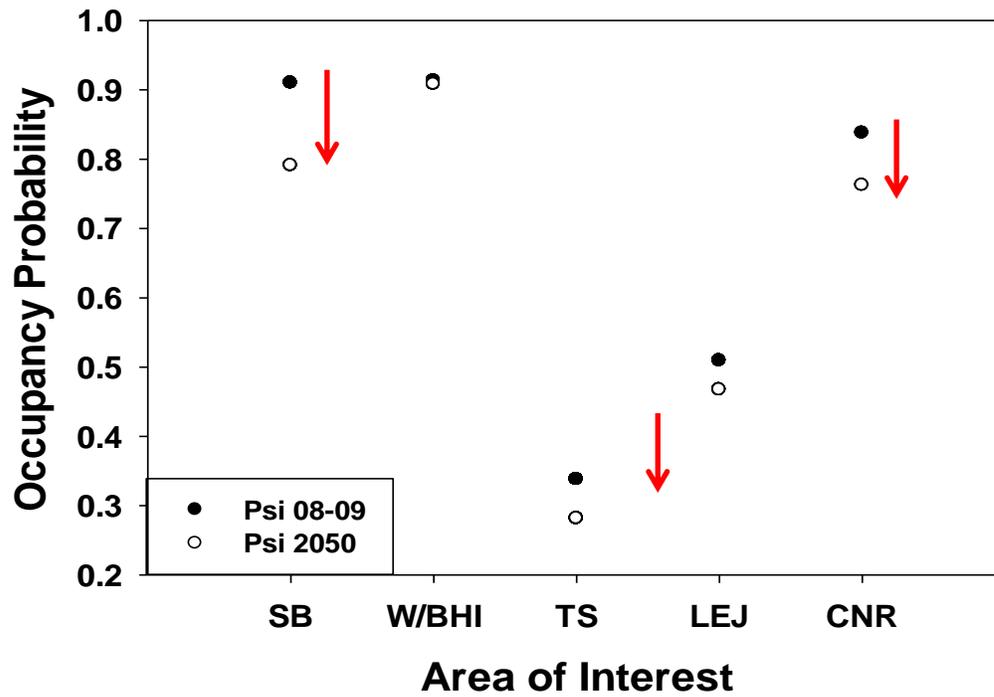


Figure 6. Projected change in occupancy probability of Painted Buntings from 2008-09 to 2050 in southeastern coastal North Carolina. Estimates of occupancy assume the species-habitat relationship developed for Painted Buntings using 2008-2009 survey data holds through 2050.

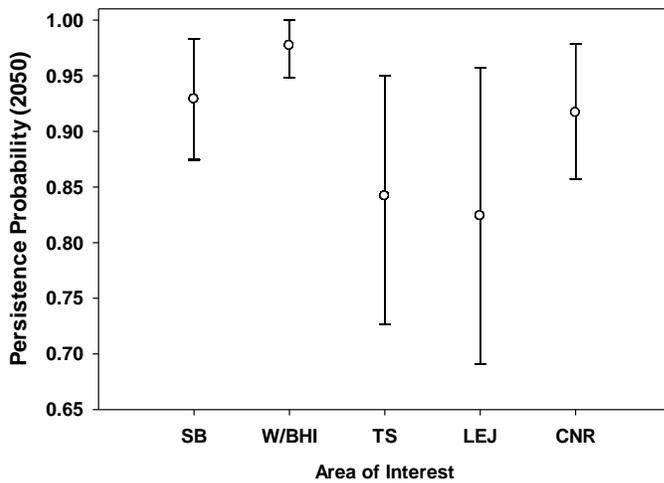
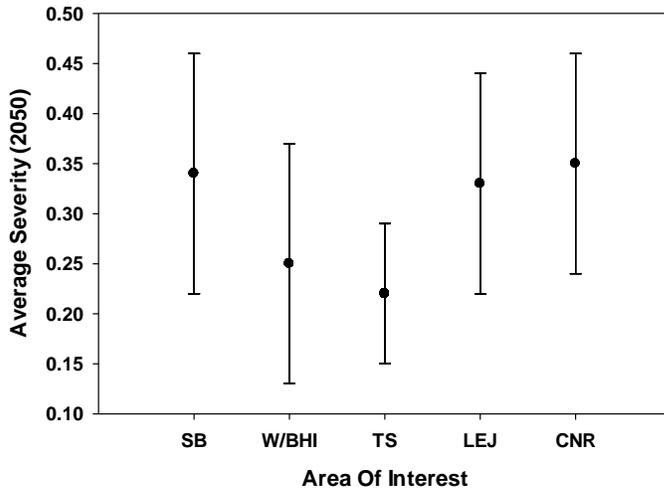


Figure 7. Average severity (\pm SE) that loss of maritime forest of a specified magnitude occurs in 2050 at five areas of conservation interest for Painted Bunting in southeastern North Carolina (top panel). Probability (95% CIs) that an area of conservation interest (AOI) will remain occupied (persist) in 2050. Magnitude of loss of habitat is depicted in figure 5. SB = Sunset Beach, W/BHI= Bald Head Island/Wilmington, TS = Topsail/Hammocks Beach State Park, LEJ = Camp Lejeune, and CNF = Croatan National Forest.

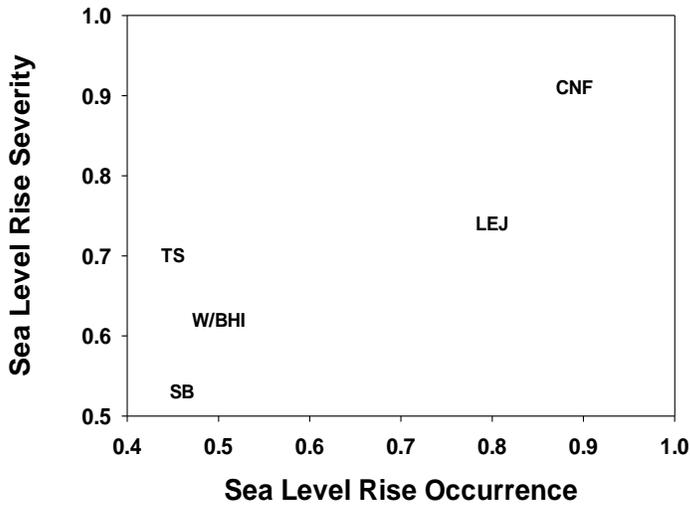


Figure 8. Risk rating matrix of projected changes in 2050 due to sea level rise in 5 areas of conservation interest for Painted Buntings in southeastern North Carolina. Rating is based on occurrence (e.g., proportion of HUCs within each AOI where terrestrial land cover changes were detected) and severity (e.g., probability of change equal or less than a specified value (x) of terrestrial land cover change). Terrestrial land cover change caused by sea level rise is the loss of all land cover types (including marshes) to open water since 2000. SB = Sunset Beach, W/BHI= Bald Head Island/Wilmington, TS = Topsail/Hammocks Beach State Park, LEJ = Camp Lejeune, and CNF = Croatan National Forest.

APPENDICES

APPENDIX 1

Description of data layers used in the Habitat Threat and Risk Assessment Tool

(Williams et al. 2015). The ecological basis for their inclusion (e.g., potential effects) is detailed below for each threat.

Habitat loss (2010-2050) – defined as the conversion of 5 broad land cover categories into urban or impervious surfaces from 2000 to 2015 on a decadal time step (i.e., 2010, 2020, 2030, 2040 and 2050). These land cover categories were upland forest, wet forest, open, wet herbaceous and scrub-shrub. We do not include metrics of habitat fragmentation because species respond to scales of fragmentation differently and the effect of fragmentation on many species is poorly understood. However, the loss of habitat is almost universally associated with a decline of species, and persistence. The threat scaled score was based on the amount (hectares) of land cover category lost over time within each sub-watershed.

Urban growth (2010-2050) – projections of urban growth throughout the southeastern United States were made using the SLEUTH (Slope, Land use, Excluded, Urban, Transportation and Hillshade) modeling protocol. SLEUTH uses a flexible cellular automata urban-growth model based on past urban growth and transportation networks. Restrictions to the probability of urbanization are implemented by natural and social land use controls such as topology or regulatory restrictions on sensitive environmental areas.

The model produces a probability surface of urbanization. A 50% probability threshold was used to delineate urban areas, which is consistent with previous applications of SLEUTH models. The threat scaled score was based on the amount of urban development (hectares) within each sub-watershed.

Fire suppression (2010-2050) - Fire suppression is a threat to wildlife habitat in that it removes one of the key mechanisms that regulate fire depended systems which are so prevalent in North Carolina, especially in the Piedmont and coastal plain. An index to fire suppression was based on the density of urban land cover within a 8 km radius. It follows from the general practice that that more heavily populated areas have a greater propensity to suppress wildfires. The threat scaled score was based on the urban density (hectares) within each sub-watershed.

Transportation corridors (Divided center line highways 2010-2050) - Major transportation corridors disrupt wildlife habitat by creating barriers to movement, fragmentation and habitat loss when initially constructed. We used North Carolina Department of Transportation data to identify existing divided center line (DCL) highways as well as proposed DCL highways (NC-DOT, 2014). The threat scaled score was based on the total miles of DCL roadways within each sub-watershed, then divided by the size of the sub-watershed.

Nutrient loading (Manure application) - The application of manure from confined animal feeding operations is an important source of organic fertilizer which can increase crop production. However, misapplication can increase eutrophication of freshwater and coastal marine systems which negatively impact aquatic systems. The EPA EnviroAtlas project compiled the mean rate of manure application for each sub-watershed based on county-scale estimates of livestock manure production. The threat scaled score was based on the application rate for each sub-watershed.

Nutrient loading (Synthetic Nitrogen Fertilizer) - Synthetic nitrogen fertilizer is the largest source of reactive nitrogen in the environment for the United States and is a critical component for maintaining high levels of food production. However, roughly half of the fertilizer is not utilized by the intended crop and ends up entering the surrounding landscape leading to increased eutrophication of water resources. The EPA EnviroAtlas project compiled the mean rate of synthetic fertilizer application for each sub-watershed based on county-level data describing total farm-level inputs. The threat scaled score was based on the application rate for each sub-watershed.

Annual Atmospheric Deposition (Total nitrogen deposition) - Atmospheric deposition of nitrogen can negatively impact both terrestrial and aquatic ecosystems. It is the primary source of acidifying chemicals lowering soil and water pH to the detriment of both flora and fauna. Atmospheric deposition of nitrogen also contributes to excess nutrients which can lead to eutrophication, lower dissolved oxygen, fish kills and decreased productivity.

The EPA EnviroAtlas project compiled annual total nitrogen deposition for 2003 based on data from the Community Multiscale Air Quality modeling system and includes wet and dry oxidized and reduced nitrogen. The threat scaled score was based on the average annual kilograms per hectare for each sub-watershed.

Annual Atmospheric Deposition (Total sulfur deposition) - Atmospheric deposition of sulfur can negate the inherent buffering capacity of soils leading to increased acidification of both soil and water with negative consequences for both flora and fauna. Increased sulfur deposition can also lead to increases in methane gas and methyl mercury. The EPA EnviroAtlas project compiled annual total sulfur deposition for 2006 based on data from the Community Multiscale Air Quality modeling system and includes wet and dry oxidized and reduced nitrogen. The threat scaled score was based on the average annual kilograms per hectare for each sub-watershed.

Energy development (Triassic basin) - The USGS has identified several Triassic basins in North Carolina as having moderate to high potential for hydrocarbon source rocks which could be accessed with drilling operations known as hydraulic fracturing. Identified threats to biota include: surface and groundwater contamination; diminished stream flow; stream siltation; habitat loss and fragmentation; localized air, noise and light pollution; and contribution to climate change. North Carolina Geological Survey data was used to quantify the areal extent of the Deep River and Dan River-Danville basins within each

sub-watershed. The threat scaled score was based on the percentage of Triassic basin geology (hectares) within each sub-watershed.

Energy development (Wind power) - Potential wind resource (50 m height) is represented by data from the National Renewable Energy Laboratory. This threat is included because it could lead to wildlife mortality. Wind power potential is classified in 8 categories (0 – Unassessed, 1 – Poor, 2 – Marginal, 3 – Fair, 4 – Good, 5 – Excellent, 6 – Outstanding, 7 – Superb). The threat scaled score was based on the average wind potential category of each sub-watershed.

Forest health (Forest insect/disease risk) - The North Carolina Forest Service developed a Forest Health Priority map as part of their Forest Action Plan in 2010. Moderate to high risk of damage from insects and diseases (native and/or established and imminent invasive threats) were assessed and mapped for the entire state. Specific pests included; southern pine beetle (*Dendroctonus frontalis*), littleleaf disease, annosus root rot, fusiform rust (*Cronartium quercuum* f.sp. *fusiforme*), hemlock woolly adelgid (*Adelges tsugae*), balsam woolly adelgid (*Adelges piceae*), beech bark disease, redbay ambrosia beetle-laurel wilt (*Raffaelea lauricola*), emerald ash borer (*Agrilus planipennis*), Asian longhorned beetle (*Anoplophora glabripennis*), and sirex woodwasp (*Sirex noctilio*). The threat scaled score was based on the presence or absence of the “moderate to high risk” in each sub-watershed.

Sea level rise – The effect of sea level rise on coastal environments was used to project change in terrestrial land and undeveloped upland land cover. Projections were made at the Biodiversity and Spatial Information Center at the North Carolina Fish and Wildlife Cooperative Research Unit using the Sea Level Affection Marshes Model (SLAMM). Projections in habitat change were made on a decada time-step from 2000 to 2050 based on IPCC’s Fourth Assessment A1B climate scenario. SLAMM simulates transforming coastal environments by accounting for nearshore geomorphological processes such as accretion, erosion, and marsh migration dynamics due to long-term seas level rise. Terrestrial land cover change is the loss of all land cover types (including marshes) to open water since 2000. Undeveloped upland change monitors the loss of undeveloped upland habitat (forests, woodlands, open) to wetlands and open water. The threat scaled score was based on the amount of habitat loss (hectares) within each sub-watershed.

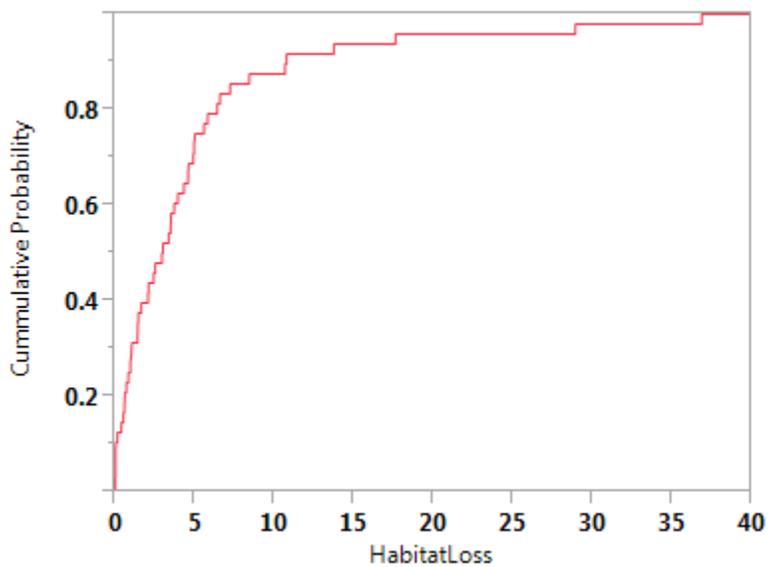
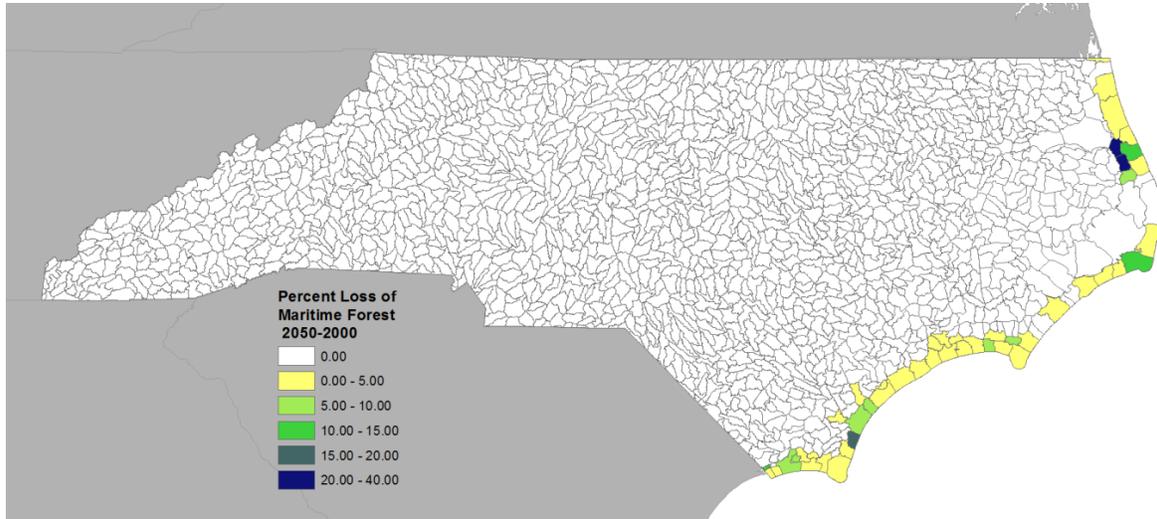
Hydrologic alteration (Number of dams) - The National Inventory of Dams is a US Army Corps of Engineers database that documents dams more than 25 feet high, more than 50 acre-feet of water, or are considered a significant hazard if they fail. The potential impacts of these structures and associated management is well documented . The EPA EnviroAtlas project summarized the number of dams within each sub-watershed. The threat scaled score was based on the number of dams present within each sub-watershed.

Impaired waters (biota and metal impairments) - The EPA monitors impaired waters and their causes as outlined by section 303(d) of the Clean Water Act. Impairments include pathogens, metals, turbidity, pH imbalance, algal growth, polychlorinated biphenyls,

nutrients, temperature, dioxin, and others. The EnviroAtlas project compiled the length of stream impacted and impairment source for sub-watersheds within the continental US. These data set represent the combined sum of all impairments. The threat scaled score was based on the total length of the stream impaired, weighted by stream density in each sub-watershed.

APPENDIX 2

Location of USGS HUCs containing maritime forest in coastal North Carolina, and projected percent loss of maritime forest by 2050 relative to levels in 2000 (top panel). Cumulative distribution function (CDF) for percent loss of maritime forest in in USGS 12 digit Hydrologic Unit (HUC) in southeastern North Carolina (bottom panel).



APPENDIX 3

Model selection table to assess influence of inter-annual (YR) difference in occupancy and detection probability of Painted Buntings in southeastern North Carolina. Surveys were conducted in 133 stations in 2008-2009.

Model	AIC	deltaAIC	AIC wgt	no.Par.	-2*LogLike
psi(.),p(YR)	677.21	0	0.33	3	671.21
psi(YR),p(.)	678.02	0.81	0.22	3	672.02
psi(YR),p(YR)	678.05	0.84	0.22	4	670.05
psi(.),p(.)	678.44	1.23	0.18	2	674.44
psi(.),p(t)	682.25	5.04	0.03	4	674.25
psi(.),p(t*YR)	684.44	7.23	0.01	7	670.44