12-14-2018

Effects of Green Tree Retention on Birds of Southern Pine Plantations

Michael Clay Parrish

Effects of green tree retention on birds of Southern pine plantations

By

Michael Clay Parrish

A Dissertation
Submitted to the Faculty of
Mississippi State University
in Partial Fulfillment of the Requirements
for the Degree of Doctor of Philosophy
in Forest Resources
in the Department of Wildlife, Fisheries, and Aquaculture

Mississippi State, Mississippi

December 2018

Effects of green tree retention on birds of Southern pine plantations

By

Michael Clay Parrish

Approved:

____
Stephen Demarais
(Major Professor)

____
Andrew W. Ezell
(Committee Member)

____
T. Bently Wigley
(Committee Member)

____
Phillip D. Jones
(Committee Member)

____
Kevin M. Hunt
(Graduate Coordinator)

____
George M. Hopper
Dean
College of Forest Resources

Name: Michael Clay Parrish

Date of Degree: December 14, 2018

Institution: Mississippi State University

Major Field: Forest Resources

Major Professor: Stephen Demarais

Title of Study: Effects of green tree retention on birds of Southern pine plantations

Pages in Study 181

Candidate for Degree of Doctor of Philosophy

In the southern United States, institutional forest owners engaged in forest certification programs often retain unharvested or less-intensively harvested vegetation when clearcut harvesting intensively managed pine (*Pinus* spp.) forests ("IMPFs"), a practice called 'green tree retention'. I investigated resultant patterns of land cover and retained structural elements in recently-harvested IMPF management units ("MUs") and related them to avian biodiversity to provide information to support harvest decisions. First, to provide forest managers baseline data on retention, I screen-digitized land cover on 1187 MUs (totaling 51646 ha) and characterized green tree retention levels and internal land cover attributes (Chapter 2). I found MU land cover was dominated by regenerating clearcuts (mean: 80.5%), streamside management zones ("SMZs"; vegetated buffers surrounding intermittent and perennial streams; 14.0%) and stringers (buffers surrounding ephemeral streams; 3.3%). Next, I surveyed 60 MUs for vegetation stem density and cover (Chapter 3). Concurrently, I surveyed avian community density and richness (Chapter 4). Vegetation and avian metrics were compared and contrasted across the dominant cover types (with emphasis on stringer/SMZ similarity) to understand impacts of retained structural elements on biodiversity outcomes. I found that snag and

log density, midstory pine density, understory deciduous cover, and ground cover were not different in stringers and SMZs; however, overstory (pine and deciduous) and midstory (deciduous) tree density was lower in stringers than in SMZs, and understory pine density was greater in SMZs. Species overlap between cover types was high (74% to 84%), but SMZs and stringers provided 27% of MU species richness. Stringers appeared to benefit both shrubland- and forest-associated birds. Finally, I sampled land cover across 4450 sq-km surrounding the 60 MUs, and performed ordination analyses to identify associations between local-scale (MU interiors) and landscape-scale (3-km buffers around MUs) land cover and avian guild diversity (Chapter 5). I found the region to be >90% forested. Cover type data explained 41% of the partial variation in avian density and total species richness. Local-scale MU characteristics appeared more important than landscape-scale characteristics in explaining avian biodiversity responses. My results suggest that retained structural features support and enhance MU biodiversity in harvested IMPFs.

DEDICATION

A lot of wonderful people have stood by me and supported me as I completed this dissertation. Dissertations are written by one person, but many special people make them possible. My wife, Lindsay, encouraged me throughout the process. She was as committed to me getting my doctorate as I was. She must have told me a hundred times that I could accomplish this. Our sweet children, Benjamin, Gemma, and Ellie, had absolutely no idea what I was working on, but they gave me more motivation than perhaps anyone else, and their beautiful smiles frequently lifted my spirits.

Without Mom's and Fred's help with the kids, I'd probably still be trying to find time to write. I especially can't thank Mom enough for moving in with us and providing the kids with such a loving presence in their life, as well as for always being understanding when I growled around the house in a Ph.D. candidate mood. Mom always built me up, and I can't thank her enough. Dad and Caryn and Jeff gave me a perspective from the other side of the degree and encouraged me frequently (and somehow made me less perplexed about the statistical side of things). Dad never once showed any doubt that I wouldn't finish, and his confidence in me was tremendous.

How I wish Stephen Maturin had been here.

E pinetis scientia avium.

From the pinelands, knowledge of birds

ACKNOWLEDGEMENTS

This document represents contributions of the Forest and Wildlife Research Center (FWRC) at Mississippi State University (MSU). This material is based upon work supported by the National Institute of Food and Agriculture, U.S. Dept. of Agriculture, McIntire-Stennis project under accession number MISZ 082090. I wish to thank: the MSU Dept. of Wildlife, Fisheries, and Aquaculture, the Dept. of Forestry, and the FWRC; the National Council for Air and Stream Improvement (NCASI); Forest Capital Partners, LLC; Plum Creek Timber Company, Inc.; Potlatch Corporation; Resource Management Service, LLC; Weyerhaeuser NR Company. For kind suggestions that helped improve this work, I thank (in alphabetical order): J. Duguay (LA Dept. of Wildlife and Fisheries, Wildlife Division; J. Fleming (MSU); J. Hepinstall-Cymerman (Univ. of Georgia); K. Hunt (MSU); D. Miller (Weyerhaeuser); R. Parrish; C. Thompson; P. Šmilauer (Univ. of South Bohemia in České Budějovice); J. Verschuyl (NCASI); and anonymous reviewers and editors at *Forest Ecology and Management* and *Forest Science.* I gratefully acknowledge the hard work of technicians J. Benson (Ch. 3), T. Campbell (Ch. 2 - 5), A. Conrad (Ch. 5), D. Dowdy (Ch. 3 - 5), G. Gover (Ch. 4 - 5), J. Luke (Ch. 3), J. Owen (Ch. 5), J. Parrish (Ch. 3), and J. Wilson (Ch. 2 - 5) - whose (mostly) cheerful suffering led to the completion of this dissertation. Finally, I thank my committee members – S. Demarais, A. Ezell, P. Jones, T. B. Wigley, and the late S. Riffell - who patiently encouraged me as I worked to complete my research.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

CHAPTER I

INTRODUCTION

The southern region of the United States [a] (henceforth: the South) is home to vast timber resources. As of 2012, 39.4% of the South's land cover was in timberland[b], accounting for 85.8% of the forestland in the region (Oswalt et al. 2014). Roughly one third of that timberland consisted of loblolly-shortleaf pine (*Pinus taeda* L.[c] - *P. echinata* Mill.) or longleaf-slash pine (*P. palustris* Mill. - *P. elliottii* Engelm.) forest (Oswalt et al. 2014). Beginning in the 1950s, global demand for forest products increased drastically (Demarais et al. 2017), and by the turn of the twenty-first century, approximately 130,000 km^2 of the South was maintained as intensively managed pine forest ("IMPF"; Fox et al. 2007). Regenerating IMPFs (<7 years post-establishment) in the South represent a major reservoir of early-successional habitat, an ephemeral habitat type that is declining in much of the eastern United States, and which is crucial to a variety of shrubland-associated species (Wilson and Watts 2000; Trani et al. 2001; Legrand et al. 2007).

Pine forests on IMPFs are frequently managed using chemical and/or mechanical site preparation, fertilization, and thinning, allowing for maximum productivity on their constituent management units ("MUs"; Yin and Sedjo 2001; Zhao et al. 2016; Demarais

---

[a] US Southern region includes: AL, AR, FL, GA, KY, LA, MI, NC, OK, SC, TN, TX, and VA.
[b] Timberland is forestland capable of producing commercial timber at a rate of 1.4 $m^3 \cdot ha^{-1} \cdot y^{-1}$ (Bechtold and Patterson 2005).
[c] Botanical taxonomic nomenclature follows Little (1979).

et al. 2017). While efficient production of forest products is a primary goal of land owners, biodiversity conservation is frequently integrated into management plans, often under the aegis of forest certification programs (Forest Stewardship Council US 2010; American Forest Foundation 2015; Sustainable Forestry Initiative 2015). Because commercial forests in the South are intensively managed, occupying a vast amount of the region, they likely exert a substantial influence over regional biodiversity, and therefore represent noteworthy opportunities for conservation efforts (Greene et al. 2016; Demarais et al. 2017).

In the southern United States, intensively managed pine forest MUs are usually harvested commercially by a process of clearcutting with green tree retention. Pine stands within the MU are clearcut, while patches of live trees and vegetation (i.e., 'green trees') are left uncut or are partially harvested, and are usually retained as vegetated buffers along perennial and intermittent streams[d]. These buffers are called streamside management zones, or 'SMZ.' Frequently, additional vegetated buffers (termed 'stringers') are also retained along ephemeral drains[e] or gullies to help further mitigate sediment entering SMZs. The primary purpose of retaining land cover is to limit sediment and solar inputs into waterways to protect water quality (Aust and Blinn 2004), but SMZs have also been shown to offer conservation advantages to terrestrial plants and wildlife (Blinn and Kilgore 2001; Rosenvald and Lõhmus 2008; Jones et al. 2010). Retention land cover may support wildlife through: (a) enhancement of vegetative

[d] Perennial streams flow continuously essentially year-round, and occupy well-defined channels. Intermittent streams have more seasonal flow patterns, and may dry up for 1 to 3 months each year, but nevertheless maintain well-defined channels.

[e] Ephemeral drains or streams generally experience flow only for hours or days after rain events. They typically have a poorly-defined channel, that is often overgrown by vegetation.

structural complexity; (b) support of biological legacies; and (c) augmented post-harvest landscape connectivity (Franklin et al. 1997; Lindenmayer and Franklin 2002; Aubry et al. 2009).

Forest products companies engaged in sustainable forest certification programs are tasked with sustaining and enhancing wildlife diversity within harvested management units. One important tool used to pursue that goal is green tree retention harvesting as described above, as there is interest in retaining structural elements in IMPF landscapes to benefit wildlife (Rudolph and Dickson 1990; Dickson et al. 1995; Rosenvald and Lõhmus 2008; Jones et al. 2009; Perry et al. 2011; Gustafsson et al. 2012; Greene et al. 2016; Warrington et al. 2016; Demarais et al. 2017) . However, the technical foundation underpinning our understanding of wildlife responses to green tree retention is limited, leading regional commercial timber companies to express interest in developing a better understanding of biodiversity responses to green tree retention effects. Furthermore, land owners experience operational and opportunity costs arising from the retention of unharvested trees on MUs (Lickwar et al. 1992; Cubbage 2004; Miller et al. 2009; Lakel et al. 2015). Therefore, an improved understanding of the impacts of green tree retention on wildlife diversity in harvested MUs would represent a welcome tool for supporting biodiversity in harvest plans.

Prior to this research project, the operational range of green tree retention in the Southern US (i.e., the typical area of retained forest cover associated with clearcuts on IMPF management units) had not previously been examined. This lack of basic knowledge represented an obstacle to understanding effects on biodiversity of the common practice of green tree retention harvesting. My first objective (Chapter 2) was to remedy this knowledge gap by conducting a survey of land cover composition and configuration on management units across a large spatial extent: the South Central Plains ecoregion of Arkansas, Louisiana, and Texas.

Plant and animal communities associated with SMZs are relatively well studied, but while stringers are commonly retained on management units alongside SMZs, there have been no substantive studies examining the structural and faunal similarity of stringers to SMZs in the South. My second objective (Chapter 3) was to survey and compare vegetative structure in SMZs and stringers to improve forest managers' understanding of the potential for stringers to contribute to structural diversity on harvested management units. Concurrently, my third objective (Chapter 4) was to survey and compare avian communities in the three dominant cover types found on three-year-old management units (i.e., regenerating clearcuts, SMZs, and stringers) to provide managers with information on relationships between retained structures and bird community density and diversity.

Intensively managed pine forests represent a large proportion of land area in the South Central Plains ecoregion, and consequently, there exists potential for land managers to coordinate conservation efforts at scales larger than single management units. However, little research attention has been given to the association of landscape characteristics at local- (internal management unit characteristics) and regional-scales (landscape buffers of several kilometers surrounding management units) with patterns of avian species diversity inside management units. My fourth objective (Chapter 5), intended to inform future regional-scale management decisions, was to classify regional land cover data and use it to determine if local-scale and/or regional-scale landscape characteristics could best explain observed patterns of avian species diversity.

1.1 Literature Cited

American Forest Foundation. 2015. American Forest Foundation (AFF) 2015-2020 Standards of Sustainability. Available online at www.treefarmsystem.org/view-standards; last accessed Oct. 15, 2016.

Aubry, K.B., C.B. Halpern, and C.E. Peterson. 2009. Variable-retention harvests in the Pacific Northwest: A review of short-term findings from the DEMO study. Forest Ecology and Management 258:398-408.

Aust, W.M., and C.R. Blinn. 2004. Forestry best management practices for timber harvesting and site preparation in the eastern United States: An overview of water quality and productivity research during the past 20 years (1982–2002). Water, Air and Soil Polution: Focus 4(1):5-36.

Bechtold, W.A., and P.L. Patterson. 2005. The enhanced forest inventory and analysis program-national sampling design and estimation procedures. Gen. Tech. Rep. SRS-80. Asheville, NC: US Department of Agriculture, Forest Service, Southern Research Station. 85 p.

Blinn, C.R., and M.A. Kilgore. 2001. Riparian management practices: A summary of state guidelines. Journal of Forestry 99(8):11-17.

Cubbage, F.W. 2004. Costs of forestry best management practices in the south: A review. Water, Air and Soil Pollution: Focus 4(1):131-142.

Demarais, S., J.P. Verschuyl, G.J. Roloff, D.A. Miller, and T.B. Wigley. 2017. Tamm review: Terrestrial vertebrate biodiversity and intensive forest management in the U.S. Forest Ecology and Management 385:308-330.

Dickson, J.G., J.H. Williamson, R.N. Conner, and B. Ortego. 1995. Streamside zones and breeding birds in eastern Texas. Wildlife Society Bulletin 23(4):750-755.

Forest Stewardship Council US. 2010. FSC - US Forest Management Standard (v.1.0). Available online at https://us.fsc.org/download.fsc-us-forest-management-standard-v1-0.95.htm. Last accessed Oct. 15 2016.

Fox, T.R., E.J. Jokela, and H.L. Allen. 2007. The development of pine plantation silviculture in the southern United States. Journal of Forestry 105(7):337-347.

Franklin, J.F., D.R. Berg, D.A. Thornburgh, and J.C. Tappeiner. 1997. Alternative silvicultural approaches to timber harvesting: Variable retention harvest systems. P. 111-136 in Creating a Forestry for the 21st Century: The Science of Ecosystem Management, Kohm, K.A., and J.F. Franklin (eds.). Island Press, Washington, D.C.

Greene, R.E., R.B. Iglay, K.O. Evans, K.V. Miller, T.B. Wigley, and S.K. Riffell. 2016. A meta-analysis of biodiversity responses to management of southeastern pine forests - Opportunities for open pine conservation. Forestry Ecology and Management 360:30-39.

Gustafsson, L., S.C. Baker, J. Bauhus, W.J. Beese, A. Brodie, J. Kouki, D.B. Lindenmayer, et al. 2012. Retention forestry to maintain multifunctional forests: A world perspective. BioScience 62(7):633-645.

Jones, P.D., B.B. Hanberry, and S. Demarais. 2009. Stand-level wildlife habitat features and biodiversity in Southern pine forests: A review. Journal of Forestry 107:398-404.

Jones, P.D., B.B. Hanberry, and S. Demarais. 2010. Managing the Southern pine forest - Retained wetland interface for wildlife diversity: Research priorities. Wetlands 30:381-391.

Lakel, W.A., W.M. Aust, C.A. Dolloff, and P.D. Keyser. 2015. Residual timber values within piedmont streamside management zones of different widths and harvest levels. Forest Science 61(1):197-204.

Legrand, H.G., M.J. Chamberlain, and E.B. Moser. 2007. Diversity and abundance of breeding birds in a managed loblolly pine forest in Louisiana. The American Midland Naturalist 157(2):329-344.

Lickwar, P., C. Hickman, and F.W. Cubbage. 1992. Costs of protecting water quality during harvesting on private forestlands in the Southeast. Southern Journal of Applied Forestry 16(1):13-20.

Lindenmayer, D.B., and J.F. Franklin. 2002. Conserving Forest Biodiversity: A Comprehensive Multiscaled Approach. Island Press, Washington, D.C. 352 p.

Little, E.L., Jr. 1979. Checklist of United States Trees (Native and Naturalized). US Department of Agriculture Agriculture Handbook no. 541. 375 p.

Miller, D.A., T.B. Wigley, and K.V. Miller. 2009. Managed forests and conservation of terrestrial biodiversity in the southern United States. Journal of Forestry 107(4):197-203.

Oswalt, S.N., W.B. Smith, P.D. Miles, and S.A. Pugh. 2014. Forest Resources of the United States, 2012: A Technical Document Supporting the Forest Service 2015 Update of the RPA Assessment. Gen. Tech. Rep. WO-91. Washington, DC: U.S. Department of Agriculture, Forest Service, Washington Office. 218 p.

Perry, R.W., T.B. Wigley, M.A. Melchiors, R.E. Thill, P.A. Tappe, and D.A. Miller. 2011. Width of riparian buffer and structure of adjacent plantations influence occupancy of conservation priority birds. Biodiversity and Conservation 20:625-642.

Rosenvald, R., and A. Lõhmus. 2008. For what, when, and where is green-tree retention better than clear-cutting? A review of the biodiversity aspects. Forest Ecology and Management 255(1):1-15.

Rudolph, D.C., and J.G. Dickson. 1990. Streamside zone width and amphibian and reptile abundance. The Southwestern Naturalist 35(4):472-476.

Sustainable Forestry Initiative. 2015. SFI 2015-2019 standard and rules: Standards, rules for label use, procedures, and guidance. Available online at www.sfiprogram.org/files/pdf/2015-2019-standardsandrules-web-lr-pdf/; last accessed on Oct 15. 2016.

Trani, M.K., R.T. Brooks, T.L. Schmidt, V.A. Rudis, and C.M. Gabbard. 2001. Patterns and trends of early successional forests in the eastern United States. Wildlife Society Bulletin 29(2):413-424.

Warrington, B.M., W.M. Aust, S.M. Barrett, W.M. Ford, M.C. Bolding, and C.A. Dolloff. 2016. Potential effects of forestry operations and associated best management practices on riparian wildlife species in the southeastern United States. P. 106-112 in Proc. of the 18th biennial Southern silvicultural research conference. Schweitzer, C.J., W.K. Clatterbuck and C. M. Oswalt (eds.). e–Gen. Tech. Rep. SRS–212. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 614 p.

Wilson, M.D., and B.D. Watts. 2000. Breeding bird communities in pine plantations on the coastal plain of North Carolina. The Chat 64(1):1-14.

Yin, R., and R.A. Sedjo. 2001. Is this the age of intensive management? A study of loblolly pine on Georgia's Piedmont. Journal of Forestry 99(1):10-17.

Zhao, D., M. Kane, R. Teskey, T.R. Fox, T.J. Albaugh, H.L. Allen, and R. Rubilar. 2016. Maximum response of loblolly pine plantations to silvicultural management in the southern United States. Forest Ecology and Management 375:105-111.

CHAPTER II

OPERATIONAL GREEN TREE RETENTION AND LAND COVER PATTERNS IN INTENSIVELY MANAGED PINE FOREST LANDSCAPES OF THE SOUTHERN UNITED STATES

2.1 Author's Note

This chapter was originally published in *Forest Science* as 'Operational green tree retention and land cover patterns in intensively managed pine forest landscapes of the southeastern U.S.' (Parrish et al. 2018). It is reproduced here with only minor editorial changes.

2.2 Abstract

Southern U.S. landowners participating in forest certification programs sometimes use green tree retention to promote structural diversity in intensively managed pine (*Pinus* spp.) forests (IMPF) and benefit wildlife species. However, the operational extent of green tree retention practices is poorly understood. Therefore, I classified land cover on 1188 South Central Plains IMPF management units ('MUs'; totaling 51646.2 ha), defined as contiguous, forested areas containing one or more IMPF patches, harvested and established as a cohort, plus associated green tree retention areas. For each MU, I characterized green tree retention levels and land cover attributes. As expected, given my sampling frame, MU land cover was dominated by regenerating clearcut (mean ± SD: 80.5% ± 14.4% of land cover) and green tree retention cover (mean 18.7% ± 14.3% of

land cover). Retention cover consisted mostly of streamside management zones (mean 14.0 % ± 13.1% of land cover) bordering perennial and intermittent streams and stringers (mean 3.3% ± 4.3% of land cover) buffering ephemeral streams. Green tree retention land cover represented a substantial proportion of the IMPF landscape in the region and potentially enhances habitat conditions for many wildlife species.

2.3 Introduction

In the southern United States (henceforth, the South), timberland[a] covered 39.4% (85.0 Mha) of the region in 2012, accounting for 85.8% of the region's forested land cover (Oswalt et al. 2014). Approximately 33% of southern timberland in 2012 was either loblolly-shortleaf pine (*Pinus taeda* L. - *P. echinata* Mill.[b]) or longleaf-slash pine (*P. palustris* Mill. - *P. elliottii* Engelm.), 56% of which was planted (Oswalt et al. 2014). Timber production in the region largely occurs on planted timberland that is intensively managed using tools such as chemical and/or mechanical site preparation, fertilization, and thinning (Yin and Sedjo 2001; Jones et al. 2009a). Management intensity coupled with spatial extent of managed forests provide opportunities for commercial timberland to contribute to regional biodiversity (Greene et al. 2016; Demarais et al. 2017).

During early-rotation (prior to initial canopy closure) and post-thinning phases, intensively managed pine forests (IMPF) represent an important reservoir of early-successional land cover, which is an ephemeral and declining resource in the eastern United States (Wilson and Watts 2000; Askins 2001; Trani et al. 2001; King and

[a] Timberland: forestland productive enough to produce 1.4 $m^3 \cdot ha^{-1} \cdot y^{-1}$ of commercial timber (Bechtold and Patterson 2005).
[b] Botanical taxonomic nomenclature follows Little, Jr. (1979).

Schlossberg 2014; Owens et al. 2014). Southern forests contain the greatest proportion of young (seedling/sapling stand-size class) forestland (21.4% of total timberland) of any U.S. region (Oswalt et al. 2014). Thus, there is interest in improved understanding of biodiversity outcomes related to management of southern IMPF landscapes (Brawn et al. 2001; Fink et al. 2006; Demarais et al. 2017). An important preliminary step toward this goal is developing baseline data on land cover patterns created on IMPF landscapes.

Landowners participating in forest certification programs are asked to consider biodiversity in their management plans (Forest Stewardship Council US 2010; American Forest Foundation 2015; Sustainable Forestry Initiative 2015). One approach for addressing biodiversity-related goals is retention of structural features (e.g., green tree retention), which has the potential to enhance stand- and landscape-level diversity (Franklin et al. 1997; Bauhus et al. 2009; Miller et al. 2009; Gustafsson et al. 2012). Proposed benefits of green tree retention include: (a) enrichment of structural complexity; (b) retention of biological legacies; and (c) enhanced connectivity across the harvested landscape (Franklin et al. 1997; Lindenmayer and Franklin 2002; Aubry et al. 2009).

Southern IMPF management units (MUs[c]) are typically harvested using clearcutting in conjunction with green tree retention (Parrish et al. 2017a). Patches of live trees and vegetation are retained as buffers along perennial and intermittent streams[d]

[c] I defined management units (MUs) as contiguous areas containing one or more intensively managed pine forest stand, uniform in age, that was harvested and reestablished as a unified cohort, plus any associated areas of green tree retention (e.g., streamside management zones or stringers).

[d] Perennial streams generally flow year-round, or nearly so, and have well-defined channels. Intermittent streams have seasonal flow and may dry up in hot weather, and also have well-defined channels. See state best management practices documents for state-specific definitions (Louisiana Department of Agriculture and Forestry 1999; Arkansas Forestry Commission 2002; Texas Forest Service 2004).

(streamside management zones, 'SMZs') and as buffers along ephemeral streams[e] (commonly referred to as 'stringers'). Scattered live trees and/or patches of unharvested vegetation may also be retained in some MUs, particularly in wet soils areas. Research interest in the relative importance of retained structural features to fauna in IMPFs in the South is evident (Rudolph and Dickson 1990; Dickson et al. 1995; Rosenvald and Lõhmus 2008; Jones et al. 2009b; Perry et al. 2011; Gustafsson et al. 2012; Greene et al. 2016; Warrington et al. 2016; Demarais et al. 2017). Retention of unharvested patches may result in operational and opportunity costs to landowners, making it critical to improve understanding of green tree retention practices in the South that can potentially be used to develop cost-efficient biodiversity conservation plans (Miller et al. 2009). Opportunity costs associated with SMZ implementation may be minimized by judicious partial harvesting in SMZs, and long-term SMZ thinning policies may provide both economic and ecological benefits to the management unit (Lickwar et al. 1992; Woodman and Cubbage 1994; Cubbage 2004; Lakel et al. 2015).

Best management practice (BMP) guidelines for green tree retention in the South take the form of targets for SMZ width and, in some cases, overstory structure, rather than overall percent green tree retention on the post-harvested landscape (Table 2.1). Guidelines for SMZ width are primarily intended to protect water quality (Aust and Blinn 2004), but SMZs also provide ancillary conservation benefits to terrestrial plants and wildlife (Blinn and Kilgore 2001; Rosenvald and Lõhmus 2008; Jones et al. 2010).

[e] Ephemeral streams usually experience flow only for short durations after rain events, and may have poorly-developed channels. See state best management practices documents for state-specific definitions (Louisiana Department of Agriculture and Forestry 1999; Arkansas Forestry Commission 2002; Texas Forest Service 2004).

Stringers potentially supplement these conservation benefits, but this topic has been poorly studied. To my knowledge, the operational range for area in retained forest cover associated with clearcut harvest units in IMPF in the South remains unexamined. Therefore, my goal was to document operational green tree retention practices in the South (specifically, the South Central Plains ecoregion) to inform management decisions. My objectives were to: (1) assess the operational level of green tree retention on three-year-old IMPF MUs established in 2008 and 2009 and (2) identify patterns of land cover composition and configuration within IMPF landscapes.

2.4 Methods

2.4.1 Study Area

I selected MUs in Arkansas, Louisiana, and Texas in the South Central Plains ecoregion of the U.S. (Fig. 2.1), an area that extends south from central Arkansas and extreme southeastern Oklahoma into northwestern Louisiana and eastern Texas (Omernik 1987; US Environmental Protection Agency 2011). The South Central Plains ecoregion is contained within the North American Bird Conservation Initiative 'West Gulf Coastal Plain / Ouachitas' Bird Conservation Region (Omernik 1987; US Environmental Protection Agency 2011; Bird Studies Canada and North American Bird Conservation Initiative 2014). Historically, upland areas of the South Central Plains were dominated by longleaf pine forests and savannas in the south and mixed shortleaf pine-hardwood forest in the north (Wilkin et al. 2011). Bottomland and riparian sites were dominated by hardwood or mixed pine-hardwood forest (Wilkin et al. 2011). The region represents the western edge of the southern pine belt and modern land cover currently consists of about

67% forest and woodlands, including extensive area in IMPF, and less than 20% cropland (Daigle et al. 2006; Griffith et al. 2007; Wilkin et al. 2011).

Winters are mild and summers are hot, with mean temperatures ranging between approximately 1º C in winter to 34º C in summer (Griffith et al. 2007). Mean annual precipitation ranges between 105 and 170 cm across the ecoregion (Wilkin et al. 2011). Perennial streams (streams flowing year-round) are abundant and mostly of low to moderate gradient (Wilkin et al. 2011), and many smaller streams experience limited or no flow during hot summer months (Woods et al. 2004). In Arkansas, Louisiana, and Texas, BMP guidelines recommended retention of SMZs adjacent to intermittent streams (streams with seasonal flow), perennial streams, ponds, and lakes, primarily for the purpose of water quality protection (Louisiana Department of Agriculture and Forestry 1999; Arkansas Forestry Commission 2002; Texas Forestry Association and Texas Forest Service 2010).

2.4.2 Management Unit and Land Cover Delineation

In 2010, five corporate or institutional landowners (henceforth: 'cooperators'; Forest Capital Partners, LLC; Plum Creek Timber Company, Inc.; Potlatch Corporation; Resource Management Service, LLC, and Weyerhaeuser Company) provided us with land ownership spatial data from harvested stands for 1187 MUs in three Texas counties, 12 Arkansas counties, and 16 Louisiana parishes located in the South Central Plains ecoregion. I requested data on MUs established in 2008 or 2009 so that my results would represent recent patterns in harvesting and stand establishment. I used cooperators' spatial data to guide my identification of ownership boundaries and stand establishment histories, but I delineated stand boundaries based on aerial photographs to ensure a greater conformity to actual post-harvest stand configurations. While all cooperators agreed that MUs consisted of one or more pine stands, harvested and re-established as a cohort, minor inconsistencies existed between cooperators as to whether MUs were considered to be inclusive of any internal and adjacent retention patches. To ensure consistency in my MU delineations across cooperators, I applied an independent set of delineation criteria to all MUs, regardless of ownership. Retention patches[f] internal to MUs were included in the MU. Retention patches separating two adjacent MUs were divided down the stream channel (or the patch midline, if a channel was not visible), and each half was assigned to its MU. I also defined MU external bounds based on adjacent roads, ownership lines, and borders with neighboring MUs.

I classified MU boundaries and land cover by screen digitization in ArcGIS Desktop 9.3. My primary base layer for MU delineations and land cover classifications was aerial imagery from the 2010 high-resolution, true color National Agricultural Imagery Program (NAIP) dataset (Table 2.2). Where available (Arkansas and Texas), I used color infrared (CIR) NAIP imagery from several recent years (Table 2.2) to support my interpretation of the base layer imagery. Features were examined and identified at a variety of spatial scales, but I performed digitization at 1:5000 scale. The high-resolution National Hydrography Dataset (NHD; Table 2.2) provided hydrography data that I used in some cases to help us identify intermittent and perennial streams and to differentiate some SMZs and stringers, but I relied primarily on high-resolution aerial imagery to identify water features. Spatial data on ephemeral stream locations and intermittent/perennial stream width were unavailable at a statewide extent.

---

[f] 'Patches' were defined as areas of distinctive land cover composition greater than 100 m^2 in area.

I defined 10 land cover classes based on visual interpretation of aerial photos and hydrography data and used them to classify land cover patches on each MU. Regenerating pine clearcuts had been harvested and re-established in 2008 or 2009. Four land cover classes contributed to total area in green tree retention: SMZs, stringers (retained trees and vegetation along ephemeral drains that feed into SMZs), non-riparian retention (scattered clumps of trees left standing in clearcuts or along substantial fencerows), and vegetated wet-soil areas. I also identified patches of impervious surface, petrochemical extraction plots, utility rights-of-way (ROWs), and small experimental forest plots. For each MU, I calculated the total area in green tree retention as the sum of area in SMZs, stringers, non-riparian retention, and wet-soil areas. I calculated percent area in green tree retention as total area in green tree retention divided by MU area.

2.4.3 Land Cover Analysis

I used the ZonalMetrics Toolbox (Adamczyk and Tiede 2017) in conjunction with ArcMap 10.3 to analyze my vector land cover dataset to produce a set of land cover pattern metrics for each MU, calculating metrics[g] at the class and landscape[h] levels (Table 2.3) . The class-level metrics, calculated across all patches of the same land cover class in a MU, included: total class area, overall class percentage of landscape, number of patches in class, class patch density per 100 ha, mean patch area, and class internal edge[i] density, omitting edge shared with the MU external boundary. I calculated two additional class-level metrics - mean radius of gyration (a measure of within-patch linear

[g] Equations for spatial metrics can be found in the FRAGSTATS program documentation (McGarigal et al. 2012).
[h] I use 'landscape' and 'landscape-level' to refer to the management unit and its internal mosaic of habitat patches.
[i] I defined 'edge' as boundaries between patches of different cover type.

extent) and mean Euclidean nearest neighbor distance - using FRAGSTATS 4.1 (McGarigal et al. 2012). For the FRAGSTATS analyses, I converted my vector-based land cover data for each MU into ERDAS IMAGINE rasters with 1.0-m cells, matching the resolution of my base layer. Class-level mean radius of gyration and Euclidean nearest neighbor distance were both calculated in FRAGSTATS using simple Euclidean straight line geometry between cell centers (McGarigal 2015), thus avoiding 'stair-step' bias associated with distance calculations using rasterized data. McGarigal (2015) offers extensive background on the calculation and interpretation of these and other spatial metrics.

I calculated landscape-level metrics using vector data across all patches present in a MU landscape: total area of MU landscape; number of patches in MU landscape; edge density of internal patches (of all classes) in MU landscape, omitting edges overlapping the MU external boundary edges; external perimeter of the MU; and external edge density of the MU boundary. I produced summary statistics for each metric across all MUs using SAS software[j], but I omitted class-level metric summaries if fewer than 60 MUs (5% of n=1187 sampled) were available to calculate a given metric. I calculated Pearson's correlation coefficient (*r*) between percent green tree retention and total area of MU landscape using the CORR procedure in SAS to explore relationships between percent green tree retention and MU total area.

The above metrics were chosen for their prevalence across landscapes and consistency in ecological interpretation (Cushman et al. 2008). Metrics of patch or

---

[j] The data analysis for this paper was generated using SAS software, version 9.4 of the SAS System for Windows, copyright © 2002-2012 SAS Institute Inc., Cary, NC, USA.

landscape area, extent, and dominance (i.e., total class area, class percentage of landscape, class mean patch area, total area of MU landscape, percent green tree retention, and total area in green tree retention) address the amount and type of habitat space available to organisms, and may be particularly relevant to area-sensitive species. Patch continuity metrics (i.e., class mean radius of gyration) relate to the within-patch linear extent. Edge-related metrics (i.e., class edge density, landscape internal and external edge density, and landscape perimeter) are relevant to edge effects on wildlife and microclimate. Patch subdivision (i.e., class number of patches, class patch density, landscape number of patches) and patch isolation (i.e., class nearest neighbor distance) metrics deal with land cover fragmentation into patches and patch separation distance. Some metrics may have interpretive value among more than one category above. My intent was to provide a group of metrics potentially useful to a variety of projects, from which relevant metrics could be thoughtfully selected for a particular application.

2.5 Results

2.5.1 Management Unit Land Cover Composition

The 1187 management units covered 51245.7 ha. Mean MU area was 43.2 ha ± 30.6 (mean ± sd), ranging from 1.3 ha to 232.4 ha (Fig. 2.2). By definition, all MU landscapes contained one or more patches of regenerating clearcut, but presence of other cover classes varied (Table 2.4). Of the 1187 MUs, 87 (7%) contained neither SMZ nor stringer patches (because they did not contain apparent perennial or intermittent stream features), 284 (24%) contained one or more patches of SMZ but not stringer, 98 (8%) contained one or more patches of stringer but not SMZ, and 718 (60%) contained one or more patches of both SMZ and stringer. I observed non-riparian retention in 333 MUs

(28%) and unharvested wet soils areas were present in 126 MUs (11%). At least one type of retention land cover was present on 1,142 MUs (96%). The remaining land cover classes (utility rights-of-way, petrochemical extraction plots, small experimental forest plots, and impervious surfaces) were each detected on fewer than 100 MUs, and accounted for less than 1% of total land area sampled. I therefore focused my further analyses on regenerating clearcuts and green tree retention land cover types.

Management unit land cover was dominated by regenerating clearcuts (mean percent of MU: 80.5% ± 14.3%) with most remaining land area in green tree retention cover classes, including 14.0% ± 13.1% in SMZs; 3.4% ± 4.3% in stringers; 0.7% ± 2.3% in non-riparian retention; and 0.6% ± 3.3% in wet-soil areas (Table 2.4). The average 43.5-ha MU (Table 2.5) thus comprised 34.6 ha of regenerating clearcut, 6.1 ha of SMZ, and 1.6 ha of stringer (Table 2.4). Mean MU percent green tree retention was 18.6% ± 14.2% and ranged from 0% retention (i.e., MUs lacking hydrologic features requiring buffers) to 81.4% retention (MUs with only small clearcut patches) (Table 2.5, Figs. 2.3, 2.4). Percent green tree retention was not correlated with MU landscape total area (r = 0.02, P = 0.45; Fig. 2.5).

2.5.2 Management Unit Land Cover Configuration

Management unit landscapes consisted of 1 to 50 patches (mean 7.2 patches ± 5.9; Table 2.5). Mean MU external perimeter was 3.2 km ± 1.4, and ranged from 0.5 km to 12.4 km. MU mean external edge density (93.3 $m \cdot ha^{-1}$ ± 47.2) was greater than mean internal edge density (64.0 $m \cdot ha^{-1}$ ± 38.5).

Mean number of patches was similar for stringers (2.5 ± 3.5) and regenerating clearcuts (2.4 ± 1.9). While the maximum number of regenerating clearcut patches per

MU was 14, stringer patches could sometimes be more numerous, with 34 detected on one MU (Table 2.4). Mean patch density was greatest for regenerating clearcuts (7.8 patches · $(100\ ha)^{-1}$ ± 8.2), followed by stringers (5.8 patches · $(100\ ha)^{-1}$ ± 7.0) and SMZs (4.4 patches · $(100\ ha)^{-1}$ ± 5.2), with remaining cover types exhibiting lower patch density. Grand mean patch area of regenerating clearcuts was 19.9 ha ± 16.8, but clearcut patches sometimes were substantially larger (max. class mean patch area = 110.6 ha). SMZ grand mean patch area (4.3 ha ± 6.3) were typically smaller than regenerating clearcut patches, although SMZs were a dominant component of some MUs (max. class mean patch area = 84.1 ha). Stringers (grand mean patch area 0.5 ha ± 0.6), non-riparian retention (grand mean patch area 0.2 ha ± 0.4), and wet soils areas (grand mean patch area 0.2 ± 1.1) were generally small landscape features relative to regenerating clearcuts and SMZs.

Class mean edge density was calculated only for MUs where class number of patches >1 (i.e., landscapes where a class was present; Table 2.4). Mean edge density was greatest for regenerating clearcuts (60.9 $m \cdot ha^{-1}$ ± 35.6), followed by SMZs (41.9 $m \cdot ha^{-1}$ ± 23.2). Stringer mean edge density (32.2 $m \cdot ha^{-1}$ ± 25.7) was similar to that of ROWs (31.1 $m \cdot ha^{-1}$ ± 23.5); but other land cover classes exhibited mean edge density below 14 $m \cdot ha^{-1}$. Grand mean radius of gyration was largest for clearcuts (177.6 m ± 84.6). SMZs and ROWs, both of which are typically long, linear features on the landscape, had comparable grand mean radius of gyration (149.8 m ± 90.6 and 146.0 ± 72.5, respectively). Grand mean radius of gyration for stringers, non-riparian retention, and wet-soils areas were approximately 26%-36% that of SMZs or ROWs.

Patch isolation (i.e., class Euclidean nearest neighbor distance) was calculated only for classes with >1 patches present on a given MU (Table 2.4). Patch isolation was least for clearcuts, with grand mean nearest neighbor distance of 32.1 m ± 24.3. Grand mean nearest neighbor distance for stringers was next lowest (169.4 m ± 157.2), and was greater for SMZs (221.1 m ± 216.5) and non-riparian retention patches (297.3 m ± 344.2).

2.6 Discussion

2.6.1 Operational Range of Green Tree Retention on IMPF Management Units

The 18.6% mean green tree retention level measured on my sites exceeded suggested percent retention levels from other regions of the U.S, which range from 3% to > 15% of stands (USDA Forest Service and USDI Bureau of Land Management 1994; Aubry et al. 2009; Benjamin 2010; Wisconsin Department of Natural Resources 2011; Bielecki et al. 2012). The Federal Northwest Forest Plan specifies a minimum of 15% green tree retention in riparian reserves on National Forest lands (USDA Forest Service and USDI Bureau of Land Management 1994). Wisconsin Department of Natural Resources (2011) suggests retaining 5 to 15% of crown cover as reserve trees or patches on even-aged rotations, noting that riparian buffers satisfy that requirement. Michigan Department of Natural Resources suggests 3 to 10% retention on stands harvested by clearcutting with reserves (Bielecki et al. 2012). Gustafsson et al. (2012) recommend no less than 5% to 10% retention, but caution that some ecological objectives may require greater retention. My estimate of percent retention was similar to the "approximately 15%" operational retention level reported in a study on 24,000 ha of IMPF in Mississippi (Elmore et al. 2004).

Forestry BMPs in the South recommend retention of SMZs around water bodies, with SMZ width based on local slope or stream width, but generally do not express recommendations in terms of percent retention (Louisiana Department of Agriculture and Forestry 1999; Arkansas Forestry Commission 2002; Texas Forest Service 2004; see also: Table 2.1). Management units on institutional forest lands lacking intermittent or perennial streams, or sites with minimal slope, may consequently contain little or no green tree retention cover based on BMP guidelines. Nevertheless, a high proportion (93%) of MUs I sampled contained retention land cover in SMZs and/or stringers. Forestry BMP-recommended SMZ widths in physiographic regions of the South with low relief and soils resistant to erosion, such as my study area, can be smaller than in regions where erosion risk is greater, potentially resulting in less green tree retention relative to other regions (Lee et al. 2004). However, green tree retention land cover made up a sizable proportion of the MUs that I surveyed. Mean BMP compliance rates (including lands owned by non-industrial private forest landowners) in the South are high (Southern state mean compliance was 92.4%; national mean compliance was 91%), and compliance with SMZ- and wetland-related practices within the South Central Plains States (Arkansas, Louisiana, Oklahoma, and Texas) ranges from 86% to 100% (National Association of State Foresters 2015; Cristan et al. 2017). Therefore, I expect that the practices documented in this study are representative of those throughout the region.

Opportunity costs associated with SMZ implementation and other forms of green tree retention have historically been a small component of overall BMP implementation costs (Cubbage 2004). These costs are typically shouldered wholly by land owners. Using data from 22 Alabama, Florida, and Georgia timber harvests in 1987, Lickwar et al. (1992) estimated SMZ implementation costs ranging from 0.09% to 0.38% of gross revenue depending on implementation intensity, with most costs arising from unharvested timber left in the SMZs (mean percent retention: 3.7%; range: 0 to 5.7% retention). Woodman and Cubbage (1994) estimated that in Georgia, SMZ implementation represented 3% to 7% of BMP implementation costs in industrial forests, and 10% of costs in non-industrial private forests. SMZ implementation costs may be increased due to logistical concerns (e.g., equipment availability issues) or mitigated by realizing value through partial harvesting in SMZs (Cubbage 2004; Lakel et al. 2015).

BMP guidelines commonly suggest that, where appropriate, managers expand buffers based on site-specific factors (e.g., soil erosion and sedimentation potential, or trout stream status; Table 2.1). Thus, I attribute the lack of correlation between retention level and MU area to harvesting decisions based on individual site conditions, independent of MU land area. Although partial harvest within retention areas is allowed by BMPs in many states (Table 2.1), I was unable to differentiate between partially harvested and unharvested retention areas based on aerial photographs.

Recommended SMZ widths are sometimes based on stream width: a metric currently unavailable as a regional-scale spatial dataset. Consequently, assessment of stream width requires local data collection, bypassing the advantages of using regional-scale data for remote determination of local site conditions. For some land owners, BMP implementation could be enhanced through the use of publicly-available, regional-scale geospatial metrics to aid in harvest planning tasks. Because 19.4% of retention land cover in my study was associated with ephemeral streams, access to large-scale geodata describing ephemeral stream channels (presently unavailable) could potentially strengthen efforts to address ecological objectives in IMPF landscapes (Daggupati et al. 2013).

2.6.2 Land Cover Patterns on Southern IMPFs

I detected a variety of cover types in MUs, but 97.9% of the typical MU landscape (by mean percent cover) was made up of regenerating clearcuts, SMZs, and stringers, which often are the focus of harvest planning. The landscape dominance of regenerating clearcuts on my sampled sites was expected given my sampling frame of recently harvested forest stands; however, it should be considered that the MUs I sampled are part of a larger landscape mosaic containing a variety of forest age classes. The most complex MU I examined was made up of 50 discrete patches, suggesting some harvest operators currently recognize and plan around complex pre-harvest land cover patterns (where present). Historical site management, however, may have contributed to the simpler MU configuration (landscape-level mean 7.2 patches ± 5.9 patches), that I observed.

The supplementary resources provided by patches of green tree retention adjacent to clearcuts are predicted to benefit MU species diversity, particularly species associated with mature forest, by enhancing landscape structural complexity and connectivity (Franklin et al. 1997; Lindenmayer and Franklin 2002; Aubry et al. 2009). For example, on a subset of 60 MUs selected from the 1187 MUs described in this paper, 27% of bird species were detected solely in SMZs and stringers, while 42 of 44 avian species occurring in regenerating clearcuts also occurred in retention land cover patches, highlighting contributions made by green tree retention land cover to MU species diversity (Parrish et al. 2017b). Luck and Wu (2002) previously called for a baseline quantification of landscape spatial signatures, against which future signatures can be compared, as an important step towards monitoring effects of landscape change on ecological processes. The spatial signatures I describe contribute to a baseline understanding of how these landscapes compare with other contemporary (and future) landscapes and how IMPFs can potentially play a role in landscape and harvest planning.

The regenerating clearcuts that dominated MUs in this study were young forests that are important contributors to the ecological network of early-successional forestland patches in the South (Trani et al. 2001). The configuration of clearcuts into extensive patches with associated large radii of gyration relative to overall MU dimensions, suggests that my study sites may benefit area-sensitive species associated with early-successional conditions. Evidence for area-sensitivity in eastern shrubland birds of southern forest systems and eastern hardwood forest systems is mixed, with some studies showing weak or no evidence for area-sensitivity (Krementz and Christie 2000; Rodewald and Vitz 2005; Lehnen and Rodewald 2009) and others arguing that some shrubland species do exhibit sensitivity (Schlossberg and King 2008; McDermott and Wood 2011; Shake et al. 2012). Shake et al. (2012) called for patches greater than five ha in coastal North Carolina, McDermott and Wood (2011), working in Appalachian hardwood clearcuts, recommend cuts larger than 10 to 15 ha, and several studies suggest few and large clearcut patches minimizing edge and maximizing core area (Schlossberg and King 2008; Lehnen and Rodewald 2009) to benefit shrubland species.

My clearcut mean patch area was just under 20 ha and mean total area in clearcuts per MU (the combined area of regenerating clearcut patches in a MU) was nearly 35 ha, which meets or exceeds the above recommendations, but is below harvest size limits of some forest certification programs (Forest Stewardship Council US 2010; Sustainable Forestry Initiative 2015) and individual companies (Boston and Bettinger 2001). Although my estimate of clearcut mean patch area has relevance within the context of landscape ecology, mean total area in clearcuts may be a more relevant metric for land owners. For certification and planning purposes, landowners generally consider clearcut size at the MU level, and all patches of clearcut in a MU are treated as one 'operational clearcut', without regard to potential subdivision by patches of other cover types. Any other cover types that may be present, therefore, are typically regarded as "inclusions" within clearcuts.

Where multiple clearcut patches occurred in MUs, they were located in close proximity on the landscape (mean nearest neighbor distance = 32.1 m ± 24.3 m), potentially contributing to internal MU landscape permeability for organisms moving between clearcuts. However, I note that long, thin SMZs dividing some clearcuts served to increase the number of clearcut patches, potentially leading to reductions in core area (Azevedo et al. 2008). Schlossberg and King (2008), evaluated seven studies of birds of eastern hardwood clearcuts and concluded that clearcut patches exhibiting low shape complexity (i.e., more compactly-shaped patches) may benefit shrubland birds by limiting edge. Edge shared with external boundaries was not included in edge density estimates, so edge density for clearcuts and SMZs (the two classes most often contacting MU external borders) was probably underestimated.

The high mean radius of gyration for SMZs suggest that organisms averse to traversing clearcuts could use SMZs to move through young MUs. Researchers in Canada and Australia have reported that some forest bird species prefer traveling longer routes dominated by mature forest to moving shorter distances through regenerating forest (Machtans et al. 1996; Desrochers and Hannon 1997; Belisle and Desrochers 2002; Robertson and Radford 2009). However, the high interpatch distance between SMZs that I observed may constrain movement between nearby SMZs. Crossing between widely-separated SMZs on young MUs may require choosing between safer (but more energetically costly) travel around a clearcut versus potentially riskier travel through the clearcut (Belisle and Desrochers 2002). There is some evidence for Florida Scrub-Jays (*Aphelocoma coerulescens*)[k] that habitat gaps greater than 2 to 3 km can cause gene flow disruptions among populations (Coulon et al. 2012), but SMZ separation on my sites was considerably lower than that threshold. As regenerating clearcuts age, overall MU landscape permeability for interior forest species would be expected to increase.

Some forest certification programs limit harvesting in adjacent MUs until trees in regenerating clearcuts have reached a minimum height or age (e.g., Sustainable Forestry Initiative 2015), which may result in relatively high-contrast ('hard') edges between recently clearcut areas and neighboring unharvested forest stands (Stamps et al. 1987). Linear forest patches, such as SMZs and stringers, may exhibit high edge density as a product of their elongated shape and relatively small patch size; when they are adjacent to a regenerating clearcut, that edge may be considered high-contrast. Lehnen and Rodewald (2013) reported lower shrubland bird abundance in habitat edges bordering mature oak-hardwood forest in Ohio, but attributed it to passive displacement rather than edge avoidance. Forest interior birds may also exhibit lower abundance near hard edges

[k] Avian taxonomic nomenclature follows American Ornithological Society Checklist of North and Middle American Birds (American Ornithological Society 2017) through the fifty-seventh supplement (Chesser et al. 2016).

with open conditions (King et al. 1997). Bats, however, appear to use hard edges on IMPFs for orientation and as movement corridors (Hein et al. 2009; Kalcounis-Rueppell et al. 2013).

Although mean interpatch distances were > 150 m for stringers, SMZs frequently provided forested corridors connecting stringers, potentially reducing travel costs for some species below what might be suggested by simple estimates of nearest neighbor distance. Stringers shared many similarities with SMZs in regard to tree density and vegetation cover on 60 IMPFs in the South Central Plains (Parrish et al. 2017a), suggesting that where stringers are configured as protrusions from SMZs, combining to form essentially contiguous patches of forest, simple estimates of MU patch density may suggest greater fragmentation on the MU landscape than is functionally present. On those 60 IMPF management units, stringers and SMZs shared 84% of bird species, suggesting that stringers augmented SMZ contributions to MU avian diversity, while diversity of shrubland specialist bird guilds was also very similar between stringers and regenerating clearcuts (Parrish et al. 2017b). I therefore expect that stringers supplement avian diversity in both regenerating clearcuts and SMZs, by facilitating access to resources in clearcuts by forest interior bird species and to mature forest resources by shrubland specialists. Forest birds in Ohio, Virginia, and West Virginia have been observed utilizing clearcuts, particularly during the post-breeding season (Marshall et al. 2003; Vitz and Rodewald 2006). Stringers that contain snags may also enhance the value of clearcuts in IMPFs for cavity nesting birds and other shrubland wildlife (Dickson et al. 1983; Homyack et al. 2011).

Canopy closure of IMPFs, which typically occurs at around 8 to 9 years in the South Central Plains, drastically alters stand conditions (Miller et al. 2009; Jones et al. 2012; Campbell et al. 2015) and the landscape relationships described in this paper should be interpreted in that context. For example, edge contrast between regenerating clearcuts and retention patches decreases with stand age, leading to expected decreases in edge effect intensity. Also, changes in structural conditions during regeneration of IMPFs may influence small mammal community makeup within adjacent SMZs (Miller et al. 2004). In this paper, I addressed internal landscape characteristics of early-rotation IMPF management units. However, a comprehensive view of the landscape context of young IMPFs should also include their role as part of the larger, regional-scale landscape mosaic that contains forests of varying age classes and compositions as well as areas of other land use.

2.7 Management Implications

Early successional land cover has declined in abundance in the eastern U.S. (King and Schlossberg 2014), and the shrubby conditions of early-rotation IMPFs represent an important reservoir of young forestland in the region. Concurrently, retaining substantial patches in SMZs and stringers increases structural complexity within IMPF landscapes and promotes landscape connectivity that likely enhances habitat conditions for species associated with older forests. The extensive edges formed by interspersion of RCCs with SMZs, stringers, and other forms of green tree retention in IMPFs seem likely to contribute to the structural diversity and biodiversity value of MUs (Rodewald and Brittingham 2004), but it is worth noting that some early-successional species may respond positively to larger clearcuts with sizeable core area. Thus, if management objectives include support for species of conservation concern, consideration for each species' life history is necessary. Although a single MU cannot be managed to provide optimal habitat conditions for all species, IMPF landscapes in the U.S. South provide a variety of land cover types and opportunities for collaboration among landowners on conservation goals. Improved understanding by stakeholders of the ecological benefits of a diversely-aged, forested landscape mosaic would likely strengthen these collaborative efforts (Ribe 1999; Brockerhoff et al. 2008).

Improved availability of remotely-sensed data relevant to stream width could help landowners determine SMZ width with minimal ground-truthing, particularly owners with large holdings and access to a geographic information systems (GIS). Ideally, these geodata should be (a) freely available (preferably online) and easily accessed; (b) accurately mapped with a fine grain and large extent (state-wide at minimum); and (c) based on static geomorphological features that exist independently of climatological conditions. Authors of future BMP updates may wish to consider these factors when recommending methods for delineating retention areas.

Table 2.1 Best management practices (BMP) documents for states in the US Southern region in effect during study period.

Jurisdiction	Reference
Alabama	Alabama Forestry Commission. 2009. Alabama's Best Management Practices for Forestry. 31 p.
Arkansas	Arkansas Forestry Commission. 2002. Arkansas forestry best management practices for water quality protection. 58 p.
Florida	Florida Department of Agriculture and Consumer Services. 2008. Silviculture Best Management Practices. 116 p.
Georgia	Georgia Forestry Commission. 2009. Georgia's Best Management Practices for Forestry. 72 p.
Kentucky	Stringer, J. W., and C. Perkins. 2001. Kentucky Forest Practice Guidelines for Water Quality Management. Cooperative Extension Service, University of Kentucky, College of Agriculture. 111 p.
Louisiana	Louisiana Department of Agriculture and Forestry. 1999. Recommended forestry best management practices for Louisiana. 83 p.
Mississippi	Mississippi Forestry Commission. 2008. Best Management Practices for Forestry in Mississippi. 44 p.
North Carolina	North Carolina Forest Service. 2006. North Carolina Forestry Best Management Practices Manual To Protect Water Quality. 137 p.
Oklahoma	Oklahoma Forestry Services. 2008. Forestry Best Management Practice Guidelines for Water Quality Management in Oklahoma. 21 p.
South Carolina	South Carolina Forestry Commission. 1994. South Carolina's Forestry Best Management Practices. 65 p.
Tennessee	Tennessee Department of Agriculture: Division of Forestry. 2003. Guide to Forestry Best Management Practices in Tennessee. 50 p.
Texas	Texas Forestry Association and Texas Forest Service. 2004. Texas Forestry Best Management Practices. 109 p.
Virginia	Virginia Department of Forestry. 2011. Virginia's Forestry Best Management Practices for Water Quality, Fifth Ed. 165 p.

Table 2.2 Spatial data layers used to identify and characterize intensively managed pine forest management units in the South Central Plains ecoregion of the USA.

Layer [a] **(Coverage Area)** [b]	**Data Source Description**
2010 TC digital orthophotos (AR, LA, TX [c])	USDA FSA National Agricultural Imagery Program (NAIP) ArcGIS server. 1 m pixels. Available online at *gis.apfo.usda.gov/arcgis/services/;* last accessed Nov. 2010.
2008-2009 TC / CIR digital orthophotos (TX)	Texas Natural Resources Information System, Texas Water Development Board. 1 m pixels. Available online at *www.tnris.org/datadownload/*; last accessed Jan. 2011.
2006 TC / CIR digital orthophotos (AR)	GeoStor: Arkansas State Land Information Board. 1 m pixels. Available online at *ftp://ftp.geostor.arkansas.gov/*; last accessed Jan. 2011.
2010 Hydrography [d] (AR, LA, TX)	USGS National Geospatial Program: The National Map: National Hydrography Data Set (NHD), high-resolution version. 1:24,000 scale. Available online at *nhd.usgs.gov/data.html*; last accessed Nov. 2010.
Level III Ecoregions (USA)	US Environmental Protection Agency (EPA): Western Ecology Division. Level III Ecoregions. 1:250,000 scale. Available online at *www.epa.gov/wed/pages/ecoregions.htm*; last accessed Sep. 2011.
Bird Conservation Regions (USA)	North American Bird Conservation Initiative – United States. Scale not specified. Available online at *www.nabci-us.org/bcrs.htm*; last accessed Oct. 2011.
2006 Digital Elevation Models (AR)	GeoStor: Arkansas State Land Information Board. 5 m pixels. Available online at *ftp://ftp.geostor.arkansas.gov/*; last accessed Mar. 2011.

Proprietary data on ownership and harvest cycles was provided to the authors by cooperating land owners and used to supplement the publicly-available datasets.
[a] Coverage areas: Arkansas (AR), Louisiana (LA), and Texas (TX).
[b] TC = True Color (3-band); CIR = Color Infrared (4-band imagery).
[c] Texas 2010 NAIP imagery contains both CIR and TC.
[d] Stream flow codes: 46003 (intermittent), 46006 (perennial).

Table 2.3 General descriptions and interpretation notes for spatial attributes (metrics) used to describe 1187 management units (MUs) in intensively managed pine (*Pinus* spp.) forests located in the South Central Plains ecoregion of Arkansas, Louisiana, and Texas, USA.

Spatial Attribute	Abbrev.	Description and Interpretation Notes
Class-level		
Total area (ha)	CA	Sum of total area in class on a MU. Class total area and percent of landscape (below) are basic measures of landscape composition (McGarigal 2015).
Percent of landscape (%)	CPL	Percent of MU made up by a land cover class. Indicates relative dominance of cover type classes in a MU.
Number of patches (count)	CNP	Number of patches in a class on a MU. It is one estimator of fragmentation / subdivision, but should be interpreted along with measures of patch proximity or extent.
Patch density (patches•(100 ha)$^{-1}$)	CPD	Number of patches in a per-area basis for MU. Estimator of fragmentation / subdivision and facilitates comparisons among different-sized landscapes (McGarigal 2015). Provides context to interpretations of class number of patches and class mean patch area.
Mean patch area (ha)	CMPA	Average patch extent by class in a MU. Should be interpreted in conjunction with class-level total area, number of patches, patch density, and patch area variability (McGarigal 2015).
Mean edge density (m•ha^{-1})	CED	MU average ratio of patch edge length to patch area, omitting edges shared with MU external boundary. Of interest in studies of edge effects on wildlife.
Mean radius of gyration (m)	CMGYR	An estimator of within-patch linear extent (MU mean across all patches of a class). Greater values indicate greater within-patch traversal distance.
Mean Euclidean nearest-neighbor distance (m)	CENN	Shortest edge-to-edge distance between two patches of same class (mean across patches of a given class in a MU). Quantifies patch isolation, where greater values indicate greater average distances between patches.
Landscape-level		
Total area of MU (ha)	LTA	Indicates total extent of MU landscape.
Number of patches (count)	LNP	Count of all patches within MU (all classes). Provides an indication of fragmentation / subdivision of the MU.
Internal edge density (m•ha^{-1})	LIED	Ratio of internal patch edge to patch area (across classes; omitting edges shared with MU external boundary). Allows for comparison between differently-sized landscapes (McGarigal 2015). Aids in interpretation of edge effects internal to MU, but does not consider contrast levels between patches.
External perimeter (m)	LPERI	Basic measure of MU perimeter length. Suggests potential for edge effects between MU and adjacent lands.

Table 2.3 (continued)

Spatial Attribute	Abbrev.	Description and Interpretation Notes
Landscape-level		
External edge density ($m \cdot ha^{-1}$)	LEED	Ratio of MU perimeter length to MU total area, allowing for comparisons between landscapes of differing sizes (McGarigal 2015).
Total area in green tree retention (ha)	RET_{TOT}	Total area of green tree retention patches (classes: streamside management zone (SMZ), stringer, non-riparian retention, wet-soils area) retained on MU after harvest.
Percent green tree retention (%)	RET_{PCT}	Percent of MU landscape made up of green tree retention cover.

Note: McGarigal (2015) offers a more extensive discussion of metric calculations (in a raster context) and metric interpretation considerations.

Table 2.4 Class-level summary statistics describing spatial attributes of land cover patches within 1187 intensively managed pine (*Pinus* spp.) forest management unit (MU) landscapes, established 2008-2009 in the South Central Plains ecoregion of Arkansas, Louisiana, and Texas, USA.

Spatial Attribute [a]	Land cover class [b]						
	RCC	SMZ [c]	STR [c]	NRR [c]	WET [c]	ROW	OIL
CA (ha)							
$\bar{x} \pm sd$	34.6 ± 24.7	6.1 ± 7.7	1.6 ± 2.5	0.3 ± 0.8	0.2 ± 1.2	0.2 ± 0.9	0.1 ± 0.4
range	[0.4, 173.4]	[0, 84.1]	[0, 25.3]	[0, 8.9]	[0, 19.2]	[0, 13.9]	[0, 7.4]
n	1187	1187	1187	1187	1187	1187	1187
total	41057.6	7227.7	1877.1	310.0	256.2	196.6	63.6
CPL (%)							
$\bar{x} \pm sd$	80.5 ± 14.3	14.0 ± 13.1	3.4 ± 4.3	0.7 ± 2.3	0.6 ± 3.3	0.4 ± 1.7	0.1 ± 0.6
range	[18.6, 100]	[0, 77.3]	[0, 34.4]	[0, 43.9]	[0, 47.3]	[0, 23.3]	[0, 8.4]
n	1187	1187	1187	1187	1187	1187	1187
CNP (count)							
$\bar{x} \pm sd$	2.4 ± 1.9	1.4 ± 1.1	2.5 ± 3.5	0.5 ± 1.0	0.1 ± 0.4	0.1 ± 0.3	0.1 ± 0.3
range	[1, 14]	[0, 9]	[0, 34]	[0, 9]	[0, 4]	[0, 3]	[0, 4]
n	1187	1187	1187	1187	1187	1187	1187
CPD (patches • (100 ha)$^{-1}$)							
$\bar{x} \pm sd$	7.8 ± 8.2	4.4 ± 5.2	5.8 ± 7.0	1.5 ± 3.7	0.4 ± 1.8	0.3 ± 1.3	0.1 ± 0.7
range	[0.7, 83.2]	[0, 74.6]	[0, 49.1]	[0, 27.9]	[0, 24.9]	[0, 18.6]	[0, 13.5]
n	1187	1187	1187	1187	1187	1187	1187
CMPA (ha)							
$\bar{x} \pm sd$	19.9 ± 16.8	4.3 ± 6.3	0.5 ± 0.6	0.2 ± 0.4	0.2 ± 1.1	0.2 ± 0.9	0.0 ± 0.2
range	[0.4, 110.6]	[0, 84.1]	[0, 6.4]	[0, 4]	[0, 19.2]	[0, 13.9]	[0, 3]
n	1187	1187	1187	1187	1187	1187	1187
CED (m • ha^{-1})							
$\bar{x} \pm sd$	60.9 ± 35.6	41.9 ± 23.2	32.2 ± 25.7	13.0 ± 17.4	13.5 ± 15.0	31.1 ± 23.5	...
range	[0, 217.6]	[2.4, 154.6]	[0.8, 148.5]	[0.7, 217.8]	[0.4, 85.8]	[3.6, 161.2]	...
n	1187	1002	816	333	126	87	...

Table 2.4 (continued)

Spatial Attribute [a]	Land cover class [b]						
	RCC	SMZ [c]	STR [c]	NRR [c]	WET [c]	ROW	OIL
CMGYR (m)							
$\bar{x} \pm sd$	177.6 ± 84.6	149.8 ± 90.6	45.7 ± 25.6	39.1 ± 32.3	52.1 ± 47.4	146.0 ± 72.5	...
range	[32.9, 638.2]	[9.7, 597]	[5.8, 216.7]	[5.4, 249.1]	[5.5, 248.1]	[24.7, 376.8]	...
n	1187	1002	816	333	126	87	...
CENN (m)							
$\bar{x} \pm sd$	32.1 ± 24.3	222.1 ± 216.5	169.4 ± 157.2	297.3 ± 344.2	...	...	...
range	[4.3, 280.3]	[3.6, 1708.8]	[6.3, 1176.7]	[14.9, 2682.1]	...	...	...
n	675	430	549	132	...	...	...

Note: Metrics were calculated from vector sources using ZonalMetrics Toolbox in ArcGIS 10.3.1 and SAS 9.4 (CA, CPL, CNP, CPD, CMPA, CED) or from 1-m rasters using FRAGSTATS v. 4.1 (CMGYR, CENN). Values omitted if $n < 60$ sites were available from which to calculate a metric for a class. CED and CMGYR were calculated if site CNP > 0; CENN was calculated if site CNP > 1. Very uncommon land cover types (small internal stands and impervious surface) were omitted for brevity.

[a] Spatial attributes: CA: area in class per MU (and total area over all MUs); CPL: class percentage of landscape; CNP: number of patches in class; CPD: class patch density; CMPA: class mean patch area per MU; CED: class edge density, excluding external edge; CMGYR: class mean radius of gyration; CENN: class mean Euclidean nearest neighbor distance (edge-to-edge).

[b] Land cover classes: regenerating clearcut (RCC); streamside management zone (SMZ), stringer (STR), non-riparian retention (NRR), wet soils (WET), utility rights-of-way (ROW), petrochemical extraction areas (OIL).

[c] Class contributed to total area in green tree retention.

Table 2.5 Landscape-level summary statistics describing land cover spatial attributes within n = 1187 intensively managed pine (*Pinus* spp.) forest management units (MUs), established 2008-2009 in the South Central Plains ecoregion of Arkansas, Louisiana, and Texas.

Spatial attribute [a]	$\bar{x} \pm sd$	**Range**
LTA (ha)	43.2 ± 30.6	[1.3, 232.4]
LNP (patches)	7.2 ± 5.9	[1, 50]
LIED ($m \cdot ha^{-1}$)	64.0 ± 38.5	[0, 258.6]
LPERI (m)	3167.5 ± 1414.4	[516.1, 12414.6]
LEED ($m \cdot ha^{-1}$)	93.3 ± 47.2	[29.6, 703.8]
$\mathbf{RET_{TOT}}$ (ha)	8.1 ± 9.3	[0, 90.8]
$\mathbf{RET_{PCT}}$ (%)	18.6 ± 14.2	[0, 81.4]

Metrics were calculated from vector sources using ZonalMetrics Toolbox in ArcGIS 10.3.1 and SAS 9.4.
[a] Spatial attributes: LTA: MU total area; LNP: number of patches (all classes) on MU; LIED: edge density of MU internal land cover patches, omitting external boundary edge; LPERI: perimeter of MU; LEED: edge density of MU exterior boundary; RET_{TOT}: area of MU landscape in green tree retention land cover classes (classes: streamside management zone (SMZ), stringer, non-riparian retention, and wet soil area); RET_{PCT}: percent of MU landscape in green tree retention.

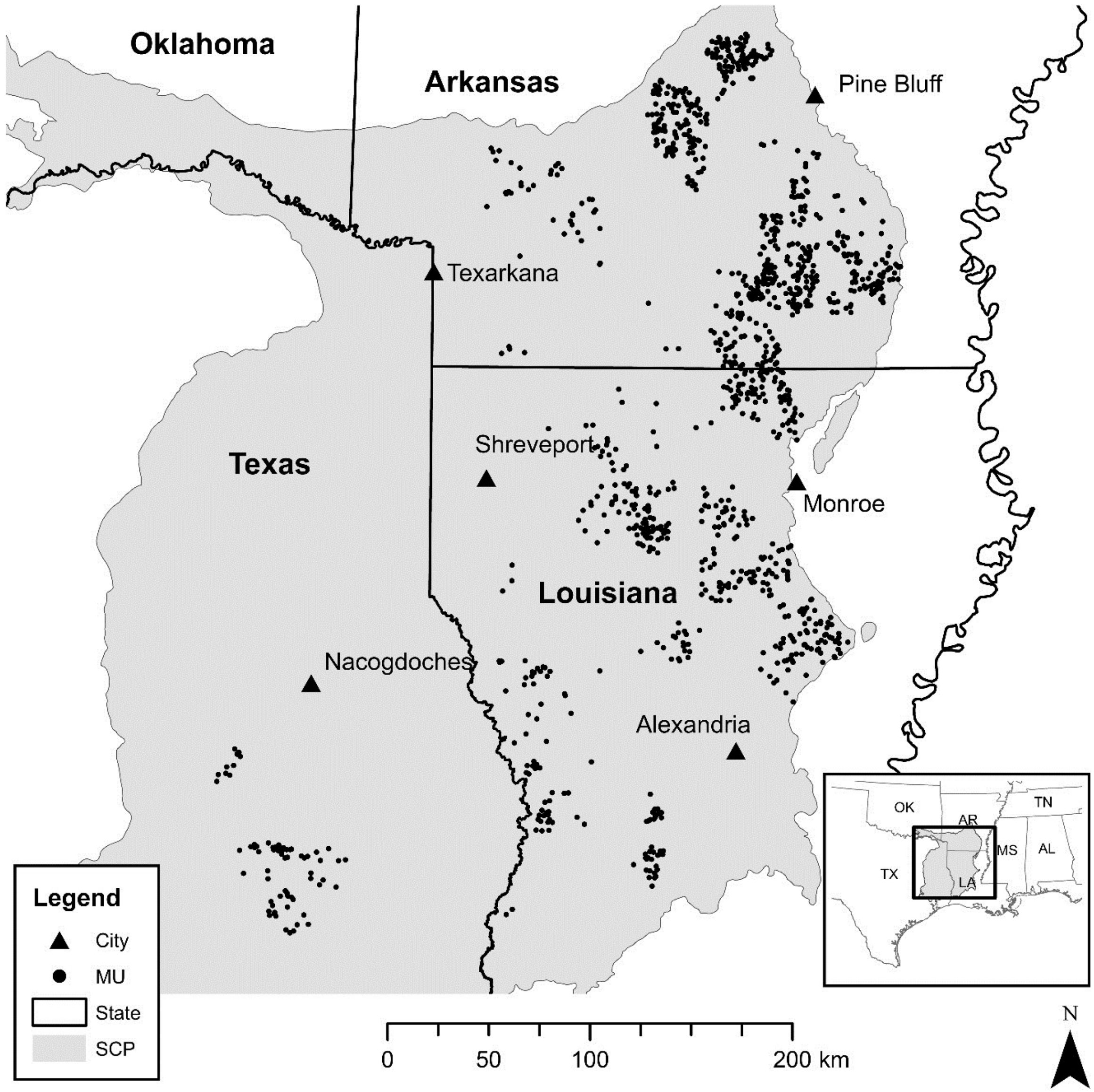

Figure 2.1 Location of 1187 forest management units in Arkansas, Louisiana, and Texas, USA

This map illustrates the locations of 1187 intensively managed pine (*Pinus* spp.) forest management units (MUs) that were established in 2008 (n = 675) and in 2009 (n = 512) in the level III ecoregion South Central Plains (SCP) of Arkansas, Louisiana, and Texas, USA.

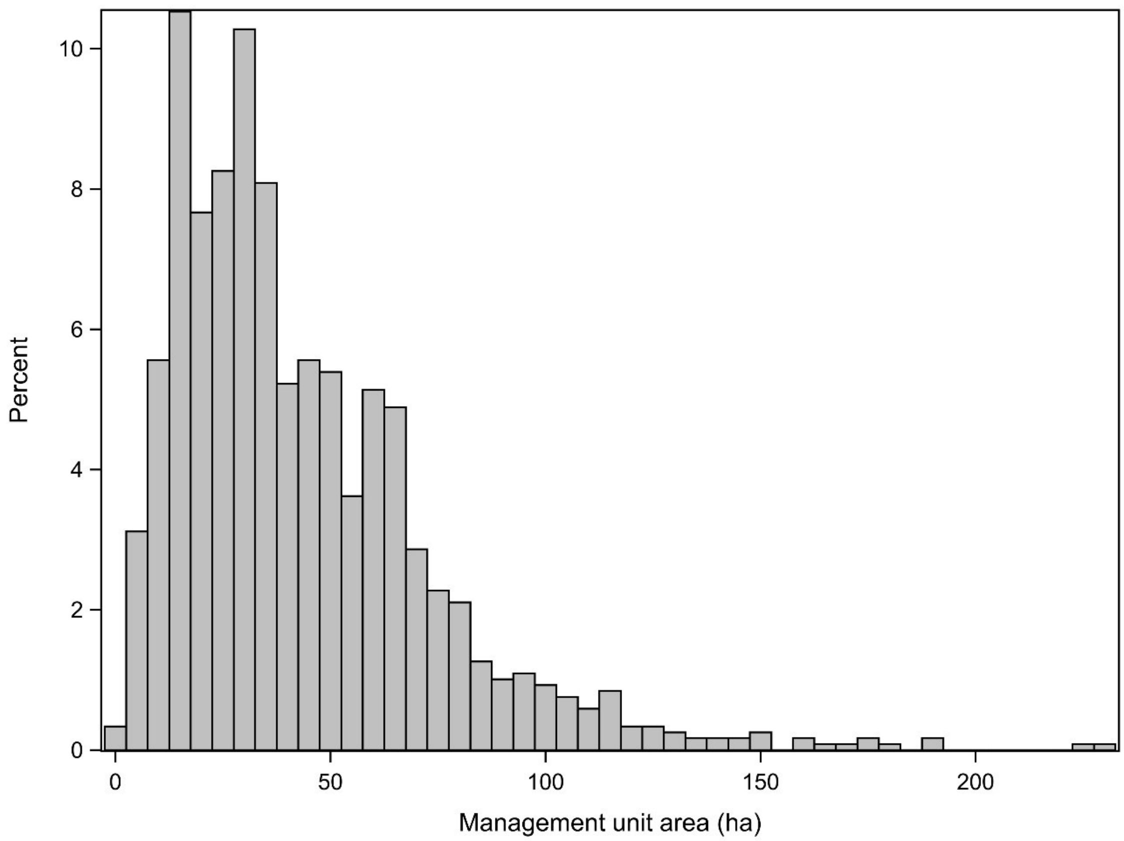

Figure 2.2 Relative frequency histogram of distribution of management unit land area

This relative frequency histogram illustrates the distribution of management unit land area (ha) of 1187 intensively managed pine (*Pinus* spp.) forest management units established in 2008 (n = 675) and in 2009 (n = 512). Management unit area ranged from 1.3 to 232.4 ha (mean 43.2 ha ± 30.6 ha). Management units were located in the South Central Plains ecoregion of Arkansas, Louisiana, and Texas, USA.

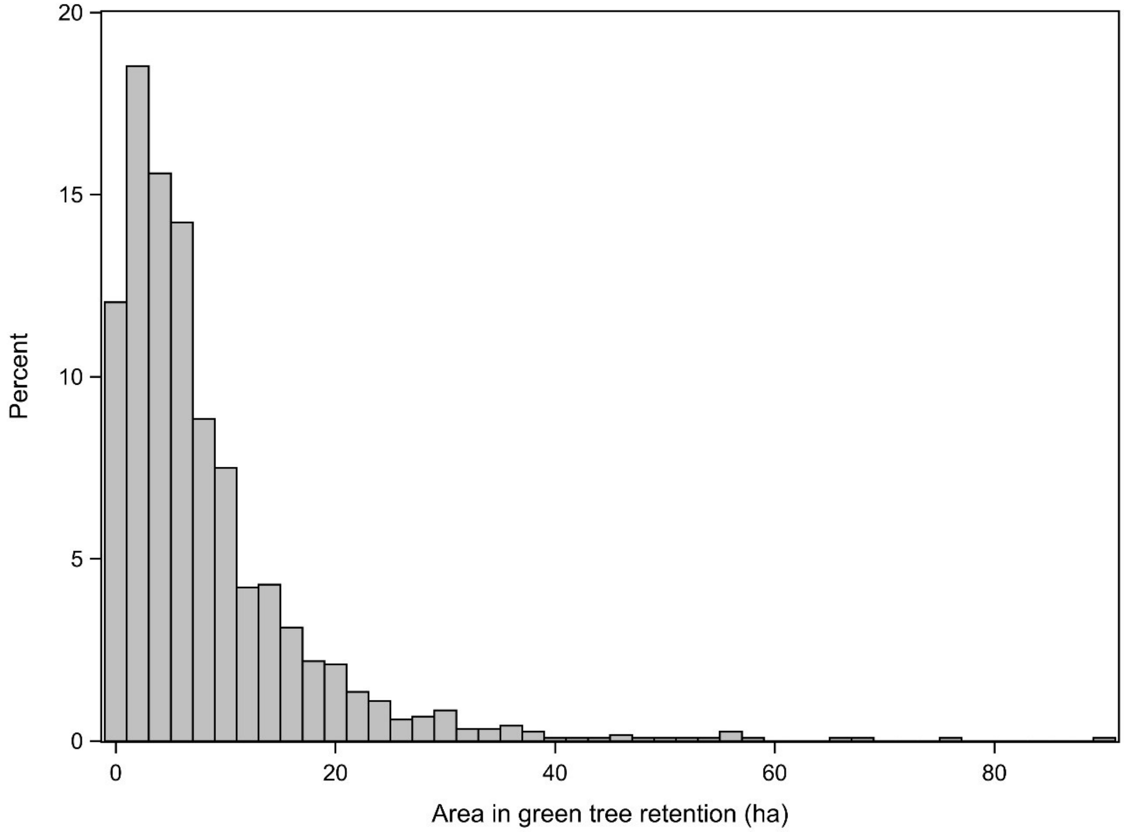

Figure 2.3 Relative frequency histogram of distribution of management unit land area in green tree retention cover classes

This relative frequency histogram illustrates the distribution of management unit land area in green tree retention land cover classes on 1187 intensively managed pine (*Pinus* spp.) forest management units, established in 2008 (n = 675) and in 2009 (n = 512), that were located in the South Central Plains ecoregion of Arkansas, Louisiana, and Texas, USA. Total area in retention per management unit ranged from 0.0 to 90.8 ha (mean: 8.1 ha, sd: 9.3 ha). Land cover classes contributing to area in retention consisted of streamside management zones (SMZs), stringers, non-riparian retention, and wet soils areas. The tail of this plot was truncated at 40 ha for brevity, omitting 17 observations ranging from 40.4 to 90.8 ha.

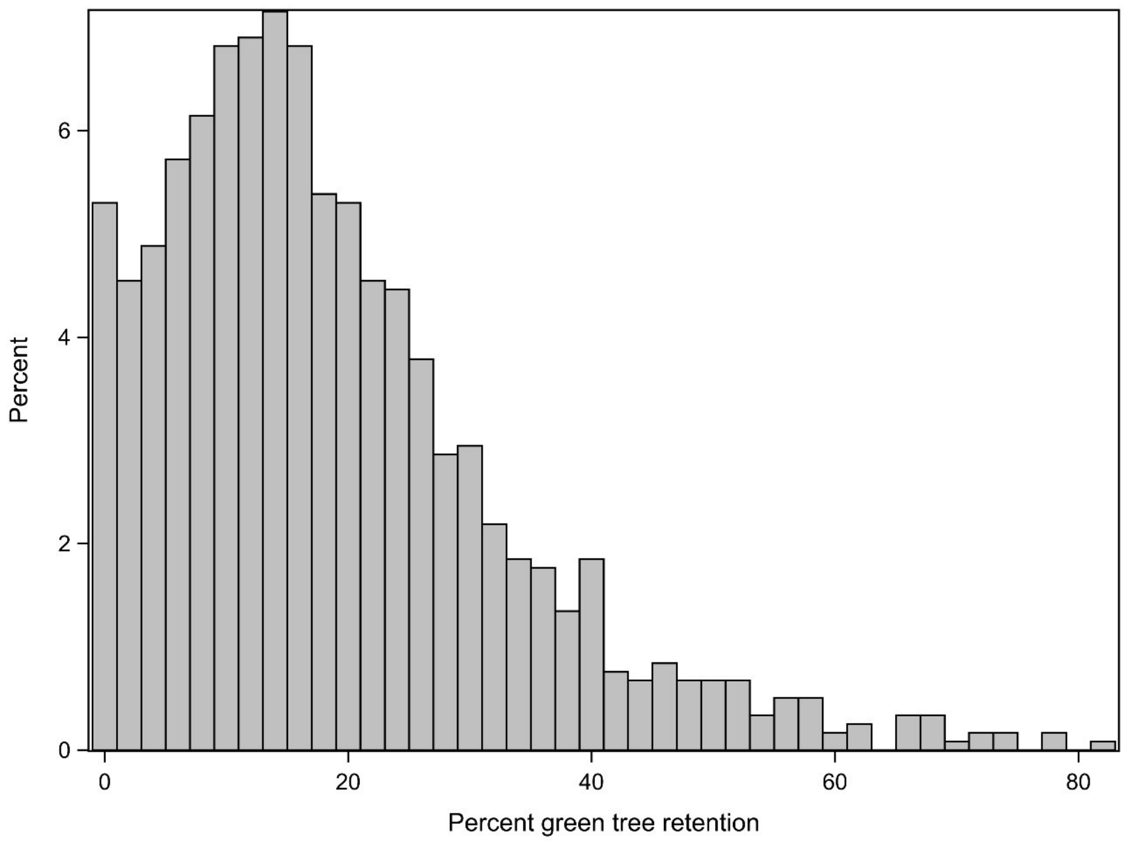

Figure 2.4 Relative frequency histogram of distribution of percent of management unit land area in green tree retention land cover

This relative frequency histogram illustrates the distribution of percent land area in green tree retention land cover classes on 1187 intensively managed pine (*Pinus* spp.) forest management units, established in 2008 (n = 675) and in 2009 (n = 512), and located in the South Central Plains ecoregion of Arkansas, Louisiana, and Texas, USA. Percent green tree retention ranged from 0.0% to 81.4% (mean: 18.6%, sd: 14.2%) and was calculated as the sum of management unit area in retention land cover classes (SMZs, stringers, wet soils areas, and non-riparian retention) / total management unit area · 100.

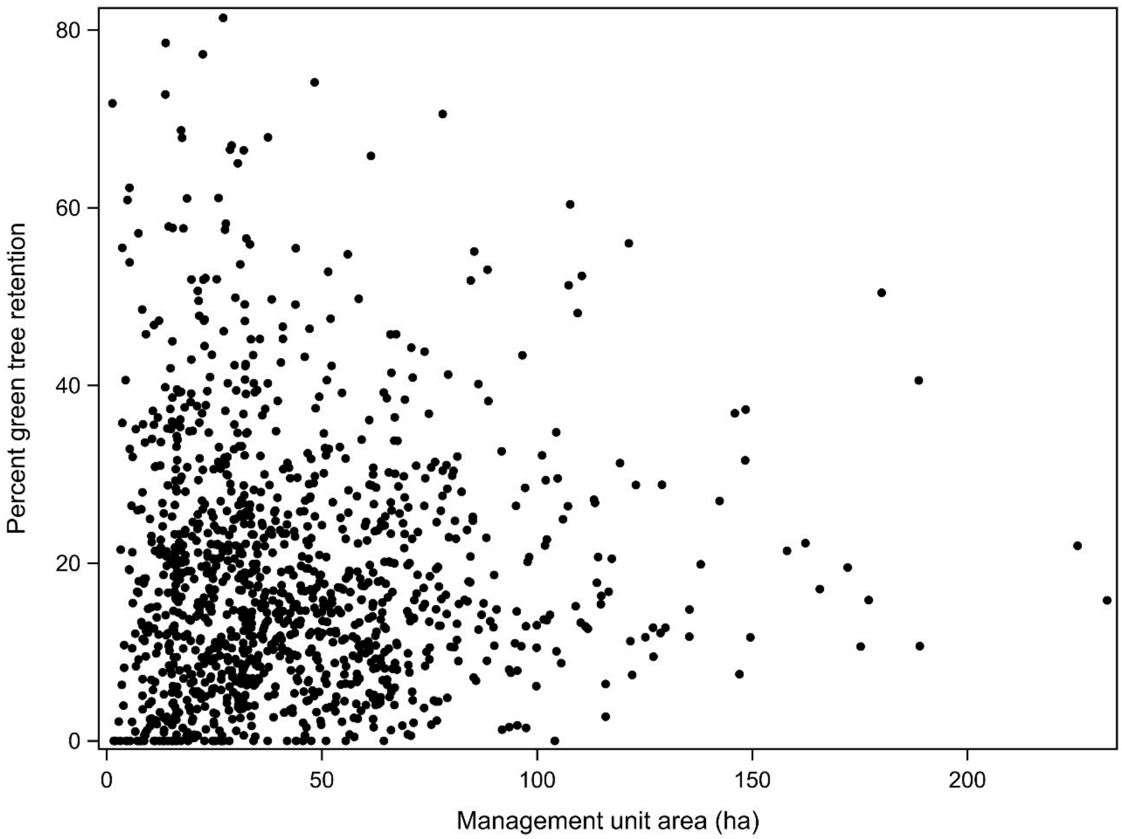

Figure 2.5 Scatterplot illustrating non-correlation of management unit total land area with percent of management unit in green tree retention land cover.

Management unit total land area was not correlated with percent of management unit in green tree retention land cover (r = 0.02; p=0.447) for 1187 pine (*Pinus* spp.) management units established in 2008 (n = 675) and in 2009 (n = 512) on institutional forest lands.

2.8 Literature Cited

Adamczyk, J., and D. Tiede. 2017. ZonalMetrics-a Python toolbox for zonal landscape structure analysis. Computers & Geosciences 99:91-99.

American Forest Foundation. 2015. American Forest Foundation (AFF) 2015-2020 Standards of Sustainability. Available online at www.treefarmsystem.org/view-standards; last accessed Oct. 15, 2016.

American Ornithological Society. 2017. Checklist of North and Middle American Birds. Available online at http://checklist.aou.org/taxa/; Last accessed January 18, 2017.

Arkansas Forestry Commission. 2002. Arkansas forestry best management practices for water quality protection. 58 p. Available online at http://forestry.arkansas.gov/Services/ManageYourForests/Documents/bmpbookrevise.pdf; last accessed on Feb. 16, 2017.

Askins, R.A. 2001. Sustaining biological diversity in early successional communities: The challenge of managing unpopular habitats. Wildlife Society Bulletin 29(2):407-412.

Aubry, K.B., C.B. Halpern, and C.E. Peterson. 2009. Variable-retention harvests in the Pacific Northwest: A review of short-term findings from the DEMO study. Forest Ecology and Management 258:398-408.

Aust, W.M., and C.R. Blinn. 2004. Forestry best management practices for timber harvesting and site preparation in the eastern United States: An overview of water quality and productivity research during the past 20 years (1982–2002). Water, Air and Soil Polution: Focus 4(1):5-36.

Azevedo, J.C., X.B. Wu, M.G. Messina, J.R. Williams, and R.F. Fisher. 2008. The role of the Sustainable Forestry Initiative in forest landscape changes in Texas, USA. P. 273-295 in Patterns and processes in forest landscapes: Multiple use and sustainable management, Lafortezza, R., G. Sanesi, J. Chen, and T.R. Crow (eds.). Springer Science & Business Media.

Bauhus, J., K. Puettmann, and C. Messier. 2009. Silviculture for old-growth attributes. Forest Ecology and Management 258:525-537.

Bechtold, W.A., and P.L. Patterson. 2005. The enhanced forest inventory and analysis program-national sampling design and estimation procedures. Gen. Tech. Rep. SRS-80. Asheville, NC: US Department of Agriculture, Forest Service, Southern Research Station. 85 p.

Belisle, M., and A. Desrochers. 2002. Gap-crossing decisions by forest birds: an empirical basis for parameterizing spatially-explicit, individual-based models. Landscape Ecology 17:219-231.

Benjamin, J.G. 2010. Considerations and recommendations for retaining woody biomass on timber harvest sites in Maine. Maine Agricultural and Forest Experiment Station Miscellaneous Publication 761.

Bielecki, J., J. Ferris, K. Kintigh, M. Koss, D. Kuhr, S. MacKinnon, S. Throop, et al. 2012. Within-Stand Retention Guidance. Michigan Department of Natural Resources Forest Resources Division. Available online at https://www.michigan.gov/documents/dnr/WithinStandRetentionGuidelines-IC4110_175766_7.pdf; last accessed: March 5, 2018.

Bird Studies Canada, and North American Bird Conservation Initiative. 2014. Bird Conservation Regions. Published by Bird Studies Canada on behalf of the North American Bird Conservation Initiative. Available online at http://www.birdscanada.org/research/gislab/index.jsp?targetpg=bcr. Accessed: 31 March 2014.

Blinn, C.R., and M.A. Kilgore. 2001. Riparian management practices: A summary of state guidelines. Journal of Forestry 99(8):11-17.

Boston, K., and P. Bettinger. 2001. The economic impact of green-up constraints in the southeastern United States. Forest Ecology and Management 145(3):191-202.

Brawn, J.D., S.K. Robinson, and F.R. Thompson, III. 2001. The role of disturbance in the ecology and conservation of birds. Annual Review of Ecology and Systematics 32:251-276.

Brockerhoff, E.G., H. Jactel, J.A. Parrotta, C.P. Quine, and J. Sayer. 2008. Plantation forests and biodiversity: Oxymoron or opportunity? Biodiversity and Conservation 17:925-951.

Campbell, T.N., P.D. Jones, S. Demarais, and A.W. Ezell. 2015. Vegetation communities in intensively established loblolly pine plantations at crown closure. Journal of Forestry 113(3):298-307.

Chesser, R.T., R.C. Banks, C. Cicero, J.L. Dunn, A.W. Kratter, I.J. Lovette, P.C. Rasmussen, et al. 2016. Fifth-seventh supplement to the American Ornithologists' Union check-list of North American birds. The Auk 133(3):544-560.

Coulon, A., J.W. Fitzpatrick, R. Bowman, and I.J. Lovette. 2012. Mind the gap: Genetic distance increases with habitat gap size in Florida Scrub Jays. Biology Letters 8:582-585.

Cristan, R., W. Michael Aust, M. Chad Bolding, S.M. Barrett, and J.F. Munsell. 2017. National status of state developed and implemented forestry best management practices for protecting water quality in the United States. Forest Ecology and Management (Article in press) https://doi.org/10.1016/j.foreco.2017.07.002.

Cubbage, F.W. 2004. Costs of forestry best management practices in the south: A review. Water, Air and Soil Pollution: Focus 4(1):131-142.

Cushman, S.A., K. McGarigal, and M.C. Neel. 2008. Parsimony in landscape metrics: Strength, universality, and consistency. Ecological Indicators 8(5):691-703.

Daggupati, P., K.R. Douglas-Mankin, and A.Y. Sheshukov. 2013. Predicting ephemeral gully location and length using topographic index models. Transactions of the American Society of Agricultural and Biological Engineers 56(4):1427-1440.

Daigle, J.J., G.E. Griffith, J.M. Omernik, P.L. Faulkner, R.P. McCulloh, L.R. Handley, L.M. Smith, et al. 2006. Ecoregions of Louisiana (color poster with map, descriptive text, summary tables, and photographs). U.S. Geological Survey: Reston, VA (map scale 1:1,000,000).

Demarais, S., J.P. Verschuyl, G.J. Roloff, D.A. Miller, and T.B. Wigley. 2017. Tamm review: Terrestrial vertebrate biodiversity and intensive forest management in the U.S. Forest Ecology and Management 385:308-330.

Desrochers, A., and S.J. Hannon. 1997. Gap crossing decisions by forest songbirds during the post-fledging period. Conservation Biology 11(5):1204-1210.

Dickson, J.G., R.N. Conner, and J.H. Williamson. 1983. Snag retention increases bird use of a clear-cut. Journal of Wildife Management 47(3):799-804.

Dickson, J.G., J.H. Williamson, R.N. Conner, and B. Ortego. 1995. Streamside zones and breeding birds in eastern Texas. Wildlife Society Bulletin 23(4):750-755.

Elmore, L.W., D.A. Miller, and F.J. Vilella. 2004. Selection of diurnal roosts by red bats (*Lasiurus borealis*) in an intensively managed pine forest in Mississippi. Forest Ecology and Management 199:11-20.

Fink, A.D., F.R. Thompson, III, and A.A. Tudor. 2006. Songbird use of regenerating forest, glade, and edge habitat types. Journal of Wildife Management 70(1):180-188.

Forest Stewardship Council US. 2010. FSC - US Forest Management Standard (v.1.0). Available online at https://us.fsc.org/download.fsc-us-forest-management-standard-v1-0.95.htm. Last accessed Oct. 15 2016.

Franklin, J.F., D.R. Berg, D.A. Thornburgh, and J.C. Tappeiner. 1997. Alternative silvicultural approaches to timber harvesting: Variable retention harvest systems. P. 111-136 in Creating a Forestry for the 21st Century: The Science of Ecosystem Management, Kohm, K.A., and J.F. Franklin (eds.). Island Press, Washington, D.C.

Greene, R.E., R.B. Iglay, K.O. Evans, K.V. Miller, T.B. Wigley, and S.K. Riffell. 2016. A meta-analysis of biodiversity responses to management of southeastern pine forests - Opportunities for open pine conservation. Forestry Ecology and Management 360:30-39.

Griffith, G., S. Bryce, J.M. Omernik, and A. Rogers. 2007. Ecoregions of Texas: Project Report to Texas Commission on Environmental Quality. Available online at ftp://newftp.epa.gov/EPADataCommons/ORD/Ecoregions/tx/TXeco_Jan08_v8_Cmprsd.pdf. Last accessed Feb. 16, 2017.125 p.

Gustafsson, L., S.C. Baker, J. Bauhus, W.J. Beese, A. Brodie, J. Kouki, D.B. Lindenmayer, et al. 2012. Retention forestry to maintain multifunctional forests: A world perspective. BioScience 62(7):633-645.

Hein, C.D., S.B. Castleberry, and K.V. Miller. 2009. Site-occupancy of bats in relation to forested corridors. Forest Ecology and Management 257(4):1200-1207.

Homyack, J.A., B.J. Paxton, M.D. Wilson, B.D. Watts, and D.A. Miller. 2011. Snags and cavity-nesting birds within intensively managed pine stands in eastern North Carolina, USA. Southern Journal of Applied Forestry 35(3):148-154.

Jones, P.D., S. Demarais, and A.W. Ezell. 2012. Successional trajectory of loblolly pine (*Pinus taeda*) plantations established using intensive management in Southern Mississippi, USA. Forest Ecology and Management 265:116-123.

Jones, P.D., S.L. Edwards, and S. Demarais. 2009a. Vegetation community responses to different establishment regimes in loblolly pine (*Pinus taeda*) plantations in southern MS, USA. Forest Ecology and Management 257:553-560.

Jones, P.D., B.B. Hanberry, and S. Demarais. 2009b. Stand-level wildlife habitat features and biodiversity in Southern pine forests: A review. Journal of Forestry 107:398-404.

Jones, P.D., B.B. Hanberry, and S. Demarais. 2010. Managing the Southern pine forest - Retained wetland interface for wildlife diversity: Research priorities. Wetlands 30:381-391.

Kalcounis-Rueppell, M.C., K.M. Briones, J.A. Homyack, R. Petric, M.R. Marshall, and D.A. Miller. 2013. Hard forest edges act as conduits, not filters, for bats. Wildlife Society Bulletin 37(3):571-576.

King, D.I., C.R. Griffin, and R.M. DeGraaf. 1997. Effect of clearcut borders on distribution and abundance of forest birds in northern New Hampshire. The Wilson Bulletin 109(2):239-245.

King, D.I., and S. Schlossberg. 2014. Synthesis of the conservation value of the early-successional stage in forests of eastern North America. Forest Ecology and Management 324:186-195.

Krementz, D.G., and J.S. Christie. 2000. Clearcut stand size and scrub-successional bird assemblies. The Auk 117(4):913-924.

Lakel, W.A., W.M. Aust, C.A. Dolloff, and P.D. Keyser. 2015. Residual timber values within piedmont streamside management zones of different widths and harvest levels. Forest Science 61(1):197-204.

Lee, P., C. Smyth, and S. Boutin. 2004. Quantitative review of riparian buffer width guidelines from Canada and the United States. Journal of Environmental Management 70:165-180.

Lehnen, S.E., and A.D. Rodewald. 2009. Investigating area-sensitivity in shrubland birds: Responses to patch size in a forested landscape. Forest Ecology and Management 257(11):2308-2316.

Lehnen, S.E., and A.D. Rodewald. 2013. Daily and seasonal movements of a shrubland-obligate breeder in relation to mature forest edge habitat. Forest Ecology and Management 305:112-119.

Lickwar, P., C. Hickman, and F.W. Cubbage. 1992. Costs of protecting water quality during harvesting on private forestlands in the Southeast. Southern Journal of Applied Forestry 16(1):13-20.

Lindenmayer, D.B., and J.F. Franklin. 2002. Conserving Forest Biodiversity: A Comprehensive Multiscaled Approach. Island Press, Washington, D.C. 352 p.

Little, E.L., Jr. 1979. Checklist of United States Trees (Native and Naturalized). US Department of Agriculture Agriculture Handbook no. 541. 375 p.

Louisiana Department of Agriculture and Forestry. 1999. Recommended Forestry Best Management Practices for Louisiana. 83 p. Available online at http://www.ldaf.state.la.us/wp-content/uploads/2014/04/BMP.pdf. Last accessed Feb. 16, 2017.

Luck, M., and J. Wu. 2002. A gradient analysis of urban landscape pattern: A case study from the Phoenix metropolitan region, Arizona, USA. Landscape Ecology 17:327-339.

Machtans, C.S., M.-A. Villard, and S.J. Hannon. 1996. Use of riparian buffer strips as movement corridors by forest birds. Conservation Biology 10(5):1366-1379.

Marshall, M.R., J.A. DeCecco, A.B. WIlliams, G.A. Gale, and R.J. Cooper. 2003. Use of regenerating clearcuts by late-successional bird species and their young during the post-fledging period. Forest Ecology and Management 183(1-3):127-135.

McDermott, M.E., and P.B. Wood. 2011. Post-breeding bird responses to canopy tree retention, stand size, and edge in regenerating Appalachian hardwood stands. Forest Ecology and Management 262:547-554.

McGarigal, K. 2015. Fragstats Help. University of Massachusetts, Amherst. Available online at: http://www.umass.edu/landeco/research/fragstats/documents/fragstats.help.4.2.pdf . Last accessed 16 May 2017.

McGarigal, K., S.A. Cushman, and E. Ene. 2012. FRAGSTATS v4: Spatial Pattern Analysis Program for Categorical and Continuous Maps. University of Massachusetts, Amherst. Available online at: www.umass.edu/landeco/research/fragstats/fragstats.html; last accessed 16 May 2017.

Miller, D.A., R.E. Thill, M.A. Melchiors, T.B. Wigley, and P.A. Tappe. 2004. Small mammal communities of streamside management zones in intensively managed pine forests of Arkansas. Forest Ecology and Management 203(1-3):381-393.

Miller, D.A., T.B. Wigley, and K.V. Miller. 2009. Managed forests and conservation of terrestrial biodiversity in the southern United States. Journal of Forestry 107(4):197-203.

National Association of State Foresters. 2015. Protecting water quality through state forestry best management practices. National Association of State Foresters. 8 p. Available online at https://stateforesters.org/sites/default/files/issues-and-policies-document-attachments/Protecting_Water_Quality_through_State_Forestry_BMPs_FINAL.pdf; last accessed March 5, 2018.

Omernik, J.M. 1987. Ecoregions of the conterminous United States. Annals of the Association of American Geographers 77(1):118-125.

Oswalt, S.N., W.B. Smith, P.D. Miles, and S.A. Pugh. 2014. Forest Resources of the United States, 2012: A Technical Document Supporting the Forest Service 2015 Update of the RPA Assessment. Gen. Tech. Rep. WO-91. Washington, DC: U.S. Department of Agriculture, Forest Service, Washington Office. 218 p.

Owens, F.L., P.C. Stouffer, M.J. Chamberlain, and D.A. Miller. 2014. Early-successional breeding bird communities in intensively managed pine plantations: Influence of vegetation succession but not site preparations. Southeastern Naturalist 13(3):423-443.

Parrish, M.C., S. Demarais, A.W. Ezell, T.B. Wigley, P.D. Jones, and S.K. Riffell. 2017a. Retained vegetation density of streamside management zones and stringers in Southern intensively managed pine forests. Forest Ecology and Management 397:89-96.

Parrish, M.C., S. Demarais, T.B. Wigley, P.D. Jones, A. Ezell, and S.K. Riffell. 2017b. Breeding bird communities associated with land cover in intensively managed pine forests of the southeastern U.S. Forest Ecology and Management 406:112-124.

Parrish, M.C., S. Demarais, T.B. Wigley, S.K. Riffell, A. Ezell, and P.D. Jones. 2018. Operational green tree retention and land cover patterns in intensively managed pine forest landscapes of the southeastern U.S. Forest Science 00(00):1-13.

Perry, R.W., T.B. Wigley, M.A. Melchiors, R.E. Thill, P.A. Tappe, and D.A. Miller. 2011. Width of riparian buffer and structure of adjacent plantations influence occupancy of conservation priority birds. Biodiversity and Conservation 20:625-642.

Ribe, R. 1999. Regeneration harvests versus clearcuts: Public views of the acceptability and aesthetics of Northwest Forest Plan harvests. Northwest Science 73(Special Issue):102-117.

Robertson, O.J., and J.Q. Radford. 2009. Gap-crossing decisions of forest birds in a fragmented landscape. Austral Ecology 34:435-446.

Rodewald, A.D., and A.C. Vitz. 2005. Edge- and area-sensitivity of shrubland birds. Journal of Wildife Management 69(2):681-688.

Rodewald, P.G., and M.C. Brittingham. 2004. Stopover habitats of landbirds during fall: Use of edge-dominated and early-successional forests. The Auk 121(4):1040-1055.

Rosenvald, R., and A. Lõhmus. 2008. For what, when, and where is green-tree retention better than clear-cutting? A review of the biodiversity aspects. Forest Ecology and Management 255(1):1-15.

Rudolph, D.C., and J.G. Dickson. 1990. Streamside zone width and amphibian and reptile abundance. The Southwestern Naturalist 35(4):472-476.

Schlossberg, S., and D.I. King. 2008. Are shrubland birds edge specialists? Ecological Applications 18(6):1325-1330.

Shake, C.S., C.E. Moorman, J.D. Riddle, and M.R. Burchell, II. 2012. Influence of patch size and shape on occupancy by shrubland birds. The Condor 114(2):268-278.

Stamps, J.A., M. Buechner, and V.V. Krishnan. 1987. The effects of edge permeability and habitat geometry on emigration from patches of habitat. The American Naturalist 129(4):533-552.

Sustainable Forestry Initiative. 2015. SFI 2015-2019 standard and rules: Standards, rules for label use, procedures, and guidance. Available online at www.sfiprogram.org/files/pdf/2015-2019-standardsandrules-web-lr-pdf/; last accessed on Oct 15. 2016.

Texas Forest Service. 2004. Texas Forestry Best Management Practices. 109 p. Available online at http://txforestservice.tamu.edu/uploadedfiles/sustainable/bmp/bmpbookindd.pdf; last accessed March 5, 2018.

Texas Forestry Association, and Texas Forest Service. 2010. Texas Forestry Best Management Practices. 113 p.

Trani, M.K., R.T. Brooks, T.L. Schmidt, V.A. Rudis, and C.M. Gabbard. 2001. Patterns and trends of early successional forests in the eastern United States. Wildlife Society Bulletin 29(2):413-424.

US Environmental Protection Agency. 2011. Ecoregions of the United States (Spatial Data). 1:250,000 scale. U.S. EPA Office of Research & Development (ORD) - National Health and Environmental Effects Research Laboratory (NHEERL), Corvallis, OR. Available online at: ftp://ftp.epa.gov/wed/ecoregions/us/Eco_Level_III_US.zip; last accessed Aug. 26, 2011.

USDA Forest Service, and USDI Bureau of Land Management. 1994. Final Supplemental Environmental Impact Statement on Management of Habitat for Late-Successional and Old-Growth Forest Related Species Within the Range of the Northern Spotted Owl.

Vitz, A.C., and A.D. Rodewald. 2006. Can regenerating clearcuts benefit mature-forest songbirds? An examination of post-breeding ecology. Biological Conservation 127:477-486.

Warrington, B.M., W.M. Aust, S.M. Barrett, W.M. Ford, M.C. Bolding, and C.A. Dolloff. 2016. Potential effects of forestry operations and associated best management practices on riparian wildlife species in the southeastern United States. P. 106-112 in Proc. of the 18th biennial Southern silvicultural research conference. Schweitzer, C.J., W.K. Clatterbuck and C. M. Oswalt (eds.). e–Gen. Tech. Rep. SRS–212. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 614 p.

Wilkin, E., F. Jiménez Nava, and G. Griffith. 2011. North American Terrestrial Ecoregions - Level III. Commission for Environmental Cooperation, Montreal, Canada. 149 p.

Wilson, M.D., and B.D. Watts. 2000. Breeding bird communities in pine plantations on the coastal plain of North Carolina. The Chat 64(1):1-14.

Wisconsin Department of Natural Resources. 2011. Wisconsin forest management guidelines: Appendix A: Tree Marking and Retention Guidelines Rev 2011. Available online at http://dnr.wi.gov/topic/ForestManagement/guidelines.html; last accessed March 5, 2018. 6 p.

Woodman, J.N., and F.W. Cubbage. 1994. Potential costs of mandatory best management practices in Georgia. P. 309-322 *in* Proceedings of the 24th Annual Southern Forest Economics Workshop., Newman, D.H., and M.E. Aronow (eds.), Savannah, Georgia, USA.

Woods, A.J., T.L. Foti, S.S. Chapman, J.M. Omernik, J.A. Wise, E.O. Murray, W.L. Prior, et al. 2004. Ecoregions of Arkansas (Color Poster with Map, Descriptive Text, Summary Tables, and Photographs). U.S. Geological Survey: Reston, VA (map scale 1:1,000,000).

Yin, R., and R.A. Sedjo. 2001. Is this the age of intensive management? A study of loblolly pine on Georgia's Piedmont. Journal of Forestry 99(1):10-17.

CHAPTER III

RETAINED VEGETATION DENSITY OF STREAMSIDE MANAGEMENT ZONES AND STRINGERS IN SOUTHERN INTENSIVELY MANAGED PINE FORESTS

3.1 Author's Note

This chapter was originally published in *Forest Ecology and Management* as 'Retained vegetation density of streamside management zones and stringers in Southern intensively managed pine forests' (Parrish et al. 2017). It is reproduced here with only minor editorial changes.

3.2 Abstract

In the southern U.S. (hereafter, South), institutional forest owners engaged in forest certification programs often retain unharvested or less-intensively harvested vegetation when clearcut harvesting intensively managed pine (*Pinus* spp.) forests (IMPF). As a result, IMPF landscapes consist of regenerating forests and associated retained streamside management zones (SMZs), stringers (buffer strips along ephemeral streams), and other forest types and structural classes. Although studies in the South have documented plant and animal communities associated with SMZs, there is a lack of information about stringers. To improve understanding of the potential for stringers to contribute to biodiversity-related management objectives, I characterized stem density and vegetation cover in SMZs and stringers associated with 60 IMPF management units

(MUs) in the South Central Plains ecoregion of Arkansas and Louisiana, USA. Snag and log density, midstory pine density, understory deciduous cover, and ground cover were not statistically different in stringers and SMZs; however, overstory (pine and deciduous) and midstory (deciduous) tree density was significantly lower in stringers than in SMZs, and understory pine density was significantly greater in SMZs.

3.3 Introduction

Global demand for wood products has led to intensified forest management practices in some regions of the world, including the southern U.S. (hereafter, South; Demarais *et al.*, 2017). There, IMPF increased from 7,280 km^2 to 129,000 km^2 during 1950-2000, and is expected to supply 50% of the softwood removals from all forests in the South (Fox et al. 2007; Smith et al. 2009). Because IMPFs are extensive in area and actively managed, they have significant potential to contribute to biodiversity maintenance and enhancement (Miller et al. 2009; Demarais et al. 2017). Forest landowners engaged in forest certification programs are encouraged to consider conservation goals in their management plans (Miller et al. 2009; Forest Stewardship Council US 2010; American Forest Foundation 2015; Sustainable Forestry Initiative 2015).

Retention of structural features (e.g., snags and green trees) is a common strategy to enhance stand-level structural diversity (Franklin et al. 1997; Gustafsson et al. 2012) after forest harvest. Green trees can be retained within harvest units as scattered individual stems, aggregated groups of stems, or as larger patches. In the South, IMPFs are commonly harvested using clearcutting with green trees retained in SMZs and other aggregated areas. As a result, IMPF management units [A] ('MUs') in the early-establishment phase are dominated by three cover types: (1) RCC, (2) SMZs buffering perennial or intermittent streams, and (3) stringers[B], vegetated buffers retained along ephemeral drains. A recent study in the SCP ecoregion found that 1188 commercial MUs averaged 43.5 ha, with mean land cover of 80.5% RCC, 14.0% SMZ, and 3.3% stringer (Parrish et al. 2018). By aggregating structural features along riparian corridors in SMZs and stringers, managers are able to provide stream buffering and water quality protection services (Lindenmayer and Franklin 2002; Seconges et al. 2013), reduce interference with regeneration and subsequent stand treatments, and provide ecosystem services, including: (a) refugia and biological legacies for species associated with mature forests; (b) structural complexity that potentially benefits recolonizing organisms; and (c) enhanced landscape connectivity (Franklin et al. 1997; Lindenmayer and Franklin 2002; Aubry et al. 2009).

Early-successional forest, which is an important component of IMPF MUs in the pre-canopy closure stage, is an ephemeral resource that is declining in the eastern U.S. (Wilson and Watts 2000; Trani et al. 2001; King and Schlossberg 2014). However, retained structural elements, such as SMZs and stringers, also can play a role in conservation of biodiversity on actively managed landscapes by enhancing structural heterogeneity (Poulsen 2002; Culbert et al. 2013). Most studies assessing biodiversity-

---

[A] I defined management units (MUs) as contiguous areas containing one or more intensively managed pine forest ('IMPF') stands that are uniform in age, that were harvested and reestablished as a unified cohort, plus any associated areas of green tree retention (e.g., streamside management zones or stringers).
[B] Stringers are vegetated buffer zones located around ephemeral drains and gullies. Watercourses associated with stringers usually feed into larger waterways, which are themselves often protected by SMZs. State best management practices guidelines in the southern US generally address stringers separately from SMZs.

SMZ relationships in IMPFs have examined associations with buffer width rather than vertical-axis vegetation structural characteristics (Rudolph and Dickson 1990; Thurmond et al. 1995; Kilgo et al. 1998; Constantine et al. 2004; Lee et al. 2004; Miller et al. 2004; Perry et al. 2011). Dickson (1989), working in eastern Texas, USA IMPFs, noted positive associations between SMZ vegetation structural complexity and width (i.e., narrow SMZs had pronounced understory layers with minimal canopy or shade, while wider SMZs had open understories and shady, closed canopies), and reported greater small mammal captures in narrow SMZs and larger numbers of reptiles, amphibians, and tree squirrels in wider SMZs. In North Carolina, USA, agricultural SMZ structural diversity positively affected bird community diversity (Smith et al. 2008).

Stringers are commonly retained in many IMPF MUs in the South. A regional-scale study in the SCP found stringers were retained on 69% of MUs (Parrish et al. 2018). In many cases, stringers are connected to SMZs and thereby can provide an extension of mature forest cover and habitat for mature forest-associated species into adjacent RCCs. Although I was unable to locate studies addressing biodiversity relationships with stringers in IMPFs, studies in other forest types in the South suggest that some biodiversity responses to stringers may be similar to those in SMZs. In southeastern Kentucky, USA, stringers around ephemeral streams in harvested deciduous forests maintained larger salamander populations than non-buffered ephemeral streams (Maigret et al. 2014). However, the degree of structural similarity between stringers and SMZs has hitherto remained unclear. Thus, my objective was to characterize and compare the patch-scale stem density, understory deciduous (shrub) cover, and ground cover of the two dominant green tree retention cover types occurring on IMPF MUs (i.e., SMZs and stringers) in the South. Of particular interest was the degree of similarity of stem density between stringers and SMZs, as I hypothesized that stringers provide benefits to MU structural diversity that are complementary to those provided by SMZs (Radabaugh et al. 2004).

3.4 Methods

3.4.1 Study Area

The South Central Plains ecoregion (Fig. 3.1) extends south from central Arkansas and southeastern Oklahoma into northwestern Louisiana and eastern Texas (Omernik 1987; US Environmental Protection Agency 2011). Historically, the SCP was dominated by longleaf (*P. palustris* Mill.) pine forests and savannas in the south and mixed shortleaf pine (*P. echinata* Mill.)-hardwood forest in the north (Little 1979; Wilkin et al. 2011); the region represents the western edge of the southern pine belt and modern land cover still consists of around 67% woodlands and less than 20% cropland, with extensive area in commercial IMPF (Daigle et al. 2006; Griffith et al. 2007; Wilkin et al. 2011). Winters are mild and summers are hot, particularly in southern parts of the SCP, with mean low and high temperatures of 1º C and 34º C, respectively (Griffith et al. 2007). Annual mean precipitation ranges from 105 cm to 170 cm across the ecoregion (Wilkin et al. 2011). Perennial streams are abundant and mostly of low to moderate gradient (Wilkin et al. 2011), and many smaller streams experience limited or no flow during hot summer months (Woods et al. 2004).

Locally-intensive pine silvicultural activity in the South Central Plains (SCP) ecoregion of Arkansas and Louisiana suggests it as an ideal area for studying green tree retention in IMPF. In 2012, 85.0 Mha (39.4%) of the South was in timberland, 16.1 Mha of which was planted loblolly-shortleaf pine (*Pinus taeda* L.[C] - *P. echinata* Mill.) or longleaf-slash pine (*P. palustris* Mill. - *P. elliottii* Engelm.) (Oswalt et al. 2014). Regional timber production typically occurs on planted timberland that is intensively managed using techniques such as chemical and/or mechanical site preparation, fertilization, and thinning (Jones et al. 2009a; Demarais et al. 2017). In Arkansas and Louisiana, managers typically retain SMZs adjacent to intermittent streams, perennial streams, ponds, and lakes as recommended by state BMP guidelines (Louisiana Department of Agriculture and Forestry 1999; Arkansas Forestry Commission 2002).

3.4.2 Site Selection

I selected 60 IMPF MUs in the SCP ecoregion of Arkansas and Louisiana from an initial set of 1188 MUs[D] in the SCP established in 2008 or 2009 that would be 3 years post-establishment when sampled during 1 June to 13 July 2011 (n = 35) or during 7 May to 19 June 2012 (n = 25). Land cover on each of the 1188 MUs was previously classified as part of a related study, using recent, high-resolution aerial imagery, and a dataset containing class-level spatial metrics was derived for each MU landscape using FRAGSTATS [E] software (Parrish et al. 2018). From that dataset, I extracted the following spatial metrics relating to my 60 study sites for use in characterizing the dimensions and contributions to stand composition made by SMZs and stringers: (1) total area in land cover class (CA), (2) class percentage of MU landscape (CPL), (3) patch density per 100 ha (CPD), (4) class mean patch area within MU (CMPA), and (5) mean

[C] Tree taxonomic nomenclature follows Little, Jr. (1979).
[D] MUs were managed by Forest Capital Partners, LLC; Plum Creek Timber Company, Inc.; Potlatch Corporation; Resource Management Service, LLC, and Weyerhaeuser Timber Companies.
[E] FRAGSTATS 4.1 software was used to derive spatial metrics; equations may be found in the program documentation (McGarigal et al. 2012).

radius of gyration (CMGYR; a measure of linear extent of patch). Criteria for sampling consideration were: (1) MU contained both SMZ and stringer cover (as determined by aerial photo interpretation); (2) MU ranked in the central 80% of the range of MU area (Parrish et al. 2018); and (3) MU was within 60 km of one of 6 logistical hubs located across the SCP (Fig. 3.1). I ranked the remaining sites by percent green tree retention and stratified them into 6 equally-sized groups, from which I randomly selected sample sites. I established two 30-m-radius plots, 200 m or more apart, in both SMZ and stringer cover on each MU; however, because area in stringers and SMZs was limited in some MUs, two plots could not be established in stringers on 10 MUs and in SMZs in 18 MUs.

3.4.3 Vegetation Surveys

I characterized vegetation in each sampling station using 4 height strata: overstory (uppermost canopy layer), midstory (3 m to underside of canopy), understory (1 m to 3 m), and ground cover (less than 1 m). I counted overstory stems of deciduous trees, pines, and snags (dead standing timber at > 45 º angle to the ground) within a 16.1-m-radius circular plot established at the station center (Fig. 3.2). I counted midstory stems (deciduous trees, pines, and snags) within five 5.1-m-radius plots, one located at the station center, and the others located at four points located 30 m from the station center at every 90°.

I characterized ground cover and understory vegetation at 12 sampling points located 10 m, 20 m, and 30 m from the station center along four 30-m transects originating from the station center every 90°. (Fig. 3.2). I counted understory stems of live pines, snags, and logs (dead timber > 10 cm laying at < 45º to the ground) within 3.6-m-radius plots established at each of the 12 sampling points. I estimated understory deciduous percent cover using an elevated nadir (downward-facing) photographic point-intercept (NP) method (Booth et al. 2006) at 217 sampling stations located in 42 of my 60 MU study sites. At each station, I captured 12 13-megapixel digital images depicting nadir views of understory vegetation cover (approximately 15.1 m^2 of ground area) from a height of 3.9 m above the forest floor (Fig. 3.2). Using program SamplePoint[F] (Booth et al. 2006), I overlaid each photograph with a 12x12 virtual grid for classification. I visually assessed vegetation cover at each of the 144 grid intersection points and classified these to generate estimates of patch-level vegetation cover characteristics at survey stations.

I also estimated percent ground cover at 12 points per station (Fig. 3.2) using a 20 cm x 50 cm Daubenmire frame (Daubenmire 1959). I estimated percent ground cover in four overlapping classes: (a) grasses and grass-like plants; (b) forbs; (c) woody plants (including pines) and canes; and (d) vines, using Daubenmire's (1959) classes (0% to 5%; 5% to 25%; 25% to 50%; 50% to 75%; 75% to 95%; and 95% to 100%). I assigned the mid-point of the range class to each sampling point.

3.4.4 Statistical Analyses

I fit Poisson and negative binomial distributions to stem count data (overstory, midstory, and understory classes) for SMZ and stringer cover types to make a preliminary determination of which distribution better fit these data. The negative binomial distribution fit the data best in each case, according to estimates of Akaike's information

[F] Nadir imagery analysis for this paper was conducted using SamplePoint software ver. 1.55. Samplepoint was developed by Berryman Consulting in cooperation with the USDA Agricultural Research Service and the USDI Bureau of Land Management. Available online at : http://www.samplepoint.org/; last accessed Jan. 17, 2017.

criterion corrected for finite sample sizes (AIC_C) that I calculated using the GENMOD procedure in SAS[G]. Using the GLIMMIX procedure in SAS, with a log link function, I created generalized linear mixed models for non-normal responses using negative binomial distributions. I used my models to test for an effect of cover type on stem count, to estimate least squares (LS) means and associated 95% confidence intervals, and to calculate contrasts between cover type pairings used to test for significant differences in LS means between cover types ($\alpha = 0.05$).

I normalized understory deciduous percent cover data using a log transformation and I back-transformed the statistics before reporting them. I used the MIXED procedure in SAS to create a mixed-effects linear model that I used to test for an effect of cover type on understory deciduous cover, to estimate LS means and associated 95% confidence intervals, and to calculate contrasts between cover type pairings to assess differences in mean percent cover between cover types.

Using the UNIVARIATE procedure in SAS, I tested untransformed and log-transformed ground cover data in each class to determine whether a transformation improved data normality. Based on my findings, I log-transformed ground cover for forbs, woody plants, and vines, but did not transform grasses. I fitted mixed effects linear models to the ground cover data using the MIXED procedure in SAS to test for cover type effects. Using my models, I obtained least squares means and 95% confidence intervals by cover type for each ground cover class and I tested hypotheses ($\alpha = 0.05$) comparing LS means between SMZ and stringer cover types within each ground cover

---

[G] The data analysis for this paper was generated using SAS/STAT software, Version 9.4 of the SAS System for Windows, copyright © 2002-2012 SAS Institute Inc. SAS and all other SAS Institute Inc. product or service names are registered trademarks or trademarks of SAS Institute Inc., Cary, NC, USA.

class. Using the CORR procedure in SAS, I examined all pairwise combinations of vegetation metrics for correlations within cover types.

3.5 Results

SMZ cover accounted for 13.8% (SD 10.9) or 7.4 ha (SD 6.5) of the average MU, while cover in stringers was 30% that of SMZs (Table 3.1). Mean patch area was nearly nine times greater in SMZs than in stringers. Mean patch density per 100 ha was greater in stringers (6.9 *patches; SD* 6.0) than in SMZs (3.8 *patches; SD* 3.3). Mean radius of gyration was much larger in SMZs (162.1 m; SD 107.6) compared with stringers (47.4 m; SD 20.9).

I detected cover type effects on stem density in four classes: overstory deciduous trees, overstory pines, midstory deciduous trees, and understory pines (Table 3.2). At the overstory level, SMZ deciduous tree stem density (52.3 stems·ha^{-1}) was approximately 1.5 times that of stringers ($P = 0.0028$) and SMZ pine stem density (20.1 stems·ha^{-1}) was twice that of stringers ($P = 0.0193$). Midstory deciduous tree stem density was greater in SMZs (285.4 stems·ha^{-1}) than in stringers ($P = 0.0364$); however, stringer stem density was only about 15% below the level observed in SMZs. Understory pine stem density was approximately three times greater in stringers (216.9 stems·ha^{-1}) than in SMZs ($P < 0.0001$).

No cover type effect was observed on stem density of snags (of any strata), midstory pines, or understory logs (Table 3.2). Snags made up a greater proportion of overstory stems in stringers (1 in 14 stems was a snag) compared with SMZs (1 in 20 stems) due to overall lower live stem density in stringers. But, overstory snag density in both cover types was < 4 stems·ha^{-1}. In both SMZs and stringers, the midstory was dominated (90 to 92% of total stems) by deciduous trees (285.5 stems·ha^{-1} and 239.8 stems·ha^{-1}, respectively), with pines and snags accounting for the balance of midstory stems. I did not observe an effect of cover type on percent cover for understory deciduous vegetation (surveyed on 17 MUs in 2011 and 25 MUs in 2012) or for any ground cover classes (Table 3.3). Vegetative ground cover in SMZs and stringers was dominated by grasses (approximately 34% cover), with forbs, woody plants, and vines each accounting for 5% to 11% cover.

Most vegetation metrics were weakly correlated ($-0.5 > r < 0.5$). In SMZs, overstory deciduous stem density was negatively correlated with understory deciduous percent cover ($r = -0.55$; $p = 0.001$), and I observed positive correlations between overstory pine stem density and midstory pine stem density ($r = 0.78$; $p < 0.001$) and between understory log and snag density ($r = 0.70$; $p < 0.001$). In stringers, I observed positive correlations between woody percent ground cover and understory deciduous percent cover ($r = 0.51$; $p = 0.006$), understory snag and log stem density ($r = 0.67$; $p < 0.001$), understory snag and midstory snag density ($r = 0.59$; $p < 0.001$), understory log and midstory snag density ($r = 0.54$; $p < 0.001$).

3.6 Discussion

3.6.1 Vegetation on Recently Harvested Sites

My sample set of 60 MU landscapes was fairly similar with regard to SMZ and stringer components to the larger set of 1188 MUs from which I selected my study sites, but most spatial metrics in my study sites (CA, CPL, CPD, CMPA) tended to average about 20% greater than the general population (Parrish et al. 2018). This is likely an artifact of my sampling criteria, which required that MUs selected for sampling include both SMZs and stringers (thus excluding sites missing one or more cover types). Because >90% of the 1188 MUs I selected sites from contained SMZs and/or stringers (Parrish et al. 2018), I believe my results are still generally applicable within that population.

The large geographic extent of this study coupled with its exploration of relative stem density and vegetation cover in the two dominant green tree retention cover types of IMPF distinguishes it from previous, smaller-scale studies carried out in the South. Dickson and Williamson (1988) surveyed six pine plantation SMZs in the coastal plain of East Texas, reporting overstory pine stem density (14 stems·ha^{-1}) and understory pine stem density (21 stems·ha^{-1}) somewhat lower than I observed. In nine SMZs located in IMPF stands established in the interior flatwoods region of Mississippi, canopy tree stem density (pines and deciduous combined) averaged 53 stems·ha^{-1}, which falls within my 95% confidence interval for overstory tree stem density (Burk et al. 1990).

I found substantial similarities between SMZs and stringers: snag and log density, midstory pine density, understory deciduous percent cover, and ground cover. Although midstory deciduous stem density was greater in SMZs than in stringers, the difference was not strongly pronounced. The key differences between the two cover types occurred in overstory and understory pine stem density and in overstory deciduous stem density. Because stringer vegetation characteristics are similar to those of SMZs, stringers essentially augment the total amount of retained mature forest structure on the landscape, increasing it from a mean of 14.0% to 17.4% of land cover (Parrish et al. 2018).

Forest harvesting effects may contribute to my observed lower overstory pine stem density in stringers compared with SMZs. Stringers are typically narrow and small in area compared to more spatially extensive SMZs (Parrish et al. 2018), potentially allowing easier access to merchantable pine stems in stringers during clearcut harvest in IMPF MUs and resulting in partial harvest of stringers. In east Texas, narrower riparian buffers were associated with more complete partial harvest operations, and narrow riparian buffers were harvested of nearly all merchantable pines, while wider riparian areas were subject to minimal partial harvest levels (Burns et al. 1999). In Louisiana, BMP guidelines allow harvesting in stringers, provided vegetative filtration is not compromised, and harvesting in SMZs may take place provided SMZ objectives (e.g., adequate canopy and soil erosion prevention) are not compromised (Louisiana Department of Agriculture and Forestry 1999). Arkansas BMP guidelines allow all harvesting systems in stringers, but require ≥ 4.6 m^2 of retained basal area in SMZs (Arkansas Forestry Commission 2002). Due to concerns about access and BMP recommendations, operators may harvest fewer stems in SMZs compared to stringers, or may refrain from cutting in SMZs altogether. The observed low midstory pine stem density, which did not differ between SMZs or stringers, was likely a consequence of the poor growing conditions for shade intolerant pines in the shaded buffers prior to harvest. I attribute the greater midstory and overstory deciduous stem density in SMZs versus stringers to management history (i.e., less intensive harvesting operations in SMZs relative to stringers). I observed a strong, positive correlation between overstory and midstory pine stem density in SMZs, and minimal correlation between them in stringers, which is consistent with long-term, consistent growth conditions in SMZs and substantial removal of merchantable trees in stringers.

Understory pine stem density was greater in stringers than in SMZs, which I attribute to the typically narrow, elongated shape of stringers (Parrish et al. 2018) resulting in greater area in close proximity to seed sources (i.e., adjacent, pre-harvest pine stands) and greater post-harvest light penetration into the stringers than into SMZs. Sixteen to 25-year-old loblolly pines generally disperse seeds from 50 m to 140 m (Williams et al. 2006). Narrower riparian buffer strips are associated with greater levels of photosynthetically active radiation, which decreases rapidly with distance from the clearcut edge (Kiffney et al. 2003). I observed a negative correlation between overstory deciduous stem density and understory deciduous percent cover in SMZs, but not stringers; however, although I expected that greater light penetration in narrower buffer strips (Kiffney et al. 2003) would promote greater shrub cover in stringers, understory deciduous cover did not differ between stringers and SMZS, Differences in shrub cover between cover types may not yet have had sufficient time to be realized, given the young age (three years post-harvest) of the regenerating stand and considering the likelihood that low vegetation in stringers probably experienced harvest impacts requiring a recovery period before continued expansion.

I did not detect differences in snag density between stringers and SMZs at any vertical strata, perhaps because my sites were only 3 years post-harvest and potential differences in snag attrition rates between SMZs and stringers had not expressed themselves. I attribute the lower ratio of live trees to snags in the overstory of stringers versus SMZs to greater harvest impacts stemming from more complete partial harvest operations in stringers. Snag and log densities appeared to be moderately correlated in SMZ understories and in stringer understories and midstories, suggesting similarity in recruitment processes. Snag retention on pine stands in the Ouachita mountains of Arkansas and Oklahoma was greater in wider riparian buffers, while buffers surrounded by clearcuts appeared to experience a greater snag attrition rate compared to buffers surrounded by intact forest (Liechty and Guldin 2009).

Over time, stringers may experience greater snag attrition from wind throw due to their narrow configuration of stringers (Liechty and Guldin 2009) Moorman et al. (1999) suggested that managers actively recruit snags on one- to two-year intervals to maintain and augment snag-derived biodiversity benefits. Wildlife also respond to snag characteristics such as diameter at breast height (DBH), height, and species more than to snag density (Moorman et al. 1999). Fallen log density did not differ between stringers or SMZs I sampled, but the degree of log microhabitat similarity between cover types is unclear. Stringers tend to be higher in elevation, less shaded, and drier than the SMZs which they drain into (pers. obs.), suggesting that microclimate and decay rates of downed logs are likely to differ between cover types. I recommend further investigation into the physical and spatial characteristics of snags and coarse woody debris that optimize biodiversity benefits on the post-harvest landscape.

Stringers and SMZs did not differ in ground cover, and were not densely vegetated, with grasses dominating other ground cover classes. Heavy ground litter conditions may favor grasses over forbs (Shelton 1994), and shade and dense litter inhibit growth of forbs, woody ground cover, and vines (Harrington and Edwards 1999). Although stringers, being narrow, probably experienced greater light penetration for the prior three years than did SMZs, it did not appear to be sufficient to differentiate percent ground cover between the two cover types.

3.6.2 Stringer Functionality on Young IMPF Landscapes

From a biodiversity standpoint, stringers may enhance connectivity between and access to SMZs and RCCs. Vegetative structure and cover on stringers I sampled were similar in most respects to SMZs and the spatial configuration of stringers may expand access into RCCs from SMZs (Parrish et al. 2018). Stringers potentially function as forested corridors for some species that normally limit their travel distance across RCCs (Desrochers and Hannon 1997; Robertson and Radford 2009). Limited evidence suggests some bird species used forested corridors for natal dispersion and adult movement more frequently than adjacent clearcuts in Alberta, Canada (Machtans et al. 1996). Whereas wider SMZs on harvested landscape may provide sufficient habitat to serve as breeding territories for some mature forest species, narrower stringers may be used mostly as movement corridors, perches, and foraging surfaces (Shirley 2006). Retained snags and green trees in RCCs were positively associated with avian biodiversity (Johnson and Landers 1982; Dickson et al. 1983; Hanberry 2007; Jones et al. 2009b), suggesting that the structural diversity provided by stringers may enhance the wildlife value of RCCs.

Effects of riparian buffer width likely vary by taxa and/or functional guilds. For example, on IMPFs in east Texas, narrow riparian buffers hosted abundant populations of small mammals and shrubland-associated birds, but reptiles, amphibians, and forest interior birds were scarce in narrow buffers and more abundant in wider ones (Dickson and Williamson 1988; Rudolph and Dickson 1990; Dickson et al. 1995). However, species typically assigned to 'interior forest' groups sometimes utilize RCC and edges during certain life stages. For example, many mature forest breeding birds used clearcuts and edges during the post-fledging phase of breeding (Pagen et al. 2000; Marshall et al.

2003; Vitz and Rodewald 2006). Thus, stringers may increase landscape permeability and provide cover, foraging opportunities, and other resources to interior forest species accessing RCC. Because of their configuration (Parrish et al. 2018) and vegetative structure, stringers may facilitate access to the RCC interior by SMZ-based forest species that may otherwise be reluctant to cross large distances across of open early successional habitat. However, forest edges associated with SMZs and stringers abutting RCCs may be associated with greater nest predation and parasitism (Haegen and Degraaf 1996; Benson et al. 2013), and greater avian, mammalian, and ophidian predator abundance (Chalfoun et al. 2002; Cox et al. 2012), though there is evidence that the intensity of these edge effects varies according to interactions with landscape factors such as overall forest cover (Cox et al. 2012) or patch size (Benson et al. 2013) and prey species life history characteristics (Flaspohler et al. 2001).

Stringers may also enhance biodiversity in RCCs by creating greater vertical heterogeneity (Suárez-Seoane et al. 2002) by providing structure used as perches, snags, cover, and foraging substrates in areas typically dominated by ground cover, understory vegetation, and young pine trees; even small patches of retained green trees may convey these benefits (Lindenmayer et al. 2015). On IMPFs in east Texas, early-successional bird species used forest edges for foraging and singing perches and avian abundance and diversity were approximately three times greater in forest strips adjacent to clearcuts (Strelke and Dickson 1980). Small mammal relative abundance and species richness were lower in SMZs adjacent to closed canopy IMPFs in Arkansas (Miller et al. 2004), and age of adjacent stands was an important predictor of bat occupancy on South Carolina IMPFs (Hein et al. 2009). Thus, I hypothesize that avian communities in SMZs and stringers may also respond to structure of adjacent forests. Landscape permeability for early-successional species would likely not be negatively affected by the presence of stringers, which are typically narrow (frequently with a clear line of sight through to the other side) and short. Furthermore, RCC had very low interpatch traversal distances when they were subdivided by retention patches (Parrish et al. 2018). Stringers represent a unique cover type, similar in many ways to SMZs, but providing a somewhat more open overstory.

3.6.3 Management Implications

Although stringers are in many ways structurally similar to SMZs, managers may wish to consider their distinct contributions to biodiversity, which suggest their classification separate from SMZs (Burns et al. 1999; Radabaugh et al. 2004). Compared with SMZs, stringers may be smaller, narrower, and more 'scrubby' and open, perhaps leading managers to disregard their ecological role on the management unit. However, even small riparian zones offer potential biodiversity support (Lindenmayer et al. 2015). Furthermore, the vegetation structure provided by stringers complements that of SMZs and RCCs, potentially increasing the value to biodiversity of the latter two cover types. Land owners should consider the potential contributions to their site's biodiversity that may result from retaining stringers and SMZs, where appropriate.

Table 3.1 Class-level summary statistics describing spatial attributes of streamside management zone (SMZ) and stringer (STR) land cover patches within *n*=60 intensively managed pine (*Pinus spp.*) forest (IMPF) management unit (MU) landscapes established 2008-2009 in the South Central Plains ecoregion of Arkansas and Louisiana

	Land cover class	
Spatial attribute [a]	**SMZ**	**STR**
CA (*ha*)		
$\bar{x} \pm sd$	7.4 ± 6.5	2.1 ± 2.0
range	[0.0, 25.6]	[0.0, 10.3]
total	442.9	124.1
CPL (*%*)		
$\bar{x} \pm sd$	13.8 ± 10.9	4.0 ± 3.6
range	[0.0, 46.0]	[0.0, 21.0]
CPD ($patches \cdot (100 \cdot ha)^{-1}$)		
$\bar{x} \pm sd$	3.8 ± 3.3	6.9 ± 6.0
range	[0.0, 20.7]	[0.0, 27.4]
CMPA (*ha*)		
$\bar{x} \pm sd$	5.3 ± 6.0	0.6 ± 0.4
range	[0.0, 25.6]	[0.0, 2.7]
CMGYR (*m*)		
$\bar{x} \pm sd$	162.1 ± 107.6	47.4 ± 20.9
range	[30.1, 597.0]	[15.4, 108.4]

Note: Metrics were derived using FRAGSTATS version 4.1.
[a] Spatial attributes: CA: class area per MU (ha), including total area over all sampled MUs; CPL: class percentage of landscape; CPD: class patch density (patches·$(100 \cdot ha)^{-1}$); CMPA: class mean patch area (ha); CMGYR: class mean radius of gyration (m).

Table 3.2 Least squares mean estimates [a] of stem density (stems·ha^{-1}) with upper and lower 95% C.I. by vegetation class and vertical strata in two cover types (streamside management zone, 'SMZ'; stringer, 'STR') occurring on 60 three-year-old intensively managed pine (*Pinus* spp.) forest management units located in the South Central Plains of Arkansas and Louisiana.

Strata[c]	**Veg. Class**	**Stem Density (95% C.I.)**		**Contrast P-values [b]**
		SMZ	**STR**	**SMZ-STR**
Overstory	DV [d]	52.3 (40.6, 67.3)	35.3 (26.9, 46.3)	0.0028
	Pine	20.1 (12.3, 33.0)	10.0 (5.8, 17.3)	0.0193
	Snag	3.9 (2.6, 6.0)	3.5 (2.2, 5.5)	0.6459
Midstory	DV	285.4 (250.1, 325.8)	239.8 (208.1, 276.5)	0.0364
	Pine	10.3 (6.3, 16.7)	11.6 (7.0, 19.1)	0.6866
	Snag	14.2 (10.0, 20.1)	15.0 (10.5, 21.4)	0.7273
Understory	Pine	71.7 (52, 98.8)	216.9 (158.7, 296.5)	<0.0001
	Snag	10.5 (6.4, 17.3)	8.5 (5.1, 14.3)	0.4039
	Log	7.3 (4.4, 12.3)	6.0 (3.5, 10.2)	0.2584

[a] Estimates obtained by generalized linear model using negative binomial distribution using the GLIMMIX procedure in SAS with a log link function.
[b] Contrast pairing P-values indicate whether stem density distribution differs among cover types.
[c] Height strata: overstory (canopy trees); midstory (3 m to underside of canopy); understory (1 to 3 m).
[d] 'DV' = deciduous vegetation.

Table 3.3 Least-squares estimates of cover (%) and contrasts among cover types (streamside management zone, 'SMZ'; stringer, 'STR') by vertical strata and vegetation class for 60 three-year-old intensively managed pine (*Pinus* spp.) forest management units ('MUs') in the South Central Plains of Arkansas and Louisiana.

Strata [b]	**Veg. Class** [c]	**Percent cover (95% C.I.)**				**Contrast P-values** [a]
		SMZ		**STR**		**SMZ-STR**
Understory	DV [d]	5.0	(3.7, 6.8)	5.2	(3.8, 7.0)	0.8744
Ground Cover	Grasses	31.6	(26.0, 37.2)	37.2	(31.4, 43.1)	0.1569
	Forbs	4.8	(4.0, 5.7)	5.0	(4.2, 5.9)	0.7391
	Woody	9.6	(7.9, 11.8)	10.8	(8.8, 13.2)	0.3485
	Vines	7.2	(6.0, 8.5)	7.1	(6.0, 8.5)	0.9718

[a] Contrast P-values indicate whether percent cover distribution differs among between cover types.
[b] Vegetative strata height ranges: understory (1 to 3 m); ground cover (below 1 m).
[c] Vegetative growth form classes 'DV', 'forbs', 'woody', and 'vines' were log transformed for analysis.
[d] Deciduous vegetation ('DV') percent cover was estimated at n=42 MU sites using nadir photographic data collection.

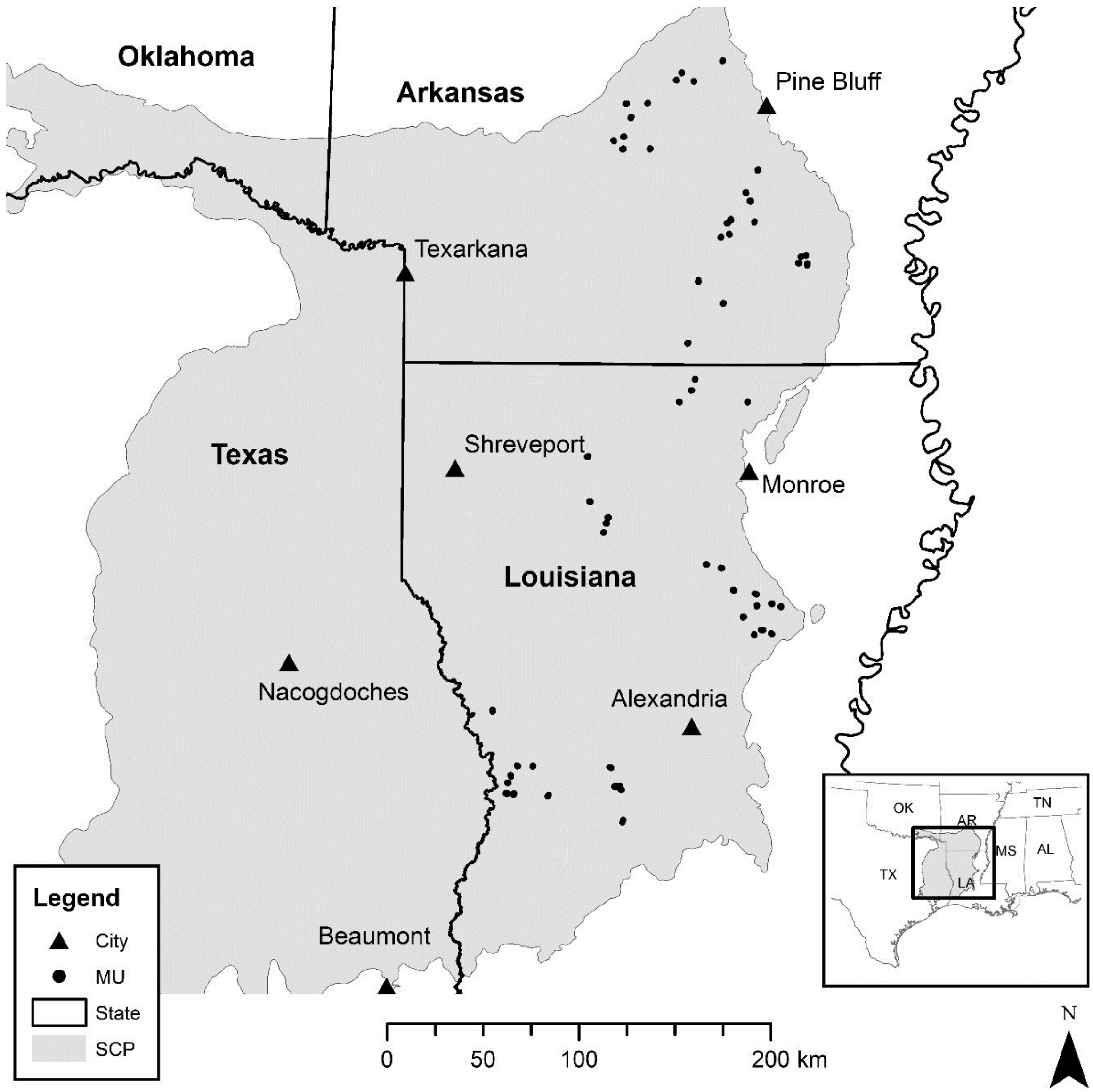

Figure 3.1 Locations of study sites in the South Central Plains (SCP) ecoregion

Sixty intensively managed pine (*Pinus* spp.) forest management units (MUs) were selected for sampling. MUs were reestablished in 2008 or 2009. Each MU was surveyed at three years post-establishment. The SCP roughly overlaps the North American Bird Conservation Initiative bird conservation region *West Gulf Coastal Plain / Ouachitas*.

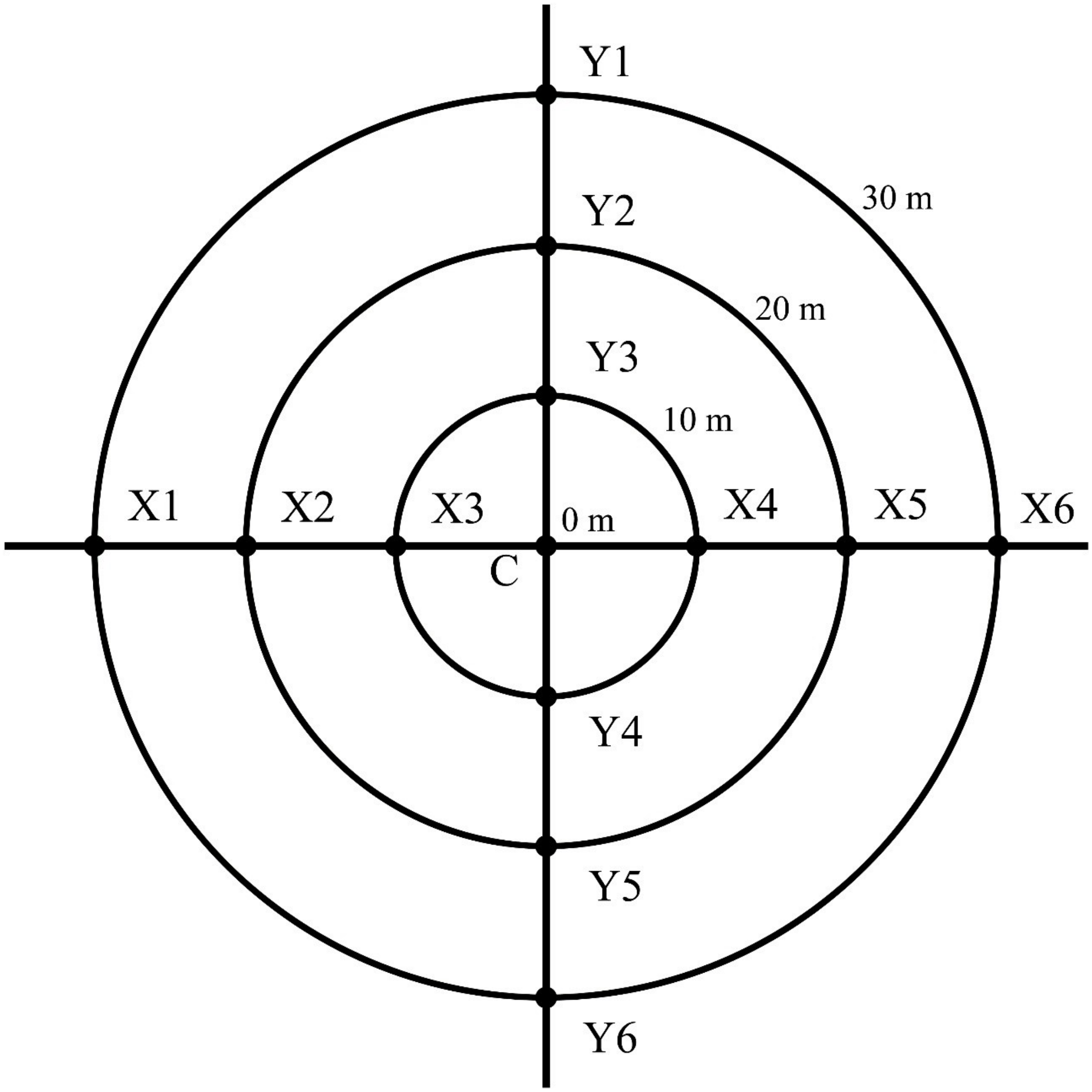

Figure 3.2 Schematic of survey station vegetation subsampling points

Concentric distance rings are shown at 10 meter intervals from the station center, C. Ground cover and understory vegetation were surveyed at points X1:X6 and Y1:Y6. Midstory vegetation was surveyed at points X1, X6, C, Y1, and Y6. Overstory vegetation was surveyed at point C.

3.7 Literature Cited

American Forest Foundation. 2015. American Forest Foundation (AFF) 2015-2020 Standards of Sustainability. Available online at www.treefarmsystem.org/view-standards; last accessed Oct. 15, 2016.

Arkansas Forestry Commission. 2002. Arkansas forestry best management practices for water quality protection. 58 p. Available online at http://forestry.arkansas.gov/Services/ManageYourForests/Documents/bmpbookrevise.pdf; last accessed on Feb. 16, 2017.

Aubry, K.B., C.B. Halpern, and C.E. Peterson. 2009. Variable-retention harvests in the Pacific Northwest: A review of short-term findings from the DEMO study. Forest Ecology and Management 258:398-408.

Benson, T.J., S.J. Chiavacci, and M.P. Ward. 2013. Patch size and edge proximity are useful predictors of brood parasitism but not nest survival of grassland birds. Ecological Applications 23(4):879-887.

Booth, D.T., S.E. Cox, and R.D. Berryman. 2006. Point sampling digital imagery with 'SamplePoint'. Environmental Monitoring and Assessment 123:97-108.

Burk, J.D., G.A. Hurst, D.R. Smith, B.D. Leopold, and J.G. Dickson. 1990. Wild Turkey use of streamside management zones in loblolly pine plantations. National Wild Turkey Symposium 6:84-89.

Burns, G.A., R.M. Whiting, Jr., G.M. LeGrande, and J.G. Dickson. 1999. Wildlife linkages: volumes and values of residual timber in riparian zones in eastern Texas. Forest Ecology and Management 114:321-327.

Chalfoun, A.D., M.J. Ratnaswamy, and F.R. Thompson, III. 2002. Songbird nest predators in forest-pasture edge and forest interior in a fragmented landscape. Ecological Applications 12(3):858-867.

Constantine, N.L., T.A. Campbell, W.M. Baughman, T.B. Harrington, B.R. Chapman, and K.V. Miller. 2004. Effects of clearcutting with corridor retention on abundance, richness, and diversity of small mammals in the Coastal Plain of South Carolina, USA. Forest Ecology and Management 202:293-300.

Cox, W.A., F.R. Thompson, III, and J. Faaborg. 2012. Landscape forest cover and edge effects on songbird nest predation vary by nest predator. Landscape Ecology 27(5):659-669.

Culbert, P.D., V.C. Radeloff, C.H. Flather, J.M. Kellndorfer, C.D. Rittenhouse, and A.M. Pidgeon. 2013. The influence of vertical and horizontal habitat structure on nationwide patterns of avian biodiversity. The Auk 130(4):656-665.

Daigle, J.J., G.E. Griffith, J.M. Omernik, P.L. Faulkner, R.P. McCulloh, L.R. Handley, L.M. Smith, et al. 2006. Ecoregions of Louisiana (color poster with map, descriptive text, summary tables, and photographs). U.S. Geological Survey: Reston, VA (map scale 1:1,000,000).

Daubenmire, R.F. 1959. A canopy-coverage method of vegetational analysis. Northwest Science 33(1):43-64.

Demarais, S., J.P. Verschuyl, G.J. Roloff, D.A. Miller, and T.B. Wigley. 2017. Tamm review: Terrestrial vertebrate biodiversity and intensive forest management in the U.S. Forest Ecology and Management 385:308-330.

Desrochers, A., and S.J. Hannon. 1997. Gap crossing decisions by forest songbirds during the post-fledging period. Conservation Biology 11(5):1204-1210.

Dickson, J.G. 1989. Streamside zones and wildlife in Southern U. S. forests. P. 131-133 in Practical approaches to riparian resource management: An educational workshop: May 8-11, 1989, Billings, Montana, Parts 89-4351, Gresswell, R.E., B.A. Barton, and J.L. Kershner (eds.). U. S. Bureau of Land Management, Billings, MT, USA.

Dickson, J.G., R.N. Conner, and J.H. Williamson. 1983. Snag retention increases bird use of a clear-cut. Journal of Wildife Management 47(3):799-804.

Dickson, J.G., and J.H. Williamson. 1988. Small mammals in streamside management zones in pine plantations. Szaro RC, Severson KE, Patton DR (coordinators) Management of Amphibians, Reptiles, and Small Mammals in North America. USDA Forest Service General Technical Report RM-166 (1988): 375-378.

Dickson, J.G., J.H. Williamson, R.N. Conner, and B. Ortego. 1995. Streamside zones and breeding birds in eastern Texas. Wildlife Society Bulletin 23(4):750-755.

Flaspohler, D.J., S.A. Temple, and R.N. Rosenfield. 2001. Species-specific Edge Effects on Nest Success and Breeding Bird Density in a Forested Landscape. Ecological Applications 11(1):32-46.

Forest Stewardship Council US. 2010. FSC - US Forest Management Standard (v.1.0). Available online at https://us.fsc.org/download.fsc-us-forest-management-standard-v1-0.95.htm. Last accessed Oct. 15 2016.

Fox, T.R., E.J. Jokela, and H.L. Allen. 2007. The development of pine plantation silviculture in the southern United States. Journal of Forestry 105(7):337-347.

Franklin, J.F., D.R. Berg, D.A. Thornburgh, and J.C. Tappeiner. 1997. Alternative silvicultural approaches to timber harvesting: Variable retention harvest systems. P. 111-136 in Creating a Forestry for the 21st Century: The Science of Ecosystem Management, Kohm, K.A., and J.F. Franklin (eds.). Island Press, Washington, D.C.

Griffith, G., S. Bryce, J.M. Omernik, and A. Rogers. 2007. Ecoregions of Texas: Project Report to Texas Commission on Environmental Quality. Available online at ftp://newftp.epa.gov/EPADataCommons/ORD/Ecoregions/tx/TXeco_Jan08_v8_Cmprsd.pdf. Last accessed Feb. 16, 2017.125 p.

Gustafsson, L., S.C. Baker, J. Bauhus, W.J. Beese, A. Brodie, J. Kouki, D.B. Lindenmayer, et al. 2012. Retention forestry to maintain multifunctional forests: A world perspective. BioScience 62(7):633-645.

Haegen, W.M.V., and R.M. Degraaf. 1996. Predation on artificial nests in forested riparian buffer strips. The Journal of Wildlife Management 60(3):542-550.

Hanberry, B.B. 2007. Birds and small mammals, intensively established pine plantations, and landscape metrics of the Coastal Plain. Ph.D. dissertation. Department of Wildlife and Fisheries, Mississippi State University, Mississippi State, MS. Advisor: S. Demarais. 140 p.

Harrington, T.B., and M.B. Edwards. 1999. Understory vegetation, resource availability, and litterfall responses to pine thinning and woody vegetation control in longleaf pine plantations. Canadian Journal of Forest Research 29:1055-1064.

Hein, C.D., S.B. Castleberry, and K.V. Miller. 2009. Site-occupancy of bats in relation to forested corridors. Forest Ecology and Management 257(4):1200-1207.

Johnson, A.S., and J.L. Landers. 1982. Habitat relationships of summer resident birds in slash pine flatwoods. The Journal of Wildlife Management 46(2):416-428.

Jones, P.D., S.L. Edwards, and S. Demarais. 2009a. Vegetation community responses to different establishment regimes in loblolly pine (*Pinus taeda*) plantations in southern MS, USA. Forest Ecology and Management 257:553-560.

Jones, P.D., B.B. Hanberry, and S. Demarais. 2009b. Stand-level wildlife habitat features and biodiversity in Southern pine forests: A review. Journal of Forestry 107:398-404.

Kiffney, P.M., J.S. Richardson, and J.P. Bull. 2003. Responses of periphyton and insects to experimental manipulation of riparian buffer width along forest streams. Journal of Applied Ecology 40:1060-1076.

Kilgo, J.C., R.A. Sargent, B.R. Chapman, and K.V. Miller. 1998. Effect of stand width and adjacent habitat on breeding bird communities in bottomland hardwoods. Journal of Wildife Management 62(1):72-83.

King, D.I., and S. Schlossberg. 2014. Synthesis of the conservation value of the early-successional stage in forests of eastern North America. Forest Ecology and Management 324:186-195.

Lee, P., C. Smyth, and S. Boutin. 2004. Quantitative review of riparian buffer width guidelines from Canada and the United States. Journal of Environmental Management 70:165-180.

Liechty, H.O., and J.M. Guldin. 2009. Structure and composition of streamside management zones following reproduction cutting in shortleaf pine stands. Forest Ecology and Management 258:1407-1413.

Lindenmayer, D.B., and J.F. Franklin. 2002. Conserving Forest Biodiversity: A Comprehensive Multiscaled Approach. Island Press, Washington, D.C. 352 p.

Lindenmayer, D.B., R.J. Hobbs, and D. Salt. 2015. Plantation forests and biodiversity conservation. Australian Forestry 66(1):62-66.

Little, E.L., Jr. 1979. Checklist of United States Trees (Native and Naturalized). US Department of Agriculture Agriculture Handbook no. 541. 375 p.

Louisiana Department of Agriculture and Forestry. 1999. Recommended Forestry Best Management Practices for Louisiana. 83 p. Available online at http://www.ldaf.state.la.us/wp-content/uploads/2014/04/BMP.pdf. Last accessed Feb. 16, 2017.

Machtans, C.S., M.-A. Villard, and S.J. Hannon. 1996. Use of riparian buffer strips as movement corridors by forest birds. Conservation Biology 10(5):1366-1379.

Maigret, T.A., J.J. Cox, D.r. Schneider, C.D. Barton, S.J. Price, and J.L. Larkin. 2014. Effect of timber harvest within streamside management zones on salamander populations in ephemeral streams of southeastern Kentucky. Forest Ecology and Management 324:46-51.

Marshall, M.R., J.A. DeCecco, A.B. WIlliams, G.A. Gale, and R.J. Cooper. 2003. Use of regenerating clearcuts by late-successional bird species and their young during the post-fledging period. Forest Ecology and Management 183(1-3):127-135.

McGarigal, K., S.A. Cushman, and E. Ene. 2012. FRAGSTATS v4: Spatial Pattern Analysis Program for Categorical and Continuous Maps. University of Massachusetts, Amherst. Available online at: www.umass.edu/landeco/research/fragstats/fragstats.html; last accessed 16 May 2017.

Miller, D.A., R.E. Thill, M.A. Melchiors, T.B. Wigley, and P.A. Tappe. 2004. Small mammal communities of streamside management zones in intensively managed pine forests of Arkansas. Forest Ecology and Management 203(1-3):381-393.

Miller, D.A., T.B. Wigley, and K.V. Miller. 2009. Managed forests and conservation of terrestrial biodiversity in the southern United States. Journal of Forestry 107(4):197-203.

Moorman, C.E., K.R. Russell, G.R. Sabin, and D.C. Guynn, Jr. 1999. Snag dynamics and cavity occurrence in the South Carolina Piedmont. Forest Ecology and Management 118:37-48.

Omernik, J.M. 1987. Ecoregions of the conterminous United States. Annals of the Association of American Geographers 77(1):118-125.

Oswalt, S.N., W.B. Smith, P.D. Miles, and S.A. Pugh. 2014. Forest Resources of the United States, 2012: A Technical Document Supporting the Forest Service 2015 Update of the RPA Assessment. Gen. Tech. Rep. WO-91. Washington, DC: U.S. Department of Agriculture, Forest Service, Washington Office. 218 p.

Pagen, R.W., F.R. Thompson, III, and D.E. Burhans. 2000. Breeding and post-breeding habitat use by forest migrant songbirds in the Missouri Ozarks. The Condor 102(4):738-747.

Parrish, M.C., S. Demarais, A.W. Ezell, T.B. Wigley, P.D. Jones, and S.K. Riffell. 2017. Retained vegetation density of streamside management zones and stringers in Southern intensively managed pine forests. Forest Ecology and Management 397:89-96.

Parrish, M.C., S. Demarais, T.B. Wigley, S.K. Riffell, A. Ezell, and P.D. Jones. 2018. Operational green tree retention and land cover patterns in intensively managed pine forest landscapes of the southeastern U.S. Forest Science 00(00):1-13.

Perry, R.W., T.B. Wigley, M.A. Melchiors, R.E. Thill, P.A. Tappe, and D.A. Miller. 2011. Width of riparian buffer and structure of adjacent plantations influence occupancy of conservation priority birds. Biodiversity and Conservation 20:625-642.

Poulsen, B.O. 2002. Avian richness and abundance in temperate Danish forests: Tree variables important to birds and their conservation. Biodiversity and Conservation 11:1551-1566.

Radabaugh, D.K., H.O. Leichty, and J.M. Guldin. 2004. A comparison of vegetation within and outside riparian areas bordering ephemeral streams in the Ouachita Mountains. Gen. Tech. Rep. SRS-74. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. pp. 21-25.

Robertson, O.J., and J.Q. Radford. 2009. Gap-crossing decisions of forest birds in a fragmented landscape. Austral Ecology 34:435-446.

Rudolph, D.C., and J.G. Dickson. 1990. Streamside zone width and amphibian and reptile abundance. The Southwestern Naturalist 35(4):472-476.

Seconges, J.M., W.M. Aust, J.R. Seiler, C.A. Dolloff, and W.A. Lakel. 2013. Streamside management zones affect movement of silvicultural nitrogen and phosphorus fertilizers to piedmont streams. Southern Journal of Applied Forestry 37(1):26-35.

Shelton, M.G. 1994. Effects of the amount and composition of the forest floor on the emergence and early establishment of loblolly pine seedlings. Canadian Journal of Forest Research 25:480-486.

Shirley, S.M. 2006. Movement of forest birds across river and clearcut edges of varying riparian buffer strip widths. Forest Ecology and Management 223:190-199.

Smith, T.A., D.L. Osmond, C.E. Moorman, J.M. Stucky, and J.W. Gilliam. 2008. Effect of vegetation managment on bird habitat in riparian buffer zones. Southeastern Naturalist 7(2):277-288.

Smith, W.B., tech coord., P.D. Miles, data coord., C.H. Perry, map coord., and S.A. Pugh, Data CD coord. 2009. Forest Resources of the United States, 2007. U.S. Department of Agriculture, Forest Service, Washington Office, Washington D.C., USA. Gen. Tech. Rep. WO-78. 336 p.

Strelke, W., and J.G. Dickson. 1980. Effect of forest clear-cut edge on breeding birds in east Texas. The Journal of Wildlife Management 44(3):559-567.

Suárez-Seoane, S., P.E. Osborne, and J. Baudry. 2002. Responses of birds of different biogeographic origins and habitat requirements to agricultural land abandonment in northern Spain. Biological Conservation 105:333-344.

Sustainable Forestry Initiative. 2015. SFI 2015-2019 standard and rules: Standards, rules for label use, procedures, and guidance. Available online at www.sfiprogram.org/files/pdf/2015-2019-standardsandrules-web-lr-pdf/; last accessed on Oct 15. 2016.

Thurmond, D.P., K.V. Miller, and T.G. Harris. 1995. Effect of streamside management zone width on avifauna communities. Southern Journal of Applied Forestry 19(4):166-169.

Trani, M.K., R.T. Brooks, T.L. Schmidt, V.A. Rudis, and C.M. Gabbard. 2001. Patterns and trends of early successional forests in the eastern United States. Wildlife Society Bulletin 29(2):413-424.

US Environmental Protection Agency. 2011. Ecoregions of the United States (Spatial Data). 1:250,000 scale. U.S. EPA Office of Research & Development (ORD) - National Health and Environmental Effects Research Laboratory (NHEERL), Corvallis, OR. Available online at: ftp://ftp.epa.gov/wed/ecoregions/us/Eco_Level_III_US.zip; last accessed Aug. 26, 2011.

Vitz, A.C., and A.D. Rodewald. 2006. Can regenerating clearcuts benefit mature-forest songbirds? An examination of post-breeding ecology. Biological Conservation 127:477-486.

Wilkin, E., F. Jiménez Nava, and G. Griffith. 2011. North American Terrestrial Ecoregions - Level III. Commission for Environmental Cooperation, Montreal, Canada. 149 p.

Williams, C.G., S.L. LaDeau, R. Oren, and G.G. Katul. 2006. Modeling seed dispersal distances: Implications for transgenic *Pinus taeda*. Ecological Applications 16(1):117-124.

Wilson, M.D., and B.D. Watts. 2000. Breeding bird communities in pine plantations on the coastal plain of North Carolina. The Chat 64(1):1-14.

Woods, A.J., T.L. Foti, S.S. Chapman, J.M. Omernik, J.A. Wise, E.O. Murray, W.L. Prior, et al. 2004. Ecoregions of Arkansas (Color Poster with Map, Descriptive Text, Summary Tables, and Photographs). U.S. Geological Survey: Reston, VA (map scale 1:1,000,000).

CHAPTER IV

BREEDING BIRD COMMUNITIES ASSOCIATED WITH LAND COVER IN INTENSIVELY MANAGED PINE FORESTS OF THE SOUTHERN UNITED STATES

4.1 Author's Note

This chapter was originally published in *Forest Ecology and Management* as 'Breeding bird communities associated with land cover in intensively managed pine forests of the southeastern U.S.' (Parrish et al. 2017b). It is reproduced here with only minor editorial changes.

4.2 Abstract

Intensively managed pine (*Pinus* spp.) forests (IMPFs) in the southern U.S. are often harvested by clearcutting in conjunction with green tree retention (i.e., retention of unharvested or less-intensively harvested trees and other vegetation), which is thought to promote structural diversity and to benefit wildlife. Management units in IMPFs thus primarily consist of regenerating pine clearcuts (RCCs) plus retained cover in streamside management zones (SMZs: vegetative buffers along perennial and intermittent streams) and/or stringers (forested buffers along ephemeral drains). To understand relationships between retained structures and avian communities, I documented and compared species diversity of breeding bird species and avian guilds in three-year-old RCCs and associated SMZs and stringers on 60 South Central Plains IMPF management units within the South

Central Plains ecoregion of the southern U.S. I detected 5617 individuals of 60 species. Eight species were considered "common birds in steep decline," one of which was frequently detected (Prairie Warbler, *Setophaga discolor*). Three of my 15 most frequently detected species were identified by Partners in Flight as "warranting management attention" for "moderate or high regional declines," and 19 other species were listed as "warranting management attention" but were uncommonly detected. Forty-two of 44 species documented in RCCs were also observed accessing retention cover areas. SMZs and stringers comprised an average of 17.4% of management unit area, but 27% of species were detected solely in retention cover types. There was an 84% species overlap between SMZs and stringers. Stringers augmented SMZ contributions to site avian diversity by hosting forest specialist guilds. Diversity of early-successional specialists was similar between stringers and RCCs, suggesting stringers also enhanced RCC contributions to site bird diversity. Furthermore, I detected several species only within stringers. Green tree retention land cover contributed to stand scale avian diversity disproportionately to its area, and in particular, stringer cover appeared to enhance the value to avifaunal species diversity of RCC and SMZ patches.

4.3 Introduction

In 2012, timberland [a] made up 39.4% (85.0 Mha) of land cover in the United States Southern region, where approximately one third of timberland was either loblolly-shortleaf pine (*Pinus taeda* L. - *P. echinata* Mill. [b]) or longleaf-slash pine (*P. palustris*

[a] Timberland: Forestland that is producing or capable of annual production in excess of 1.4 $m^3 \bullet ha^{-1}$ of industrial wood in natural stands, and is not withdrawn from timber utilization by statute or administrative regulation.
[b] Tree taxonomic nomenclature follows Little, Jr. (1979)

Mill. - *P. elliottii* Engelm.) forests, more than half of which were planted (Oswalt et al. 2014). Many of these pine forests are managed with practices such as chemical and/or mechanical site preparation, fertilization, and thinning, enabling intensively managed pine forests (IMPFs) to produce much of the region's timber products (Yin and Sedjo 2001; Zhao et al. 2016; Demarais et al. 2017). However, many land owners also consider biodiversity conservation in their forest management plans, with some addressing biodiversity objectives under the auspices of forest certification programs (e.g., Forest Stewardship Council US 2010; American Forest Foundation 2015; Sustainable Forestry Initiative 2015) to promote conservation goals on their lands (Miller et al. 2009). Because IMPFs make up a substantial proportion of the Southern landscape, the opportunity exists for these actively managed lands to furnish landscape components beneficial to local and regional biodiversity (Miller et al. 2009; Henry et al. 2015; Demarais et al. 2017).

Early-successional forest is one component of IMPF landscapes, where it is established first as regenerating clearcuts (RCCs) in early-rotation stands prior to canopy closure (at about year 8-9), and where it often reoccurs for several years following subsequent, commercial thinnings. Open, shrubby, early-successional cover is an ephemeral resource that is declining in much of the eastern U.S., although the Southern region contains the largest remaining areas of early-successional cover (Trani et al. 2001). Greater than 50% to 70% of shrubland bird species exhibited declines from 1966 to 2011 in the bird conservation regions Western Gulf Coastal Plain/Ouachitas, Mississippi Alluvial Valley, and Southeastern Coastal Plain (King and Schlossberg

2014). In addition to regenerating clearcuts, patches of unharvested forest and vegetation are commonly retained on IMPF landscapes, further diversifying land cover.

Retention of individual live trees or unharvested patches of vegetation (e.g., green tree retention) is commonly recommended to promote post-harvest structural diversity (Franklin et al. 1997; Gustafsson et al. 2012), and benefits include: (a) retention of biological legacies; (b) greater structural complexity; and (c) improved landscape connectivity (Abernethy and Turner 1987; Franklin et al. 1997; Lindenmayer and Franklin 2002; Aubry et al. 2009). Southern forest managers typically retain unharvested or less-intensively managed vegetation as buffers around perennial and intermittent streams, termed streamside management zones (SMZs), and as buffers around ephemeral drains, commonly called 'stringers' (Parrish et al. 2017a). In addition to protecting water quality, this form of retention limits overtopping vegetation in the regenerating clearcut that could subsequently interfere with forest regeneration and later forestry operations. Best management practices (BMPs) for southern states generally offer limited guidance on retention of stringers. However, 68% of 1187 commercial management units [c] sampled in the South Central Plains ecoregion contained stringers, accounting for approximately 20% of all green tree retention land cover, indicating that commercial land owners frequently implement stringers voluntarily (Parrish et al. 2018).

Birds are an important component of overall biological diversity in forest communities, and they respond readily to changes in forest structure (Mazerolle and

[c] Management units: contiguous areas containing one or more intensively managed pine (*Pinus* spp.) forest stands that are uniform in age, having been harvested and reestablished as a unified cohort; plus any associated areas of green tree retention (e.g., streamside management zones or stringers; Parrish et al. 2017a).

Villard 1999; Balestrieri et al. 2015). Conservation of species dependent on early-successional forestland has been identified as a high research priority (Lorimer and White 2003; King and Schlossberg 2014; North American Bird Conservation Iniative 2016). Bird community associations with RCCs and SMZs in Southern IMPFs are relatively well documented (Dickson et al. 1984, Dickson et al. 1995b, Wilson and Watts 2000, Hanberry 2005); and previous studies have focused on effects of SMZ buffer width on IMPF species diversity (Rudolph and Dickson 1990; Thurmond et al. 1995; Kilgo et al. 1998; Constantine et al. 2004; Lee et al. 2004; Miller et al. 2004; Perry et al. 2011). However, I am unaware of any investigations into avian-stringer associations.

Although stringer vegetation is structurally similar to SMZ vegetation in several respects, the two cover types are distinct in ways that could potentially influence presence of associated bird species: stringers and SMZs in young IMPFs did not differ in ground cover, understory deciduous cover, or density of snags, logs, and midstory pines; but, SMZs had greater density of overstory deciduous and pine trees and slightly greater density of midstory deciduous trees, while stringers had greater understory pine stem density (Parrish et al. 2017a). Land managers have expressed interest in better understanding associations between species diversity and retained structural elements on IMPF management units. Thus, my objective was to document and compare the breeding season avian abundance and diversity within the three dominant cover types in 60 IMPF management units: RCCs, SMZs, and stringers. I used a guild-based approach (Root 1967; Blondel 2003; Gray et al. 2006) towards assessing avian species diversity associations with management unit cover types, as areas exhibiting high within-guild species diversity can be inferred to be beneficial to most or all members of the guild

(Bishop and Myers 2005). Of particular interest, given the similarity of vegetation structure and close spatial proximity between SMZs and stringers (Parrish et al. 2017a), was the degree of avian diversity overlaps between those cover types.

4.4 Methods

4.4.1 Study Area

The South Central Plains ecoregion (Fig. 4.1) of the U.S. extends from southeastern Oklahoma and central Arkansas into eastern Texas and north-central Louisiana, coinciding with the North American Bird Conservation Initiative bird conservation region 'West Gulf Coastal Plain / Ouachitas' (Omernik 1987; US Environmental Protection Agency 2011; Bird Studies Canada and NABCI 2014). The region experiences humid subtropical climate conditions, with mild winters (220 to 290 frost free days), hot summers (mean temperatures range from 17° C to 21° C), and mean annual precipitation between 105 and 170 cm (Woods et al. 2004; Daigle et al. 2006; Wilkin et al. 2011). Perennial streams (generally of low and moderate gradient) are abundant in the region, although hot summer conditions may cause smaller reaches (i.e., intermittent streams) to partially or completely dry up (Woods et al. 2004; Wilkin et al. 2011).

The South Central Plains ecoregion is located on the western edge of the Southern "pine belt." Prior to European colonization, South Central Plains forests were composed mostly of mixed shortleaf pine-hardwood in the north and longleaf pine savannas in the south (Wilkin et al. 2011). Loblolly pine was found in wet bottomlands and pine flatwoods and was also present in shortleaf pine-hardwood forest, especially south of the Arkansas River (Bragg 2008).

Two-thirds of the modern South Central Plains consists of forestland, and commercial IMPFs maintain a strong regional presence (Daigle et al. 2006; Griffith et al. 2007; Wilkin et al. 2011). Early-successional land cover within 5 km of my study area was predominantly pre-canopy closure IMPFs, such as the RCCs that I sampled (mean ± sd: 21.2% ± 6.8% of regional land cover), and thinned IMPFs (14.8% ± 8.1% of regional land cover; see chapter five). Arkansas and Louisiana forestry BMPs prescribe SMZ retention along perennial and intermittent streams and other permanent waters, primarily for the purpose of water quality protection (Louisiana Department of Agriculture and Forestry 1999; Arkansas Forestry Commission 2002). Land cover on an average management unit in the South Central Plains was dominated by RCCs (80.5% cover), with the remaining area mostly in two retention cover types: 14.0% in SMZs and 3.4% in stringers (Parrish et al. 2018).

4.4.2 Site Selection

I selected as sample sites 60 management units in the South Central Plains ecoregion of Arkansas and Louisiana from an initial set of 1187 management units established in 2008 and 2009 (Fig. 4.1). I sampled 35 sites in 2011 and another 25 sites in 2012, when regenerating pine stands were three years old. Sample sites were selected from management units ranking in the central 80% of the range of management unit acreage, that contained SMZ and stringer cover, and were within 60 km of one of six logistical hubs I established in the study area (Parrish et al. 2017a). I selected sites that allowed two survey stations to be established 200 m apart within each of the three dominant management unit cover types (RCCs, SMZs, and stringers) (i.e., RCCs, SMZs, and stringers; Parrish et al. 2017a); however, because some sites had limited available area in stringers and SMZs, I established only one station in stringers in 10 management units and one station in SMZs in 18 management units.

4.4.3 Bird surveys

I conducted three 10-minute, 50-m-radius avian point-count surveys at each sampling station between 5 May 2011 and 13 July 2011 (2011 sampling season) or 5 May 2012 and 21 June 2012 (2012 sampling season) under the following conditions: time between sunrise and 10:00 am, no precipitation, minimal fog, and wind speed <16 km • h^{-1}. I recorded species for all birds seen or heard within a 50-m radius. In narrower riparian areas, my 50-m count circles sometimes slightly overlapped RCC cover at the edges, but I judged that those transitional areas, which were strongly influenced by their association with proximate riparian retention patches, were best included with the adjacent areas of retention. Vegetation density sometimes confounded the ability of surveyors to approach stations silently, necessitating surveyors to observe an initial one-minute, quiet 'settling down' period prior to counts, to allow bird activity to resume more normal levels (Buckland et al. 2008). Birds observed to move in response to an approaching observer were recorded at the initial point of detection and were included in analyses.

4.4.4 Assignment to Avian Guilds

I defined my set of "target species" as breeding land birds meeting most of their reproductive season needs within the bounds of the management unit. I excluded from the target set: (1) birds seen outside count intervals or 50-m count circles, (2) late migrants, (3) non-landbirds, (4) nocturnal species, and (5) wide-ranging species (e.g., members of order *Accipitriformes,* families *Hirundinidae* and *Apodidae,* and genus *Corvus*). I defined functional guilds based on three life history characteristics: (1) foraging substrate; (2) nest placement location; and (3) dominant breeding habitat type (Appendix A). I reviewed species accounts in The Birds of North America (Rodewald 2015) for all target species, and assigned to them a membership within each group of guilds. Foraging substrate guilds were: (1) canopy (including midstory); (2) tree bark; (3) shrub layer (i.e., understory); and (4) ground. Nest placement guilds were: (1) canopy (including midstory); (2) tree cavity; (3) shrub; and (4) ground. Dominant breeding habitat type guilds were: (1) mature forest; (2) shrubland (i.e., early-successional forest); and (3) grassland.

4.4.5 Community Metrics

I computed community metrics using only target species recorded during point-counts. I calculated management unit-level species richness (S_{MU}) within guilds, defined as the count of target species assigned to a guild, regardless of cover type. I computed species richness by cover type (S_{CT}) within guilds. I also computed the percent of guild members present by cover type ($(S_{CT} \bullet S_{MU}^{-1}) \bullet 100$) to better understand how guild member species were represented among cover types.

I computed relative diversity (J'), by cover type within individual guilds and at the site level, as the ratio of the Shannon diversity index to the theoretical maximum Shannon diversity (Zar 2010). Relative diversity assesses how evenly individuals are allocated among species in a group. It reaches its maximum at 1 (all individuals evenly distributed among species) and decreases as individuals are less-evenly distributed among species.

I calculated Jaccard's similarity index (Jaccard 1902; Levandowsky and Winter 1971) to compare similarity of avian communities among the three cover types. I calculated Jaccard similarity for each pairing as the ratio of the intersection to the union of species present in the two cover types. Jaccard's similarity index ranges between 0 (no common species between groups) and 1 (makeup of both groups is identical). I compared my results to critical values of Jaccard's index to assess whether my observed Jaccard's indices were significantly different from those expected with random assortment of species among cover types (Real 1999).

I computed site-level relative density of birds (birds • ha^{-1}) as the mean across stations by cover type within each site. I defined station-level density as the maximum density recorded across three visits. I calculated mean, standard deviation, and range of relative density by cover type over *n*=60 sites for each guild. I used the GLM procedure in SAS[d] to perform analysis of variance (ANOVA) for comparing mean relative density as a function of cover type within each guild with a significance threshold of 0.05.

4.5 Results

4.5.1 Management Unit-Level Avian Communities

I detected 13137 birds representing 104 species, of which 60 species (n = 5617 birds) were designated as target species (Appendix A) to be used in analyses. Henceforth, I restrict my scope to those 60 species. Species richness was greatest in SMZs (S_{CT} = 55) and stringers (S_{CT} = 52), and was 15% to 20% lower in RCCs (S_{CT} = 44;

[d] The data analysis for this paper was generated using SAS software, Version 9.4 of the SAS System for Windows, copyright © 2002-2012 SAS Institute Inc. SAS and all other SAS Institute Inc. product or service names are registered trademarks or trademarks of SAS Institute Inc., Cary, NC, USA.

Table 4.1). Relative diversity was greatest in SMZs (J' = 0.81) and stringers (J' = 0.80) and approximately 15% lower in RCCs (J' = 0.69; Table 4.1).

Jaccard similarity indices indicated a high degree of similarity in species makeup between cover types, with only a 10% difference between the cover type pairings of least and greatest Jaccard similarity (Table 4.2; Appendix A). The SMZ-stringer pairing exhibited the greatest Jaccard similarity (84% similarity). Jaccard similarity indices for each pairing were significant (P <0.001 for all pairings), indicating the similarity indices were different from those expected under a random assortment of species among cover types (Real 1999). Forty-two of the 60 target species were detected at least once in each of the three cover types (Appendix A). Two infrequently observed species (Killdeer [e] and Scissor-tailed Flycatcher) were detected only in RCCs. SMZs and stringers contributed 27% of species to the management unit communities: six species were detected only in SMZs, three were detected only in stringers, and seven species were detected in both retention cover types but not in RCCs.

Five target species (Indigo Bunting n=776 detections; Yellow-breasted Chat, n=671; White-eyed Vireo, n=523; Northern Cardinal, n=478; and Prairie Warbler, n=337) accounted for 49.6% of all detections during counts, while the top 15 target species, ranked by number of detections, accounted for 79.3% of all detections. Eight target species (Northern Bobwhite, Yellow-billed Cuckoo, Red-headed Woodpecker, Northern Flicker, Prairie Warbler, Field Sparrow, Grasshopper Sparrow, and Eastern

[e] Avian taxonomic nomenclature follows American Ornithological Society Checklist of North and Middle American Birds (2017) through the fifty-seventh supplement (Chesser et al. 2016). I refer to standardized common names throughout this article. Please see Appendix A for scientific names.

Meadowlark) were classified by Partners in Flight (PIF) as 'common birds in steep decline' (Appendix A), due to estimated population declines >50% during the previous 40 years (Partners in Flight Science Committee 2012). Of those eight species, Prairie Warbler was detected frequently during counts. However, the remaining seven each accounted for <0.6% of detections. Three of the top 15 species were identified as experiencing "warranting management attention" due to either "moderate regional threats and moderate regional declines" (Blue-gray Gnatcatcher) or "high regional declines" (Prairie Warbler, Orchard Oriole; Partners in Flight Science Committee 2012). A further 19 target species were classified by PIF as "warranting management attention," and one species (Bachman's Sparrow) was classified as being "in need of immediate management attention" (Partners in Flight Science Committee 2012; Appendix A). However, those species were uncommon on my sites, each accounting for ≤1% of detections.

The Louisiana Department of Wildlife and Fisheries (LDWF) classified 19 of my target species as "demonstrably secure (S5)" or "apparently secure (S4)," five species as "rare and local (S3)" and two species as "critically imperiled (S1)" (Appendix A; Lester et al. 2005; Holcomb et al. 2015). The S3- and S1-ranked species were rarely detected during counts (<0.1% of detections), except for Northern Bobwhite (ranked S3; 21 detections). The Arkansas Game and Fish Commission (2004) classified one target species (Red-headed Woodpecker) as 'special concern'. I also detected several White-tailed Kites (a non-target species), ranked S1B, "critically imperiled in Louisiana," in close proximity on three occasions (23 May, 12 June, and 06 July 2011) at a site in Vernon Parish, Louisiana, suggesting possible breeding activity at the periphery of that species' range.

4.5.2 Foraging Substrate Guilds

Of the 60 target species, 70% were either ground foragers (S_{MU} = 24) or canopy foragers (S_{MU} = 18), ten species were shrub foragers, and the remaining six species were bark foragers (Table 4.1, Appendix A). Ground forager species richness was greater in SMZs (S_{CT} = 21) and stringers (S_{CT} = 19) than in clearcuts (S_{CT} = 15), but relative diversity of ground foragers was similar across cover types (J' = 0.80, 0.78, and 0.79, respectively). Ground forager relative density did not significantly differ between cover types ($P \geq 0.176$ for all contrasts; Tables 4.3 and 4.4). Canopy forager species richness was greatest in SMZs and stringers (S_{CT} = 16 in both) and lower in RCCs (S_{CT} = 12), and relative diversity of canopy foragers was 21% to 26% greater in retention cover types than in RCCs (Table 4.1). Canopy forager relative density was greatest in SMZs (5.1 ± 2.3 birds • ha^{-1}), followed by stringers (4.0 ± 2.2 birds • ha^{-1}), and was lowest in RCCs (1.0 ± 1.2 birds • ha^{-1}; $P \leq 0.001$ for all contrasts; Tables 4.3 and 4.4). Nearly all shrub forager and all bark forager guild members were detected at least once in all three cover types, and relative diversity within both guilds was similarly high (J' range: 0.77 to 0.89). Bark forager relative density did not significantly differ among SMZs and stringers (P = 0.836); contrasts against bank forager density in RCCs were not performed due to infrequent detections in RCCs. Shrub forager relative density was significantly greater in RCCs (6.2 ± 1.8 birds·ha^{-1}) than in stringers (5.2 ± 2.5 birds • ha^{-1}) or SMZs (3.9 ± 2.5 birds • ha^{-1}; $P \leq 0.013$ for all contrasts; Tables 4.3 and 4.4).

4.5.3 Nest location guilds

Species richness in SMZs and stringers was greater than in RCCs for all nest location guilds (Table 4.1). Relative diversity was similar between all three cover types within nest location guilds. Canopy nester relative density was more than three times greater in SMZs than in RCCs and density in stringers was approximately 2.5 times greater than in RCCs. Cavity nesters were frequently absent from RCCs. Therefore I only compared relative density between SMZs and stringers, finding no significant difference in mean relative density ($P = 0.076$; Tables 4.3 and 4.4). Shrub nester relative density was lower in SMZs than in either RCCs ($P = 0.040$) or stringers ($P = 0.004$), and I did not detect a significant difference in relative density between RCCs and stringers ($P = 0.375$; Tables 4.3 and 4.4). Relative density of ground nesters was not significantly different between any cover type pairing (Tables 4.3 and 4.4).

4.5.4 Dominant Breeding Cover Guilds

Most birds detected on management units were mature forest breeders ($S_{MU} = 36$) or shrubland breeders ($S_{MU} = 18$). Stringers and SMZs hosted more species in the mature forest breeder guild (89% and 94% of S_{MU}, respectively) compared with RCCs (69% of S_{MU}), but most members of the shrubland breeder guild were detected in all three cover types (S_{MU} ranged from 89% (RCC) to 100% (SMZ); Table 4.1). Mature forest breeder relative density was greatest in SMZs (7.0 ± 2.8 birds • ha^{-1}) and was 4.7 times the density in RCCs, while density in stringers was 24% lower than in SMZs ($P < .001$ for all contrasts; Tables 4.3 and 4.4). Shrubland breeder relative density was not significantly different between RCCs and stringers ($P = 0.645$), and was 35% to 37% larger in stringers and RCCs than in SMZs ($P < .001$ for both contrasts; Tables 4.3 and 4.4). Grassland breeders were poorly represented in all cover types, with only three of six potential species present in each cover type (Table 4.1). Because grassland breeders were rare and largely absent from management units, I did not compare density between cover types for that guild.

4.6 Discussion

4.6.1 Notable Species

Recently harvested IMPFs (e.g., my study sites in the South Central Plains) represent a primary source of breeding habitat for shrubland birds (e.g., Prairie Warbler, Yellow-breasted Chat, Indigo Bunting) in the coastal plains of the southern U.S. Smaller studies in RCCs located in comparable IMPFs in eastern Texas (Dickson et al. 1984), Louisiana (Legrand et al. 2007; Owens et al. 2014), Mississippi (Hanberry et al. 2012), and North Carolina, USA (Wilson and Watts 2000) reported site avian species richness between 29 and 42 species, although I note that the above studies did not sample in associated riparian retention areas. I detected essentially all species reported in the above studies in addition to a number of other, less common species, resulting from my study's larger sample size, more extensive geographic scope, and consideration of green tree retention areas associated with RCCs. Most species that I detected frequently were also abundant in other studies (Dickson et al. 1984; Wilson and Watts 2000). One exception was Northern Bobwhite, ranked 7th of 33 species on a single IMPF site in the South Central Plains region of eastern Texas, intensively sampled from 1977-1981 (Dickson et al. 1984). In my study, Northern Bobwhite was ranked far lower in relative abundance, perhaps due to regional declines in that species reported over the previous decades (Hernández et al. 2012).

I detected eight species that were listed by PIF as 'common birds in steep decline' (Appendix A; Partners in Flight Science Committee 2012), one of which, Prairie

Warbler, was commonly encountered on my sites. Prairie Warbler has been identified by PIF as "experiencing moderate regional threats and large declines during breeding" (Partners in Flight Science Committee 2012). LDWF classified the species as apparently secure (S4B) during breeding (Lester et al. 2005). Prairie Warbler is a complete migrant that nests and forages in shrubs and is a disturbance-dependent breeder that prefers early-successional cover types (Hunter et al. 2001). Young, regenerating forests represent a primary source of breeding habitat for Prairie Warbler in the South Central Plains ecoregion. King and Schlossberg (2014) reported that from 2005 to 2008, the amount of young forest (seedling-sapling successional stage) was increasing slightly in Arkansas ($0.83\% \cdot y^{-1}$), but was decreasing rapidly in Louisiana ($-16.7\% \cdot y^{-1}$), which is a potential concern facing shrubland specialists in the ecoregion.

I detected limited numbers of species designated by state wildlife agencies as being of greater conservation concern, suggesting that land cover types on the management units that I sampled were perhaps of marginal utility to the uncommon species, or simply illustrating the rarity of those species. For example, Bachman's Sparrow was one of the species of greatest conservation concern encountered in this study, having been listed as 'S3 / vulnerable' in Louisiana (Lester et al. 2005). I detected only six individuals during point-counts on a single site and I attributed the detections to spillover from a population associated with a nearby longleaf pine savanna (Dunning et al. 1995). Three-year-old RCCs, which dominated the young management units that I surveyed, are not prime Bachman's Sparrow habitat (Tirpak et al. 2009) and I concluded that the species was probably not a typical user of my sites.

4.6.2 Guild Associations with Cover Type

Early-successional specialists (i.e., shrub foragers, shrub nesters, and shrubland breeders) were well-represented in all cover types, and often >90% of potential guild members were present in a given cover type. Relative density in each guild were greatest in RCCs and stringers, and lowest in SMZs, perhaps reflecting the expected preference among shrubland specialists for cover with reduced canopy and midstory tree density, or greater understory pine density - characteristics of RCCs and stringers (Parrish et al. 2017a). This suggests that breeding shrubland birds make regular use of retained mature forest (i.e., SMZs) in IMPFs, albeit to a lesser extent than they use RCCs and stringers. A meta-analysis of seven studies of edge effects on shrubland birds in the northeastern U.S. found evidence that shrubland bird abundance was greater in clearcut interiors than at clearcut-forest edges (Schlossberg and King 2008). Shrubland patch size predicted patch occupancy by several area-sensitive early-successional bird species better than shape index or regional percent forested land cover on 43 early-successional forested buffers in the South Atlantic Coastal Plain of North Carolina (Shake et al. 2012). Conversely, Krementz and Christie (2000) did not observe an area effect on avian variation in richness, abundance, or reproductive effort in clearcuts on South Carolina, USA longleaf pine plantations. Additional research into area sensitivity and edge effects on shrubland birds is warranted.

Concurrently, forest specialist guilds (forest breeder, canopy nester, and canopy forager guilds) exhibited greater species richness in SMZs and stringers than in RCCs, while relative density for those guilds was greatest in SMZs, intermediate in stringers, and lowest in RCCs. The greater numbers of forest specialists in retention cover types is not surprising, due to those guilds' exploitation of vertically structured vegetation layers, which is largely absent from RCCs (Parrish et al. 2017a). However, the fact that forest specialists were often observed in RCCs suggests that SMZs and stringers may indirectly contribute to the species diversity of RCCs through spillover of forest guild members making forays into regenerating clearcuts. Forest interior specialists found in clearcuts could potentially be non-territorial, non-breeding floaters (Penteriani et al. 2011) drawn to the high-quality food resources of RCCs (Greenberg et al. 2011), and such individuals may go undetected in point-counts. For example, Pagen et al. (2000) captured non-singing Acadian Flycatchers in Missouri, USA hardwood system clearcuts during the breeding season, noting that migrants commonly used clearcuts but some forest species moved through quietly, making point-count detections difficult. It is therefore a possibility that my estimates of diversity of migrant forest specialists in RCCs were biased downward to an unknown extent. Use of clearcuts by forest interior bird species reportedly continued with some regularity into the post-breeding period in hardwood forest systems of Missouri, Ohio, Virginia, and West Virginia, USA (Pagen et al. 2000; Marshall et al. 2003; Vitz and Rodewald 2006), and a New Hampshire, USA study suggested that birds fledged in mature forests may, in fact, prefer shrublands to more mature forest types during the post-fledging period (Chandler et al. 2012).

Species richness was similar and relative density was not significantly different between SMZs and stringers for the bark forager and cavity nester guilds. In a previous study on my sites, bark foraging substrates and cavity trees were primarily found in SMZs and stringers, and no difference in snag density between SMZs and stringers was observed on my study sites in a previous study (Parrish et al. 2017a). I conclude that stringers increase the amount of forested land cover suitable for those guilds beyond what would be provided solely by SMZs. Bark foragers and cavity nesters (including Red-headed Woodpecker, an Arkansas species of concern) are likely to benefit from retention of stringers and snags in harvested portions of management units (Dickson et al. 1983).

Southern IMPFs may provide useful areas for grassland breeders during the seedling stage of the first few years of RCC succession (Thill and Koerth 2005), but as management units progress into later successional stages, their utility to grassland species quickly diminishes. My three-year-old RCCs were mostly aged out of the grassy stage and I detected very few grassland breeding birds in my study (e.g., Dickcissel, with 48 detections during point-counts). However, because a mosaic of IMPFs of varying age classes is maintained across the region, very young IMPFs likely remain consistently available to grassland birds. IMPF landscapes provide a multi-aged patchwork of forests important to regional bird diversity (Legrand et al. 2007).

4.6.3 Role of SMZs and Stringers on Management Units

Stringers and SMZs comprised an average of 17.4% of land area on management units in the ecoregion (Parrish et al. 2018); however, I detected 27% of target species (n = 16) solely within those two cover types. Of the 44 target species detected in RCCs, only two were exclusive to RCCs, whereas the remaining 42 species found in RCCs also occurred in SMZs and/or stringers. This illustrates that while most 'clearcut' species also exploit resources found in the two retention cover types (at least occasionally), many interior forest species only occupy young management units when provided sufficient area in green tree retention land cover. Retention of SMZs and stringers thereby serves to enhance post-harvest species diversity on young IMPFs. Also, juxtaposition of patches of riparian forest (i.e., SMZs), scattered trees and shrubs (i.e., stringers), and shrubby, open-canopy cover (i.e., RCCs) on young IMPFs creates a diversity of landscape components that was exploited by a variety of more generalist species which I detected within all three primary cover types.

Stringers clearly supplemented SMZ contributions to site avian species diversity: there was an 84% species overlap between SMZs and stringers, and stringers hosted seven species otherwise detected only in SMZs. Additionally, three uncommon species (Yellow-throated Warbler, Chipping Sparrow, and Eastern Meadowlark) were found exclusively in stringers, albeit in very low numbers, suggesting that stringers may provide their own unique resources to management units. Density of forest specialists (forest-breeder, canopy-forager, and canopy-nester guilds) was greatest in SMZs, which I attribute to high stem density of overstory trees and midstory hardwoods maintained in SMZs (Parrish et al. 2017a), and/or larger forest patch size of SMZs (Parrish et al. 2018). Forest specialist species richness was similar between stringers and SMZs, and relative density in stringers was intermediate between SMZs and RCCs. Thus, while stringers may not support densities of forest birds as high as in SMZs, they appeared to increase forest specialist numbers on management units, thereby augmenting SMZ contributions to site-specific avian diversity. Furthermore, I found that diversity of early-successional specialists in stringers was similar to that in RCCs, suggesting that stringers serve to enhance avian species diversity of RCCs as well as of SMZs.

Stringers may supplement other ecological functions provided by SMZs. Riparian buffer strips and forested corridors facilitate species movement across harvested landscapes, particularly for forest species reluctant to cross expansive, open areas

(Machtans et al. 1996; Desrochers and Hannon 1997; Robertson and Radford 2009). The presence of stringers on the management unit landscape may enhance SMZ-RCC connectivity for forest interior guilds by providing vegetated corridors that facilitate access to clearcut interiors. However, stringers with greater canopy cover, by extending into RCC core areas, could potentially reduce the value of open, shrubby patches to area-sensitive early-successional specialists, and more research attention to that topic is warranted. SMZs and stringers provide greater vertical heterogeneity adjacent to RCCs, including structures used as perches, snags, cover, and foraging substrates which may enhance RCC species richness (Suárez-Seoane et al. 2002). Increases in composite variables representing vegetation structure, evergreen cover, and ground cover in IMPF RCCs <5 years old were positively associated with avian diversity metrics (Owens et al. 2014). Although riparian buffers of greater width (i.e., SMZs) may offer superior conservation value to area-sensitive forest species compared to narrower buffer strips (i.e., stringers), smaller cover features likely serve to supplement resources found in SMZs (Perry et al. 2011). Even small patches of retained green tree land cover may convey wildlife benefits (Lindenmayer et al. 2015).

The extensive amount of mature forest-shrubland edge associated with stringers and SMZs has prompted research investigating the potential for negative edge effects on birds in IMPFs. Indigo Buntings in South Carolina were attracted to narrow, linear patches of mature forest extending into early-successional cover, but breeding birds in those patches experienced lower fecundity and greater predation risk, with the patches effectively acting as ecological traps (Weldon and Haddad 2005). Conversely, mature treelines with high edge-to-area ratios appeared to benefit nesting success in Painted

Bunting, an early-successional species that co-occurs with Indigo Bunting in Louisiana (Vasseur and Leberg 2015). A study of nest failure in Acadian Flycatchers (a forest specialist) nesting in mature IMPF stands in South Carolina found no evidence that proximity to clearcut edges increased nest failure rates and no nest parasitism was observed (Hazler et al. 2006). Nest parasitism rates by Brown-headed Cowbird on Yellow-breasted Chat, Indigo Bunting, and Prairie Warbler varied from 3% to 24% in RCCs with abundant edges in IMPFs located in Mississippi (Hackemack et al. 2016). Additional research comparing reproductive success to landscape configuration would contribute to a better understanding of edge effects on survival outcomes. Regardless, it must be noted that in my study sites, SMZs were prescribed under BMP guidelines primarily for protection and maintenance of water quality in perennial and intermittent water features (Louisiana Department of Agriculture and Forestry 1999; Arkansas Forestry Commission 2002). Stringers were often voluntarily implemented for the same purpose. Therefore, water quality protection can be projected to take management priority over potential edge effects on avian diversity.

4.7 Management Implications

Regenerating clearcuts in IMPFs of the southern U.S. represent an important cover type for a variety of early-successional specialists in the region, some of which are of heightened conservation concern (Partners in Flight Science Committee 2012). Members of early-successional specialist guilds, while obviously favoring RCCs, nevertheless also made regular use of green tree retention areas (i.e., stringers and SMZs), and stringers particularly appeared to enhance species diversity in young management units. Green tree retention practices contributed to overall site species diversity by providing vertical vegetation structure that supported mature forest specialists in addition to shrubland-associated birds. Retained stringers appeared to augment diversity contributions made by SMZs, and stringers provided cover used by both early-successional and forest interior guilds. Further research on the effects of local and regional landscape context on species diversity and reproductive success in IMPFs is warranted.

IMPFs with rotations of 25 to 30 years (Miller et al. 2009) typically experience peak avian diversity between years two and six (a secondary peak often occurs post-thinning), and changes in relative species prevalence are expected during this period (Dickson et al. 1984, 1995; Keller et al. 2003). The grass-shrub stage in regenerating clearcuts is brief, generally lasting less than nine years of the average Southern pine stand rotation. Modern forest management practices that produce a shifting, patchy, regional landscape of differently-aged forests are critical to maintaining the region's early-successional forestland. By sustaining the regional mosaic of forests in varying successional stages, land managers ensure that ephemeral, early-successional cover remains continuously available to wildlife in the southern United States.

Table 4.1 Management unit-level species richness (SMU) and cover type-level species richness (SCT), percent of SMU represented (% SMU) a, and species relative diversity (J'), by functional guild group affiliation (foraging substrate, nesting location, and breeding cover) on 60 intensively managed pine (*Pinus* spp.) forest management units in the South Central Plains ecoregion of Arkansas and Louisiana, USA.

		Cover type								
Grouping		**Regenerating Clearcut** [b]			**Streamside Management Zone**			**Stringer**		
	S_{MU}	S_{CT}	% S_{MU}	*J'*	S_{CT}	% S_{MU}	*J'*	S_{CT}	% S_{MU}	*J'*
Management unit										
total	60	44	73%	0.69	55	92%	0.81	52	87%	0.80
Foraging substrate [c]										
Canopy	18	12	67%	0.61	16	89%	0.82	16	89%	0.77
Bark	6	6	100%	0.87	6	100%	0.83	6	100%	0.89
Shrub	10	9	90%	0.77	10	100%	0.81	9	90%	0.82
Ground	24	15	63%	0.79	21	88%	0.80	19	79%	0.78
Nesting location										
Canopy	19	14	74%	0.79	17	89%	0.82	16	84%	0.85
Cavity	12	10	83%	0.88	12	100%	0.85	11	92%	0.90
Shrub	16	13	81%	0.75	15	94%	0.77	16	100%	0.77
Ground	13	7	54%	0.69	11	85%	0.69	9	69%	0.60
Breeding cover										
Forest	36	25	69%	0.77	34	94%	0.82	32	89%	0.82
Shrubland	18	16	89%	0.75	18	100%	0.74	17	94%	0.77
Grassland	6	3	50%	0.20	3	50%	0.92	3	50%	0.89

[a] Rounded to nearest integer.
[b] Regenerating clearcuts were three-years-old during avian sampling.
[c] Two foraging location generalists were not assigned guild membership.

Table 4.2 Jaccard similarity indices comparing avian communities associated with three cover types (three-year-old regenerating clearcut, 'RCC'; streamside management zone, 'SMZ'; and stringers, 'STR') that dominate intensively managed pine (*Pinus* spp.) forest management units in the South Central Plains ecoregion of Arkansas and Louisiana, USA.

Cover type pairing	**Jaccard similarity index**	**Critical values [a] (α = 0.001)**	***P*-value [b]**
RCC - SMZ	0.74	(0.1404, 0.5439)	<0.001
RCC - STR	0.78	(0.1296, 0.5556)	<0.001
SMZ - STR	0.84	(0.1379, 0.5517)	<0.001

[a] Critical values based on Real (1999).
[b] H_0: Jaccard similarity was the result of random chance.

Table 4.3 Mean, standard deviation, and range of avian relative density (birds • ha^{-1}) by functional guild affiliation in three dominant cover types of 60 intensively managed pine (*Pinus* spp.) forest management units in the South Central Plains ecoregion of Arkansas and Louisiana, USA.

	Avian relative density (birds • ha^{-1})					
	Regenerating Clearcut [a]		**Streamside Mgmt. Zone**		**Stringer [b]**	
Guild Affiliation	***$\bar{x} \pm sd$***	**range**	***$\bar{x} \pm sd$***	**range**	***$\bar{x} \pm sd$***	**range**
Foraging location [c]						
Canopy	1.0 ± 1.2	[0.0, 4.5]	5.1 ± 2.3	[1.3, 10.2]	4.0 ± 2.2	[0.0, 10.8]
Bark	0.1 ± 0.4	[0.0, 1.9]	0.9 ± 0.9	[0.0, 3.8]	0.9 ± 1.0	[0.0, 3.8]
Shrub	6.2 ± 1.8	[2.5, 10.2]	3.9 ± 2.5	[0.0, 9.5]	5.2 ± 2.5	[1.9, 16.6]
Ground	1.9 ± 1.3	[0.0, 6.4]	1.9 ± 1.4	[0.0, 6.4]	2.2 ± 1.5	[0.0, 6.4]
Nesting location						
Canopy	1.1 ± 1.2	[0.0, 5.1]	3.7 ± 1.9	[0.0, 7.6]	2.7 ± 1.6	[0.0, 7.6]
Cavity	0.5 ± 0.7	[0.0, 2.5]	2.7 ± 1.6	[0.0, 7.6]	2.2 ± 1.8	[0.0, 6.4]
Shrub	6.9 ± 2.0	[2.5, 13.4]	6.1 ± 2.8	[0.6, 14.0]	7.3 ± 2.8	[2.5, 14.0]
Ground	1.1 ± 1.1	[0.0, 5.1]	0.9 ± 0.7	[0.0, 3.2]	0.9 ± 0.9	[0.0, 3.8]
Breeding cover						
Forest	1.5 ± 1.5	[0.0, 5.7]	7.0 ± 2.8	[2.5, 13.4]	5.3 ± 2.9	[0.0, 15.3]
Shrubland	7.1 ± 2.0	[3.8, 12.1]	5.2 ± 2.7	[0.0, 12.1]	7.0 ± 2.7	[2.5, 12.7]
Grassland	0.4 ± 1.0	[0.0, 5.1]	0.1 ± 0.3	[0.0, 1.9]	0.1 ± 0.6	[0.0, 3.8]

[a] Regenerating clearcuts were three-years-old during avian sampling.
[b] n = 58 management units for stringer cover type.
[c] Two foraging location generalists were not assigned guild membership.

Table 4.4 ANOVA contrasts of mean avian relative density (birds • ha^{-1}) by functional guild affiliation between pairings of the three dominant cover types found in 60 intensively managed pine (*Pinus* spp.) forest management units in the South Central Plains ecoregion of Arkansas and Louisiana, USA.

	Cover type contrast pairings [a]								
	RCC-SMZ			**RCC-STR**			**SMZ-STR**		
Guild Affiliation	**Est.**	***SE***	***P*-val**	**Est.**	***SE***	***P*-val**	**Est.**	***SE***	***P*-val**
Foraging location									
Canopy	-4.1	0.30	<0.001	-3.0	0.30	<0.001	1.1	0.30	0.001
Bark	...	...	...	...	...	...	0.0	0.16	0.836
Shrub	2.3	0.39	<0.001	1.0	0.39	0.013	-1.3	0.39	0.001
Ground	0.0	0.23	0.891	-0.3	0.23	0.223	-0.3	0.23	0.176
Nesting location									
Canopy	-2.6	0.26	<0.001	-1.6	0.26	<0.001	1.0	0.26	<0.001
Cavity	...	...	...	...	...	...	0.4	0.24	0.076
Shrub	0.8	0.39	0.040	-0.4	0.40	0.375	-1.2	0.40	0.004
Ground	0.2	0.14	0.090	0.2	0.14	0.199	-0.1	0.14	0.690
Breeding cover									
Forest	-5.5	0.37	<0.001	-3.8	0.38	<0.001	1.7	0.38	<0.001
Shrubland	1.9	0.41	<0.001	0.2	0.41	0.645	-1.7	0.41	<0.001
Grassland	...	...	...	...	...	...	...	...	...

Note: Significant P-values indicate a difference in avian mean density between cover types in pair, with positive estimates indicating the first pair member is greater than the second member, and vice-versa. I did not compare cover types that had strongly zero-dominated density data (e.g., bark foragers were largely absent from RCC cover) and indicate those cases with an ellipsis.

[a] Cover types: 'RCC': regenerating clearcut (*n*=60); 'SMZ': streamside management zone (*n*=60); 'STR': stringer (*n*=58).

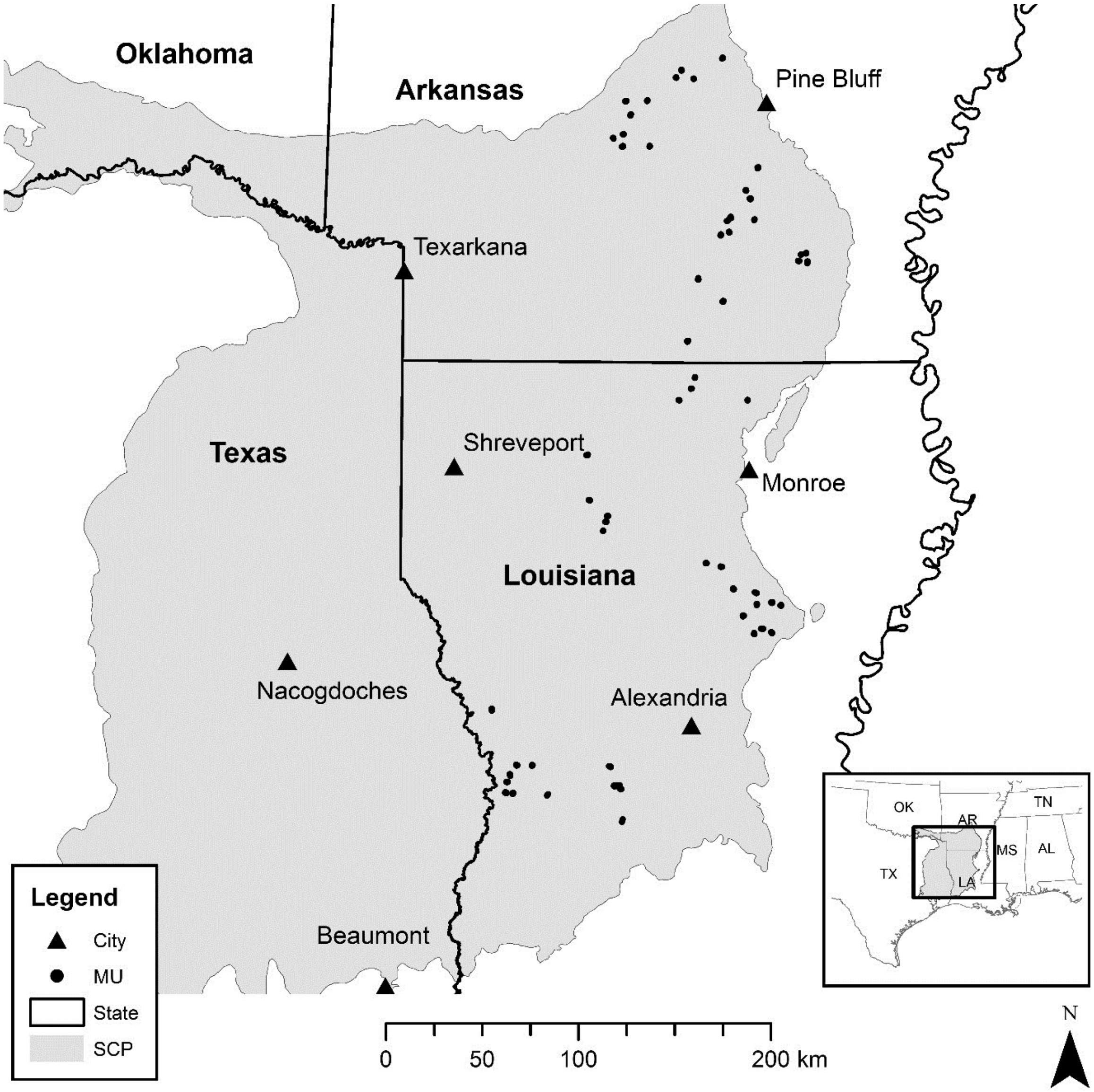

Figure 4.1 Study site locations within the South Central Plains (SCP) ecoregion in Arkansas and Louisiana, USA.

Sixty intensively managed pine (*Pinus* spp.) forest management units (MUs) established in 2008 or 2009 were selected for breeding bird point-count sampling when the MUs were 3 years post-establishment. The SCP roughly coincides with the North American Bird Conservation Initiative bird conservation region *West Gulf Coastal Plain / Ouachitas*.

4.8 Literature Cited

Abernethy, Y., and R.E. Turner. 1987. US forested wetlands: 1940-1980. BioScience 37(10):721-727.

American Forest Foundation. 2015. American Forest Foundation (AFF) 2015-2020 Standards of Sustainability. Available online at www.treefarmsystem.org/view-standards; last accessed Oct. 15, 2016.

Arkansas Forestry Commission. 2002. Arkansas forestry best management practices for water quality protection. 58 p. Available online at http://forestry.arkansas.gov/Services/ManageYourForests/Documents/bmpbookrevise.pdf; last accessed on Feb. 16, 2017.

Arkansas Game and Fish Commission. 2004. Arkansas endangered, threatened, and species of special concern: January 9, 2004. Little Rock, AR, USA.

Aubry, K.B., C.B. Halpern, and C.E. Peterson. 2009. Variable-retention harvests in the Pacific Northwest: A review of short-term findings from the DEMO study. Forest Ecology and Management 258:398-408.

Balestrieri, R., M. Basile, M. Posillico, T. Altea, B. De Cinti, and G. Matteucci. 2015. A guild-based approach to assessing the influence of beech forest structure on bird communities. Forest Ecology and Management 356:216-223.

Bird Studies Canada and NABCI. 2014. Bird Conservation Regions. Published by Bird Studies Canada on behalf of the North American Bird Conservation Initiative. http://www.birdscanada.org/research/gislab/index.jsp?targetpg=bcr Accessed: 31 March 2014.

Bishop, J.A., and W.L. Myers. 2005. Associations between avian functional guild response and regional landscape properties for conservation planning. Ecological Indicators 5:33-48.

Blondel, J. 2003. Guilds or functional groups: does it matter? Oikos 100(2):223-231.

Bragg, D.C. 2008. The prominence of pine in the Upper West Gulf Coastal Plain during historical times. P. 29-54 in Freeman and Custis Red River Expedition of 1806: Two Hundred Years Later, Hardy, L.M. (ed.). Bulletin of the Museum of Life Sciences #13, Louisiana State University in Shreveport, Shreveport, LA.

Buckland, S.T., S.J. Marsden, and R.E. Green. 2008. Estimating bird abundance: making methods work. Bird Conservation International 18(Supplement S1):S91-S108.

Chandler, C.C., D.I. King, and R.B. Chandler. 2012. Do mature forest birds prefer early-successional habitat during the post-fledging period? Forest Ecology and Management 264:1-9.

Constantine, N.L., T.A. Campbell, W.M. Baughman, T.B. Harrington, B.R. Chapman, and K.V. Miller. 2004. Effects of clearcutting with corridor retention on abundance, richness, and diversity of small mammals in the Coastal Plain of South Carolina, USA. Forest Ecology and Management 202:293-300.

Daigle, J.J., G.E. Griffith, J.M. Omernik, P.L. Faulkner, R.P. McCulloh, L.R. Handley, L.M. Smith, et al. 2006. Ecoregions of Louisiana (color poster with map, descriptive text, summary tables, and photographs). U.S. Geological Survey: Reston, VA (map scale 1:1,000,000).

Demarais, S., J.P. Verschuyl, G.J. Roloff, D.A. Miller, and T.B. Wigley. 2017. Tamm review: Terrestrial vertebrate biodiversity and intensive forest management in the U.S. Forest Ecology and Management 385:308-330.

Desrochers, A., and S.J. Hannon. 1997. Gap crossing decisions by forest songbirds during the post-fledging period. Conservation Biology 11(5):1204-1210.

Dickson, J.G., R.N. Conner, and J.H. Williamson. 1983. Snag retention increases bird use of a clear-cut. Journal of Wildife Management 47(3):799-804.

Dickson, J.G., R.N. Conner, and J.H. Williamson. 1984. Bird community changes in a young pine plantation in east Texas. Southern Journal of Applied Forestry 8(1):47-51.

Dickson, J.G., R.N. Conner, and J.H. Williamson. 1995. Winter birds in a developing pine plantation. P. 303-313 *in* Annual Conference of the Southeastern Association of Fish and Wildlife Agencies.

Dunning, J.B., R. Borgella, K. Clements, and G.K. Meffe. 1995. Patch isolation, corridor effects, and colonization by a resident sparrow in a managed pine woodland. Conservation Biology 9(3):542-550.

Forest Stewardship Council US. 2010. FSC - US Forest Management Standard (v.1.0). Available online at https://us.fsc.org/download.fsc-us-forest-management-standard-v1-0.95.htm. Last accessed Oct. 15 2016.

Franklin, J.F., D.R. Berg, D.A. Thornburgh, and J.C. Tappeiner. 1997. Alternative silvicultural approaches to timber harvesting: Variable retention harvest systems. P. 111-136 in Creating a Forestry for the 21st Century: The Science of Ecosystem Management, Kohm, K.A., and J.F. Franklin (eds.). Island Press, Washington, D.C.

Gray, M.A., S.L. Baldauf, P.J. Mayhew, and J.K. Hill. 2006. The response of avian feeding guilds to tropical forest disturbance. Conservation Biology 21(1):133-141.

Greenberg, C.H., R.W. Perry, C.A. Harper, D.J. Levey, and J.M. McCord. 2011. The role of young, recently disturbed upland hardwood forest as high quality food patches. P. 121-142 in Sustaining Young Forest Communities: Ecology and Management of Early-successional Habitats in the Central Hardwood Region, USA., Greenberg, C.H., B.S. Collins, and W.L. Thompson (eds.), Springer, New York.

Griffith, G., S. Bryce, J.M. Omernik, and A. Rogers. 2007. Ecoregions of Texas: Project Report to Texas Commission on Environmental Quality. Available online at ftp://newftp.epa.gov/EPADataCommons/ORD/Ecoregions/tx/TXeco_Jan08_v8_Cmprsd.pdf. Last accessed Feb. 16, 2017.125 p.

Gustafsson, L., S.C. Baker, J. Bauhus, W.J. Beese, A. Brodie, J. Kouki, D.B. Lindenmayer, et al. 2012. Retention forestry to maintain multifunctional forests: A world perspective. BioScience 62(7):633-645.

Hackemack, T.F., Z.G. Loman, S.K. Riffell, and S.A. Rush. 2016. Do nest size and shape characteristics affect nest parasitism rates? Wildlife Society Bulletin 40(4):722-726.

Hanberry, B.B., P. Hanberry, S.K. Riffell, S. Demarais, and J.C. Jones. 2012. Bird assemblages of intensively established pine plantations in Coastal Plain Mississippi. The Journal of Wildlife Management 76(6):1205-1214.

Hazler, K.R., A.J. Amacher, R.A. Lancia, and J.A. Gerwin. 2006. Factors influencing Acadian flycatcher nesting success in an intensively managed forest landscape. Journal of Wildlife Management 70(2):532-538.

Henry, D.R., D.A. Miller, and T.W. Sherry. 2015. Integrating wildlife conservation with commercial silviculture - Demography of the Swainson's Warbler (*Limnothlypis swainsonii*), a migrant bird of conservation concern in Southern pine forests, USA. in Precious Forests - Precious Earth, Zlatic, M. (ed.). InTech, DOI: 10.5772/61168. Available online at: https://www.intechopen.com/books/precious-forests-precious-earth/integrating-wildlife-conservation-with-commercial-silviculture-demography-of-the-swainson-s-warbler-. Last accessed: June 21, 2017.

Hernández, F., L.A. Brennan, S.J. DeMaso, J.P. Sands, and D.B. Wester. 2012. On reversing the Northern Bobwhite population decline: 20 years later. Wildlife Society Bulletin 37(1):177-188.

Holcomb, S.R., A.A. Bass, C.S. Reid, M.A. Seymour, N.F. Lorenz, B.B. Gregory, S.M. Javed, et al. 2015. Louisiana Wildlife Action Plan (Draft, October 2015). Louisiana Department of Wildlife and Fisheries, Baton Rouge, LA, USA, p. 661 pp.

Hunter, W.C., D.A. Buehler, R.A. Canterbury, J.L. Confer, and P.B. Hamel. 2001. Conservation of disturbance-dependent birds in eastern North America. Wildlife Society Bulletin 29(2):440-455.

Jaccard, P. 1902. Lois de Distribution Florale dans la Zone Alpine. Bulletin de la Societe Vaudoise des Sciences Naturelles 38(144):69-130.

Keller, J.K., M.E. Richmond, and C.R. Smith. 2003. An explanation of patterns of breeding bird species richness and density following clearcutting in Northeastern USA forests. Forest Ecology and Management 174(1-3):541-564.

Kilgo, J.C., R.A. Sargent, B.R. Chapman, and K.V. Miller. 1998. Effect of stand width and adjacent habitat on breeding bird communities in bottomland hardwoods. Journal of Wildlife Management 62(1):72-83.

King, D.I., and S. Schlossberg. 2014. Synthesis of the conservation value of the early-successional stage in forests of eastern North America. Forest Ecology and Management 324:186-195.

Krementz, D.G., and J.S. Christie. 2000. Clearcut stand size and scrub-successional bird assemblies. The Auk 117(4):913-924.

Lee, P., C. Smyth, and S. Boutin. 2004. Quantitative review of riparian buffer width guidelines from Canada and the United States. Journal of Environmental Management 70:165-180.

Legrand, H.G., M.J. Chamberlain, and E.B. Moser. 2007. Diversity and abundance of breeding birds in a managed loblolly pine forest in Louisiana. The American Midland Naturalist 157(2):329-344.

Lester, G.D., S.G. Sorensen, P.L. Faulkner, C.S. Reid, and I.E. Maxit. 2005. Louisiana Comprehensive Wildlife Conservation Strategy. Louisiana Department of Wildlife and Fisheries, Baton Rouge, LA, USA, 455 pp.

Levandowsky, M., and D. Winter. 1971. Distance between sets. Nature 234:34-35.

Lindenmayer, D.B., and J.F. Franklin. 2002. Conserving Forest Biodiversity: A Comprehensive Multiscaled Approach. Island Press, Washington, D.C. 352 p.

Lindenmayer, D.B., R.J. Hobbs, and D. Salt. 2015. Plantation forests and biodiversity conservation. Australian Forestry 66(1):62-66.

Little, E.L., Jr. 1979. Checklist of United States Trees (Native and Naturalized). US Department of Agriculture Agriculture Handbook no. 541. 375 p.

Lorimer, C.G., and A.S. White. 2003. Scale and frequency of natural disturbances in the Northeastern US: Implications for early successional forest habitats and regional age distributions. Forest Ecology and Management 185:41-64.

Louisiana Department of Agriculture and Forestry. 1999. Recommended Forestry Best Management Practices for Louisiana. 83 p. Available online at http://www.ldaf.state.la.us/wp-content/uploads/2014/04/BMP.pdf. Last accessed Feb. 16, 2017.

Machtans, C.S., M.-A. Villard, and S.J. Hannon. 1996. Use of riparian buffer strips as movement corridors by forest birds. Conservation Biology 10(5):1366-1379.

Marshall, M.R., J.A. DeCecco, A.B. WIlliams, G.A. Gale, and R.J. Cooper. 2003. Use of regenerating clearcuts by late-successional bird species and their young during the post-fledging period. Forest Ecology and Management 183(1-3):127-135.

Mazerolle, M.J., and M.-A. Villard. 1999. Patch characteristics and landscape context as predictors of species presence and abundance: A review. Ecoscience 6(1):117-124.

Miller, D.A., R.E. Thill, M.A. Melchiors, T.B. Wigley, and P.A. Tappe. 2004. Small mammal communities of streamside management zones in intensively managed pine forests of Arkansas. Forest Ecology and Management 203(1-3):381-393.

Miller, D.A., T.B. Wigley, and K.V. Miller. 2009. Managed forests and conservation of terrestrial biodiversity in the southern United States. Journal of Forestry 107(4):197-203.

North American Bird Conservation Iniative. 2016. The State of North America's Birds 2016. Environment and Climate Change Canada: Ottawa, Ontario. 8 p. Available online at http://www.stateofthebirds.org/2016/wp-content/uploads/2016/05/SoNAB-ENGLISH-web.pdf. Last accessed Jun. 27, 2017.

Omernik, J.M. 1987. Ecoregions of the conterminous United States. Annals of the Association of American Geographers 77(1):118-125.

Oswalt, S.N., W.B. Smith, P.D. Miles, and S.A. Pugh. 2014. Forest Resources of the United States, 2012: A Technical Document Supporting the Forest Service 2015 Update of the RPA Assessment. Gen. Tech. Rep. WO-91. Washington, DC: U.S. Department of Agriculture, Forest Service, Washington Office. 218 p.

Owens, F.L., P.C. Stouffer, M.J. Chamberlain, and D.A. Miller. 2014. Early-successional breeding bird communities in intensively managed pine plantations: Influence of vegetation succession but not site preparations. Southeastern Naturalist 13(3):423-443.

Pagen, R.W., F.R. Thompson, III, and D.E. Burhans. 2000. Breeding and post-breeding habitat use by forest migrant songbirds in the Missouri Ozarks. The Condor 102(4):738-747.

Parrish, M.C., S. Demarais, A.W. Ezell, T.B. Wigley, P.D. Jones, and S.K. Riffell. 2017a. Retained vegetation density of streamside management zones and stringers in Southern intensively managed pine forests. Forest Ecology and Management 397:89-96.

Parrish, M.C., S. Demarais, T.B. Wigley, P.D. Jones, A. Ezell, and S.K. Riffell. 2017b. Breeding bird communities associated with land cover in intensively managed pine forests of the southeastern U.S. Forest Ecology and Management 406:112-124.

Parrish, M.C., S. Demarais, T.B. Wigley, S.K. Riffell, A. Ezell, and P.D. Jones. 2018. Operational green tree retention and land cover patterns in intensively managed pine forest landscapes of the southeastern U.S. Forest Science 00(00):1-13.

Partners in Flight Science Committee. 2012. Species Assessment Database, Version 2012 (Bird Conservation Region 25). Available online at http://rmbo.org/pifassessment. Last accessed Mar. 16, 2017.

Penteriani, V., M. Ferrer, and M.M. Delgado. 2011. Floater strategies and dynamics in birds, and their importance in conservation biology: towards an understanding of nonbreeders in avian populations. Animal Conservation 14(3):233-241.

Perry, R.W., T.B. Wigley, M.A. Melchiors, R.E. Thill, P.A. Tappe, and D.A. Miller. 2011. Width of riparian buffer and structure of adjacent plantations influence occupancy of conservation priority birds. Biodiversity and Conservation 20:625-642.

Real, R. 1999. Tables of significant values of Jaccard's index of similarity. Miscellània Zoològica 22(1):29-40.

Robertson, O.J., and J.Q. Radford. 2009. Gap-crossing decisions of forest birds in a fragmented landscape. Austral Ecology 34:435-446.

Rodewald, P., (Ed.). 2015. The Birds of North America. Cornell Laboratory of Ornithology, Ithaca, NY. Available online at http://bna.birds.cornell.edu/BNA/. Last accessed Jun. 27, 2017.

Root, R.B. 1967. The niche exploitation pattern of the Blue-gray Gnatcatcher. Ecological Monographs 37(4):317-350.

Rudolph, D.C., and J.G. Dickson. 1990. Streamside zone width and amphibian and reptile abundance. The Southwestern Naturalist 35(4):472-476.

Schlossberg, S., and D.I. King. 2008. Are shrubland birds edge specialists? Ecological Applications 18(6):1325-1330.

Shake, C.S., C.E. Moorman, J.D. Riddle, and M.R. Burchell, II. 2012. Influence of patch size and shape on occupancy by shrubland birds. The Condor 114(2):268-278.

Suárez-Seoane, S., P.E. Osborne, and J. Baudry. 2002. Responses of birds of different biogeographic origins and habitat requirements to agricultural land abandonment in northern Spain. Biological Conservation 105:333-344.

Sustainable Forestry Initiative. 2015. SFI 2015-2019 standard and rules: Standards, rules for label use, procedures, and guidance. Available online at www.sfiprogram.org/files/pdf/2015-2019-standardsandrules-web-lr-pdf/; last accessed on Oct 15. 2016.

Thill, R.E., and N.E. Koerth. 2005. Breeding birds of even- and uneven-aged pine forests of Eastern Texas. Southeastern Naturalist 4(1):153-176.

Thurmond, D.P., K.V. Miller, and T.G. Harris. 1995. Effect of streamside management zone width on avifauna communities. Southern Journal of Applied Forestry 19(4):166-169.

Tirpak, J.M., D.T. Jones-Farrand, F.R. Thompson, III, D.J. Twedt, and W.B. Uihlein, III. 2009. Multiscale habitat suitability index models for priority landbirds in the Central Hardwoods and West Gulf Coastal Plain / Ouachitas bird conservation regions. Gen. Tech. Rep. NRS-49. Newtown Square, PA: USDA Forest Service Northern Research Station. 195 p.

Trani, M.K., R.T. Brooks, T.L. Schmidt, V.A. Rudis, and C.M. Gabbard. 2001. Patterns and trends of early successional forests in the eastern United States. Wildlife Society Bulletin 29(2):413-424.

US Environmental Protection Agency. 2011. Ecoregions of the United States (Spatial Data). 1:250,000 scale. U.S. EPA Office of Research & Development (ORD) - National Health and Environmental Effects Research Laboratory (NHEERL), Corvallis, OR. Available online at: ftp://ftp.epa.gov/wed/ecoregions/us/Eco_Level_III_US.zip; last accessed Aug. 26, 2011.

Vasseur, P.L., and P.L. Leberg. 2015. Effects of habitat edges and nest site characteristics on Painted Bunting nest success. Journal of Field Ornithology 86(1):27-40.

Vitz, A.C., and A.D. Rodewald. 2006. Can regenerating clearcuts benefit mature-forest songbirds? An examination of post-breeding ecology. Biological Conservation 127:477-486.

Weldon, A.J., and N.M. Haddad. 2005. The effects of patch shape on Indigo Buntings: evidence for an ecological trap. Ecology 86(6):1422-1431.

Wilkin, E., F. Jiménez Nava, and G. Griffith. 2011. North American Terrestrial Ecoregions - Level III. Commission for Environmental Cooperation, Montreal, Canada. 149 p.

Wilson, M.D., and B.D. Watts. 2000. Breeding bird communities in pine plantations on the coastal plain of North Carolina. The Chat 64(1):1-14.

Woods, A.J., T.L. Foti, S.S. Chapman, J.M. Omernik, J.A. Wise, E.O. Murray, W.L. Prior, et al. 2004. Ecoregions of Arkansas (Color Poster with Map, Descriptive Text, Summary Tables, and Photographs). U.S. Geological Survey: Reston, VA (map scale 1:1,000,000).

Yin, R., and R.A. Sedjo. 2001. Is this the age of intensive management? A study of loblolly pine on Georgia's Piedmont. Journal of Forestry 99(1):10-17.

Zar, J.H. 2010. Biostatistical Analysis. Pearson Prentice Hall, Upper Saddle River, New Jersey, USA. 944 p.

Zhao, D., M. Kane, R. Teskey, T.R. Fox, T.J. Albaugh, H.L. Allen, and R. Rubilar. 2016. Maximum response of loblolly pine plantations to silvicultural management in the southern United States. Forest Ecology and Management 375:105-111.

CHAPTER V

AVIAN RESPONSES TO LOCAL AND LANDSCAPE CHARACTERISTICS IN SOUTHERN U.S. INTENSIVELY MANAGED PINE FORESTS

5.1 Abstract

Forest landowners in the southern U.S. typically retain unharvested or partially-harvested patches of trees and vegetation along streams and ephemeral drains when clearcutting intensively managed pine (*Pinus* spp.) stands to protect water quality and enhance biological diversity. This approach to retention results in post-clearcut management units (MUs) dominated by three cover types: (1) regenerating clearcuts plus associated vegetated riparian buffers surrounding (2) perennial/intermittent streams (i.e., streamside management zones; "SMZs") and (3) ephemeral drains (i.e., "stringers"). However, relationships between species diversity and attributes of vegetation within MUs and of the surrounding landscape are poorly understood. Thus, I investigated the potential for local-scale (internal to MUs) characteristics and landscape-scale (1- to 5- km buffers around MUs) attributes to influence avian species diversity in 3-year-old MUs. I classified and assessed land cover (imagery year: 2010) within 5 km of 60 MUs and found 90.7% of the landscape dominated by pine forests (pre-canopy closure, 21.2%; mature thinned, 14.9%; and mature closed canopy, 25.2%) and mixed pine-hardwood forest (29.4%). I used partial redundancy analysis to investigate associations of local- and landscape-scale characteristics with total species richness and density of six avian

guilds. Cover type alone explained substantial variation (partial R-sq: 0.41) in the response variables and may be sufficient to support some management decisions; however, incorporating additional local-scale characteristics into models may refine estimates of species diversity. Predicted density of mature forest-associated guilds and cover type-level species richness were greater in SMZs and stringers than in regenerating clearcuts. Local-scale vegetation and MU internal configuration entered avian species diversity models much more frequently than did landscape-scale characteristics. Birds in SMZs responded most strongly to local-scale vegetation characteristics associated with stand openness/light penetration (e.g., grassy ground cover) and mature forest structural conditions (e.g., MU percent green tree retention, overstory hardwood density). Landscape-scale metrics were less effective at explaining patterns of avian species richness and density in regenerating clearcuts and stringers. Infrequent inclusion of landscape-scale characteristics in models may have been due to the highly forested character of the region. Thus, forest managers in such regions may choose to focus on site-level considerations for sustaining MU biodiversity. Local- and broad-scale characteristics have value for explaining variation in local-scale avian diversity, particularly within SMZs; however, the substantial amount of unexplained variation in my models suggests that my pool of potential predictors was insufficient to account for the apparent influence of other, important drivers of avian diversity in recently harvested management units.

5.2 Introduction

In the southern U.S., intensively managed pine (*Pinus* spp.) forests are a dominant cover type representing 19.9% [a] of timberland in the region (Oswalt et al. 2018). Regenerating (<7 years post-establishment) intensively managed pine forests are a particularly important source of breeding habitat for shrubland-associated birds in the southern U.S. coastal plains (Legrand et al. 2007; Parrish et al. 2017b). Shrubland birds are a group that includes species in decline, such as Prairie Warbler [b] (Partners in Flight Science Committee 2012). In addition to providing early-successional land cover, intensively managed pine forest management units [c] (MUs) in the South often contain patches of retained live trees and vegetation, which support mature forest-associated birds (Parrish et al. 2017b; Parrish et al. 2018).

Avian species diversity (species richness and density) at the local patch level can be influenced by characteristics of land cover composition and configuration at multiple spatial scales [d]. Avian diversity responses to local-scale vegetation and land cover characteristics are relatively well-studied (e.g., Howell et al. 2000; Mitchell et al. 2006; Azpiroz and Blake 2016; Parrish et al. 2017b). Patch area dependencies have been identified for a number of shrubland bird species (e.g., Shake et al. 2012; Roberts and

---

[a] Based on preliminary 2017 estimates of proportion of area in planted pine (*Pinus* spp.) forests to total area in timberland in southern US (Oswalt et al. 2018).

[b] Avian taxonomic nomenclature follows American Ornithological Society (2017) Checklist of North and Middle American Birds through the fifty-seventh supplement (Chesser et al. 2016). I refer to standardized avian common names throughout this article. See Table 5.1 for scientific names.

[c] I defined 'management unit' as a pine stand (or group of stands) harvested and replanted as a unified cohort, plus any green tree retention areas associated with the harvested areas (Parrish et al. 2018).

[d] Herein, I refer to several spatial extents, paralleling the terminology of Lee and Carroll (2014). "Local scale" refers to the interior area of a management unit. "Landscape scale" may refer to several larger extents: 1-, 3-, or 5-km buffers surrounding management units.

King 2017). Retained vertical structure (i.e., live trees and vegetation), positively influenced local-scale avian richness and abundance, by providing refugia, perches, foraging and nesting substrates, and enhancing connectivity (Rosenvald and Lõhmus 2008; Culbert et al. 2013). Vegetation cover, including hardwood basal area and grassy ground cover influenced occupancy of mature forest- and shrubland-associated birds in pine forests in Georgia, USA (Lee and Carroll 2014).

Less-frequently explored is the potential for landscape context surrounding focal patches to influence avian species diversity at the focal patch level (Lee et al. 2002; Holland et al. 2004; Parrish and Hepinstall-Cymerman 2012; Lee and Carroll 2014). For example, occupancy of shrubland birds in small forest openings was positively associated with the presence of adjacent, larger patches of suitable habitat in Massachusetts, USA (Roberts and King 2017) and in Rhode Island, USA (Buffum and McKinney 2014). Avian richness responded positively to landscape-level diversity of forest age and type in forests in Arkansas, South Carolina, and West Virginia, USA (Mitchell et al. 2006). Post-breeding mature-forest birds may benefit from access to nearby, resource-rich, early-successional patches present on the landscape (Vitz and Rodewald 2006; Bowen et al. 2007; Schlossberg et al. 2018). Within managed pine forests in Georgia, USA, occupancy of pine-grassland bird species responded more strongly to local-scale vegetation characteristics, whereas interior forest species were more strongly associated with landscape-scale variables (Lee and Carroll 2014). The above examples suggest the need for a multiscale approach when assessing responses of avian community measures to forest characteristics.

To date, however, there has been little research on the effects of land cover context on birds of Southern intensively managed pine forests, even in ecoregions such as the South Central Plains. Thus, the nature of relationships between landscape attributes and bird community measures in intensively managed pine forests is not yet fully understood. Relationships between landscape characteristics and mature forest-associated species (e.g., Robinson et al. 1995; Howell et al. 2000; Betts et al. 2010; Suarez-Rubio et al. 2013) are perhaps better understood than for shrubland-associated species (Schlossberg and King 2008; Roberts and King 2017). The relative strength of avian responses to local- versus larger-scale environmental characteristics appears to be strongly species- or group-specific (Mitchell et al. 2006; Galitsky and Lawler 2015). An improved understanding of wildlife-landscape relationships would inform management of intensively managed pine forests (Miller et al. 2009; Lee and Carroll 2014). Because birds are widespread, easily observed, and responsive to landscape alteration, they are popular subjects in ecological studies exploring relationships between species diversity and landscape context (Mazerolle and Villard 1999; Rosenvald and Lõhmus 2008).

Research conclusions about the influence of land cover characteristics on local-scale avian communities are mixed (e.g., Mitchell et al. 2001; Hagan and Meehan 2002; Rodewald 2003; Betts et al. 2006; Lee and Carroll 2014). However, bird communities within primarily forested matrices may be less influenced by broad-scale conditions compared to communities within matrices more fragmented by agricultural and urban land use (Andrén 1994; Miller et al. 2004; Lee and Carroll 2014). Landowners most often have opportunities to alter internal MU conditions through retention of SMZs and stringers (Parrish et al. 2018). Improved understanding of the relative importance of plot-level vegetation, internal MU characteristics, and surrounding landscape characteristics to avian communities could enhance harvest planning decisions. To provide needed fine-scale, classified land cover for the areas surrounding my sites, my first objective was to classify and assess broad-scale land cover makeup surrounding MUs. To inform future management decisions, my second objective was to assess avian community responses to: (a) internal MU characteristics, including vegetation structural conditions as well as area and pattern of cover types within MUs, and (b) area and pattern of land cover surrounding MUs.

5.3 Methods

5.3.1 Study Area

Sometimes called the Piney Woods region (Jordan 1978; Ricketts and Dinerstein 1999), the South Central Plains ecoregion (Fig. 5.1) lies on the western edge of the U.S. southern pine belt, encompassing an area from southeastern Oklahoma and southwestern Arkansas south into eastern Texas and western Louisiana, and roughly overlapping the North American Bird Conservation Initiative's conservation region 'West Gulf Coastal Plain / Ouachitas' (Omernik 1987; US Environmental Protection Agency 2011; Bird Studies Canada and North American Bird Conservation Initiative 2014). The ecoregion has a humid subtropical climate, experiencing hot summers (mean temperatures 17° C to 21° C) and mild winters (220 to 290 frost-free days), with yearly precipitation ranging from 105 cm to 170 cm (Woods et al. 2004; Daigle et al. 2006; Wilkin et al. 2011). Perennial streams (usually low to moderate gradient) are common in the region, but despite mean rainfall of 130 cm•y^{-1}, hot summer conditions result in many smaller (intermittent) creeks experiencing seasonal dry periods (Woods et al. 2004; Wilkin et al. 2011).

Historically, forests of the northern South Central Plains were mostly comprised of mixed loblolly pine (*Pinus taeda* L.) [e] / shortleaf pine (*P. echinata* Mill.) / and various hardwoods while the southern portion of the ecoregion was dominated by longleaf pine (*P. palustris* Mill.) savannas with loblolly pine also present in creek bottoms and pine flatwoods (Reynolds et al. 1984; Bragg 2008; Wilkin et al. 2011). Today, intensively managed pine forests have a strong presence in the ecoregion (Sleeter et al. 2013; Parrish et al. 2018). In 2015, 82% of southwestern Arkansas was forestland (35% of which was planted forest), and intensive forest management was commonly employed (Rosson 2016). In 2014, 70% to 78% of the South Central Plains in Louisiana was forested, and about 63% of Louisiana forestland was planted, mostly in loblolly pine (Oswalt 2017). Arkansas and Louisiana forestry best management practices recommended retention of SMZs along perennial and intermittent waterways, primarily to maintain healthy water quality via runoff control (Louisiana Department of Agriculture and Forestry 1999; Arkansas Forestry Commission 2002).

5.3.2 Study Site Selection and Typical Management Practices

I selected 60 MUs as sample sites (Fig. 5.1) from an initial set of 1187 located in the South Central Plains ecoregion (Parrish et al. 2017a; Parrish et al. 2017b). I chose my MUs to represent the operational range of percent green tree retention in the ecoregion and the central 80% of the regional range of MU acreage (Parrish et al. 2017a). Given

[e] Tree taxonomic nomenclature follows Little (1979).

that sufficient area existed in a MU, I established two survey stations in each of its three dominant cover types: 3-year-old regenerating clearcuts (RCCs; n=118), SMZs, (n=106), and stringers (n=107), however, because some sites had limited available area in stringers and SMZs, I established only one station in stringer cover in 10 MUs and one station in SMZ cover in 18 MUs (Parrish et al. 2017b). I positioned stations >200 m apart to promote avian sampling independence.

Forest landowners in the South Central Plains ecoregion of Arkansas and Louisiana commonly harvest management units (MUs) by clearcutting with concurrent retention of patches of unharvested or partially harvested vegetation along streams and ephemeral drains, a practice commonly referred to as “green tree retention” harvesting (Parrish et al. 2018). Thus, although recently harvested MUs in the ecoregion are dominated by regenerating pine forest (mean: 80.5% of MU land cover), approximately 19% of land cover on a typical MU is in patches of retained vegetation, primarily streamside management zones ("SMZs"; 14.0% of MU land cover) and vegetated ephemeral drains ("stringers"; 3.4% of MU land cover), as well as, to a much lesser extent, vegetated wet soils areas and non-riparian patches (Parrish et al. 2018).

5.3.3 Avian Sampling

I sampled avian communities on 35 MUs between 5 May 2011 and 13 July 2011, and also on 25 other MUs between 5 May 2012 and 21 June 2012 (Parrish et al. 2017b). Avian sampling was performed between sunrise and 10:00 am, during periods of no precipitation, minimal fog, and wind speed <16 km•h^{-1} (Parrish et al. 2017b). I visited each MU three times during a single breeding season, and on each visit, conducted 10-minute 50-m-radius avian point-count surveys at each station (Parrish et al. 2017b). My “target species” were identified by excluding: (1) birds detected incidentally or beyond the 50-m count circles; (2) early and late migrants, (3) non-landbirds and/or wading birds; (4) nocturnal species; (5) broadly-ranging species, such as members of order *Accipitriformes*, families *Hirundinidae* and *Apodidae*, and genus *Corvus*; and (6) species detected fewer than 10 times (Parrish et al. 2017b).

I assigned my target species to functional guilds (Table 5.1) based on species accounts in The Birds of North America (Rodewald 2015; Parrish et al. 2017b). Shrubland-associated species were assigned to three guilds including shrub-foragers (n = 9), shrub-nesters (n = 13), and early-successional / shrubland-breeders (n = 15), and mature forest-associated species were assigned to three guilds including canopy-feeders (n = 14), canopy-nesters (n =14), and forest-breeders (n = 27). I chose shrubland-associated and mature forest-associated guilds to represent species with a broad range of life history characteristics. For each guild, I calculated maximum relative density (avian detections • ha^{-1}) by station and used those values to calculate a site-level mean by cover type for each guild. I also calculated MU-level total species richness (*SCT*) by cover type, as the number of unique target species observed over all stations in a MU within a given cover type. I thus derived seven avian species community metrics (Table 5.2) that I used as response variables in subsequent modeling.

5.3.4 Local-scale Cover Sampling

I defined my “local scale” of interest as the area encompassed by the bounds of a given MU. In an earlier phase of my study, I estimated the following vegetation metrics at each avian sampling station: stem density of trees (classes: hardwood [f] and pine) at the overstory and midstory strata [g], stem density of pines at the understory strata, and percent grassy ground cover (Parrish et al. 2017a). I calculated station-level mean stem densities by class and mean percent grassy ground cover across stations. I treated the station-level means as subsamples within MUs, and used them to calculate MU-level means by cover type, to be used as local-scale metrics.

I previously delineated land cover within each of my 60 MUs using screen digitization in ArcGIS Desktop 9.3 [h] at a 1:5000 digitization scale (Parrish et al. 2018). My primary base layer for classifications was aerial imagery from the 2010 high-resolution, true color National Agricultural Imagery Program dataset, although other imagery products were sometimes used to inform my interpretation of the base layer (Parrish et al. 2018). Regenerating clearcuts, SMZs, and stringers were the primary cover types within MUs, accounting for approximately 98% of the typical MU (Parrish et al. 2018). Therefore, my analyses focused on those three land cover classes.

Using ZonalMetrics Toolbox in ArcGIS 10.3.1 (Adamczyk and Tiede 2017) and FRAGSTATS v4 (McGarigal et al. 2012), I calculated the following internal MU pattern metrics (Parrish et al. 2018): class mean patch area (units: ha; classes: RCC, SMZ); class mean radius of gyration [i] (a measure of within-patch linear traversal extent; units: m;

---

[f] For simplicity herein, I refer to deciduous tree species collectively as 'hardwoods.'
[g] I previously defined vegetation strata as follows: overstory (uppermost canopy layer); midstory (3 m height to underside of overstory canopy); understory (1 m to 3 m height); ground cover (< 1 m height; Parrish et al. 2017b). Overstory- and midstory-level vegetation was virtually absent from regenerating clearcuts, so I therefore omitted stem density variables from those strata in clearcut models.
[h] ArcGIS® 9 and 10 for Desktop software is copyright © 1999-2015 by Environmental Systems Research Institute (ESRI), Redlands, CA, and is used herein under license.
[i] Mean radius of gyration was the only metric derived using FRAGSTATS. I used a 1-m raster cell size to match my base layer resolution. Remaining metrics were derived using ZonalMetrics Toolbox and were directly calculated using vector data.

classes: RCC, SMZ); class mean edge density (units: m • ha^{-1}; class: SMZ); number of patches of all classes within MU; MU percent land cover in green tree retention, and MU total area (units: ha). I also determined latitude and longitude at each avian sampling point. I assumed that internal MU pattern metrics were representative of all avian sampling stations located within a MU.

5.3.5 Landscape-scale Cover Sampling

I delineated patches of 2010 land cover in 1-km, 3-km, and 5-km buffers surrounding each MU using screen digitization in ArcGIS 10.3.1, at a 1:15000 digitization scale. Landscape-scale land cover was delineated into eight classes (Table 5.3) based on visual interpretation of aerial imagery and hydrography data. Classes consisted of: pre-closure pine forest (*PE*); thinned pine forest (*PT*); closed-canopy pine forest (*PC*); utility right-of-way; hardwood and mixed pine-hardwood forest (*MPH*); agricultural / cultivated land use; urban / residential land use; and lakes, ponds, and wetlands. My primary aerial photo base layer was true-color, high-resolution (1-m pixels) National Agricultural Imagery Program imagery acquired in 2010, although I sometimes also used several other contemporary (2006-2013) aerial photo datasets in both true-color and color infrared to inform my interpretation of landscape features in the 2010 imagery (Table 5.4).

Terrestrial land cover features were directly screen digitized. Ponded aquatic features were imported using hydrography data from the high-resolution (1:24000) National Hydrography Dataset, which were current through September 2013 in Arkansas and February 2014 in Louisiana (Table 5.4). Certain aquatic features were poorly represented by the National Hydrography Dataset, and in those cases, I corrected my map by screen-digitizing those features using aerial imagery. I digitized rights-of-way by identifying their center lines from aerial imagery, measuring width at five equidistant points along the rights-of-way, and buffering the right-of-way centerline by the estimated mean right-of-way width . After completing digitization, I eliminated sliver polygons (terrestrial polygons <1000 m^2 and water polygons <500 m^2) by collapsing them into neighboring polygons. I validated my final map topology using the topology toolset in ArcGIS to identify and correct any topological errors (e.g., slivers or gaps in land cover). Using ZonalMetrics Toolbox in ArcGIS 10.3.1, I calculated percent of the landscape in each cover class at each of the three buffer scales (1-km, 3-km, and 5-km). I assumed that landscape-scale metrics were equally representative of all points within management units.

5.3.6 Selection of Forest Metrics

Using the CORR procedure in SAS, I calculated Pearson correlation coefficients between forest metrics to identify and remove redundant, highly-correlated terms ($|r| > 0.65$) from the following set of predictor metrics. Local-scale characteristics consisted of: MU percent retention; MU area; SMZ edge density, mean patch area and mean radius of gyration; regenerating clearcut mean patch area and mean radius of gyration; MU number of patches; and vegetation characteristics (percent ground cover (class: grasses) and stem density (classes: understory pine, midstory pine and hardwood, and overstory pine and hardwood). Landscape-scale (1-, 3-, and 5-km buffers) characteristics were comprised of land cover (classes: PE, PT, PC, and MPH). I excluded from further analyses landscape-scale cover classes that were poorly-represented (accounting for <4% of land cover).

5.3.7 Ordination Analysis

Redundancy analysis ("RDA"; Legendre and Legendre 2012; Šmilauer and Lepš 2014) is a form of constrained, linear ordination (i.e., a direct gradient analysis) that allows the user to simultaneously analyze linear relationships between a set of multiple explanatory variables (i.e., environmental / landscape characteristics) and a set of multiple response variables (i.e., avian density and species richness). I conducted partial RDA, a variant of RDA in which some covariate effects are initially specified and then removed before the analysis is carried out against the remaining variability (Davies and Tso 1982; Legendre and Legendre 2012; Šmilauer and Lepš 2014). In all analyses, I included as response variables the following avian metrics, which I henceforth refer to collectively as my set of 'responses': cover type-level species richness (*SCT*) and relative density of shrub-foragers (*DSF*), shrub-nesters (*DSN*), shrubland breeders (*DSB*), canopy-foragers (*DCF*), canopy-nesters (*DCN*), and mature forest-breeders (*DFB*). I centered and standardized response variables in all analyses, and defined significance levels at $\alpha = 0.05$.

From my RDA results, I developed ordination biplot figures to graphically depict associations between my sets of response and explanatory variables. In a biplot, continuous response variables and continuous explanatory variables are displayed as lines emanating from the origin, with arrows pointing in the direction of the maximum increase in the variable along the ordination axes, where longer lines indicate a greater rate of change (ter Braak and Prentice 1988; Šmilauer and Lepš 2014). Longer arrows denote more important variables (ter Braak and Prentice 1988). The angle between an explanatory variable and a response variable is representative of their correlation

(Legendre and Legendre 2012): narrower angles indicate greater correlation; a right angle indicates no correlation; and more obtuse angles indicate negative correlation. (Šmilauer and Lepš 2014). Factor-type explanatory variables are displayed as points representing class-level centroids (means) of case scores (Šmilauer and Lepš 2014). Projecting the centroid point onto a response variable line at a right angle again suggests the relationship between the two (Legendre and Legendre 2012). Higher values of the response variable are associated with centroids that are a shorter distance from the response's arrowhead. CANOCO 5 software was utilized to perform analyses and develop biplot figures (ter Braak and Šmilauer 2017). Within the above framework, the biplot may be interpreted to provide relative strength of associations between explanatory variables and responses. Because the ordination axes in constrained ordinations such as RDA represent linear combinations (i.e., weighted sums) of explanatory variables, ecological meaning can be attributed to the axes based on how the explanatory variables vary along each axis (Šmilauer and Lepš 2014). For an excellent technical discussion of the interpretation of ordination distance biplots, refer to Legendre and Legendre (2012).

After I eliminated redundant metrics, I developed a pool of land cover characteristics at the local scale (within MU) and the landscape scale (3-km buffers around MUs) that I utilized as potential explanatory variables in a series of ordination analyses. At the local scale, explanatory variables included: latitude, MU area, MU percent green tree retention, MU number of patches, mean patch area of regenerating clearcuts, hardwood stem density (overstory and midstory strata), pine stem density (overstory, midstory, and understory strata), and percent grassy ground cover. Landscape-scale variables consisted of percent land cover measures (classes: pre-canopy closure pine forest, thinned pine forest, closed-canopy pine forest, and mixed pine-hardwood forest).

I ran a partial RDA to test for an effect of cover type classes (three classes: regenerating clearcut, SMZ, and stringer) on response variables. I defined cover type as the explanatory variable, the six avian density metrics and total richness metric as response variables, and used site (i.e., MU) and latitude as covariates. I specified unrestricted permutation tests to be carried out in blocks defined by site. Concurrently, I ran two partial RDAs by year (2011 and 2012 observations), using identical model parameters to those described above. My purpose in generating the year-effect models was to help us assess whether year-to-year variations made a noteworthy difference in model outcomes, or whether the model lacking a year-effect was sufficient to provide more generalized conclusions. I compared partial R^2 of the year-effect models to that of the non-year-effect model, and also examined resulting biplots to view consistency of variable relationships among models.

Because cover type was included ($P < 0.03$) in the previous avian responses model, I was interested in whether cover type might interact with the remaining members of my explanatory variable pool in affecting avian responses. Most of the potential explanatory variables in my pool did not vary, or varied only slightly, within sites, precluding the use of cover type as a covariate alongside site (since most or all variation would be explained by site). I conducted a partial RDA that tested for significant interactions between cover type and each member of the explanatory variable pool. In that analysis, I defined explanatory variables as interaction terms (i.e., cover type interacting with each of the other landscape metrics), and specified latitude, cover type, and each of the 14 individual landscape metrics as covariates. I carried out forward model selection (α_{enter} = 0.05) on my seven response variables with unrestricted permutations.

To address potential interactions between cover type and several variables, I opted to conduct further partial RDA by cover type. I carried out three separate partial RDAs, with unrestricted permutations, using forward model selection on data from (1) regenerating clearcuts, (2) SMZs, and (3) stringers. I used my pool of 17 potential explanatory variables in all models, with the exception that I omitted midstory and overstory vegetation variables from my regenerating clearcut model, because those vegetative strata were functionally absent in that cover type. I defined latitude as a covariate in each RDA. As with the earlier cover type-only model, I again generated parallel sets of models (for each cover type) by year (2011 and 2012) to assess whether accounting for year-to-year variations made noteworthy differences in model outcomes, or whether models lacking a year-effect were sufficient to provide more generalized conclusions. I compared partial R^2 of the year-effect models to that of the non-year-effect model, and also examined resulting biplots to view consistency of variable relationships among models.

5.4 Results

5.4.1 Internal Management Unit Characteristics

My 60 study sites were bounded by latitudes 30.9°N and 34.4° N and longitudes 91.8° W and 93.5° W. Estimated internal MU land cover characteristics are presented in Table 5.5. Regenerating clearcuts were dominated by planted understory pine (mean ± sd: 930.6 stems•ha^{-1} ± 725.3) and grassy ground cover (59.1% ground cover ± 22.8%;

Table 5.6). Regenerating clearcuts were functionally without a midstory or overstory. Compared with regenerating clearcuts, SMZs and stringers were characterized by the presence of well-developed midstory and overstory vegetative strata and lower levels of understory pine stem density and grassy ground cover.

In a separate analysis of the SMZ and stringer dataset (Parrish et al. 2017a), overstory hardwood and pine stem density was greater in SMZs than in stringers. Midstory stem density of hardwood trees was greater in SMZs than stringers, but there was no difference in midstory pine density. Understory pine density was greater in stringers than in SMZs, but there was no significant difference in percent grassy ground cover between SMZs and stringers.

5.4.2 Landscape Characteristics Surrounding Management Units

Pine forest (classes: *PE*, *PT*, and *PC*) and mixed pine-hardwood forest (class: *MPH*) were the dominant land cover classes at 1-km, 3-km, and 5-km scales, with small changes in relative ranking among the three most extensive classes (classes: *PE*, *PC*, and *MPH*; Table 5.7). At the 5-km scale, pine forests accounted for an average of 61.3% of broad-scale land cover, while total forested land cover (classes: *PE*, *PT*, *PC*, and *MPH*) accounted for an average of 90.7% of broad-scale land cover (Table 5.7). At the 1-km scale, mean total forested land cover was 94.6% (Table 5.7). The remaining four uncommon cover classes (i.e., agriculture; urban/high intensity; open water and wetlands; and utility rights-of-way) each accounted for <4% of land cover at any scale (Table 5.7) and I did not include them in further analyses.

5.4.3 Spatial Metric Correlations and Predictor Pool Development

Within my pool of predictors, correlations between retained metrics pairs were all less than $|r| = 0.57$ ($P < 0.03$ for all significant correlations). Based on correlation results, I retained in my pool of predictors the following local-scale (internal MU) characteristics: MU area, MU number of patches, MU mean clearcut patch area, MU percent green tree retention, tree stem density (classes: understory pine, midstory pine and hardwood; overstory pine and hardwood) and percent grassy ground cover. I also retained as predictors landscape-scale (3-km buffers) percent land cover estimates (classes: *PE*, *PT*, *PC*, and *MPH*).

5.4.4 Avian Characteristics

I included 42 target bird species (Table 5.1) in my analyses based on their assignment to one or more guilds of interest (Parrish et al. 2017b). Each species was assigned membership in up to three guilds: foraging level (shrub, n=9 spp.; canopy, n=14 spp.); nesting level (shrub, n=13 spp.; canopy, n=14 spp.); and predominant breeding habitat (shrubland, n=15 spp.; mature forest, n=27 spp.). I assigned 19 species to three guilds, 12 species to two guilds, and 11 species to a single guild.

Mean *SCT* was similar between SMZs and stringers, and was greater in both retention cover types than in RCCs (Table 5.2; Parrish et al. 2017b). I previously reported ANOVA contrasts between mean avian relative density by functional guild affiliation between pairings of the three dominant cover types (Parrish et al. 2017b), which I summarize below and in Table 5.2 for the guilds analyzed herein. Shrub-forager mean density was greatest in RCCs, followed by stringers then SMZs. Mean densities of shrub-nesters and shrubland-breeders were not significantly different between RCCs and stringers, and were lower in SMZs. Mean density of canopy-forager, canopy-nester, and mature-forest breeder guilds all were greatest in SMZs, followed by stringers, then RCCs.

5.4.5 Ordination Analysis Results

I detected a significant effect of cover type on my response variables (P = 0.002; Fig. 5.2) in the cover type model lacking a year effect. Cover type accounted for 41% of the partial variation (partial R^2 = 0.410) in the avian responses after latitude and site effects were accounted for. Partial R^2 estimates for the 2011 and 2012 year effect models were 0.380 and 0.480, respectively, and biplots indicated no substantial differences in variable relationships compared with those in the non-year-effect model. Therefore, I retained the model lacking a year effect, and discuss it below. My assessment of the resulting biplot suggested that density of shrubland guild birds (*DSN*, *DSB*, *DSF*) was relatively greater (and similar) in regenerating clearcut and stringer cover types, compared with in SMZ cover. Total species richness by cover type (*SCT*) and density of mature forest-associated guilds (*DCF*, *DCN*, *DFB*) were greatest in SMZ patches, followed closely by stringers, and lowest in regenerating clearcut patches.

I identified five interaction terms involving cover type that significantly affected my set of avian responses: cover type X landscape-scale percent early successional pine forest (P = 0.018); (2) cover type X local-scale percent green tree retention (P = 0.016); (3) cover type X local-scale percent grassy ground cover (P = 0.024); (4) cover type X understory pine stem density (P = 0.032); and (5) cover type X midstory pine stem density (P = 0.032). Because cover type influenced associations between other explanatory variables and my set of avian responses, I performed further analyses by cover type. Because each cover type-level model was derived from a different data subset, I interpreted each resulting model independently by cover type class.

The explanatory value of my SMZ model lacking a year effect term (partial R^2 = 0.290) was nearly identical to explanatory value of the 2011 and 2012 year-effect models (partial R^2: 0.296 and 0.291, respectively), and my examination of biplot relationships suggested minimal differences between models. Therefore, I retained the SMZ model without a year effect (Fig. 5.3) and discuss it below. My SMZ model (Fig. 5.3) included percent green tree retention (P = 0.002), percent grassy ground cover (P = 0.004), overstory hardwood stem density (P = 0.02), understory pine stem density (P = 0.034), and regenerating clearcut mean patch area (P = 0.042). My assessment of the resulting biplot suggested that density of shrubland birds (*DSF*, *DSB*, *DSN*) was negatively associated with greater percent green tree retention and overstory hardwood stem density, and positively associated with variables associated with more open forest characteristics (i.e., higher levels of understory pine stem density, percent grassy ground cover, and mean patch area of regenerating clearcuts). In SMZs, mature forest-associated guilds (*DCN*, *DFB*, *DCF*) and total species richness (*SCT*) had strong, negative associations with understory pine density, percent grassy ground cover, and mean patch area of regenerating clearcuts, comparatively moderate negative associations with overstory hardwood stem density, but were positively associated with percent green tree retention. The SMZ model accounted for 29.0% of variation in avian diversity metrics after accounting for latitude. The first two axes of the resulting biplot accounted for 53.2% and 97.8% of the cumulative explained fitted variation, respectively.

My regenerating clearcut and stringer models both performed weakly, respectively explaining only 13.1% and 17.7% of variation in avian guild diversity metrics after accounting for latitude effects. The two year-effect model pairs for clearcuts and stringers did not improve model explanatory value. I discuss the non-year-effect clearcut and stringer models below.

My regenerating clearcut model (Fig. 5.4) included regenerating clearcut mean patch area ($P = 0.01$) and understory pine stem density ($P = 0.046$). My assessment of the resulting biplot suggested that densities of forest breeders (*DFB*) and forest canopy nesters (*DCN*) were negatively associated with understory pine density and had little meaningful association with mean clearcut patch area. Density of shrubland-associated guilds (*DSB*, *DSN*, *DSF*), forest canopy foragers (*DCF*) and total species richness by cover type (*SCT*) appeared to have strong, negative associations with mean clearcut patch area and moderate negative associations with understory pine stem density. The first two axes of the resulting biplot accounted for 83.29% and 100% of the cumulative explained fitted variation, respectively.

My stringer model (Fig. 5.5) included landscape-scale percent early successional pine forest ($P = 0.016$), overstory hardwood stem density ($P = 0.006$), and percent grassy ground cover ($P = 0.04$). Percent of the landscape in young pine MUs was associated with lower stringer density for all guilds, but the strongest negative association was with total species richness in stringers (*SCT*). Shrubland-associated guilds had positive associations with percent grassy ground cover. Density of forest-associated guilds and total species richness in stringers exhibited a negative association with percent grassy ground cover. The first two axes of the biplot accounted for 70.30% and 96.81% of the cumulative explained fitted variation, respectively.

5.5 Discussion

5.5.1 Landscape-scale Land Cover Assessment

The landscape-scale land cover dataset I created was unique in having a fine-grained resolution relative to its broad spatial extent. My estimate of mean percent forested land cover (90.7% within 5-km buffers [j]) was in close agreement with county/parish-level [k] estimates (85.1% forested land cover) obtained through the U.S. Forest Service's Forest Inventory & Analysis (FIA) EVALIDator tool (USDA Forest Service 2018). The 3-year-old commercial pine MUs that served as my study sites defined the centers of the 5-km buffers in which I sampled land cover; consequently, they were located in areas dominated by intensively managed forest, which likely accounts for my slightly greater estimate of forested land cover compared to FIA data. Agriculture, urban uses, and water bodies made up only a small proportion of the sampled landscape in my study area. My estimates are therefore most applicable to understanding regional forest makeup within landscapes dominated by institutional forestland, such as to the South Central Plains.

At 5-km scales, I found that 36.5% of the landscape mosaic was maintained in early-successional land cover, in the form of pre-canopy closure and recently-thinned

---

[j] In my assessment of landscape-scale land cover, I refer to estimates made using 5-km buffers. In later avian modeling, I utilized 3-km buffers.

[k] Arkansas counties: Bradley, Calhoun, Clark, Cleveland, Dallas, Drew, Grant, Hot Spring, Jefferson, and Union. Louisiana parishes: Allen, Bienville, Caldwell, Catahoula, Claiborne, LaSalle, Rapides, Sabine, Union, Vernon, and Winn parishes. Clark and Jefferson counties (AR) and Rapides and Catahoula parishes (LA) contained only small portions of my study area and were omitted.

pine forests and shrubby utility rights-of-way. Early-successional cover is an ephemeral and declining resource critical to the conservation of a wide variety of organisms in the South (Trani et al. 2001; King and Schlossberg 2014; Owens et al. 2014; Greene et al. 2016; Demarais et al. 2017), and these intensively managed pre-closure and recently-thinned pine forests thereby represent a primary sources of early-successional cover in the region. Peak avian diversity is expected during the early-rotation phase (typically years two to six), with a smaller peak often occurring post-thinning (Miller et al. 2009), making these early-successional MUs an important source of regional bird diversity (Parrish et al. 2017b). These open forests are the result of active forest management practices, through which forest owners maintain a perpetual, regional mosaic of early-successional forest patches that is continually replenished through rotating harvests.

One quarter of the sampled landscape was in closed-canopy pine forest. In the South, large private landowners typically manage pine forests for multiple products using even-aged rotations with a first thinning at about age 15, a second thinning at about age 21, and a final harvest around age 33 (Lang et al. 2016). Therefore, closed-canopy conditions are typically ephemeral, and a significant portion of my landscape is likely to be in an open-canopy condition throughout much of the rotation. Although studies of managed pine forests in the South have noted declines in species diversity in closed canopy stands (e.g., Childers et al. 1986; Dickson et al. 1993; Conner and Dickson 1997; Krementz and Christie 1999; Loehle et al. 2005), some bird species such as Swainson's Warbler (*Limnothlypis swainsonii*) that are usually associated with deciduous forests have been recorded breeding in closed-canopy managed pine forests (Graves 2015; Henry et al. 2015).

On average, 29.4% of my study landscape was comprised of hardwood or mixed-pine hardwood forest (i.e., MPH forest). FIA forest cover data (USDA Forest Service 2018) classified 26.7% of the counties and parishes within which I sampled [k] as MPH forest classes [l], closely agreeing with my estimate. Mature forest-associated bird species have been documented making intensive use of early-successional habitats during spring migration as well as during the late-summer post-natal period and fall migration. Examples include forest birds using bottomland forest gaps in South Carolina (Bowen et al. 2007) and hardwood system clearcuts in Missouri, Ohio, New Hampshire, Virginia, and West Virginia, USA (Pagen et al. 2000; Marshall et al. 2003; Vitz and Rodewald 2006; Chandler et al. 2012). In hardwood bottomland stands within a managed pine forest matrix in South Carolina, total avian density was greater within the hardwoods compared with the pine stands, but mature forest-associated species were expected to cross over and utilize structurally similar patches in mid- to late-rotation pine management units, and vice-versa (Turner et al. 2002). In hardwood-pine systems in Massachusetts, USA, breeding season Wood Thrush (a mature forest-associated species) abundance was greater in mature forests with associated patches of early-successional cover (Schlossberg et al. 2018). MPH forests proximate to managed pine forests likely represent a source of recolonizing forest-associated species, particularly when linked to the pine forests via SMZ corridors.4.2 Management Unit Cover Type Associations with Avian Community Measures

[l] FIA land cover classes: oak/pine (OkP), oak/hickory (OkH), oak/gum/cypress (OkGC), elm/ash/cottonwood (EAC), other hardwoods (OtHw)

My internal MU-scale cover type model accounted for a substantial amount of partial variation (40.96%) in avian community responses after latitude and site effects were accounted for, and was my strongest model. Other researchers have previously recognized the utility of cover type associations for understanding bird communities (e.g., Hamel 1992; Conner and Dickson 1997; Kilgo et al. 2002). I detected mild year effects in my response variables, which I attributed to annual environmental variation; however, my comparative analyses yielded consistent models regardless of inclusion of year as a model parameter. Because I was most interested in identifying generalized relationships between land cover metrics and avian community measures that could offer broader applications, I opted to omit year effects from my models.

My present findings confirmed conclusions I drew from an earlier assessment of my bird dataset. Specifically, I observed that SMZ and stringer cover types exhibit similarity in bird guild density, and appear to work synergistically to augment site species richness by hosting a suite of mature forest-associated bird species in addition to the shrubland-associated guild members that are present in all three primary cover types (Parrish et al. 2017b). I suspect this is a consequence of the vegetation structural similarity between stringers and SMZs (Parrish et al. 2017a), which may result in both cover types providing similar services to birds. Sixteen of 60 species appeared to be mature forest obligates, and were only detected in SMZs and stringers where patch characteristics such as overstory tree density and shady conditions met their specialized requirements. I also noted similarities in shrubland-associated bird densities between regenerating clearcuts and stringers (Parrish et al. 2017b). Stringers are relatively narrow, open patches, and that configuration appears likely to appeal to shrubland-associated birds for use as call posts, foraging sites, and nesting sites. A number of shrubland-associated species appear to prefer taller shrubby structures (Schlossberg et al. 2010), which suggests that stringers may attract those species from nearby regenerating clearcuts. Because most shrubland-associated bird species were present in SMZs, stringers, and regenerating clearcuts, but many forest-associated species occurred only in SMZs and stringers (Parrish et al. 2017b), I expected that total species richness would be correlated with land cover characteristics that were also associated with greater density of mature forest-associated guilds. This pattern was observed in the ordination biplot for my cover type model and was also seen in biplots for my SMZ and stringer models.

5.5.2 Local and Landscape Characteristics Associated with Avian Species Community Measures

My internal MU characteristic models varied in their predictive utility. My model of avian community measures in SMZs explained nearly 30% of the partial variation in avian total species richness and guild densities. Partial R^2 values for my models of avian community measures in regenerating clearcuts and stringers were < 18%, and lacked strong explanatory value.

Interpretation of my RCC model proved challenging. Given the poor model performance (partial $R^2 = 0.131$), it seems that my pool of explanatory variables was insufficient for explaining patterns of shrubland- and mature forest-associated guild densities in clearcuts. I was surprised that density of shrubland-associated birds in regenerating clearcuts was negatively associated with mean clearcut patch size, because studies have often shown a positive association between shrubland bird abundance and early-successional habitat area and my regenerating clearcuts were considerably larger than published minimum areas for area-sensitive shrubland birds (Rodewald and Vitz 2005; Roberts and King 2017). However, the weak fit of this model suggests that this finding may be tenuous.

The SMZ model was my strongest model, accounting for 29.0% of the partial variation in avian responses after accounting for latitude. Again, total species richness and density of mature forest-associated guilds responded to environmental characteristics similarly, reflecting the increase in richness associated with forest birds exclusively occupying SMZs and stringers. I interpreted the X-axis in the ordination biplot to represent characteristics of larger SMZs dominated by mature hardwood forest (increasing to the right). I interpreted the Y-axis to reflect open-canopy conditions (increasing to the top). Density of shrubland guilds present in SMZs responded negatively to increasing overstory hardwood stem density and percent green tree retention. Density of shrubland foragers responded positively to increasing light penetration / openness in SMZs, but mature forest-associated guild densities and total species richness responded negatively along that axis. In Missouri, density of mature forest-associated avian species was greatest in wider riparian zones with high percent canopy cover and low percent grassy ground cover, while shrubland-associated species richness was greatest in narrower, shrub-bordered riparian zones with greater grassy ground cover and more open canopies (Peak and Thompson 2006), which is consistent with my interpretation.

In SMZs, mature forest-associated guild densities and total species richness were positively associated with percent green tree retention while shrubland-associated guilds responded negatively. This observation may be a function of SMZ extent being related to

MU percent green tree retention. Because SMZ count-stations tended to be located towards the patch interior, SMZ count-stations on MUs with greater percent retention might have been located deeper in the forest. In east Texas, USA, bird assemblages in wide SMZs favored interior forest species, while shrubland-associated species were more abundant in narrower patches (Dickson et al. 1995).

The stringer model accounted for < 18% (partial $R^2 = 0.177$) of the partial variation in the avian response variables. I interpreted the X-axis of the ordination biplot to reflect dominance of hardwood overstory trees in stringers and extent of early-successional forestland in the surrounding landscape (3-km buffer around MUs). The Y-axis appeared to reflect percent grassy ground cover. Shrubland-associated guild densities in stringers were positively associated with grassy ground cover and negatively associated with overstory hardwood stem density, as in my SMZ model. Peak and Thompson (2006) reported that shrubland birds responded positively to grassy ground cover and canopy openness in narrow riparian zones. Occupancy of early-successional birds in Connecticut, USA was not related to surrounding landscape context, which was attributed to adaptations by those species to exploit isolated, ephemeral patches of shrubland on a forested landscape (Askins et al. 2007). I found that all guild densities within stringers were negatively associated with percent of the broad-scale landscape in early-successional pine forest, but I consider this relationship tenuous, given the poor model explanatory value. Mature forest-associated guild density and total species richness in stringers responded negatively to percent grassy ground cover and overstory hardwood stem density, similar to the SMZ model. This might reflect a preference by mature forest-associated birds for microhabitats containing a shady, yet open understory.

5.5.3 Landscape Characteristics not Associated with Avian Species Diversity

Bird guilds in my study responded to understory and ground level vegetation (percent grassy ground cover and understory pine density) and overstory hardwood trees, but stem density of midstory trees and of overstory pines were not included in any models. Shrubland- and mature forest-associated bird species in restored savannas and woodlands in Missouri responded (mostly negatively) to sapling stem density (Reidy et al. 2014). However, only one of 40 habitat suitability indices constructed for landbird species of the Central Hardwoods and West Gulf Coastal Plain/Ouachitas Bird Conservation Regions included midstory stem density (Tirpak et al. 2009). In managed pine forests in east-central Mississippi, USA, mid-rotation herbicide application and controlled burning reduced midstory hardwoods in pine stands, increasing abundance of multiple species that favored open, early-successional conditions (Iglay et al. 2018).

Management unit area was not included in any of my models. The central 80% of MUs, when ranked by MU area, ranged from approximately 34 ha to 77 ha, with a corresponding mean SMZ patch size of 5.3 ha and mean clearcut patch size of 26 ha. Thus, I expected my MU areas to exceed minimum area requirements for breeding populations of shrubland- and mature forest-associated species on my sites (Kilgo et al. 1998; Roberts and King 2017). Exclusion of MU area from my models likely suggests that the variable generally exceeded minimum requirements for the breeding birds present.

Landscape-scale percent of landscape surrounding MUs in thinned pine, closed-canopy pine, or mixed pine-hardwood forest types were not significantly associated with guild density or total species richness in any of the three primary cover types. As a single exception, percent of the landscape (3-km buffer) in pre-canopy closure pine forest entered one model (for stringers), although that model had a weak explanatory value (partial $R^2 = 0.177$). A possible explanation for this finding is that there was relatively little contrast among patches in my forest-dominated study region especially when compared to forest patches embedded within a landscape dominated by agricultural / urban land uses.

My study landscape was >90% forested, with >60% in pine forests and approximately 30% in hardwood / mixed pine-hardwood forest. Urban land use, agricultural land use, and ponds/lakes/wetlands classes each comprised < 4% of the sampled landscape matrix. Landscape-scale effects on avian communities appear to be less important than local-scale effects on sites in lower-contrast, forest-dominated matrices such as ours, versus sites embedded in agriculture/urban-dominated matrices (Hagan and Meehan 2002; Lichstein et al. 2002; Rodewald 2003; Betts et al. 2006; Lee and Carroll 2014); but also see MacFaden and Capen (2002). For example, within managed forest-dominated landscapes in Maine, USA, occupancy of 20 bird species was explained better by stand-level characteristics than by landscape-scale characteristics, although some species also responded to some landscape-scale metrics (Hagan and Meehan 2002). In managed forest landscapes in the Greater Fundy Ecosystem, NB, Canada, 21 bird species responded more strongly to local- than to landscape-scale characteristics (Betts et al. 2006). In the South, pine forests within urban/agricultural matrices made up > 30% of the landscape, and forest patches appeared functionally connected sufficiently to contribute to avian diversity at the landscape scale (Lee and Carroll 2014). In fact, some studies have documented positive relationships between avian community measures and heterogeneity in forest age and type within working forest landscapes (e.g., Loehle et al. 2005; Mitchell et al. 2006). In light of my study region's high proportion of forested land cover, it may not have been optimal for detecting broad-scale landscape effects on birds.

5.6 Management Implications

Within a forest-dominated (>90% forested) landscape, density of six avian guilds and total species richness in three-year-old, intensively managed pine forests were most strongly associated with cover type and internal MU land cover characteristics. Retention of SMZs and stringers was associated with greater mature forest-associated avian guild densities and total species richness, highlighting the contributions of retained patches to avian communities in management units. Simple cover type information explained approximately 41% of the partial variation in guild densities and total species richness, and may be sufficient to address basic land owner questions. Internal MU land cover characteristics appeared to influence avian guild density and richness, perhaps most strongly within SMZs; but, my models incorporating internal characteristics varied in predictive strength. Landscape-scale characteristics external to MUs did not appear to strongly influence avian response variables measured within management units. Efforts to promote and sustain biodiversity in recently-harvested industrial pine forests currently emphasize manipulation of internal MU characteristics (e.g., retained structural features related to green tree retention) and my results do not suggest any change in that focus should occur at this time. The substantial amount of unexplained variation in my models suggests the influence of other important drivers of species diversity (e.g., unmeasured vegetation characteristics, topography, microclimate variables, or other temporal or spatial scale effects). Demographic outcomes related to local-scale and landscape characteristics were not addressed in my study, but might represent interesting avenues of future research. My findings are most applicable to institutional pine forests in predominantly forested regions, and could potentially be less applicable to forests in landscapes with greater regional agriculture and urban land use.

Table 5.1 Common name, species, and guild assignments for shrubland-associated and mature forest-associated species.

Common Name	**Species**	**Functional guild assignments** [a]		
		Foraging strata	**Nesting strata**	**Breeding habitat**
Northern Bobwhite	*Colinus virginianus*	-	-	S
Mourning Dove	*Zenaida macroura*	-	C	S
Yellow-billed Cuckoo	*Coccyzus americanus*	C	C	F
Ruby-throated Hummingbird	*Archilochus colubris*	S	C	F
Red-headed Woodpecker	*Melanerpes erythrocephalus*	-	-	S
Red-bellied Woodpecker	*Melanerpes carolinus*	-	-	F
Downy Woodpecker	*Picoides pubescens*	-	-	F
Northern Flicker	*Colaptes auratus*	-	-	F
Eastern Wood-Pewee	*Contopus virens*	C	C	F
Acadian Flycatcher	*Empidonax virescens*	C	C	F
Great Crested Flycatcher	*Myiarchus crinitus*	C	-	F
Eastern Kingbird	*Tyrannus tyrannus*	S	C	F
White-eyed Vireo	*Vireo griseus*	C	S	S
Yellow-throated Vireo	*Vireo flavifrons*	C	C	F
Red-eyed Vireo	*Vireo olivaceus*	C	C	F
Blue Jay	*Cyanocitta cristata*	-	C	F
Carolina Chickadee	*Poecile carolinensis*	C	-	F
Tufted Titmouse	*Baeolophus bicolor*	C	-	F
Brown-headed Nuthatch	*Sitta pusilla*	C	-	F
Carolina Wren	*Thryothorus ludovicianus*	-	-	F
Blue-gray Gnatcatcher	*Polioptila caerulea*	C	C	F
Eastern Bluebird	*Sialia sialis*	-	-	S
Wood Thrush	*Hylocichla mustelina*	-	S	F
Gray Catbird	*Dumetella carolinensis*	-	S	F
Brown Thrasher	*Toxostoma rufum*	-	S	F
Black-and-white Warbler	*Mniotilta varia*	-	-	F
Kentucky Warbler	*Geothlypis formosa*	-	-	F
Common Yellowthroat	*Geothlypis trichas*	S	S	S

Table 5.1 (continued)

Common Name	Species	Functional guild assignments [a]		
		Foraging strata	Nesting strata	Breeding habitat
Hooded Warbler	*Setophaga citrina*	S	S	S
Northern Parula	*Setophaga americana*	C	C	F
Pine Warbler	*Setophaga pinus*	C	C	F
Prairie Warbler	*Setophaga discolor*	S	S	S
Yellow-breasted Chat	*Icteria virens*	S	S	S
Eastern Towhee	*Pipilo erythrophthalmus*	-	-	S
Field Sparrow	*Spizella pusilla*	-	-	S
Summer Tanager	*Piranga rubra*	C	C	F
Northern Cardinal	*Cardinalis cardinalis*	-	S	F
Blue Grosbeak	*Passerina caerulea*	S	S	S
Indigo Bunting	*Passerina cyanea*	S	S	S
Painted Bunting	*Passerina ciris*	-	S	S
Brown-headed Cowbird	*Molothrus ater*	-	S	S
Orchard Oriole	*Icterus spurius*	S	C	F

[a] Each species was assigned membership in up to three guilds: foraging level (shrub "S", *n*=9 spp.; canopy "C", *n*=14 spp.); nesting level (shrub "S", *n*=13 spp.; canopy "C", *n*=14 spp.); and predominant breeding habitat (shrubland "S", *n*=15 spp.; mature forest "F", *n*=27 spp.). If life history characteristics did not fall within those levels (indicated with "-" sign), the species was not analyzed with the guild(s).

Table 5.2 Site-level mean, standard deviation, and range of avian total species richness and guild densities by cover type (regenerating clearcut, "RCC", *n*=60 sites; streamside management zone, "SMZ", *n*=60 sites; and stringer, *n*=58 sites).

Variable [a]	**RCC**		**SMZ**		**Stringer**	
	Mean ± SD	**Range**	**Mean ± SD**	**Range**	**Mean ± SD**	**Range**
SCT	8.6 ± 2.8	[2, 14]	13.4 ± 3.6	[5, 21]	12.7 ± 4.1	[5, 21]
DSF	6.2 ± 1.8	[2.5, 10.2]	3.9 ± 2.4	[0, 9.5]	5.2 ± 2.5	[1.9, 16.6]
DCF	1 ± 1.2	[0, 4.5]	5.1 ± 2.3	[1.3, 10.2]	4 ± 2.2	[0, 10.8]
DSN	6.9 ± 2	[2.5, 13.4]	6.1 ± 2.8	[0.6, 14]	7.2 ± 2.8	[1.9, 14]
DCN	1.1 ± 1.1	[0, 5.1]	3.6 ± 1.9	[0, 7.6]	2.6 ± 1.6	[0, 7.6]
DSB	7.1 ± 2	[3.8, 12.1]	5.2 ± 2.7	[0, 12.1]	7 ± 2.7	[2.5, 12.7]
DFB	1.5 ± 1.5	[0, 5.7]	6.9 ± 2.7	[2.5, 13.4]	5.3 ± 2.9	[0, 15.3]

Note: Data reproduced from: *Forest Ecology and Management, vol. 406, Parrish, MC et al., Breeding bird communities associated with land cover in intensively managed pine forests of the southeastern U.S., p. 116. Copyright 2017.*

[a] Total species richness (*SCT*) and guild densities (shrub foragers, *DSF*; canopy foragers, *DCF*; shrub nesters, *DSN*, canopy nesters, *DCN*, shrubland-breeders, *DSB*; mature forest breeders, *DFB*; avian detections • ha^{-1})

Table 5.3 Descriptions of land cover classes used to classify landscapes surrounding 60 intensively managed pine (*Pinus* spp.) forest management units in the South Central Plains ecoregion of Arkansas and Louisiana.

Code	Land cover class	Description
PE	Pine forest (pre-closure)	Pine forest, prior to first canopy closure, dominated by early-successional vegetation.
PT	Pine forest (thinned)	Pine forest with exposed understory (usually due to thinning operations).
PC	Pine forest (closed canopy)	Pine forest with closed canopy.
ROW	Utility right-of-way	Utility corridors maintained regularly as early-successional habitat.
MPH	Mixed pine-hardwood forest	Hardwood or mixed pine-hardwood forests, often bottomlands.
AG	Agricultural use	Agricultural use, extensive pastures or cropland.
URB	Intensive human use	Urban areas or areas of high human activity (e.g., mowed yards with buildings).
WAT	Lakes, ponds, wetlands	Open water features and wet soils areas.

Table 5.4 Spatial data sources used in characterization of 2010 land cover within 5250 m buffers surrounding 60 pine plantation management unit study sites.

Spatial layer [a] (acquisition date)	**Data Source**	**Scale / Resolution**
Color IR and true color digital orthophotographs: AR (2006)	GeoStor: Arkansas State Land Information Board. *geostor.arkansas.gov*	1-m pixels
True color aerial orthophotographs (2010)	USDA/NRCS National Agricultural Imagery Program (NAIP): Geospatial Data Gateway. *datagateway.nrcs.usda.gov*	1-m pixels
True color aerial orthophotographs (winter/summer 2010 & 2011)	ESRI World Imagery Map Streaming Service (sourced via Microsoft Corporation). *goto.arcgisonline.com/maps/World_Imagery*	30-cm pixels
True color aerial orthophotographs (2012)	Microsoft Corporation: Bird's Eye by Pictometry. *bing.com/maps*	30-cm pixels
True color aerial orthophotographs (2013)	USDA/FSA NAIP ArcGIS server. *gis.apfo.usda.gov/arcgis/services/*	1-m pixels
Ponded Water [b]: AR (Sep. 2013); LA (Feb. 2014)	USGS National Geospatial Program: The National Map: National Hydrography Dataset (NHD): High Resolution version. *nhd.usgs.gov/data.html*	1:24,000
Streams [c]: AR (Sep. 2013); LA (Feb. 2014)	NHD: High-resolution version. *nhd.usgs.gov/data.html*	1:24,000
Elevation (Jan. 2013 and Jul. 2015)	National Elevation Dataset (NED) 1/3 arc second seamless DEM. *nationalmap.gov/3dep_prodserv.htm*	10-m pixels

[a] Coverage for all layers included Arkansas (AR) and Louisiana (LA), except where noted.
[b] Waterbody FType codes: 390 ('LakePond'); 436 ('Reservoir'); 466 ('SwampMarsh').
[c] Stream FType codes: 334 ('Connector'), 336 ('CanalDitch'), 460 ('StreamRiver'), 558 ('ArtificialPath').

Table 5.5 Local-scale (within management units [a], "MUs") means and ranges of land cover characteristics for components considered for use in biodiversity modeling.

	Means and ranges	
Landscape component	$\bar{x} \pm sd$	*range*
Management unit area (ha)	53.2 ± 15.8	[24.7, 85.2]
Green tree retention cover (%)	18.3 ± 12.7	[0.7, 47.5]
Number of patches per MU (patches)	9.1 ± 5.3	[3, 28]
SMZ [b] mean patch area (ha)	5.3 ± 5.9	[0.2, 25.6]
SMZ edge density ($m \cdot ha^{-1}$)	47.2 ± 23.2	[3.0, 94.4]
SMZ mean radius of gyration (m)	160.6 ± 106.6	[30.1, 597.0]
RCC [b] mean patch area (ha)	25.8 ± 19.9	[4.2, 79.2]
RCC mean radius of gyration (m)	198.8 ± 88.8	[73.9, 391.9]

[a] Management units were selected to represent the central 80% of the regional range of management unit acreage, thus the area of sample sites varied.

Table 5.6 Site-level mean, standard deviation (SD), and range of estimated stem density and percent grassy ground cover by cover type (regenerating clearcut, *n*=60 sites; SMZ, *n*=60 sites; and stringer, *n*=58 sites).

Variable [a]	**Regenerating clearcut** [b]		**SMZ**		**Stringer**	
	Mean ± SD	**Range**	**Mean ± SD**	**Range**	**Mean ± SD**	**Range**
OSTY_H	0.1 ± 0.8	[0, 6.1]	74.5 ± 65.4	[0, 343.8]	50.4 ± 55	[0, 245.6]
OSTY_P	0 ± 0	[0, 0]	45.5 ± 84.5	[0, 546.5]	39.1 ± 70.2	[0, 307]
MSTY_H	10 ± 30.2	[0, 208.1]	414 ± 310.8	[24.5, 1089.2]	388.2 ± 348.8	[0, 1566.5]
MSTY_P	17.5 ± 57.8	[0, 318.2]	30 ± 55.3	[0, 281.5]	38.3 ± 91.6	[0, 575.2]
USTY_P	930.6 ± 725.3	[20.5, 4114]	210.8 ± 376.3	[0, 1678.3]	445.1 ± 505.5	[0, 2353.8]
%GRASS	59.1 ± 22.8	[5.9, 95.9]	33 ± 22	[2.5, 91.3]	38.5 ± 20.6	[3.5, 96.5]

Note: Data partially reproduced from: *Forest Ecology and Management, vol. 397, Parrish, M.C. et al., Retained vegetation density of streamside management zones and stringers in Southern intensively managed pine forests, p. 91. Copyright 2017.*

[a] Variables were: stem density (units: stems • ha^{-1}; classes: overstory hardwood, *OSTY_H*; overstory pine, *OSTY_P*; midstory hardwood, *MSTY_H*; midstory pine, *MSTY_P*; understory pine, *USTY_P*) and percent grassy ground cover *(%GRASS*).

[b] Within regenerating clearcuts, midstory stems were sparsely present in 20 MUs and overstory hardwoods present in one MU, resulting in strongly zero-dominated data for those variables in clearcuts. I therefore omitted midstory and overstory stem density estimates from the pool of potential explanatory variables in clearcut models, but report on them here for completeness.

Table 5.7 Landscape composition metrics measured at three landscape-level spatial scales of interest (i.e., 1-, 3-, and 5-km buffers surrounding management units) for potential use in avian biodiversity modeling.

Spatial scale	Landscape component	Percent of landscape $\bar{x} \pm sd$	*range*
1-km buffer	Pine forest (pre-1st closure)	28.9 ± 14.1	[7.6, 63.8]
	Pine forest (thinned)	14.2 ± 11.9	[0.0, 48.1]
	Pine forest (closed canopy)	26.4 ± 11.8	[5.9, 55.6]
	Utility right-of-way	0.3 ± 0.8	[0.0, 4.8]
	Hardwood / mixed pine-hardwood forest	25.1 ± 13.8	[0.2, 75.0]
	Agricultural land use	2.7 ± 5.7	[0.0, 35.9]
	Urban and high intensity use	1.2 ± 2.1	[0.0, 12.3]
	Ponds, lakes, and wetlands	1.1 ± 2.7	[0.0, 15.8]
3-km buffer	Pine forest (pre-1st closure)	23.1 ± 8.5	[7.1, 44.3]
	Pine forest (thinned)	15.1 ± 8.9	[1.0, 45.9]
	Pine forest (closed canopy)	25.9 ± 8.1	[9.3, 42.3]
	Utility right-of-way	0.4 ± 0.5	[0.0, 2.4]
	Hardwood / mixed pine-hardwood forest	28.6 ± 12.0	[7.8, 71.7]
	Agricultural land use	3.0 ± 3.4	[0.0, 14.2]
	Urban and high intensity use	2.1 ± 2.6	[0.0, 12.0]
	Ponds, lakes, and wetlands	1.4 ± 3.1	[0.0, 16.2]
5-km buffer	Pine forest (pre-1st closure)	21.2 ± 6.8	[9.5, 40.2]
	Pine forest (thinned)	14.9 ± 8.0	[3.1, 39.0]
	Pine forest (closed canopy)	25.2 ± 7.1	[9.4, 38.0]
	Utility right-of-way	0.4 ± 0.4	[0.0, 2.0]
	Hardwood / mixed pine-hardwood forest	29.4 ± 10.4	[13.6, 62.4]
	Agricultural land use	3.7 ± 3.5	[0.1, 14.8]
	Urban and high intensity use	2.7 ± 3.1	[0.0, 18.3]
	Ponds, lakes, and wetlands	1.6 ± 2.7	[0.0, 11.9]

[a] Management units were selected to represent the central 80% of the regional range of management unit acreage, thus the area of sample sites varied ($\bar{x} \pm sd$: 53.3 ha ± 15.8 ha; range: 24.7 to 85.2 ha). Consequently, the exact area of each spatial scale varied slightly between study sites.

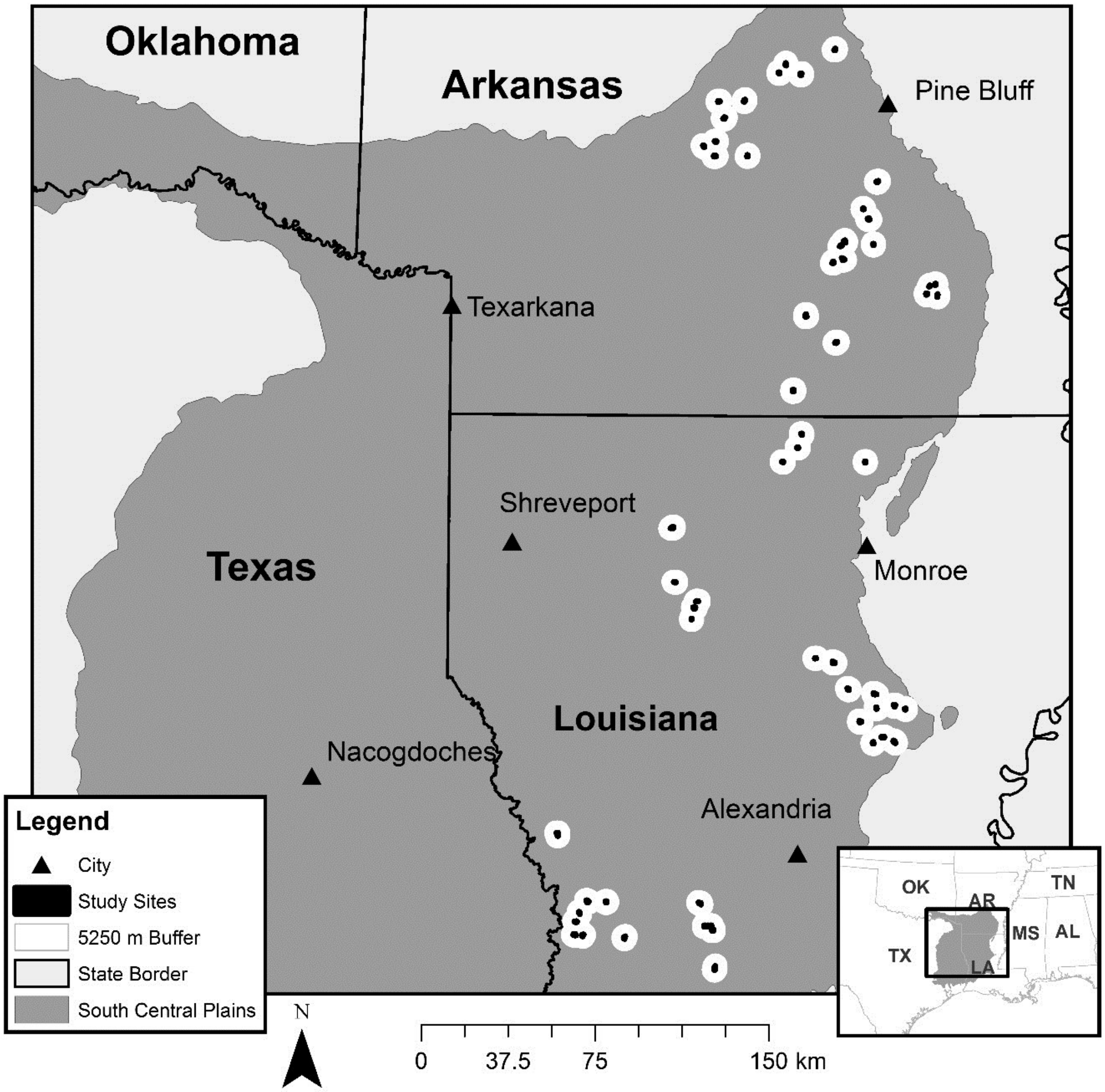

Figure 5.1 Locations of study sites and 5.25-km regional land cover buffers.

Study sites were located in Arkansas and Louisiana within the South Central Plains ecoregion and were bounded within approximately (34.4° N, 93.5° W) and (30.9° N, 91.8° W). Mean site elevation above sea level ranged between 28.6 m and 128.8 m (mean: 67.3 m; SD: 23.0). Sites were harvested in either 2008 or 2009 and were surveyed at three years post-establishment.

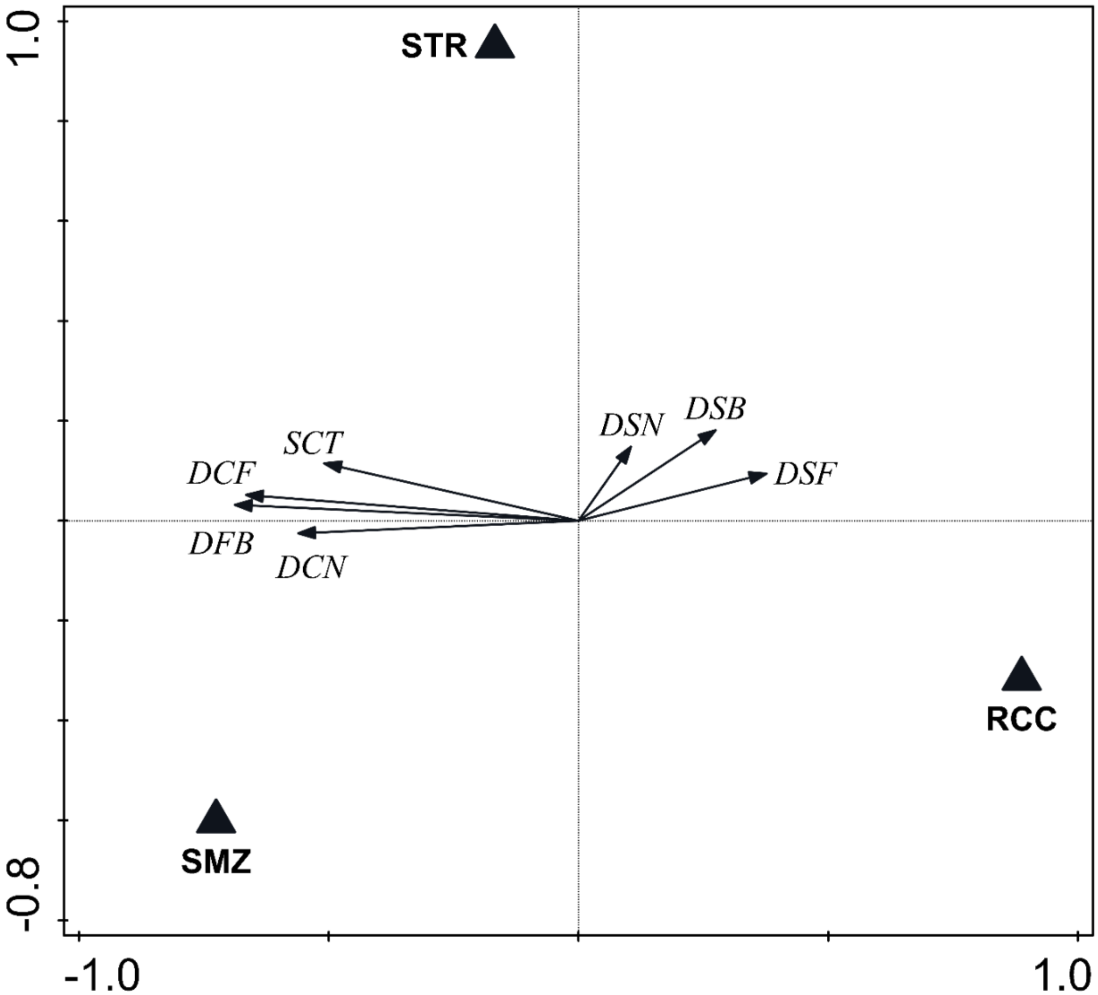

Figure 5.2 Associations between dominant management unit land cover types and avian guild densities and total avian species richness in 3-year-old intensively managed pine (*Pinus* spp.) forest management units, located in the South Central Plains ecoregion of Arkansas and Louisiana, USA.

The ordination biplot diagram from partial redundancy analysis shows response variables (shows response variables (avian guild densities and total avian species richness) as filled arrows and explanatory variables (mean case score centroids of dominant management unit cover type classes) as filled triangles. Avian diversity metrics are: total species richness (*SCT*) and avian guild densities for shrubland-associated guilds (shrub-feeders, *DSF*; shrub-nesters, *DSN*; and shrubland breeders, *DSB*) and mature forest-associated guilds (canopy-feeders, *DCF*; canopy-nesters, *DCN*; and mature forest breeders, *DFB*). Dominant cover types are: regenerating clearcut (RCC), streamside management zone (SMZ), and stringer (STR). Cover type accounted for 40.96% of the partial variation in the avian responses after latitude and site effects were accounted for.

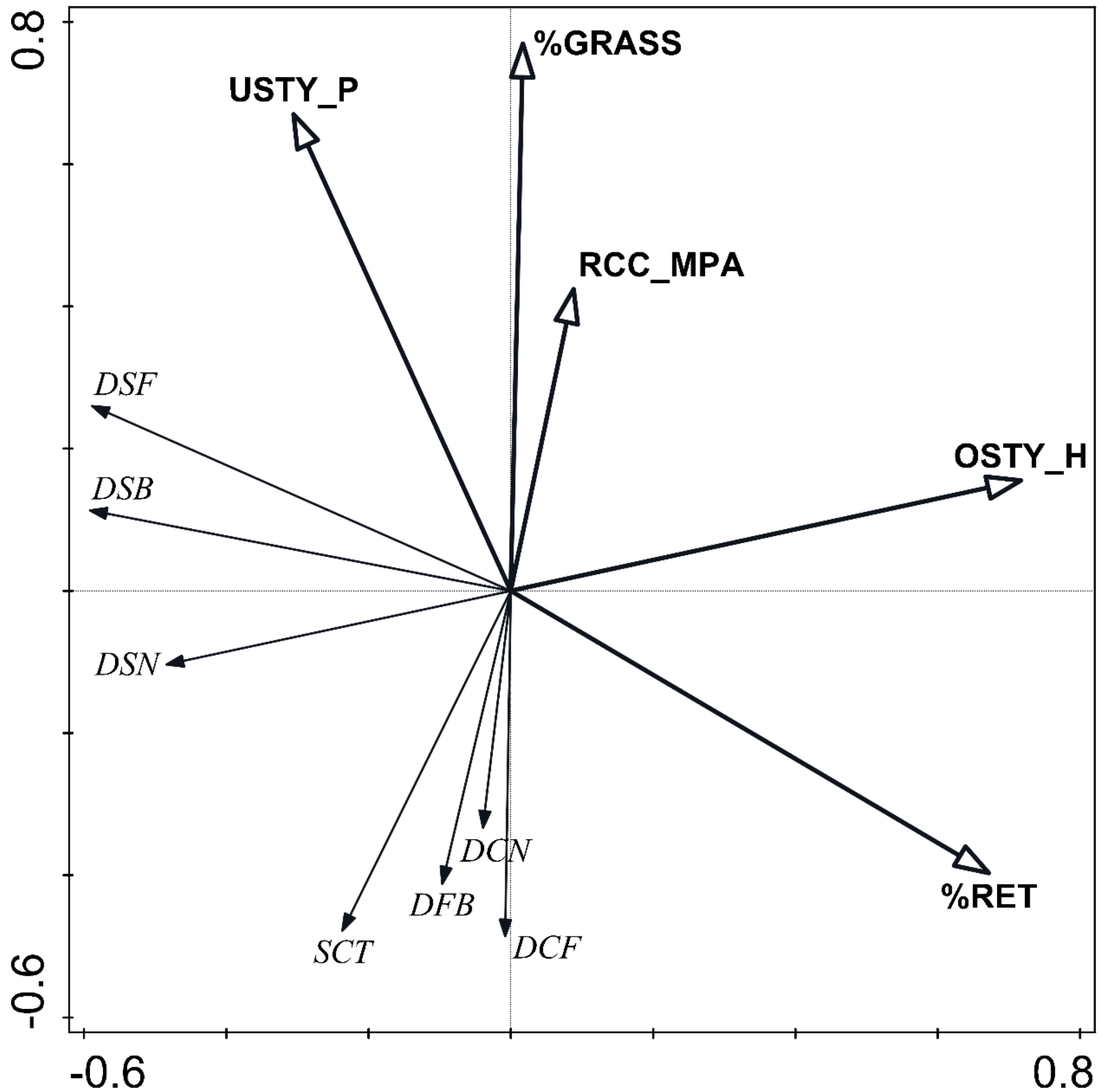

Figure 5.3 Associations between land cover characteristics and avian guild densities and total avian species richness in streamside management zones (SMZs) within 3-year-old intensively managed pine (*Pinus* spp.) forest management units, located in the South Central Plains ecoregion of Arkansas and Louisiana, USA.

The ordination biplot diagram from partial redundancy analysis shows response variables (avian guild densities and total avian species richness) as filled arrows and explanatory variables (land cover characteristic gradients) as open arrows. Avian response variables are: total species richness (*SCT*) and avian guild densities for shrubland-associated guilds (shrub-feeders, *DSF*; shrub-nesters, *DSN*; and shrubland breeders, *DSB*) and mature forest-associated guilds (canopy-feeders, *DCF*; canopy-nesters, *DCN*; and mature forest breeders, *DFB*). Land cover characteristics are: understory pine stem density (*USTY_P*), percent grasses and grass-like ground cover (*%GRASS*), mean patch area of regenerating clearcuts (*RCC_MPA*), overstory hardwood tree stem density (*OSTY_H*), and management unit percent green tree retention (*%RET*). The explanatory landscape variables included in this model accounted for 29.02% of the partial variation in the avian responses after latitude effects were accounted for.

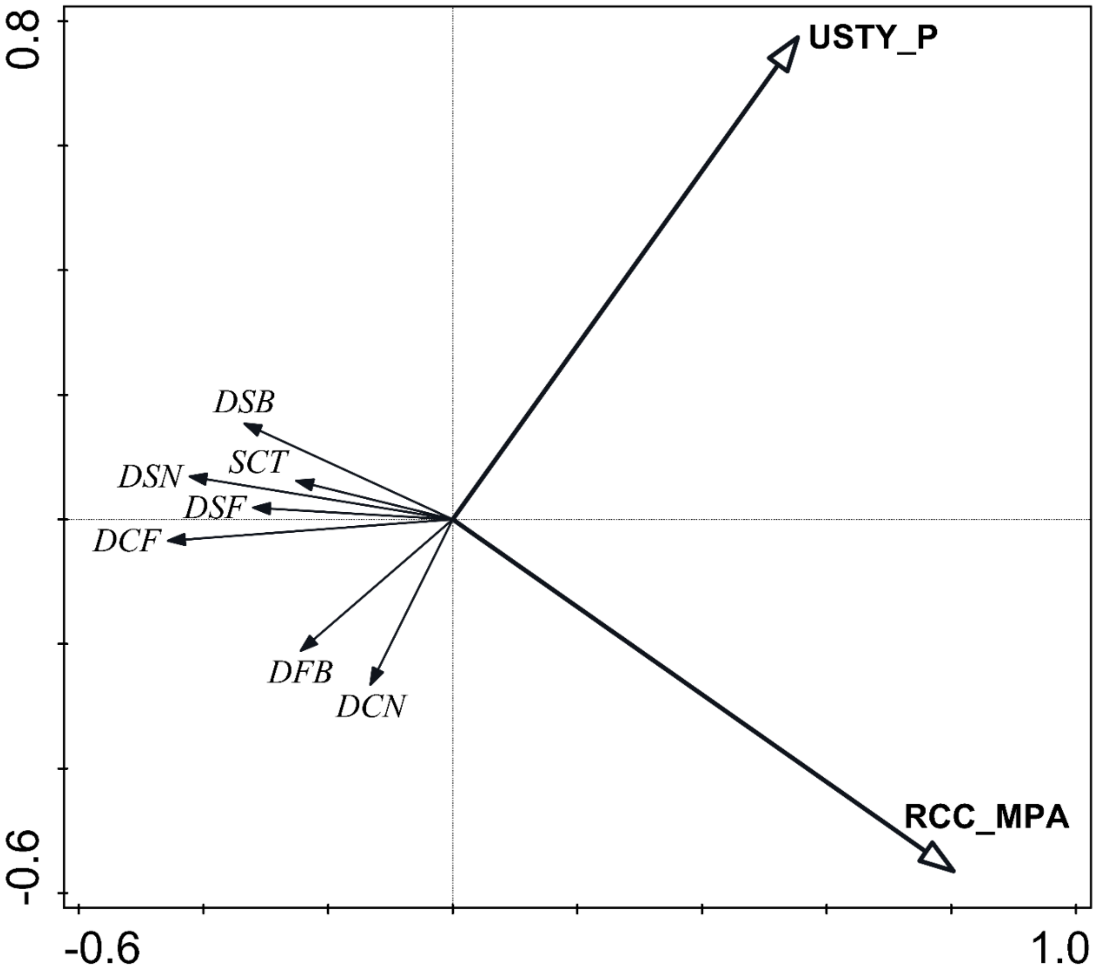

Figure 5.4 Associations between land cover characteristics and avian guild densities and total avian species richness in regenerating clearcuts within 3-year-old intensively managed pine (*Pinus* spp.) forest management units, located in the South Central Plains ecoregion of Arkansas and Louisiana, USA.

The ordination biplot diagram from partial redundancy analysis shows response variables (avian guild densities and total avian species richness) as filled arrows and explanatory variables (land cover characteristic gradients) as open arrows. Avian response variables are: total species richness (*SCT*) and avian guild densities for shrubland-associated guilds (shrub-feeders, *DSF*; shrub-nesters, *DSN*; and shrubland breeders, *DSB*) and mature forest-associated guilds (canopy-feeders, *DCF*; canopy-nesters, *DCN*; and mature forest breeders, *DFB*). Land cover characteristics are: understory pine stem density (*USTY_P*) and mean patch area of regenerating clearcuts (*RCC_MPA*). The explanatory landscape variables included in this model accounted for 13.14% of the partial variation in the avian responses after latitude effects were accounted for.

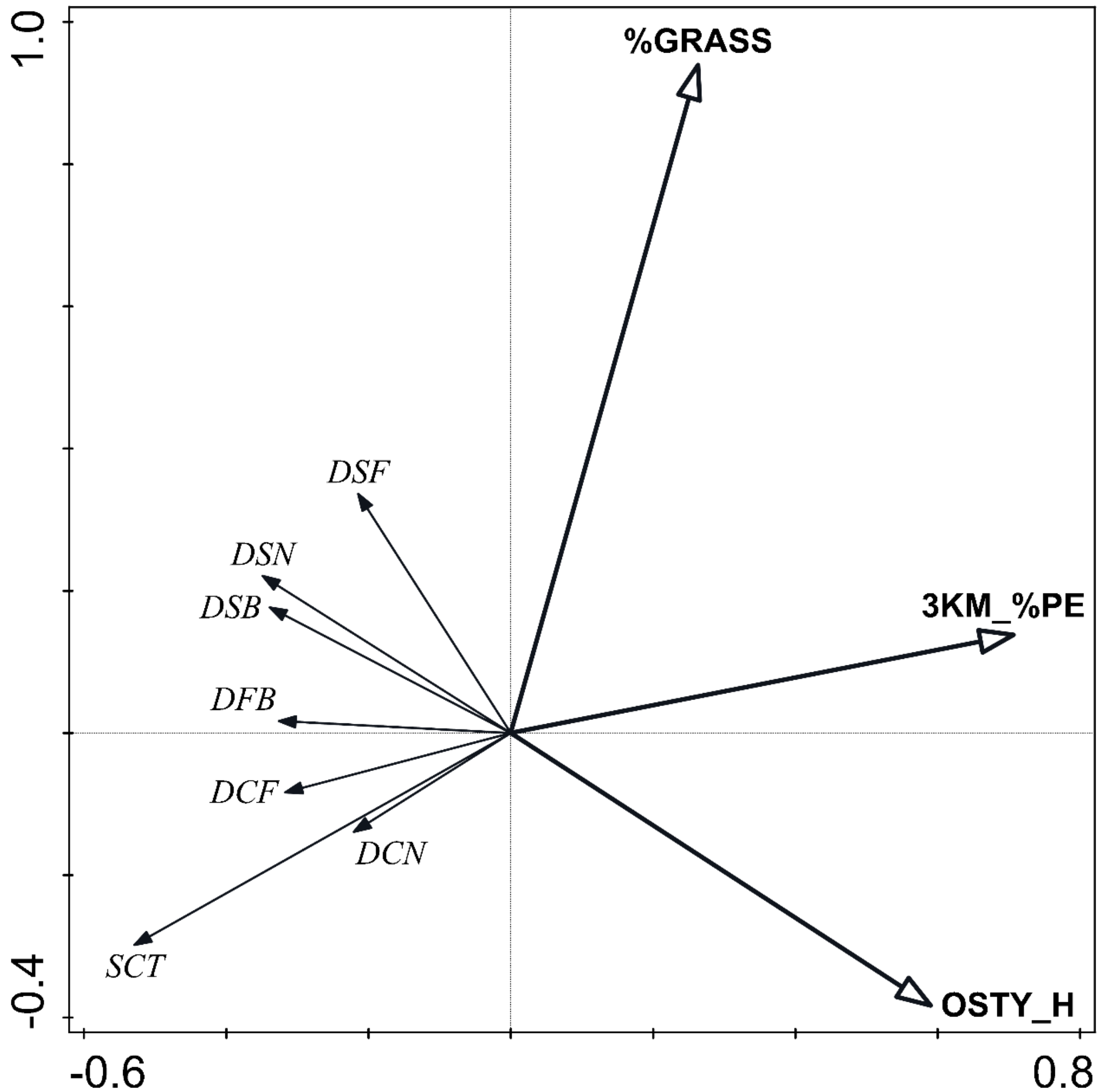

Figure 5.5 Associations between land cover characteristics and avian guild densities and total avian species richness in stringers (ephemeral stream vegetative buffers) within 3-year-old intensively managed pine (*Pinus* spp.) forest management units, located in the South Central Plains ecoregion of Arkansas and Louisiana, USA.

The ordination biplot diagram from partial redundancy analysis shows response variables (avian guild densities and total avian species richness) as filled arrows and explanatory variables (land cover characteristic gradients) as open arrows. Avian response variables are: total species richness (*SCT*) and avian guild densities for shrubland-associated guilds (shrub-feeders, *DSF*; shrub-nesters, *DSN*; and shrubland breeders, *DSB*) and mature forest-associated guilds (canopy-feeders, *DCF*; canopy-nesters, *DCN*; and mature forest breeders, *DFB*). Land cover characteristics are: local-scale percent grasses and grass-like ground cover (*%GRASS*) and overstory hardwood tree stem density (*OSTY_H*), and landscape-scale (3-km buffer around management unit) percent of the landscape in early-successional pine forest (*3KM_%PE*).. The explanatory landscape variables included in this model accounted for 17.69% of the partial variation in the avian responses after latitude effects were accounted for.

5.7 Literature Cited

Adamczyk, J., and D. Tiede. 2017. ZonalMetrics-a Python toolbox for zonal landscape structure analysis. Computers & Geosciences 99:91-99.

American Ornithological Society. 2017. Checklist of North and Middle American Birds. Available online at http://checklist.aou.org/taxa/; Last accessed January 18, 2017.

Andrén, H. 1994. Effects of habitat fragmentation on birds and mammals in landscapes with different proportions of suitable habitat: a review. Oikos 71(3):355-366.

Arkansas Forestry Commission. 2002. Arkansas forestry best management practices for water quality protection. 58 p. Available online at http://forestry.arkansas.gov/Services/ManageYourForests/Documents/bmpbookrevise.pdf; last accessed on Feb. 16, 2017.

Askins, R.A., B. Zuckerberg, and L. Novak. 2007. Do the size and landscape context of forest openings influence the abundance and breeding success of shrubland songbirds in southern New England? Forest Ecology and Management 250(3):137-147.

Azpiroz, A.B., and J.G. Blake. 2016. Associations of grassland birds with vegetation structure in the Northern Campos of Uruguay. The Condor 118(1):12-23.

Betts, M., J.C. Hagar, J.W. Rivers, J. Alexander, K. McGarigal, and B. McComb. 2010. Thresholds in forest bird occurrence as a function of the amount of early-seral broadleaf forest at landscape scales. Ecological Applications 20(8):2116-2130.

Betts, M.G., A.W. Diamond, G.J. Forbes, M.A. Villard, and J.S. Gunn. 2006. The importance of spatial autocorrelation, extent and resolution in predicting forest bird occurrence. Ecological Modelling 191(2):197-224.

Bird Studies Canada, and North American Bird Conservation Initiative. 2014. Bird Conservation Regions. Published by Bird Studies Canada on behalf of the North American Bird Conservation Initiative. Available online at http://www.birdscanada.org/research/gislab/index.jsp?targetpg=bcr. Accessed: 31 March 2014.

Bowen, L.T., C.E. Moorman, and J.C. Kilgo. 2007. Seasonal bird use of canopy gaps in a bottomland forest. The Wilson Journal of Ornithology 119(1):77-88.

Bragg, D.C. 2008. The prominence of pine in the Upper West Gulf Coastal Plain during historical times. P. 29-54 in Freeman and Custis Red River Expedition of 1806: Two Hundred Years Later, Hardy, L.M. (ed.). Bulletin of the Museum of Life Sciences #13, Louisiana State University in Shreveport, Shreveport, LA.

Buffum, B., and R.A. McKinney. 2014. Does proximity to wetland shrubland increase the habitat value for shrubland birds of small patches of upland shrubland in the northeastern United States? International Journal of Forestry Research 2014.

Chandler, C.C., D.I. King, and R.B. Chandler. 2012. Do mature forest birds prefer early-successional habitat during the post-fledging period? Forest Ecology and Management 264:1-9.

Chesser, R.T., R.C. Banks, C. Cicero, J.L. Dunn, A.W. Kratter, I.J. Lovette, P.C. Rasmussen, et al. 2016. Fifth-seventh supplement to the American Ornithologists' Union check-list of North American birds. The Auk 133(3):544-560.

Childers, E.L., T.L. Sharik, and C.S. Adkisson. 1986. Effects of loblolly pine plantations on songbird dynamics in the Virginia Piedmont. The Journal of Wildlife Management 50(3):406-413.

Conner, R.N., and J.G. Dickson. 1997. Relationships between bird communities and forest age, structure, species composition and fragmentation in the West Gulf Coastal Plain. Texas Journal of Science 49:123-138.

Culbert, P.D., V.C. Radeloff, C.H. Flather, J.M. Kellndorfer, C.D. Rittenhouse, and A.M. Pidgeon. 2013. The influence of vertical and horizontal habitat structure on nationwide patterns of avian biodiversity. The Auk 130(4):656-665.

Daigle, J.J., G.E. Griffith, J.M. Omernik, P.L. Faulkner, R.P. McCulloh, L.R. Handley, L.M. Smith, et al. 2006. Ecoregions of Louisiana (color poster with map, descriptive text, summary tables, and photographs). U.S. Geological Survey: Reston, VA (map scale 1:1,000,000).

Davies, P., and M.K.-S. Tso. 1982. Procedures for reduced-rank regression. Applied Statistics 31(3):244-255.

Demarais, S., J.P. Verschuyl, G.J. Roloff, D.A. Miller, and T.B. Wigley. 2017. Tamm review: Terrestrial vertebrate biodiversity and intensive forest management in the U.S. Forest Ecology and Management 385:308-330.

Dickson, J.G., R.N. Conner, and J.H. Williamson. 1993. Breeding bird community changes in a developing pine plantation. Bird Populations. 1: 28-35 1:28-35.

Dickson, J.G., J.H. Williamson, R.N. Conner, and B. Ortego. 1995. Streamside zones and breeding birds in eastern Texas. Wildlife Society Bulletin 23(4):750-755.

Galitsky, C., and J.J. Lawler. 2015. Relative influence of local and landscape factors on bird communities vary by species and functional group. Landscape Ecology 30(2):287-299.

Graves, G.R. 2015. Recent large-scale colonisation of southern pine plantations by Swainson's Warbler *Limnothlypis swainsonii*. Bird Conservation International 25(03):280-293.

Greene, R.E., R.B. Iglay, K.O. Evans, K.V. Miller, T.B. Wigley, and S.K. Riffell. 2016. A meta-analysis of biodiversity responses to management of southeastern pine forests - Opportunities for open pine conservation. Forestry Ecology and Management 360:30-39.

Hagan, J.M., and A.L. Meehan. 2002. The effectiveness of stand-level and landscape-level variables for explaining bird occurrence in an industrial forest. Forest Science 48(2):231-242.

Hamel, P.B. 1992. Land Manager's Guide to the Birds of the South. Gen. Tech. Rep. SE-22. Asheville, NC: US Department of Agriculture, Forest Service, Southeastern Forest Experiment Station. 437 p.

Henry, D.R., D.A. Miller, and T.W. Sherry. 2015. Integrating wildlife conservation with commercial silviculture - Demography of the Swainson's Warbler (*Limnothlypis swainsonii*), a migrant bird of conservation concern in Southern pine forests, USA. in Precious Forests - Precious Earth, Zlatic, M. (ed.). InTech, DOI: 10.5772/61168. Available online at: https://www.intechopen.com/books/precious-forests-precious-earth/integrating-wildlife-conservation-with-commercial-silviculture-demography-of-the-swainson-s-warbler-. Last accessed: June 21, 2017.

Holland, J.D., D.G. Bert, and L. Fahrig. 2004. Determining the Spatial Scale of Species' Response to Habitat. BioScience 54(3):227-233.

Howell, C.A., S.C. Latta, T.M. Donovan, P.A. Porneluzi, G.R. Parks, and J. Faaborg. 2000. Landscape effects mediate breeding bird abundance in midwestern forests. Landscape Ecology 15(6):547-562.

Iglay, R.B., R.E. Greene, B.D. Leopold, and D.A. Miller. 2018. Bird conservation potential of fire and herbicide treatments in thinned pine stands. Forest Ecology and Management 409:267-275.

Jordan, T.G. 1978. Perceptual regions in Texas. Geographical Review 68(3):293-307.

Kilgo, J.C., D.L. Gartner, B.R. Chapman, J.B. Dunning Jr, K.E. Franzreb, S.A. Gauthreaux, C.H. Greenberg, et al. 2002. A test of an expert-based bird-habitat relationship model in South Carolina. Wildlife Society Bulletin 30(3):783-793.

Kilgo, J.C., R.A. Sargent, B.R. Chapman, and K.V. Miller. 1998. Effect of stand width and adjacent habitat on breeding bird communities in bottomland hardwoods. Journal of Wildife Management 62(1):72-83.

King, D.I., and S. Schlossberg. 2014. Synthesis of the conservation value of the early-successional stage in forests of eastern North America. Forest Ecology and Management 324:186-195.

Krementz, D.G., and J.S. Christie. 1999. Scrub-successional bird community dynamics in young and mature longleaf pine-wiregrass savannahs. The Journal of Wildlife Management 63(3):803-814.

Lang, A.J., S. Baker, and B. Mendell. 2016. Forest management practices of private timberland owners and managers in the U.S. South (2016 update). Forest Resources Association Technical Release 16-R-17. Available online at: https://forestresources.org/pdf/Summer2016/16-R-17%20FOREST%20MANAGEMENT%20PRACTICES%20OF.pdf. Last accessed: 15 Sept. 2018.

Lee, M.-B., and J.P. Carroll. 2014. Relative importance of local and landscape variables on site occupancy by avian species in a pine forest, urban, and agriculture matrix. Forest Ecology and Management 320:161-170.

Lee, M., L. Fahrig, K. Freemark, and D.J. Currie. 2002. Importance of Patch Scale vs Landscape Scale on Selected Forest Birds. Oikos 96:110-118.

Legendre, P., and L. Legendre. 2012. Numerical Ecology, Third English Edition. Elsevier Science Amsterdam, The Netherlands.

Legrand, H.G., M.J. Chamberlain, and E.B. Moser. 2007. Diversity and abundance of breeding birds in a managed loblolly pine forest in Louisiana. The American Midland Naturalist 157(2):329-344.

Lichstein, J.W., T.R. Simons, and K.E. Franzreb. 2002. Landscape effects on breeding songbird abundance in managed forests. Ecological Applications 12(3):836-857.

Little, E.L., Jr. 1979. Checklist of United States Trees (Native and Naturalized). US Department of Agriculture Agriculture Handbook no. 541. 375 p.

Loehle, C., T.B. Wigley, S. Rutzmoser, J.A. Gerwin, P.D. Keyser, R.A. Lancia, C.J. Reynolds, et al. 2005. Managed forest landscape structure and avian species richness in the southeastern US. Forest Ecology and Management 214:279-293.

Louisiana Department of Agriculture and Forestry. 1999. Recommended Forestry Best Management Practices for Louisiana. 83 p. Available online at http://www.ldaf.state.la.us/wp-content/uploads/2014/04/BMP.pdf. Last accessed Feb. 16, 2017.

MacFaden, S.W., and D.E. Capen. 2002. Avian Habitat Relationships at Multiple Scales in a New England Forest. Forest Science 48(2):243-253.

Marshall, M.R., J.A. DeCecco, A.B. WIlliams, G.A. Gale, and R.J. Cooper. 2003. Use of regenerating clearcuts by late-successional bird species and their young during the post-fledging period. Forest Ecology and Management 183(1-3):127-135.

Mazerolle, M.J., and M.-A. Villard. 1999. Patch characteristics and landscape context as predictors of species presence and abundance: A review. Ecoscience 6(1):117-124.

McGarigal, K., S.A. Cushman, and E. Ene. 2012. FRAGSTATS v4: Spatial Pattern Analysis Program for Categorical and Continuous Maps. University of Massachusetts, Amherst. Available online at: www.umass.edu/landeco/research/fragstats/fragstats.html; last accessed 16 May 2017.

Miller, D.A., T.B. Wigley, and K.V. Miller. 2009. Managed forests and conservation of terrestrial biodiversity in the southern United States. Journal of Forestry 107(4):197-203.

Miller, J.R., M.D. Dixon, and M.G. Turner. 2004. Response of avian communities in large-river floodplains to environmental variation at multiple scales. Ecological Applications 14(5):1394-1410.

Mitchell, M.S., R.A. Lancia, and J.A. Gerwin. 2001. Using landscape-level data to predict the distribution of birds on a managed forest: Effects of scale. Ecological Applications 11(6):1692-1708.

Mitchell, M.S., S.H. Rutzmoser, T.B. Wigley, C. Loehle, J.A. Gerwin, P.D. Keyser, R.A. Lancia, et al. 2006. Relationships between avian richness and landscape structure at multiple scales using multiple landscapes. Forest Ecology and Management 221(1-3):155-169.

Omernik, J.M. 1987. Ecoregions of the conterminous United States. Annals of the Association of American Geographers 77(1):118-125.

Oswalt, S.N. 2017. Forests of Louisiana, 2014. Resource Update FS-117. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station.

Oswalt, S.N., P.D. Miles, S.A. Pugh, and W.B. Smith. 2018. Forest Resources of the United States, 2017: a technical document supporting the Forest Service 2020 update of the RPA Assessment. Gen. Tech. Rep. WO-xxx. Washington, DC: U.S. Department of Agriculture, Forest Service, Washington Office. xxx p.

Owens, F.L., P.C. Stouffer, M.J. Chamberlain, and D.A. Miller. 2014. Early-successional breeding bird communities in intensively managed pine plantations: Influence of vegetation succession but not site preparations. Southeastern Naturalist 13(3):423-443.

Pagen, R.W., F.R. Thompson, III, and D.E. Burhans. 2000. Breeding and post-breeding habitat use by forest migrant songbirds in the Missouri Ozarks. The Condor 102(4):738-747.

Parrish, M.C., S. Demarais, A.W. Ezell, T.B. Wigley, P.D. Jones, and S.K. Riffell. 2017a. Retained vegetation density of streamside management zones and stringers in Southern intensively managed pine forests. Forest Ecology and Management 397:89-96.

Parrish, M.C., S. Demarais, T.B. Wigley, P.D. Jones, A. Ezell, and S.K. Riffell. 2017b. Breeding bird communities associated with land cover in intensively managed pine forests of the southeastern U.S. Forest Ecology and Management 406:112-124.

Parrish, M.C., S. Demarais, T.B. Wigley, S.K. Riffell, A. Ezell, and P.D. Jones. 2018. Operational green tree retention and land cover patterns in intensively managed pine forest landscapes of the southeastern U.S. Forest Science 00(00):1-13.

Parrish, M.C., and J.A. Hepinstall-Cymerman. 2012. Associations between Multiscale Landscape Characteristics and Breeding Bird Abundance and Diversity across Urban-Rural Gradients in Northeastern Georgia, USA. Urban Ecosystems 15:559-580.

Partners in Flight Science Committee. 2012. Species Assessment Database, Version 2012 (Bird Conservation Region 25). Available online at http://rmbo.org/pifassessment. Last accessed Mar. 16, 2017.

Peak, R.G., and F.R. Thompson, III. 2006. Factors affecting avian species richness and density in riparian areas. The Journal of Wildlife Management 70(1):173-179.

Reidy, J.L., F.R. Thompson, and S.W. Kendrick. 2014. Breeding bird response to habitat and landscape factors across a gradient of savanna, woodland, and forest in the Missouri Ozarks. Forest Ecology and Management 313(1):34-46.

Reynolds, R.R., J.B. Baker, and T.T. Ku. 1984. Four decades of selection management on the Crossett Farm Forestry Forties. Bulletin. 872. Agricultural Experiment Station, Division of Agriculture, University of Arkansas.

Ricketts, T.H., and E. Dinerstein. 1999. Terrestrial Ecoregions of North America. Island Press, Washington, D.C., USA.

Roberts, H.P., and D.I. King. 2017. Area requirements and landscape-level factors influencing shrubland birds. The Journal of Wildlife Management 81(7):1298-1307.

Robinson, S.K., F.R. Thompson, III, T.M. Donovan, D.R. Whitehead, and J. Faaborg. 1995. Regional forest fragmentation and the nesting success of migratory birds. Science 267(5206):1987-1990.

Rodewald, A.D. 2003. The importance of land uses within the landscape matrix. Wildlife Society Bulletin:586-592.

Rodewald, A.D., and A.C. Vitz. 2005. Edge- and area-sensitivity of shrubland birds. Journal of Wildife Management 69(2):681-688.

Rodewald, P., (Ed.). 2015. The Birds of North America. Cornell Laboratory of Ornithology, Ithaca, NY. Available online at http://bna.birds.cornell.edu/BNA/. Last accessed Jun. 27, 2017.

Rosenvald, R., and A. Lõhmus. 2008. For what, when, and where is green-tree retention better than clear-cutting? A review of the biodiversity aspects. Forest Ecology and Management 255(1):1-15.

Rosson, J.F., Jr. 2016. Forests of Arkansas, 2015. Resource Update FS-81. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station.

Schlossberg, S., and D.I. King. 2008. Are shrubland birds edge specialists? Ecological Applications 18(6):1325-1330.

Schlossberg, S., D.I. King, R.B. Chandler, and B.A. Mazzei. 2010. Regional Synthesis of Habitat Relationships in Shrubland Birds. The Journal of Wildlife Management 74(7):1513-1522.

Schlossberg, S., D.I. King, S. Destefano, and M. Hartley. 2018. Effects of early-successional shrubland management on breeding wood thrush populations. The Journal of Wildlife Management 0(0):10 pp.

Shake, C.S., C.E. Moorman, J.D. Riddle, and M.R. Burchell, II. 2012. Influence of patch size and shape on occupancy by shrubland birds. The Condor 114(2):268-278.

Sleeter, B.M., T.L. Sohl, T.R. Loveland, R.F. Auch, W. Acevedo, M.A. Drummond, K.L. Sayler, et al. 2013. Land-cover change in the conterminous United States from 1973 to 2000. Global Environmental Change 23(4):733-748.

Šmilauer, P., and J. Lepš. 2014. Multivariate Analysis of Ecological Data using CANOCO 5, Second Edition. Cambridge University Press.

Suarez-Rubio, M., S. Wilson, P. Leimgruber, and T. Lookingbill. 2013. Threshold responses of forest birds to landscape changes around exurban development. PLoS ONE 8(6):e67593.

ter Braak, C.J., and I.C. Prentice. 1988. A theory of gradient analysis. P. 271-317 in Advances in ecological research. Elsevier.

ter Braak, C.J.F., and P. Šmilauer. 2017. Canoco 5, Windows release (5.10): Software for multivariate data exploration, testing, and summarization. Biometris, Wageningen Research Foundation, Wageningen University & Research, the Netherlands, and Petr Šmilauer, Czech Republic.

Tirpak, J.M., D.T. Jones-Farrand, F.R. Thompson, III, D.J. Twedt, and W.B. Uihlein, III. 2009. Multiscale habitat suitability index models for priority landbirds in the Central Hardwoods and West Gulf Coastal Plain / Ouachitas bird conservation regions. Gen. Tech. Rep. NRS-49. Newtown Square, PA: USDA Forest Service Northern Research Station. 195 p.

Trani, M.K., R.T. Brooks, T.L. Schmidt, V.A. Rudis, and C.M. Gabbard. 2001. Patterns and trends of early successional forests in the eastern United States. Wildlife Society Bulletin 29(2):413-424.

Turner, J.C., J.A. Gerwin, and R.A. Lancia. 2002. Influences of Hardwood Stand Area and Adjacency on Breeding Birds in an Intensively Managed Pine Landscape. Forest Science 48(2):323-330.

US Environmental Protection Agency. 2011. Ecoregions of the United States (Spatial Data). 1:250,000 scale. U.S. EPA Office of Research & Development (ORD) - National Health and Environmental Effects Research Laboratory (NHEERL), Corvallis, OR. Available online at: ftp://ftp.epa.gov/wed/ecoregions/us/Eco_Level_III_US.zip; last accessed Aug. 26, 2011.

USDA Forest Service, Forest Inventory and Analysis Program. 2018. Forest Inventory EVALIDator web-application Version 1.7.0.01. St. Paul, MN: U.S. Department of Agriculture, Forest Service, Northern Research Station. [Available only on internet: http://fsxopsx1056.fdc.fs.usda.gov:9001/Evalidator/evalidator.jsp] Accessed: 25 Sept. 2018.

Vitz, A.C., and A.D. Rodewald. 2006. Can regenerating clearcuts benefit mature-forest songbirds? An examination of post-breeding ecology. Biological Conservation 127:477-486.

Wilkin, E., F. Jiménez Nava, and G. Griffith. 2011. North American Terrestrial Ecoregions - Level III. Commission for Environmental Cooperation, Montreal, Canada. 149 p.

Woods, A.J., T.L. Foti, S.S. Chapman, J.M. Omernik, J.A. Wise, E.O. Murray, W.L. Prior, et al. 2004. Ecoregions of Arkansas (Color Poster with Map, Descriptive Text, Summary Tables, and Photographs). U.S. Geological Survey: Reston, VA (map scale 1:1,000,000)..

APPENDIX A

BIRD SPECIES DETECTED DURING POINT COUNTS

A.1 Introduction

Below, I list common and scientific names for the 104 bird species detected during 2011 and 2012 field surveys in intensively managed pine (*Pinus* spp.) forests (IMPF) located in the South Central Plains ecoregion of the United States (Table A.1). Avian taxonomic nomenclature follows the American Ornithological Society Checklist of North and Middle American Birds (American Ornithological Society 2017) through the fifty-seventh supplement (Chesser et al. 2016) and are listed in taxonomic order. I also indicate whether a species was a 'target species' used in my analyses, guild affiliations for the species, and conservation status of the species at the state and regional level, if threatened.

A.2 Targeted bird species

I defined my "target set" of species as breeding land birds meeting the majority of their reproductive season needs within the bounds of a management unit. I excluded from the target set: (1) birds seen outside count intervals or 50-m count circles; (2) late migrants, (3) non-landbirds; (4) nocturnal species; and (5) wide-ranging species (e.g., members of order *Accipitriformes* and families *Hirundinidae* and *Apodidae*). Sixty target species were detected during point-counts and were used in later analyses. Seven additional target species were detected outside of point-counts and were not analyzed. Thirty-seven non-target species were also observed in management units, although my surveys were not intended to systematically detect those species. I include both target species and non-target species in Table A.1 to provide a more comprehensive record of species that occur on IMPF management units.

A.3 Assignment to guilds

I assigned guild affiliation only to target species detected during point-counts. Guild affiliations were based on information gleaned from species accounts in The Birds of North America (Rodewald 2015). My guild assignments apply only to dominant breeding season traits, and are best applied to birds occurring in the South Central Plains ecoregion. Birds were assigned to three guilds based upon life history characteristics: (1) foraging location; (2) nest placement location; and (3) dominant breeding cover.

A.4 Conservation status at state and national levels

Conservation status remarks are included for species of special conservation concern occurring in Arkansas (Arkansas Game and Fish Commission 2004) and Louisiana (Lester et al. 2005). I have included updates to the Louisiana species of conservation concern based on the 2015 draft of the Louisiana Wildlife action plan (Holcomb et al. 2015), and have indicated them where pertinent. I also list species classified by Partners in Flight (PIF) as being of regional conservation concern during their breeding season (Partners in Flight Science Committee 2012). I include the PIF regional conservation score (RCS-b, ranging from 0 to 25) and the suggested action level (management attention, 'MA': species is moderately threatened and experiencing moderate to large declines; immediate management, 'IM': high regional threats and large population declines) for each species (Panjabi et al. 2012).

Table A.1 Common and scientific names, guild affiliations, conservation status (state and regional level), and occupancy by cover type for birds detected during and outside of point-counts in 60 intensively managed pine (*Pinus* spp.) forest management units in Arkansas and Louisiana, USA.

		Guild affiliation [a]			Conservation Status [b]			Occupancy by Cover Type [c]		
Species [d]	**Target Species [e]**	**Foraging Location**	**Nesting Location**	**Breeding Habitat**	**LA**	**AR**	**Regional Conservation Score / (Action Level) [f]**	**RCC**	**SMZ**	**STR**
Canada Goose *Branta canadensis*	...	...	...	...	...	...	...	...	...	...
Wood Duck *Aix sponsa*	...	...	...	...	...	...	...	...	...	...
Northern Bobwhite [g] *Colinus virginianus*	T	G	G	E	S3 [h]	...	14 (MA)	+	+	+
Wild Turkey *Meleagris gallopavo*	...	...	...	...	...	...	...	...	...	...
Mourning Dove *Zenaida macroura*	T	G	C	E	...	...	...	+	+	+
Yellow-billed Cuckoo [g] *Coccyzus americanus*	T	C	C	F	S5B	...	15 (MA)	+	+	+
Greater Roadrunner *Geococcyx californianus*	...	...	...	...	...	...	14 (MA)	...	...	...
Common Nighthawk [g] *Chordeiles minor*	...	...	...	...	...	...	...	...	...	...
Chuck-will's-widow *Antrostomus carolinensis*	...	...	...	...	S4B	SC	16 (MA)	...	...	...
Chimney Swift [g] *Chaetura pelagica*	...	...	...	...	S5B	...	15 (MA)	...	...	...
Ruby-throated Hummingbird *Archilochus colubris*	T	S	C	F	...	...	...	+	+	+
Killdeer *Charadrius vociferus*	T	G	G	G	...	...	...	+	-	-
Great Blue Heron *Ardea herodias*	...	...	...	...	...	...	...	...	...	...

Table A.1 (continued)

Species [d]	Target Species[e]	Guild affiliation [a]			Conservation Status [b]			Occupancy by Cover Type [c]		
		Foraging Location	Nesting Location	Breeding Habitat	LA	AR	Regional Conservation Score / (Action Level) [f]	RCC	SMZ	STR
Great Egret *Ardea alba*	...	...	...	...	...	...	...	...	...	...
Little Blue Heron *Egretta caerulea*	...	...	...	...	S3N, S4B	...	...	...	...	...
Cattle Egret *Bubulcus ibis*	...	...	...	...	...	...	...	...	...	...
Green Heron *Butorides virescens*	...	...	...	...	...	...	...	...	...	...
Black-crowned Night-Heron *Nycticorax nycticorax*	...	...	...	...	...	...	...	...	...	...
Yellow-crowned Night-Heron *Nyctanassa violacea*	...	...	...	...	S2N, S5B	...	...	...	...	...
White Ibis *Eudocimus albus*	...	...	...	...	...	...	...	...	...	...
Black Vulture *Coragyps atratus*	...	...	...	...	...	...	...	...	...	...
Turkey Vulture *Cathartes aura*	...	...	...	...	...	...	...	...	...	...
White-tailed Kite [i] *Elanus leucurus*	...	...	...	...	S1B [h]	...	...	...	...	...
Mississippi Kite *Ictinia mississippiensis*	...	...	...	...	...	...	...	...	...	...
Cooper's Hawk *Accipiter cooperii*	...	...	...	...	...	SC	...	...	...	...
Red-shouldered Hawk *Buteo lineatus*	...	...	...	...	...	...	...	...	...	...
Broad-winged Hawk *Buteo platypterus*	...	...	...	...	...	...	...	...	...	...
Red-tailed Hawk *Buteo jamaicensis*	...	...	...	...	...	...	...	...	...	...

Table A.1 (continued)

		Guild affiliation [a]			Conservation Status [b]			Occupancy by Cover Type [c]		
Species [d]	**Target Species[e]**	**Foraging Location**	**Nesting Location**	**Breeding Habitat**	**LA**	**AR**	**Regional Conservation Score / (Action Level) [f]**	**RCC**	**SMZ**	**STR**
Eastern Screech-Owl *Megascops asio*	...	...	...	...	...	...	...	...	...	...
Great Horned Owl *Bubo virginianus*	...	...	...	...	...	...	...	...	...	...
Barred Owl *Strix varia*	...	...	...	...	...	...	...	...	...	...
Red-headed Woodpecker [g] *Melanerpes erythrocephalus*	T	B	V	E	S4 [h]	SC	16 (MA)	+	+	+
Red-bellied Woodpecker *Melanerpes carolinus*	T	B	V	F	...	...	...	+	+	+
Downy Woodpecker *Picoides pubescens*	T	B	V	F	...	...	...	+	+	+
Hairy Woodpecker *Picoides villosus*	T	B	V	F	...	...	...	+	+	+
Northern Flicker [g] *Colaptes auratus*	T	G	V	F	...	...	14 (MA)	+	+	+
Pileated Woodpecker *Dryocopus pileatus*	T	B	C	F	...	...	16 (MA)	+	+	+
American Kestrel *Falco sparverius*	...	...	...	...	...	...	14 (MA)	...	...	...
Eastern Wood-Pewee *Contopus virens*	T	C	C	F	...	...	15 (MA)	+	+	+
Acadian Flycatcher *Empidonax virescens*	T	C	C	F	...	...	17 (MA)	-	+	+
Eastern Phoebe *Sayornis phoebe*	...	...	...	...	...	...	...	...	...	...
Great Crested Flycatcher *Myiarchus crinitus*	T	C	V	F	...	...	...	-	+	+
Eastern Kingbird *Tyrannus tyrannus*	T	S	C	F	...	...	14 (MA)	+	+	+

Table A.1 (continued)

		Guild affiliation [a]			Conservation Status [b]			Occupancy by Cover Type [c]		
Species [d]	Target Species[e]	Foraging Location	Nesting Location	Breeding Habitat	LA	AR	Regional Conservation Score / (Action Level) [f]	RCC	SMZ	STR
Scissor-tailed Flycatcher *Tyrannus forficatus*	T	C	C	G	S4B	...	...	+	-	-
White-eyed Vireo *Vireo griseus*	T	C	S	E	...	...	...	+	+	+
Yellow-throated Vireo *Vireo flavifrons*	T	C	C	F	S4B	...	...	+	+	+
Red-eyed Vireo *Vireo olivaceus*	T	C	C	F	...	...	...	+	+	+
Blue Jay *Cyanocitta cristata*	T	X	C	F	...	...	...	+	+	+
American Crow *Corvus brachyrhynchos*	...	...	...	...	...	...	...	...	...	...
Fish Crow *Corvus ossifragus*	...	...	...	...	...	...	...	...	...	...
Purple Martin *Progne subis*	...	...	...	...	...	...	...	...	...	...
Tree Swallow *Tachycineta bicolor*	...	...	...	...	...	...	...	...	...	...
Barn Swallow *Hirundo rustica*	...	...	...	...	...	...	...	...	...	...
Carolina Chickadee *Poecile carolinensis*	T	C	V	F	...	...	...	+	+	+
Tufted Titmouse *Baeolophus bicolor*	T	C	V	F	...	...	...	+	+	+
Brown-headed Nuthatch *Sitta pusilla*	T	C	V	F	S5 [h]	...	17 (MA)	+	+	+
Marsh Wren *Cistothorus palustris*	...	...	...	...	S4	...	...	...	...	...
Carolina Wren *Thryothorus ludovicianus*	T	G	V	F	...	...	...	+	+	+

Table A.1 (continued)

		Guild affiliation [a]			Conservation Status [b]			Occupancy by Cover Type [c]		
Species [d]	Target Species[e]	Foraging Location	Nesting Location	Breeding Habitat	LA	AR	Regional Conservation Score / (Action Level) [f]	RCC	SMZ	STR
Blue-gray Gnatcatcher *Polioptila caerulea*	T	C	C	F	...	...	14 (MA)	+	+	+
Eastern Bluebird *Sialia sialis*	T	G	V	E	...	...	...	+	+	+
Swainson's Thrush *Catharus ustulatus*	...	...	...	...	...	...	...	...	...	...
Wood Thrush *Hylocichla mustelina*	T	G	S	F	S4B	...	14 (MA)	-	+	+
American Robin *Turdus migratorius*	T	G	C	E	...	...	...	-	-	+
Gray Catbird *Dumetella carolinensis*	T	G	S	F	...	...	...	+	+	+
Brown Thrasher *Toxostoma rufum*	T	G	S	F	...	...	15 (MA)	+	+	+
Northern Mockingbird *Mimus polyglottos*	T	G	S	E	...	...	...	+	+	+
Cedar Waxwing *Bombycilla cedrorum*	...	...	...	...	...	...	...	...	...	...
American Goldfinch *Spinus tristis*	T	C	S	E	...	...	...	-	+	+
Ovenbird *Seiurus aurocapilla*	T	G	G	F	...	...	...	-	-	+
Worm-eating Warbler *Helmitheros vermivorum*	T	G	G	F	S3B [h]	...	15 (MA)	-	-	+
Louisiana Waterthrush *Parkesia motacilla*	T	G	G	F	S3B [h]	...	16 (MA)	-	+	+
Black-and-white Warbler *Mniotilta varia*	T	B	G	F	...	...	15 (MA)	+	+	+

Table A.1 (continued)

		Guild affiliation [a]			Conservation Status [b]			Occupancy by Cover Type [c]		
Species [d]	**Target Species[e]**	**Foraging Location**	**Nesting Location**	**Breeding Habitat**	**LA**	**AR**	**Regional Conservation Score / (Action Level) [f]**	**RCC**	**SMZ**	**STR**
Prothonotary Warbler *Protonotaria citrea*	T	S	V	F	S5B	...	16 (MA)	-	-	+
Swainson's Warbler *Limnothlypis swainsonii*	...	...	...	...	S4B	SC	18 (MA)	...	...	...
Mourning Warbler *Geothlypis philadelphia*	...	...	...	...	...	...	...	...	...	...
Kentucky Warbler *Geothlypis formosa*	T	G	G	F	S4B	...	17 (MA)	+	+	+
Common Yellowthroat *Geothlypis trichas*	T	S	S	E	...	...	...	+	+	+
Hooded Warbler *Setophaga citrina*	T	S	S	E	S5B	...	...	+	+	+
American Redstart *Setophaga ruticilla*	T	C	C	F	S3B	...	...	-	-	+
Northern Parula *Setophaga americana*	T	C	C	F	S5B	...	...	-	+	+
Pine Warbler *Setophaga pinus*	T	C	C	F	...	...	...	+	+	+
Yellow-throated Warbler *Setophaga dominica*	T	C	C	F	...	...	...	-	+	-
Prairie Warbler [g] *Setophaga discolor*	T	S	S	E	S4B	...	17 (MA)	+	+	+
Black-throated Green Warbler *Setophaga virens*	...	...	...	...	...	SC	...	...	...	...
Yellow-breasted Chat *Icteria virens*	T	S	S	E	...	...	...	+	+	+
Eastern Towhee *Pipilo erythrophthalmus*	T	G	G	E	...	...	...	+	+	+
Bachman's Sparrow *Peucaea aestivalis*	T	G	G	G	S3	...	19 (IM)	-	+	+

Table A.1 (continued)

Species [d]	Target Species [e]	Guild affiliation [a] Foraging Location	Nesting Location	Breeding Habitat	Conservation Status [b] LA	AR	Regional Conservation Score / (Action Level) [f]	Occupancy by Cover Type [c] RCC	SMZ	STR
Chipping Sparrow *Spizella passerina*	T	G	S	F	...	...	...	-	+	-
Field Sparrow [g] *Spizella pusilla*	T	G	G	E	S4B, S5N	...	15 (MA)	+	+	+
Grasshopper Sparrow [g] *Ammodramus savannarum*	T	G	G	G	S1B [h]	...	...	-	-	+
Song Sparrow *Melospiza melodia*	...	...	...	...	...	...	...	...	...	...
Summer Tanager *Piranga rubra*	T	C	C	F	...	...	...	+	+	+
Northern Cardinal *Cardinalis cardinalis*	T	X	S	F	...	...	...	+	+	+
Rose-breasted Grosbeak *Pheucticus ludovicianus*	...	...	...	...	...	...	...	...	...	...
Blue Grosbeak *Passerina caerulea*	T	S	S	E	...	...	...	+	+	+
Indigo Bunting *Passerina cyanea*	T	S	S	E	...	...	...	+	+	+
Painted Bunting *Passerina ciris*	T	G	S	E	S5B	...	...	+	+	+
Dickcissel *Spiza americana*	T	G	G	G	S4B	...	14 (MA)	+	+	+
Red-winged Blackbird *Agelaius phoeniceus*	...	...	...	...	...	...	...	...	...	...
Eastern Meadowlark [g] *Sturnella magna*	T	G	G	G	S4 [h]	...	15 (MA)	-	+	-
Common Grackle *Quiscalus quiscula*	...	...	...	...	...	...	...	...	...	...
Brown-headed Cowbird *Molothrus ater*	T	G	S	E	...	...	...	+	+	+

Table A.1 (continued)

Species [d]	Target Species [e]	Guild affiliation [a]			Conservation Status [b]			Occupancy by Cover Type [c]		
		Foraging Location	Nesting Location	Breeding Habitat	LA	AR	Regional Conservation Score / (Action Level) [f]	RCC	SMZ	STR
Orchard Oriole *Icterus spurius*	T	S	C	F	S5B	...	15 (MA)	+	+	+
Baltimore Oriole *Icterus galbula*	...	...	...	...	...	...	...	...	...	...

[a] Guild membership codes: Foraging location: canopy / midstory (C), tree bark (B), shrubs (S), ground (G); Nest location: canopy / midstory (C), tree cavity (V), shrubs (S); ground (G); Dominant breeding cover: mature forest (F); shrubland / early-successional (S); grassland (G).

[b] Conservation status codes for Arkansas (AR) from Arkansas Game and Fish Commission (2004): special concern (SC). Codes for Louisiana (LA) from Lester and others (2005): secure (S5), apparently secure (S4), vulnerable (S3), and 'B' suffix indicates breeding season.

[c] Occupancy status listed for target species only. Presence (+) or absence (-) was derived from point-count data.

[d] Avian taxonomic nomenclature follows American Ornithological Society Checklist of North and Middle American Birds (2017) through the fifty-seventh supplement (Chesser 2016). Changes to taxonomy as of 2018 might be warranted.

[e] Member of target set of species used in analyses (indicated by 'T'): breeding, diurnal landbirds meeting the majority of breeding needs on-site.

[f] Partners in Flight breeding season regional conservation scores and action level: warrants management attention (MA), warrants immediate management (IM). Source: Partners in Flight Science Committee (2012). Species Assessment Database, version 2012 (bird conservation region 25). Available online at http://rmbo.org/pifassessment; last accessed on Mar. 16, 2017.

[g] Denotes common bird in steep decline: populations estimated to have decreased by >50% during previous 40 years (Partners in Flight Science Committee, 2012).

[h] Based on 2015 Louisiana Wildlife Action Plan (draft) conservation rankings (Holcomb 2015).

[i] A notable, non-target species, White-tailed Kite, was ranked S1B ('critically imperiled in Louisiana') by Louisiana Department of Wildlife and Fisheries (Lester, 2005). Technician T. Campbell observed several individuals in proximity to each other on one site, suggesting possible courtship or breeding activity at the extreme edge of their range. I do not consider White-tailed Kite a typical species for my sites, but wish to note the observation.

A.5 Literature Cited

American Ornithological Society. 2017. Checklist of North and Middle American Birds. Available online at http://checklist.aou.org/taxa/; Last accessed January 18, 2017.

Arkansas Game and Fish Commission. 2004. Arkansas endangered, threatened, and species of special concern: January 9, 2004. Little Rock, AR, USA.

Chesser, R.T., R.C. Banks, C. Cicero, J.L. Dunn, A.W. Kratter, I.J. Lovette, P.C. Rasmussen, et al. 2016. Fifth-seventh supplement to the American Ornithologists' Union check-list of North American birds. The Auk 133(3):544-560.

Holcomb, S.R., A.A. Bass, C.S. Reid, M.A. Seymour, N.F. Lorenz, B.B. Gregory, S.M. Javed, et al. 2015. Louisiana Wildlife Action Plan (Draft, October 2015). Louisiana Department of Wildlife and Fisheries, Baton Rouge, LA, USA, p. 661 pp.

Lester, G.D., S.G. Sorensen, P.L. Faulkner, C.S. Reid, and I.E. Maxit. 2005. Louisiana Comprehensive Wildlife Conservation Strategy. Louisiana Department of Wildlife and Fisheries, Baton Rouge, LA, USA, 455 pp.

Panjabi, A.O., P.J. Blancher, R. Dettmers, and K.V. Rosenberg. 2012. The Partners in Flight Handbook on Species Assessment: Version 2012. Partners in Flight technical series no. 3. Rocky Mountain Bird Observatory. Available online at: http://rmbo.org/pubs/downloads/PIFHandbook2012.pdf; last accessed Mar. 16, 2017.

Partners in Flight Science Committee. 2012. Species Assessment Database, Version 2012 (Bird Conservation Region 25). Available online at http://rmbo.org/pifassessment. Last accessed Mar. 16, 2017.

Rodewald, P., (Ed.). 2015. The Birds of North America. Cornell Laboratory of Ornithology, Ithaca, NY. Available online at http://bna.birds.cornell.edu/BNA/. Last accessed Jun. 27, 2017.