ABSTRACT

FRITTS, SARAH REBECAH. Implementing Woody Biomass Harvesting Guidelines that Sustain Reptile, Amphibian, and Shrew Populations. (Under the direction of Drs. Christopher Moorman and Dennis Hazel).

To address ecological sustainability of harvesting woody biomass for renewable energy and biofuel feedstocks, Biomass Harvesting Guidelines (BHGs) have been developed to ensure recommended volumes of downed woody debris (DWD) are retained following harvest. Yet, the efficacy of BHG implementation and the effects of BHGs on wildlife have not been examined at an operational scale. We developed an experiment replicated in the southeastern United States Coastal Plain in North Carolina (n = 4) and Georgia (n = 4) in 2011-2013 to determine: 1) if BHGs could be implemented to maintain desired volumes of post-harvest DWD; and 2) if wildlife are affected by varied volumes of DWD retention following harvests with and without BHGs. Each of eight replicate clearcuts contained six treatments: 1) biomass harvest with no BHGs; 2) 15% retention with biomass dispersed; 3) 15% retention with biomass clustered; 4) 30% retention with biomass dispersed; 5) 30% retention with biomass clustered; and 6) no biomass harvest.

We sampled post-harvest DWD using a line-intersect method and a visual encounter method. We used a post-harvest operational metric to determine if BHGs were successful at retaining recommended volumes of DWD. From April–August 2011-2013, we sampled herpetofauna and shrews using drift fence arrays and compared herpetofauna species richness and shrew relative abundance among treatments using analysis of variance. We identified relationships between the abundance of commonly captured species and retained DWD and vegetation using N-mixture models and linear regression. To evaluate use of DWD by *Anaxyrus* *terrestris*, we used: 1) a controlled enclosure experiment; and 2) a field-based radio-telemetry study in 2013. In the enclosure experiment, toads equipped with passive implant transponder tags selected among four 4-m^2 treatments: 1) 100% of the area covered by coarse woody debris (CWD; 100CWD); 2) ≈ 50% of the volume of 100CWD in large pile (50CWD); 3) ≈ 25% of the volume of 100CWD dispersed evenly throughout the treatment (25CWD); and 4) no CWD (0CWD). Toad movements were recorded via an electronic antenna system as they moved between treatment plots. We ranked nocturnal use of treatments during enclosure trials using compositional analysis. In the field study, we located radio-marked female *A. terrestris* daily and used mixed-effects logistic regression models to compare toad diurnal refuge site selection with available habitat.

Adding a woody biomass harvest reduced post-harvest volume of DWD by 81%. BHGs were successful at retaining volumes of DWD similar to those recommended. Across the three years of study, we captured 7,995 amphibians, 171 reptiles, and 1,252 shrews in 43,350 trap nights. We did not detect consistent differences in amphibian or reptile species richness or shrew relative abundance among treatments. However, *Gastrophryne carolinensis* abundance was positively related to volume of retained DWD in the treatment unit in Georgia in 2012. Abundance estimates of other species were not related to DWD volume. Ranking of enclosure treatments from most to least selected by *A. terrestris* (n = 47) was 0CWD, 100CWD, 25CWD, 50CWD. Use of 100CWD increased as temperature increased during periods without rain. Radio-marked *A. terrestris* (n = 37) avoided grass, bare ground, and leaf litter/sawdust, but showed no selection for DWD.

Our results suggest: 1) BHGs can be used to retain specified volumes of DWD; and 2) BHGs may not be needed at current harvest efficiencies to sustain herpetofauna and shrews.

Clearcutting and site preparation activities are the dominant disturbance events that largely drive the wildlife response, regardless of DWD removal during biomass harvests. However, if future technological advances result in increased biomass harvest efficiencies, some wildlife species may benefit from retention of DWD piles, especially in initial phases of stand development when vegetation cover is sparse.

Implementing Woody Biomass Harvesting Guidelines that Sustain Reptile, Amphibian, and Shrew Populations

by
Sarah Rebecah Fritts

A dissertation submitted to the Graduate Faculty of
North Carolina State University
in partial fulfillment of the
requirements for the degree of
Doctor of Philosophy

Fisheries, Wildlife, and Conservation Biology

Raleigh, North Carolina

2014

APPROVED BY:

Christopher E. Moorman
Committee Co-Chair

Dennis W. Hazel
Committee Co-Chair

Steven B. Castleberry

Jessica A. Homyack

Kenneth H. Pollock

DEDICATION

To my family. This dissertation would not have been possible without the emotional and financial support from my parents, Joe and Callie Fritts, and grandmother, Marilyn Brannon. To my grandfather and uncle for encouraging my enthusiasm for the outdoors. And to my husband, Todd Evans, for providing shelter from the storm.

BIOGRAPHY

Sarah grew up outside of Atlanta, Georgia. She graduated Bachelor of Science degree in Wildlife Biology and Management from the University of Georgia in 2003 and began traveling around the country for various seasonal wildlife positions. She went back to school and graduated with a Master of Science degree in Natural Resources and Environmental Sciences from the University of Illinois at Urbana-Champaign in 2008. She met Todd while in Illinois and traveled with him to Germany for two years before coming back to the United States to begin her Ph.D. research. Sarah and Todd currently live in Austin, Texas.

ACKNOWLEDGMENTS

I would like to thank my committee members, Christopher Moorman, Dennis Hazel, Jessica Homyack, Steven Castleberry, and Ken Pollock, for their guidance and patience throughout. I thank my colleagues, particularly Morgan Elfelt, Chris Farrell, Steven Grodsky, Ryan Klimstra, Marcus Lashley, Annemarie Prince, and Paul Tallie, for their countless hours of physical labor, brainstorming, and discussing all aspects of this project. Special thanks to M. Adams, J. Kraus, E. Evans, B. Moore, J. Eells, J. Kiser, J. Bowser, and J. O'Connor for working independently for long hours in the hot sun to collect the data that made this project possible. Thanks to Jen Howze for providing expertise. Finally, I would like to thank all of my friends and family for their support and understanding during the last few years.

TABLE OF CONTENTS

LIST OF TABLES

CHAPTER 1

CHAPTER 4

LIST OF FIGURES

CHAPTER 1

Biomass Harvesting Guidelines Affect Downed Woody Debris Retention

ABSTRACT

Our objective was to determine if a retention area-based Biomass Harvesting Guideline (BHG) strategy maintained desired volumes of downed woody debris (DWD) following woody biomass harvests. We implemented six randomly-assigned treatments in four clearcuts in loblolly pine plantations in the Coastal Plain physiographic region of North Carolina during 2010-2011: 1) woody biomass harvest with no BHGs (NOBHG); 2) 15% retention with woody biomass dispersed (15DISP); 3) 15% retention with woody debris clustered (15CLUS); 4) 30% retention with woody biomass dispersed (30DISP); 5) 30% retention with woody biomass clustered (30CLUS); and 6) no woody biomass harvest (i.e., clearcut only; NOBIOHARV). Prior to harvesting, we flagged 15% or 30% of the treatment area to serve as woody biomass retention sources for those treatments, and all woody biomass from the flagged area was retained and distributed across that entire treatment unit. We examined effects of treatments on: 1) fraction of estimated volume of pre-harvest standing volume (total and woody biomass) retained as residual DWD; and 2) fraction of retained DWD in treatments 2 – 5 relative to retained DWD in the NOBHGS and NOBIOHARV treatments. Estimates based on the second metric were most similar to target retentions with retention percentages at 18.8% in 15CLUS, 14.1% in 15DISP, 39.0% in 30CLUS, and 38.0% in 30DISP. Adding a woody biomass harvest reduced volume of residual DWD by 81% in NOBHG compared to NOBIOHARV. Treatments resulted in retention of DWD fractions approximate to those prescribed, suggesting BHGs can be implemented successfully in an operational setting.

KEYWORDS: biofuels**,** biomass harvesting guidelines, coarse woody debris, downed woody debris, renewable energy, residual forest biomass

INTRODUCTION

Although woody biomass has substantial potential to provide renewable energy and liquid transportation fuels (i.e., biofuels) to meet renewable standards in the United States and Europe, intensified harvesting of woody biomass for energy production could lead to a reduction in downed woody debris (DWD) in forest systems. Woody biomass harvests extract considerable volumes of logging residues and trees that are not merchantable as roundwood, which may include treetops, limbs, slash, undesirable species, and felled small trees, during traditional timber harvests (Rudolphi and Gustafsson 2005). However, in managed forests harvested for roundwood only (i.e., do not have a woody biomass harvest component), woody biomass typically is retained on the forest floor as DWD after harvest of crop trees. As markets for wood-based energy expand and technological advances in conversion and harvest machinery occur, there may be less DWD retained following woody biomass harvests (Haynes 2003, Perlack et al. 2005, Janowiak and Webster 2010).

Changes in the volume, size, and distribution of DWD following woody biomass harvests could alter ecosystem services. Coarse woody debris (CWD; defined in this study as DWD ≥ 7.62 cm in diameter for a length of at least 0.914 m (Woodall and Monleon 2008)), is important for energy flow and nutrient cycles, atmospheric carbon balance, and wildlife habitat (Harmon et al. 1986, Loeb 1996, Currie and Nadelhoffer 2002). Nutrients used for tree regeneration are stored in CWD, and a large portion of the terrestrial carbon storage may be related to CWD accumulation (Harmon et al. 1986). Further, wildlife use CWD for nesting, feeding, travel, and refugia from extreme temperatures (Hassingger 1989, Loeb 1996, Whiles and Grubaugh 1996, Currie and Nadelhoffer 2005), and abundance of some wildlife species is connected to CWD availability (Carey and Johnson 1995, Rosenvald et al. 2011). Fine woody debris (FWD) (DWD smaller than CWD) also is an important component of forest ecosystems. Branches and foliage comprise only a small proportion of total tree biomass, but contain up to half of the nitrogen, phosphorous, calcium, potassium, and magnesium immobilized in tree biomass (Alban et al. 1978). Further, in some geographic regions, FWD provides habitat for fungi, lichens, arthropods, and bryophytes, which are vital components of forest health and productivity (Crow 1988, Crow 1990). In addition to size and volume, the distribution of residual DWD may have ecological implications, particularly for wildlife (Manning and Edge 2008, Grace et al. 2011, Bechard 2012, Grace et al. 2012). Many animals use clustered DWD for denning and protection from predators while movement of small, less vagile species, such as lizards, toads, shrews, and mice may be facilitated by dispersed DWD (Harmon et al. 1986, Loeb 1999, Butts and McComb 2000, McCay 2000).

Biomass Harvesting Guidelines (BHGs) have been developed to minimize potential environmental consequences of woody biomass harvesting (Evans et al. 2010). BHGs are voluntary recommendations that emphasize the importance of DWD to wildlife and biodiversity, water quality and riparian zones, and soil productivity (Evans et al. 2010). BHGs are based on the premise that greater volumes of DWD are better than less to maintain biological diversity and site productivity and to reduce erosion risk (Harmon and Hua 1991, Loeb 1996). Most BHGs include a target fraction of pre-harvest standing volume to be retained following woody biomass harvests scattered evenly throughout the site as DWD or in small slash piles, which are thought to benefit some wildlife species (i.e., MFRC 2007, PACDNR 2008, Herrick et al. 2009, KDF 2011). Additionally, BHGs regularly include a recommendation to retain all pre-harvest CWD in an undisturbed state, presumably to ensure retention of adequate levels of CWD in all stages of decay for wildlife habitat and nutrient cycling (Herrick et al. 2009, MFRC 2011). However, retaining specified fractions of pre-harvest biomass and sizes, distributions, and a variety of decay classes of DWD may be difficult to implement at an operational level.

BHG standards vary in terms of the type and amount of DWD to be retained, and the efficacy of BHGs in meeting designated targets has not been tested. For example, Wisconsin BHGs recommend retaining tops and limbs from 10% of harvested trees, Kentucky BHGs recommend retaining 15%-30% of logging residues, and Minnesota BHGs recommend retaining tops and limbs from 20% of harvested trees (MFRC 2007, Herrick et al. 2009, KDF 2011). Pennsylvania BHGs recommend retaining 15-30% of all pre-harvest biomass (PADCNR 2008), which equates to one out of every three to six trees harvested and results in considerably greater volumes of DWD post-harvest than retaining 30% of pre-harvest volume not merchantable as roundwood or 30% of tops and butts. However, methods for ensuring debris retention, quantifying the volume of debris to be retained, and assessing the success of retention efforts are not clearly described in published BHG documents. Instead, implementation strategies are left to individual loggers or forest managers. Yet, there are substantial logistical challenges to estimating fractions of pre-harvest standing volume to be retained as DWD (Fielding et al. 2012) and to monitoring to ensure targeted woody biomass retention goals are met.

At the time of harvest, most forest stands do not have a field-based pre-harvest inventory of available woody biomass, which poses a significant challenge to implementation of BHG standards based on pre-harvest values. Therefore, strategies that do not depend on pre-harvest inventories likely are more easily and accurately implemented. For example, one method for retaining a specific fraction of pre-harvest available woody biomass is to retain targeted biomass (e.g., small-diameter trees, tops and butts, logging residues) from an area of the forest stand of equal size to a percentage specified in BHG standards (e.g., woody biomass from 30% of the stand area). This retention area-based method promotes harvesting roundwood and woody biomass simultaneously and requires only one pass through the forest stand. Another method is to harvest all roundwood and then return later to harvest woody biomass. However, returning after the roundwood harvest to harvest woody biomass requires a second pass through the forest stand, which is less economical than a one-pass system (Stuart et al. 1981, Puttock 1986, Watson et al. 1986). A third method is to retain biomass on site from an equivalent fraction of grapple loads during the entire harvest operation (e.g., one load of every three to retain approximately 30%). Although the third method is a one-pass system, DWD may need to be delivered to the loading dock and later redistributed across the harvest site; if DWD is cut and retained where it lay, logging equipment will need to navigate around piled residual DWD, which also could increase harvest costs. The success of these methods at achieving targeted retention goals has not been quantified or compared.

If implementation of BHGs in operational settings fails to reach target goals for woody biomass retention, then BHGs potentially may not protect forest ecosystems as intended. Therefore, research is needed to test the efficacy of BHGs in an operational context to guide their future development and implementation. Our objectives were to use a pre-post randomized complete block design to: 1) determine if residual DWD is reduced by adding a woody biomass harvest to a clearcut operation; and 2) determine if retention area-based BHGs are successful at retaining specified volumes and arrangements of DWD following clearcut and woody biomass harvests on an operational scale in pine plantations in the southeastern United States. Our research is part of a large-scale study that addresses the effects of woody biomass harvests and BHGs on biodiversity and site productivity.

METHODS

Study Area and Design

Our study was conducted on four clearcut sites in intensively managed loblolly pine (*Pinus taeda*) plantations in the Coastal Plain Physiographic Region of North Carolina, USA. All sites were between 77º0'0W and 76º53'50W and 35º34'0N and 35°38'20N. Study sites were managed for conventional forestry products from loblolly pine plantations (i.e., pulpwood and sawtimber) and were 32-39 years old at the time of harvest. Stands had two commercial thinning entries before the final harvest and one broadcast herbicide treatment with Chopper© + Red River Supreme surfactant year one post-harvest. The soil series on the study sites was predominately Bayboro loam, leaf silt loam, and Pantego loam (National Resources Conservation Services 2012).

We used a pre-post randomized complete block experimental design with four replicate clearcuts as blocks to test the success of a retention area-based BHG implementation strategy at retaining desired fractions of pre-harvest standing volume as post-harvest DWD. The retention area-based implementation strategy identifies an area of the forest stand of equal size to a percentage specified in BHG standards and retains all woody biomass from that retention area while harvesting woody biomass from the remainder of the site (e.g., woody biomass from 30% of the stand area). We divided each of the four clearcuts into six treatment areas, each of which was randomly assigned to one of the following woody biomass retention treatments: 1) clearcut with a woody biomass harvest and no BHGs implemented (NOBHG); 2) clearcut with 15% retention with woody biomass dispersed (15DISP); 3) clearcut with 15% retention with woody biomass clustered (15CLUS), 4) clearcut with 30% retention with woody biomass dispersed (30DISP); 5) clearcut with 30% retention with woody biomass clustered (30CLUS); and 6) clearcut with no biomass harvest (i.e., clearcut only; NOBIOHARV), which served as a reference. Treatment classifications were based on existing BHGs in other states which often recommend retaining 10-15% or 20-30% of woody biomass on the forest floor as DWD (PADCNR 2008, Herrick 2009, KDF

2011, MFRC 2011). Treatment areas were 11.4 ± 0.4 (mean ± SE) ha in size. The four replicate clearcuts were harvested by three different loggers; one logger harvested two replicate clearcuts.

Sites were harvested (i.e., roundwood and woody biomass) and biomass harvesting treatments were implemented from November 2010 through February 2011. Large pines were harvested as sawtimber and pulpwood, large hardwoods were harvested as pulpwood or woody biomass depending on market value, and small-diameter pines and hardwoods and pine tops were harvested or retained as woody biomass. Because our study had an operational scope, we did not define woody biomass in our instructions to loggers, but instead allowed the loggers to determine what was merchantable and non-merchantable as roundwood and woody biomass. Though researchers thought all hardwoods would be considered woody biomass, we were informed after the field measurements that product replacement between hardwood woody biomass and pulpwood existed based on market values and mill quotas. Therefore, some large hardwoods were taken to the mill as pulpwood, but the exact volumes are unknown. Although limbs from large pines can be used as woody biomass, researchers were informed after treatment implementation that operators chipped only hardwoods from our study sites and left all pine limbs on the forest floor. Pine limbs were stripped at the site of felling to increase roundwood harvest efficiency. For the NOBHG treatments, we instructed loggers to follow operating procedures for a typical woody biomass harvest. Therefore, loggers harvested all roundwood and woody biomass from the entire NOBHG treatment units. For the NOBIOHARV treatments, we asked loggers to leave all woody biomass on site and sawtimber and pulpwood were harvested as normal. The four BHG treatments were implemented using retention areas that were selected based on fractions (i.e., 15% or 30%) of total treatment area calculated in ArcGIS. We located retention areas using a handheld Garmin Rino GPS (Olathe, Kansas, USA) and flagged boundaries. During clearcut and treatment implementation, all standing pines and hardwoods merchantable as roundwood were cut, delimbed, and brought to the logging deck with a skidder and grapple. Hardwoods not merchantable as roundwood (i.e., woody biomass) from the retention areas were cut, left intact, and redistributed throughout the site with the skidder. In the non-retention areas, woody biomass was recovered and chipped at the logging decks during harvesting. Each retention area was clearcut after the other 85% or 70% of the treatment area was harvested. We instructed loggers to retain woody biomass from retention areas to be either spread evenly throughout the dispersed (DISP) treatments or placed in grapple-sized clusters randomly throughout the clustered (CLUS) treatments. In February through March 2011 (prior to DWD sampling), sites received V-shearing to reduce stumps and fell any remaining stems. Shearing moved debris into 3 m-wide, long, linear debris rows. However, DWD retained in the CLUS and NOBIOHAR treatments for BHGs easily could be identified for post-harvest quantification because it was primarily hardwood and remained in large discrete piles. Piles were oriented with the linear rows and were not disturbed during V-shearing. DWD retained in the DISP treatments was pushed into long, linear debris piles along with the stripped pine limbs.

Field Sampling

In summer 2010, prior to clearcut harvesting, we estimated volumes of total standing volume, standing volume that was not merchantable as roundwood (i.e., pre-harvest standing available woody biomass), and pre-harvest downed CWD. We sampled pre-harvest total standing volume using the United States Forest Service Forest Inventory and Analysis phase two inventory protocol (Bechtold and Scott 2005) with nine inventory plots distributed across each treatment area in a systematic grid. We measured all trees with a diameter at breast height (DBH; diameter at 1.37 m above ground) ≥ 12.7 cm in a 7.32-m diameter macroplot and all tress with a DBH < 12.7 cm in a 2.1-m diameter nested microplot. The center of the nested microplot was 3.66 m and 90º from the macroplot center. We recorded DBH, total height, and species for all measured trees. For estimating volume, we sorted trees by species and three size classes: sapling, trees with a 2.5 cm ≤ DBH < 12.6 cm; pole, softwoods with 12.7 cm ≤ DBH < 22.9 cm and hardwoods with 12.7 cm ≤ DBH < 27.9 cm; and sawtimber, softwoods with a DBH ≥ 22.9 cm and hardwoods with a DBH ≥ 27.9 cm for estimating volumes. We used equations from Oswalt and Conner (2011) to calculate gross (0.30-m stump to tip) volume for each size class of each species. We estimated pre-harvest total standing volume by adding gross volume of all species and size classes. We calculated net volumes (0.30-m stump to 10-cm top) for each tree using methods found in Oswalt and Conner (2011). Although, merchantability and subsequent final products of harvested wood can fluctuate with delivery prices for fuel chips, fuel costs, and prices of other forest products at the time of harvesting, large pine boles typically are harvested for sawtimber and pulpwood, leaving hardwoods, saplings, and pine tops and limbs as available woody biomass. Therefore, we calculated volume of pre-harvest available woody biomass by summing the estimated gross volume of hardwoods, the estimated gross volume of saplings (pine and hardwood), and the difference between the estimated gross and net volumes for large pines to account for pine tops. Although we included pine tops and pine saplings in our estimated volume of pre-harvest standing available woody biomass, loggers focused on hardwoods as available woody biomass.

At the nine inventory plots per treatment area, we also sampled pre-harvest DWD using a line-intersect sampling (LIS) technique (Van Wagner 1968). We established transects from the inventory plot center point at 0º, 120º, and 240º azimuths for sampling CWD and FWD. Transects for sampling CWD were 7.32 m in length and transects for sampling FWD were 3.13 m in length and all radiated from the center point. We classified logs into one of five decay classes from class one, which is recently downed wood that is sound with intact limbs and bark, to class five, which represents the most advanced stage of decomposition with debris having soft, blocky to powdery texture (Maser et al. 1979). We extrapolated volume of CWD and FWD over each treatment area (m^3 ha^{-1}) using methods described Woodall and Monleon (2008). Residual DWD sampling occurred after V-shearing, which resulted in the majority of DWD being in long, linear debris rows; therefore, relatively low volumes of DWD was retained scattered evenly throughout the site. The linear DWD rows left after shearing were comprised of few stems scattered throughout the rows or large DWD piles. Piles created through the V-shearing process were distinguishable from piles created by treatment implementation because of the type of DWD (i.e., piles created solely by shearing were predominantly pine and the piles created by the retention treatments were predominantly hardwood), the DWD orientation within the pile (i.e., piles created by treatment implementation were deposited by the skidder and the trees within the pile had the same orientation with aligned butt ends), and the density of the piles (i.e., most piles created through V-shearing were not as dense as piles retained for treatment implementation because piles created through treatment implementation included whole trees in similar orientations). Pine piles typically were smaller than hardwood piles and were created when pine limbs that were broken or removed from tree boles during harvesting, skidding, and loading were opportunistically pushed together during shearing. We defined hardwood piles as two or more hardwood trees retained on the forest floor as a result of retention treatments (i.e., retention of 15%, 30%, or all woody biomass), whereas we defined large pine piles as any pine residue pile > 1.6 m in height or > 3.8 m in length and less than 50% soil. Because sampling occurred after V-shearing, all treatments contained piled DWD (i.e., DISP treatments contained piles created through V-shearing) and scattered DWD (i.e., DWD missed during V-shearing).

We used LIS and completed a visual encounter census to estimate volume of residual DWD after harvest, treatment implementation, and V-shearing in summer 2011. We used the LIS method to sample scattered CWD, scattered FWD, and small slash piles, and the visual encounter census for larger piles, particularly the large pine piles created by mechanical site preparation and the large hardwood piles created during the BHG retention process. Because few pine piles and no hardwood piles were within our LIS macroplots, we visually located and measured the length, width, and height and visually estimated the packing ratio (i.e., density of wood) of total woody debris and packing ratio of CWD of all hardwood and large pine piles. We measured only the large pine piles with the visual encounter method because many small pine piles were intercepted and measured with the LIS method. We recorded each pile location using a Garmin Rino GPS (Olathe, Kansas, USA). We confirmed by GIS that no piles were counted with both methods. We summed the volume of piles estimated from the visual encounter method and the volume of scattered debris estimated and extrapolated using LIS to generate total debris volume (m^3 ha^{-1}) for each treatment area. Each site contained two or three logging decks for harvesting the six treatment units. Large butts of trees from the entire replicate (i.e., all treatment units within the site) were retained at the logging deck; therefore, woody debris at logging decks was not included in the final debris volume estimates.

***Statistical Analysi*s**

Because variances were not equal among treatments, we used two-way design nonparametric Friedman tests to compare pre-harvest hardwood standing volume, pre-harvest pine standing volume, pre-harvest standing volume available as woody biomass, and pre-harvest CWD among treatments with treatment and replicate clearcut as class variables; no differences would suggest homogeneity among clearcuts. We used separate Friedman tests to compare volumes of residual piled DWD, residual piled pine DWD, residual piled hardwood DWD, residual scattered DWD, residual scattered pine DWD, residual scattered hardwood DWD, total residual DWD, residual CWD (scattered plus piled), and residual FWD (scattered plus piled) among treatments. Also, we estimated the fraction of pre-harvest and residual CWD in each decay class. We used post-hoc asymptotic general independence tests to examine pairwise differences between treatments when the Friedman test was significant. We conducted all analyses using statistical software program R and set $\alpha = 0.05$.

We calculated the percent reduction of DWD from harvesting woody biomass using the following equation:

$$(DWD_{NOBIOHARV}\ j - DWD_{NOBHG}\ j)/\ DWD_{NOBIOHARV\ j} \qquad \text{(equation 1)}$$

where $DWD_{[\text{treatment}]j}$ is the volume ($m^3\ ha^{-1}$) of DWD in either the NOBIOHARV or NOBHG treatment and block (*j*).

To assess how closely BHG treatments (i.e., 15CLUS, 15DISP, 30CLUS, and 30DISP) approximated target retention goals, we used: 1) a predictive modeling metric based on the pre-harvest inventory (equations 2 and 3); and 2) a post-harvest operational-based metric (equation 4). For the predictive modeling metric, we calculated the fraction of DWD retained based on two of the pre-harvest inventory estimates. The first estimate used pre-harvest total standing volume and was calculated using the following equation:

$$(\text{residual DWD}_{ij}/\text{pre-harvest total standing volume}_{ij})*\ 100 \qquad \text{(equation 2)}$$

where residual DWD_{ij} is the volume ($m^3\ ha^{-1}$) of residual DWD in treatment (*i*) and block (*j*) and pre-harvest total standing volume_{ij} is the volume of all standing trees ($m^3\ ha^{-1}$) in treatment (*i*) and block (*j*) prior to harvest. The second estimate used pre-harvest available woody biomass volume and was calculated using the following equation:

(residual DWD_{ij}/pre-harvest available woody biomass $volume_{ij}$) * 100 (equation 3)

where residual DWD_{ij} is the volume (m^3 ha^{-1}) of residual DWD in treatment (*i*) and block (*j*) and pre-harvest available woody biomass $volume_{ij}$ is the volume (m^3 ha^{-1}) of standing pine saplings, pine tops, and hardwoods. Although we assumed complete recovery of woody biomass in the NOBHG treatments, considerable DWD remained. We thus considered DWD retained in the NOBHG treatments to be non-merchantable as woody biomass and used this volume as a baseline below which DWD is not likely to be reduced (i.e., DWD not merchantable for any forest product). Therefore, we calculated the fraction of what we considered to be merchantable biomass retained using an operational-based metric with the following equation:

($DWDij$ – $DWD_{NOBHG}\,j$) /$DWD_{NOBIOHARV}\,j$ * 100 (equation 4)

where $DWDij$ is the volume (m^3 ha^{-1}) residual DWD in treatment (*i*) and block (*j*).

RESULTS

The treatment units were similar prior to harvesting (Table 1). Adding a woody biomass harvest component to a clearcut roundwood harvest reduced the volume of residual DWD by up to 81% in NOBHG compared to NOBIOHARV (equation 1). As expected, piled hardwood constituted the majority of woody biomass, especially in treatments with a woody biomass harvest (Figure 1). NOBIOHARV had greater volumes of residual DWD than NOBHG; however, although the 30% retention treatments had greater volumes of retained DWD than the 15% retention treatments, the volumes of retained DWD did not significantly differ among the four BHG treatments (Table 2). All residual CWD was in decay class one, whereas pre-harvest CWD was in decay classes two through five with 30.2% in decay class two, 56.4% in decay class three, 11.0% in decay class four, and 2.3% in decay class five.

Although we based retention percentages on pre-harvest available woody biomass, considerable fractions of pre-harvest total standing volume were retained as DWD (equation 2, Table 3). Retention fractions estimated by predictive modeling based on pre-harvest available woody biomass (equation 3) were greater than prescribed because substantial standing volume was considered non-merchantable (e.g., pine limbs) and was retained as DWD (Table 3). Retention fractions in the four BHG treatments estimated by the post-harvest operational metric that considered DWD retained in the NOBHG and NOBIOHARV treatments (equation 4) most closely resembled the target retentions (Table 3). Although total volume of DWD was similar in the DISP and CLUS treatments, DWD in treatments where dispersion occurred had greater densities of smaller piles compared to the clustered treatments (Figure 2).

DISCUSSION

Our study demonstrates that BHGs can be implemented in an operational setting and that targeted volumes of woody biomass retention can be achieved. Although BHGs typically do not prescribe methods for retaining target volumes of DWD, our approach using retention areas equivalent to BHG percentages met specified goals in intensively managed loblolly pine stands in the Atlantic Coastal Plain of the southeastern U.S. Further, the retention area-based implementation method was favored by loggers compared to retaining specified fractions of grapple loads (personal communication, loggers in field study area). However, the retention area-based approach may yield less accurate volumes of retained DWD on less homogenous sites where woody biomass-yielding debris within the retention area differs from average volume over the entire management unit. Additionally, implementation of retention area-based BHGs will rely on landowner or logger success at marking retention area boundaries, leaving target retention volumes, and redistributing debris throughout the site. In our study, DWD was not re-distributed evenly across the DISP treatment units, but instead retained in small debris piles. Redistributing DWD evenly was difficult because skidder operators often dropped more than one tree at a time with the grapple and because site preparation immediately following harvest moved most debris into piles. Methods other than retention area-based strategies also may be effective at retaining specified volumes of DWD, but additional site impacts including the number of passes required must be considered.

Biomass Harvesting Guidelines that specify retention levels as a percent of marketable debris during harvests may be more likely to meet retention goals than BHGs based on pre-harvest inventories of woody biomass availability. Field-based pre-harvest inventories rarely are available due to their financial and time costs, particularly inventories of non-merchantable debris such as available woody biomass. Target retentions of DWD based on pre-harvest inventories will not be met unless retention goals are quite low, especially those based on pre-harvest total standing volume. For example, BHG language based on pre-harvest total standing volume often suggests similar retention fractions as what was retained in NOBIOHARV on our study sites. An average of 26.7% of estimated pre-harvest total standing volume was retained in NOBIOHARV, which is similar to Pennsylvania's retention target of 30% of pre-harvest total standing volume (PADCNR 2008).

Volume and spatial arrangement of retained DWD likely varies with multiple factors (e.g., current markets, loggers and logging equipment, and focal wood products), and stronger demand for woody biomass or improved harvest efficiencies (mechanical or economical) could result in lower volumes of logging residue where woody biomass harvests occur, particularly without BHGs. Sawtimber was the primary product across our study sites, and optimizing volumes of harvested woody biomass was not a primary goal. Although our study focused mainly on hardwood tree boles for woody biomass, increases in demand of woody biomass for energy and biofuels could lead to higher chip prices and the expansion of harvests to include other debris such as stumps, pine limbs, and small-diameter living stems. Further, shifts in local market dynamics may influence residual DWD through product replacement. For example, paper and cellulose fiber mills near our study sites maintain a market for smaller-diameter trees, but areas without these facilities may have greater volumes of non-merchantable biomass available for other uses. Yet, when local mills reach saturation, small-diameter trees that normally go to the mill may instead be chipped as woody biomass or retained as DWD on the harvest sites. For example, large hardwoods that were merchantable as pulpwood often were retained on our study sites. While interviewing the loggers responsible for treatment implementation, we determined that large hardwoods were retained either because their harvest was not economical or because roundwood quotas had been met.

Retaining a variety of size classes of DWD, particularly large CWD (> 30-cm diameter), may be beneficial to wildlife and nutrient cycling and could be emphasized in BHGs. Study sites with no BHGs had near pre-harvest volumes of FWD, and all treatments had greater volumes of residual CWD, which is expected following a disturbance such as a clearcut harvest (Harmon et al. 1986). Although all residual CWD was in decay class one (i.e., sound) whereas various decay classes existed before harvest, decay rates of DWD are rapid in the Southeast (Harmon et al. 1986, Moormon et al. 1999). Therefore, it may be important to retain large CWD that will persist for more than a few years.

BHGs may not be needed under current operational and economic efficiencies in southeastern pine plantations. The minimum volumes of debris retained in a treatment unit was 7.81 t ha-1, so all treatment units exceeded by at least three-fold the Forest Guild's volume recommendations of 2.24 t ha-1 in pine forests of the Piedmont and Coastal Plain physiographic regions of the U.S (Perschel et al. 2012). Considerable volumes of what we defined as woody biomass may have been retained in NOBHG treatments because harvest inefficiencies due to mechanical or economic limitations prevented complete harvest of DWD. Pine limbs, which we thought would be harvested, were retained and comprised the majority of DWD in the NOBHG treatments. Skidders were used to de-limb pine trees so that more trees could be carried to the logging deck in each load, thereby resulting in greater volumes of retained DWD than expected.

Variations in site preparation activities can result in differences in residual DWD volume and arrangements (Neu et al. 2014), regardless of the BHG implementation method used. For example, we conducted a similar replicated study with the same treatment implementation in the Coastal Plain of Georgia, USA, but site preparation activities in Georgia differed. Instead of shearing DWD into 3-m-wide rows with 3-m-wide bedded rows in between, Georgia sites had few very large windrows and spot piles consisting of tops, limbs, soil, and other debris. Few individual stems or smaller piles were left scattered throughout the site. Volumes of DWD at Georgia research sites exceeded BHG target retention goals and the volume of DWD retained from retention areas was low relative to the high volume of DWD that was retained from the non-retention areas, causing differences in DWD volume among treatments to be undetectable.

Although our study addresses how well BHGs can be implemented operationally, additional research is needed on implementation in other physiographic regions, particularly when different site-preparation activities are used or in areas managed for other tree species (e.g., hardwood, naturally regenerated mixed-deciduous coniferous forest). Further, research should be conducted to determine if BHGs achieve the sustainability goals for which they were developed. We recommend including retention-area based implementation strategies into BHGs and researching site productivity and wildlife response to BHGs to set appropriate BHG target volumes.

ACKNOWLEDGMENTS

Funding was provided by the United States Department of Agriculture National Institute of Food and Agriculture Managed Ecosystems Program, the Department of the Interior Southeast Climate Science Center, the Biofuels Center of North Carolina, the North Carolina

Bioenergy Initiative, Weyerhaeuser Company, Plum Creek, Georgia Pacific, and the National Council for Air and Stream Improvement. Thanks to T. B. Wigley, E. Vance, J. Homyack, K. Pollock, S. Castleberry, and Z. Leggett for review of early drafts of the manuscript. Special thanks to N. Osborne, A. Metzger, S. Grodsky, B. Moore, J. Eells, and J. Kiser for help marking retention areas and collecting field data. We thank W. Biggs, T. Glancy, J. Harris, and T. Leggett as the forest managers

LITERATURE CITED

Alban, D.H., Perald, D.A., Schlaegel, B.E., 1978. Biomass and nutrient distribution in aspen, pine, and spruce stands on the same soil type in Minnesota. Can. J. For. Res. 8; 290-299.

Bechtold, W.A., Scott, C.T., 2005. The Forest Inventory and Analysis Plot Design. In Bechtold, WA, Patterson, PL, editors. The Enhanced Forest Inventory and Analysis Program National Sampling Design and Estimation Procedures. United States Department of Agriculture Forest Service. General Technical Report SRS-80. Southern Research Station. Asheville, North Carolina, USA. pp. 27-42.

Bechard, A.M., 2012. Influence of Row Spacing and Debris Distribution on Vegetation and Small Mammals in Louisiana Pine Plantations. Master of Science Thesis. Department of Renewable Natural Resources. Louisiana State University.

Butts S.R., McComb, W.C., 2000. Associations of forest-floor vertebrates with coarse woody debris in managed forests of western Oregon. J. Wildl. Manage. 64, 95-104.

Carey A.B., Johnson, M.L., 1995. Small mammals in managed, naturally young, and old-growth forests. Ecol. Appl. 5, 336-352.

Crow, T. R., 1988. Biological diversity: Why is it important to foresters? In: Managing north-central forests for non-timber values. Society of American Foresters. Duluth, Minnesota, USA. pp. 969-977.

Crow, T. R., 1990. Old growth and biological diversity: a basis for sustainable forestry. In: Old-growth forests: What are they? How do they work? Proceedings from the conference on old growth forests: 1990 January 20: Canadian Scholars' Press, Inc. Toronto, Ontario, Canada. pp. 49-62.

Currie, W.S., Nadelhoffer, K.J., 2002. The imprint of land-use history: Patterns of carbon and nitrogen in downed woody debris at the Harvard Forest. Ecosystems. 5; 446-60.

Evans, A.M., Perschel, R.T., Kittler, B.A., 2010. Revised assessment of biomass harvesting and retention guidelines. Forest Guile, Sante Fe, New Mexico, USA.

Fielding, D., Cubbage, F., Peterson, M., Hazel, D., Gugelmann, B., Moorman, C., 2012. Opinions of forest managers, loggers, and forest landowners in North Carolina regarding biomass harvesting guidelines. International Journal of Forestry Research.

Grace, J.L., Chamberlain, M.J., Miller, D.A., 2012. Effects of Row Spacing and Debris Distribution on Deer Forage and Carrying Capacity in Newly Established Loblolly-pine Plantations in Louisiana. Proc. Annu. Conf. Southeast Assoc. Fish and Wildl. Agencies 66; 133–139.

Grace, J.L., Chamberlain, M.J., Stouffer, P.S., Miller, D.A., 2011. Effects of row spacing and debris distribution on small mammal and vegetation communities in newly established loblolly pine plantations. Proc. Annu. Conf. Southeast Assoc. Fish and Wildl. Agencies 65; 38-44.

Harmon, M.E., Franklin, J.F., Swanson, F.J., Sollins, P., Gregory, S.V., Lattin, J.D., 1986. Ecology of coarse woody debris in temperate ecosystems. Adv. Ecol. Res., 15; 133-302.

Harmon, M.E., Hua C., 1991. Coarse woody debris dynamics in two old-growth ecosystems. Bioscience 41; 604-610.

Hassinger, J.D., 1989. Dead wood for wildlife. In Finley, JC, Brittingham MC, editors. Timber Management and its effects on wildlife: Penn State Forest Resources Issues Conference Proceedings. School of Forest Resources and Penn State Cooperative Extension, University Park, Pennsylvania, USA, pp. 171-84.

Haynes, R.W., 2003. An analysis of the timber situation in the United States: 1952 to 2050. United States Department of Agriculture Forest Service, PNW-GTR-560. Pacific Northwest Research Station, USA.

Herrick, S.K., J.A. Kovach, E.A. Padley, C.R. Wagner, D.E. Zastrow. 2009. Wisconsin's forestland woody biomass harvesting guidelines. WI DNR Division of Forestry and Wisconsin Council on Forestry, PUB-FR-435-2009. Madison, Wisconsin, USA.

Janowiak, M.K., Webster, C.R., 2010. Promoting ecological sustainability in woody biomass harvesting. J. For. 108; 16-23.

Kentucky Division of Forestry (KDOF), 2011. Recommendations for the harvesting of woody biomass. Kentucky Division of Forestry. Frankfurt, Kentucky, USA. 5pp.

Loeb, S. C., 1996. The role of coarse woody debris in the ecology of southeastern mammals. In McMinn, J. W. and D. A. Crossley Jr. (editors). Biodiversity and coarse woody debris in southern forests: proceedings of the workshop on coarse woody debris in southern forests: effects on biodiversity. United State Department of Agriculture Forest Service. General Technical Report SE-94, Southern Research Station, Asheville, North Carolina, USA, pp.108-18.

Loeb, S.C., 1999. Responses of Small Mammals to Coarse Woody Debris in a Southeastern Pine Forest. J Mammal. 80; 460-471.

Manning, J.A., Edge, W.D., 2008. Small mammal responses to fine woody debris and forest fuel reduction in southwest Oregon. J. Wildl. Manage. 72; 625-32.

Maser, C., Anderson, R.G., Cromack, Jr. K, Williams, J.T., Martin, R.E., 1979. Dead and down woody debris. In: Thomas, JW (editor). Wildlife habitats in managed forests: the Blue Mountains of Oregon and Washington. United States Department of Agriculture Forest Service Agricultural Handbook No. 553. pp 78-95.

McCay, T.S., 2000. Use of Woody Debris by Cotton Mice (*Peromyscus gossypinus*) in a Southeastern Pine Forest. J Mammal. 81; 527-535.

Minnesota Forest Resources Council (MFRC), 2007. Biomass Harvest Guidelines. Minnesota Forest Resources Council, St. Paul, Minnesota, USA.

Moorman, C.E., Russell, K.R., Sabin, G.R., Guynn Jr., D.C., 1999. Snag dynamics and cavity occurrence in the South Carolina Piedmont. For. Ecol. Manage. 118; 37-48.

Neu, J., Jones, P. D., Demaris, S., Ezell, A. W., Riffell, S. K., Wigley, T. B. in press. Retained woody structure in 1- to 2-year –old loblolly pine (*Pinus taeda* L.) plantations in Mississippi, Louisiana, and Arkansas: implications for wildlife conservation. Journal of Sustainable Forestry.

Oswalt, C.M., Conner, R.C., 2011. Southern forest inventory and analysis volume equation user's guide. General Technical Report SRS–138. United States Department of Agriculture Forest Service, Southern Research Station. Asheville, North Carolina, USA.

Pennsylvania Department of Conservation and Natural Resources (PADCNR), 2008. Harvesting woody biomass for energy. Pennsylvania Department of Conservation and Natural Resources. Harrisburg, Pennsylvania, USA.

Perlack, R.D., Wright, L.L., Turhollow, A.F., Graham, R.L., Stokes, B.J., Erback, D.C., 2005. Biomass as feedstock for a bioenergy and bioproducts industry: the technical feasibility of a billion-ton annual supply. Oakridge National Laboratory, Oak Ridge, Tennessee, USA.

Perschel, B., Evans, A., DeBonis, M., 2012. Forest biomass retention and harvesting guidelines for the Southeast. Forest Guild Southeast Biomass Working Group. Forest Guild. Santa Fe, New Mexico, USA.

Puttock, G.D., 1986. The economics of collecting and processing whole-tree chips and logging residues for energy. Forest Products Journal 37; 15-20.

Rosenvald R., Lohmus A., Kraut A., Remm L., 2011. Bird communities in hemiboreal old-growth forests: The roles of food supply, stand structure, and site type. For. Ecol. Manage. 262; 1541-50.

Rudolphi, J., L. Gustafsson. 2005. Effects of forest-fuel harvesting on the amount of deadwood on clear-cuts. Scand. J. Forest. Res. 20: 235-242.

Stuart, W.B., Porter, C.D., Walbridge, T.A., Oderwald, R.G., 1981. Economics of modifying harvesting systems to recover energy wood. Forest Products Journal 31; 37 – 42.

Watson, W.F., Stokes, B.J., Savelle, I.W., 1986. Comparisons of two methods of harvesting biomass for energy. Forest Products Journal 36; 63-68.

Whiles, M.R., Grubaugh, J.W., 1996. Importance of coarse woody debris to southern forest herpetofauna. In McMinn J. W. and J. A. Crossley Jr.(editors). Biodiversity and coarse woody debris in southern forests: proceedings of the workshop on coarse woody debris in southern forests: effects on biodiversity. United State Department of Agriculture Forest Service General Technical Report SE-94, Southern Research Station, Asheville, North Carolina, USA, pp. 94-100.

Woodall, C.W., Monleon, V.J., 2008. Sampling protocol, estimation, and analysis procedures for the down woody debris indicator of the FIA program. United States Department of Agriculture Forest Service. Newtown Square, Pennsylvania, USA. NRS-GTR-22.

Van Wagner, C.E., 1968. The line intersect method in forest fuel sampling. For. Sci., 14; 20-26.

Table 1. Mean ± SE of estimated pre-harvest standing volumes (m^3/ha) of pine, hardwood, and woody biomass (i.e., biomass not merchantable for sawtimber or pulpwood) and coarse woody debris (CWD; downed woody debris ≥ 7.62 cm in diameter for a length of at least 0.914 m) in four loblolly pine (*Pinus taeda*) plantations in the Coastal Plain of North Carolina, USA (2011). We measured standing volumes and DWD prior to clearcut harvest (2010) and implementation of six woody biomass harvest treatments (n=4): 1) clearcut with a traditional biomass harvest and no BHGs implemented (NOBHG); 2) clearcut with 15% retention with woody debris dispersed (15DISP); 3) clearcut with 15% retention with woody debris clustered (15CLUS); 4) clearcut with 30% retention with woody debris dispersed (30DISP); 5) clearcut with 30% retention with woody debris clustered (30CLUS); and 6) clearcut with no biomass harvest (NOBIOHARV). Treatment means were compared using nonparametric Friedman's tests.

	Treatment							
Pre-harvest	NOBHG	15CLUS	15DISP	30CLUS	30DISP	NOBIO HARV	$_{max}T$	P_{trt}
Pine	279.39±25.22	281.77±11.31	289.60±15.94	241.88±18.54	237.61±18.95	296.33±49.11	2.46	0.14
Hardwood	42.07±7.73	32.52±10.04	43.45±12.69	39.15±15.74	49.51±14.20	44.06±3.97	1.51	0.66
Woody Biomass	64.59±5.80	54.03±10.24	65.63±11.96	66.49±19.66	67.94±133.88	64.76±4.82	2.27	0.21
DWD	2.00±1.30	0.98±0.82	1.78±1.68	0.44±.19	2.91±2.46	2.36±1.50	2.08	0.30

Table 2. Mean ± SE of estimated downed woody debris (DWD) retention (m^3/ha) of piled pine, piled hardwood, scattered pine, scattered hardwood, total DWD, coarse woody debris (CWD; DWD ≥ 7.62 cm in diameter for a length of at least 0.914 m), and fine woody debris (FWD; DWD < CWD) in four loblolly pine (*Pinus taeda*) plantations in the Coastal Plain of North Carolina, USA (2011). We measured DWD after a 2010 clearcut harvest and implementation of six woody biomass harvest treatments (n = 4): 1) clearcut with a traditional biomass harvest and no BHGs implemented (NOBHG); 2) clearcut with 15% retention with woody debris dispersed (15DISP); 3) clearcut with 15% retention with woody debris clustered (15CLUS); 4) clearcut with 30% retention with woody debris dispersed (30DISP); 5) clearcut with 30% retention with woody debris clustered (30CLUS); and 6) clearcut with no biomass harvest (NOBIOHARV). Treatment means were compared using non-parametric Friedman's tests. Different letters indicate significantly different values at the $\alpha = 0.05$ level.

	Treatment							
DWD	NOBHG	15CLUS	15DISP	30CLUS	30DISP	NOBIOHARV	maxT	P_{trt}
Piled	5.29±1.46a	27.76±9.51ab	30.80±13.52ab	39.28±13.52ab	40.50±8.42ab	90.84±20.77b	3.40	<0.01
Pine	5.12±1.30	5.42±2.57	11.57±6.40	6.08±3.98	8.92±5.01	37.84±14.55	2.65	0.09
Hardwood	0.17±0.17a	22.34±6.98ab	19.23±7.33ab	33.21±8.63ab	31.58±2.71ab	53.01±16.56b	3.21	0.02
Scattered	15.60±0.45	12.22±1.45	12.22±3.12	19.08±4.59	18.43±5.96	23.73±11.49	1.70	0.53
Pine	12.43±0.85	8.82±1.59	7.21±1.82	9.61±2.12	11.84±6.94	7.72±3.25	1.89	0.41
Hardwood	3.17±0.61	3.39±0.27	5.00±2.01	9.47±4.32	6.60±2.44	16.01±9.29	2.08	0.30
Total DWD	20.65±1.45a	37.76±9.42ab	40.80±13.11ab	55.17±12.49ab	55.75±12.49ab	108.20±20.05b	3.59	<0.01
CWD	9.47±1.60a	17.44±4.24ab	20.68±8.14ab	24.06±5.30ab	29.06±6.32ab	38.31±10.47b	3.40	<0.01
FWD	10.09±1.06a	14.84±3.43ab	13.99±2.56ab	23.36±3.33ab	18.67±7.70ab	51.76±8.50b	3.21	0.02

Table 3. Mean ± SE of estimated fractions of downed woody debris (DWD) retention (m^3 ha^{-1}) in four loblolly pine (*Pinus taeda*) plantations in the Coastal Plain of North Carolina, USA (2011). We measured pre-harvest standing volume in 2010 and post-harvest DWD after a clearcut harvest and implementation of six woody biomass harvest treatments (n=4): 1) clearcut with a traditional woody biomass harvest and no BHGs implemented (NOBHG); 2) clearcut with 15% retention with woody biomass dispersed (15DISP); 3) clearcut with 15% retention with woody biomass clustered (15CLUS); 4) clearcut with 30% retention with woody biomass dispersed (30DISP); 5) clearcut with 30% retention with woody biomass clustered (30CLUS); and 6) clearcut with no woody biomass harvest (NOBIOHARV). We estimated the fraction of DWD retained using a predictive modeling metric based on the pre-harvest inventory of total standing volume (TSV, equation 2), a predictive modeling metric based on the pre-harvest inventory of available woody biomass volume (AWB, equation 3), and a post-harvest operational metric based on the DWD retained in the NOBHG and NOBIOHARV treatments (OPM, equation 4).

	Treatment					
	NOBHG	15CLUS	15DISP	30CLUS	30DISP	NOBIOHARV
TSV	6.1% ± 0.4%	10.1% ± 1.8%	10.2% ± 2.9%	17.0% ± 2.0%	16.2% ± 2.8%	26.7% ± 4.1%
AWB	31.0% ± 3.2%	59.0% ± 4.5%	54.5% ± 14.3%	77.5% ± 8.7%	75.7% ± 19.5%	139.2% ± 24.2%
OPM		18.8% ± 9.1%	14.2% ± 14.7%	39.0% ± 2.8%	38.0% ± 11.6%	

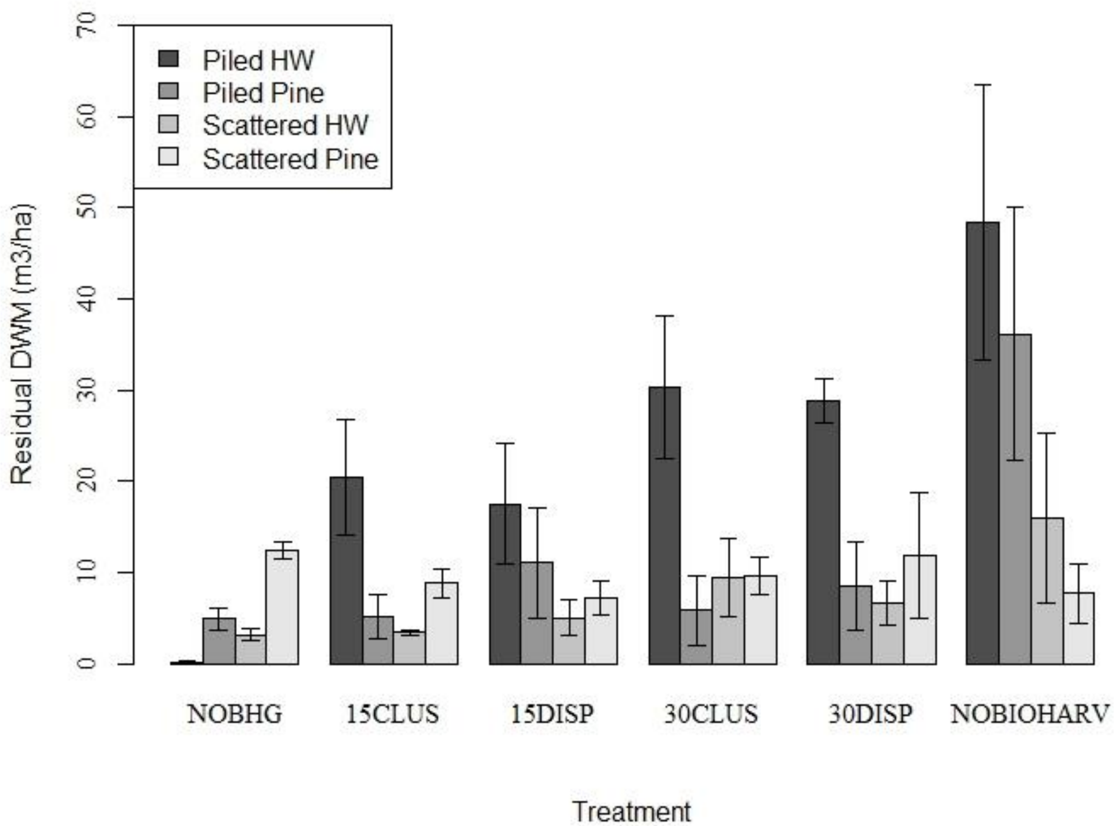

Figure 1. Mean ±SE volume (m^3/ha) of downed woody debris (DWD) of piled hardwood (HW), piled pine, scattered hardwood, and scattered pine in four loblolly pine (*Pinus taeda*) plantations in the Coastal Plain of North Carolina, USA (2011). We measured DWD volumes post-clearcut harvest and after implementation of six woody biomass harvest treatments (n=4): 1) clearcut with a traditional biomass harvest and no BHGs implemented (NOBHG); 2) clearcut with 15% retention with woody debris dispersed (15DISP); 3) clearcut with 15% retention with woody debris clustered (15CLUS); 4) clearcut with 30% retention with woody debris dispersed (30DISP); 5) clearcut with 30% retention with woody debris clustered (30CLUS); and 6) clearcut with no biomass harvest (NOBIOHARV).

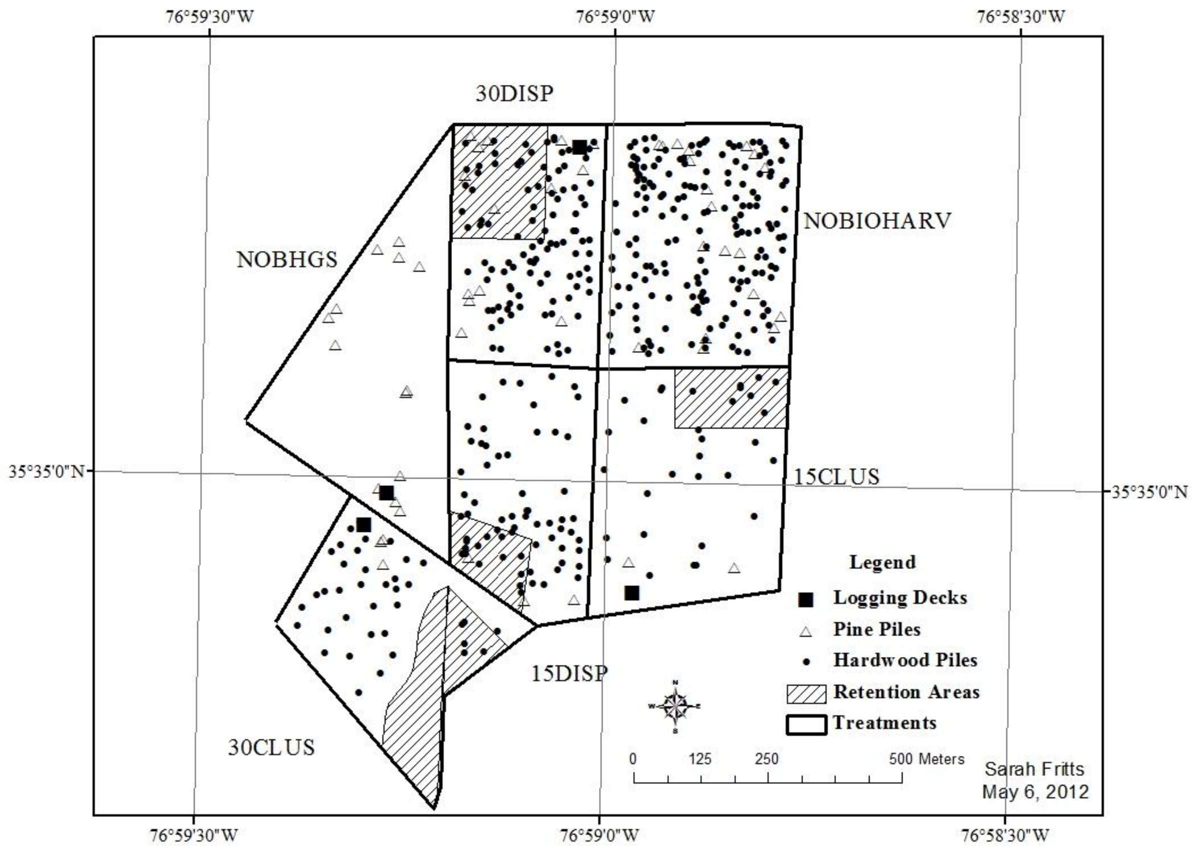

Figure 2. Piled hardwood downed woody debris (DWD) and piled pine DWD retained and logging deck locations in six woody biomass harvest treatments in one of four clearcuts in North Carolina, USA (2011). Treatments were: 1) clearcut with a traditional biomass harvest and no BHGs implemented (NOBHG); 2) clearcut with 15% retention with woody debris dispersed (15DISP); 3) clearcut with 15% retention with woody debris clustered (15CLUS); 4) clearcut with 30% retention with woody debris dispersed (30DISP); 5) clearcut with 30% retention with woody debris clustered (30CLUS); and 6) clearcut with no biomass harvest (NOBIOHARV).

CHAPTER 2

Can Biomass Harvesting Guidelines Sustain Herpetofauna Following Harvests of Logging Residues for Renewable Energy?

ABSTRACT

Wood from forests is a major supplier of renewable energy, and the southeastern United States is the largest exporter of wood pellets in the world. However, gleaning woody biomass for renewable energy could decrease downed woody debris (DWD) available for wildlife habitat. To help ensure the ecological sustainability of woody biomass harvesting, Biomass Harvesting Guidelines (BHGs) have been developed but have not been experimentally tested in an operational setting. We used herpetofauna as environmental indicators of the success of BHGs at reaching sustainability goals. We compared reptile, amphibian, and combined species richness and southern toad (*Anaxyrus terrestris*) and eastern narrowmouth toad (*Gastrophryne carolinensis*) abundance among six treatments that varied in percentage of retained DWD after clearcutting in North Carolina (n = 4) and Georgia (n = 4), USA from April to August 2011-2013. Treatments were: 1) woody biomass harvest with no BHGs; 2) 15% retention with woody biomass dispersed; 3) 15% retention with woody biomass clustered; 4) 30% retention with woody biomass dispersed; 5) 30% retention with woody biomass clustered; and 6) no woody biomass harvest. We used analysis of variance to compare species richness among treatments in each state. We used package unmarked in statistical program R to fit hierarchical abundance models while accounting for imperfect detection for *A. terrestris* and *G. carolinensis*. We included DWD volume estimates at the treatment unit scale and the drift fence array scale and vegetative structure and composition at the drift fence array scale as covariates. Sampling herpetofauna with drift fence arrays resulted in 171 reptile captures representing 21 species and 7,995 amphibian captures representing 18 species during 43,350 trap nights. Reptile, amphibian, and combined species richness estimates were similar among treatments within states. No covariates were predictors of abundance of *A. terrestris*. Eastern narrowmouth toad abundance was positively related to volume of retained DWD in the treatment unit ($P = 0.05$) and negatively related to distance to the nearest wet depression ($P = 0.04$) in 2012 in Georgia. Other relationships were weak or absent. The lack of consistent effects of retained DWD volume on herpetofauna suggests sufficient DWD volumes are retained following biomass harvests to conserve herpetofauna. However, if future technological advances result in lower volumes of DWD than what occurred on our study sites, we recommend DWD piles be retained throughout harvest sites for use by amphibians with high desiccation risk.

KEYWORDS: amphibians, biofuels, biomass harvesting guidelines, downed woody debris, reptile, woody biomass

INTRODUCTION

Climate change and depletion of fossil fuels are driving increased interest in renewable electricity and biofuels worldwide. Wood from forests (i.e., woody biomass) currently is a major supplier of renewable energy, and use of woody biomass for power, heat, and liquid biofuels is expected to increase as additional renewable energy and biofuel directives are implemented (Cook and Beyea 2000, Perlack et al. 2005, Hillring 2006, Mantau et al. 2010).

In the United States, biomass is the largest domestic source of renewable energy, accounting for approximately 3% of total primary energy consumption and 5% of production (EIA 2013). Nearly 75% of the total annual consumption of biomass is derived from forests (Perlack et al. 2005). Additionally, woody biomass is a critical source of renewable-based energy in the European Union and currently accounts for > 50%, on average by country, of the gross inland energy consumption from renewable energy sources (Mantau et al. 2010). In 2012, nearly 40% of the world's trade in wood pellets (i.e., a form of woody biomass burned for energy) was exported from North America to Europe, an increase of 50% from 2011 (REN21 2013).

However, there are concerns about the long-term sustainability of woody biomass harvests, particularly in regards to wildlife response to reductions of downed woody debris (DWD) (i.e., Riffell et al. 2011, Otto et al. 2013). The National Wildlife Federation and the Southern Environmental Law Center recently identified woody biomass extraction for use as a renewable energy feedstock as a primary risk of southeastern forest degradation (Evans et al. 2013). Woody biomass harvests extract forest harvest residues, including treetops, limbs, slash, and felled small trees, that otherwise would remain standing or on the forest floor as DWD (Rudolphi and Gustafusson 2005, Fritts et al. in review). Yet, DWD is an important nutrient source and provides habitat for a variety of wildlife (Harmon et al. 1986, Lattimore et al. 2009, Evans and Kelty 2010, Janowiak and Webster 2010, Riffell et al. 2011).

Biomass Harvesting Guidelines (BHGs) are voluntary recommendations developed to help ensure the ecological sustainability of woody biomass harvests (e.g., MFRC 2007, Röser et al

2008, PADCNR 2008, KYDOF 2011, Perschel et al. 2012). BHGs typically include a target percentage of pre-harvest woody biomass to retain as DWD following woody biomass harvests; however, suggested volumes and spatial arrangements of DWD vary among prescriptions. BHGs have been created under the general idea that “more” DWD is better than “less” to maintain biological diversity and site productivity (Harmon and Hua 1991, Ranius and Fahrig 2006), but empirical evidence on the efficacy of BHGs at reaching sustainability goals is lacking.

Herpetofauna can be used as environmental indicators of the ecological sustainability of woody biomass harvests because they require abundant DWD for thermoregulation, protection from desiccation, reproduction, and feeding substrate (Jaeger 1980, Hassinger 1989, Whiles and Grubaugh 1996, Butts and McComb 2000). Amphibians depend on ambient temperatures to regulate body temperature, have permeable skin, and have limited dispersal capabilities, and therefore are vulnerable to changes in land use (Whiles and Grubaugh 1996, Welsh and Ollivier 1998, Vitt and Caldwell 2009). Amphibians risk desiccation because their skin is subject to high rates of evaporative water loss (Jørgensen 1997), and DWD can mediate extreme temperature and moisture regimes. For example, the temperature under and inside downed wood often is lower than ambient, particularly after canopy removal (Graham 1925, Kluber et al. 2009, Homyack et al. 2011). Further, woody biomass harvests typically occur in conjunction with overstory harvests. Canopy tree removal leads to changes in air temperature and moisture regimes on the forest floor (Valigura and Messina 1994, Brooks and Kyker-Snowman 2008), which may affect some herpetofauna species. Although clearcutting may not negatively impact lizards, small snakes and amphibians often respond negatively following timber harvests (Greenberg et al. 1994, Petranka et al. 1994, McLeod and Gates 1998, Semlitsch et al. 2008, Todd and Andrews 2008, Homyack and Haas 2009). For example, juvenile southern toads (*Anaxyrus terrestris*) have lower survival and are smaller in clearcuts than in mature forests (Todd and Rothermel 2006), and the relative abundance of several snake species is reduced in clearcuts compared to unharvested or thinned stands (Todd and Andrews 2008). Thus, retaining woody biomass during timber harvests could mitigate potential negative effects of increased temperature and reduced moisture by providing suitable microclimates (Rittenhouse et al. 2008).

Concerns about sustainability of woody biomass harvesting often are focused on the southeastern United States, from where much of the wood used as forest bioenergy originates. The region is considered the “wood basket” of the United States and currently is the largest exporter of wood pellets in the world (Hanson et al. 2010, Evans et al. 2013, Goh et al. 2013). Further, the United States Department of Agriculture (USDA) predicts that ≈ 50% of second generation biofuels needed to meet the United States biofuel mandates will originate from the southeastern United States by 2020 (USDA 2010). However, the region is a global center of amphibian diversity and contains approximately half of the known herpetofauna species in the United States (Conant and Collins 1998, Gibbons and Buhlman 2001, Tuberville et al. 2005). Research in the region has demonstrated variable responses among herpetofauna species to experimental removal or manipulation of DWD (Greenberg and Waldrop 2008, Owens et al. 2008, Davis et al. 2010, Matthews et al. 2010, Homyack et al. 2013); however, no research has been conducted on herpetofauna response to operational-scale woody biomass harvests or has quantified the minimum volume of retained woody biomass needed to sustain herpetofauna populations.

If BHGs in operational settings fail to reach wildlife sustainability goals, reptiles and amphibians may not be conserved as intended. Therefore, research is needed to test the efficacy of BHGs at sustaining reptile and amphibian populations to guide their future development and implementation. The southeastern United States is an ideal location for an experimental study of herpetofauna response to woody biomass harvests because it provides an intersection between a herpetofauna biodiversity hotspot and industrial forestry supporting a woody biomass market. Because DWD decays quickly in the southeastern United States (Moorman et al. 1999), we predict the greatest effects of woody biomass harvests on DWD volume would occur within the first three to four years post-harvest, which also may represent the time period of the greatest herpetofauna response. To aid in science-based management decisions regarding the ecological sustainability of woody biomass harvesting, we examined the effects of retaining a range of DWD volumes on herpetofauna communities. Our main objectives were to use an operational-scale experiment to: 1) determine if woody biomass harvests affect amphibian and reptile species richness; 2) given an effect is observed, determine if retention of DWD as would occur under BHGs would mediate amphibian and reptile responses; and 3) increase understanding of the mechanisms affecting herpetofauna response to woody biomass harvesting. For the third objective, we used the operational-scale experiment coupled with fine-scale measures of DWD and vegetation to: 1) evaluate the relationship between volume of DWD and the abundances of the most commonly captured herpetofauna; and 2) evaluate the relationship between vegetation structure and composition and the abundances of the most commonly captured herpetofauna once vegetation was well-established in 2013.

METHODS

Study Area and Design

We designed a randomized complete block study on eight sites (i.e., replicate clearcuts): four in Beaufort County, North Carolina; three in Glynn County, Georgia; and one in Chatham County, Georgia, USA. All study sites were in intensively managed loblolly pine (*Pinus taeda*) plantations in the Coastal Plain Physiographic Region. North Carolina sites were managed for sawtimber production, had two commercial thinning entries before the final clearcut harvest, and were between 32 and 39 years old at the time of final harvest. Georgia sites were managed for chip-and-saw and pulpwood production and were between 25 and 33 years of age at the time of final clearcut harvest. Three Georgia sites had one commercial thinning entry and one site had two commercial thinning entries before harvest. North Carolina soil series were predominately loam and silt loam. Georgia soil series were predominantly loam, clay loam, and fine sandy loam.

Clearcut harvests with treatment implementations were completed in 2010-2011. North Carolina sites were 70.5 ± 6.1 ha (mean ± SE) and Georgia sites were 64.4 ± 3.1 ha. Sites were divided into six randomly-assigned woody biomass harvest treatments that were 11.7 ± 0.5 ha (range = 8.4 – 16.3 ha) in North Carolina and 10.7 ± 0.4 ha (range = 7.6 – 14.3 ha) in

Georgia. Treatments were: 1) clearcut with a traditional woody biomass harvest and no BHGs implemented (NOBHG); 2) clearcut with 15% retention with woody biomass evenly dispersed throughout the treatment unit (15DISP); 3) clearcut with 15% retention with woody biomass in large piles throughout the treatment unit (15CLUS); 4) clearcut with 30% retention with woody biomass evenly dispersed throughout the treatment unit (30DISP); 5) clearcut with 30% retention with woody biomass in large piles throughout the treatment unit (30CLUS); and 6) clearcut with no woody biomass harvest (i.e., clearcut only; NOBIOHARV), which served as a reference. Retention targets focused on hardwoods from the retention areas in the four BHG treatments or from the entire treatment area in NOBIOHARV. For the NOBHG treatments, we instructed loggers to follow normal operating procedures for a typical woody biomass harvest. For the NOBIOHARV treatments, we asked loggers to leave all woody biomass not harvested as roundwood on the ground as DWD, although sawtimber and pulpwood were harvested as normal. The four BHG treatments were implemented using retention areas that represented either 15% or 30% of the total treatment plot area using ArcGIS (ESRI, Redlands, California, USA). We located retention areas using a handheld Garmin Rino GPS (Olathe, Kansas, USA) and flagged boundaries. During clearcut and treatment implementation, all standing pines merchantable as roundwood were cut and brought to the logging deck with a skidder and grapple and most hardwoods from the retention areas were left intact and redistributed throughout the treatment unit with the skidder. Each retention area was clearcut after the non-retention treatment area was harvested. In the non-retention areas, woody biomass (i.e., hardwood) was recovered and chipped at the logging decks during harvest. In the retention areas, loggers retained woody biomass to be either spread evenly throughout the dispersed (DISP) treatments or placed in grapple-sized piles randomly throughout the clustered (CLUS) treatments.

North Carolina sites were V-sheared immediately following harvest and before herpetofauna sampling to aid in stand establishment. Shearing moved debris into the 3-m space between pine beds, and rearranged debris into long, linear rows. Sites were bedded to provide a raised growing surface for seedlings, which were planted fall-winter 2011-2012 at a density of ≈ 1,100 trees/ha. North Carolina sites received a broadcast application year one post-harvest and a banded application year two post-harvest with Chopper© (BASF, Raleigh, North Carolina, USA) with Red River Supreme surfactant.

Two of the Georgia sites were clearcut harvested with treatments implemented in winter 2010-2011 and were available for sampling in 2011. The remaining two Georgia sites were clearcut in summer-fall 2011 and were available for sampling in 2012. Immediately following harvest of the Georgia sites, most debris was windrowed or spot piled in relatively few large piles in each treatment unit; few individual stems and no smaller DWD piles were distributed throughout the treatment units. Two Georgia sites were double-bedded in August 2011 and one site in October 2011 and all three were planted in January 2012 at a density of ≈ 1,495 trees/ha. These three sites received a banded herbicide application of Arsenal© (BASF, Raleigh, North Carolina, USA) and Sulfometuron in April 2012. The fourth Georgia site was bedded, planted at a density of ≈ 726 trees/ha, and received a broadcast herbicide treatment with Chopper© (BASF, Raleigh, North Carolina, USA) in spring 2012.

Herpetofauna Sampling

We sampled amphibians and reptiles from mid-April to early-August in 2011, 2012, and 2013. In North Carolina, year one of sampling occurred after harvest and V-shearing and years two and three of sampling occurred after bedding and planting. We sampled two Georgia sites for one year after harvest and windrowing/spot piling and all Georgia sites for two years after bedding and planting. In 2011, we sampled only two Georgia sites because the others had not been harvested. In 2012 and 2013, we sampled all eight sites.

In each treatment unit, we established three 'Y'-shaped drift fence arrays with 7.6-m arms. We constructed drift fences from silt-fence material. We placed drift fences in the center of three randomly-selected quadrants in each treatment unit > 30 m apart. We placed 19-L buckets at the end of each drift fence array arm, buried flush with the ground. The center of one randomly selected drift fence in each treatment unit had a three-sided funnel trap modified from Burgdorf et al. (2005), and the other two drift fences in each treatment unit had 19-L buckets buried flush with the ground. We drilled three small holes in the bottom of each bucket for water drainage and placed sponges that were wetted daily in each trap. We used bucket lids raised ≈ 25 cm for shade to lower desiccation risk of amphibians. We sampled for ten consecutive days in 2011 and five consecutive days in 2012 and 2013. We sampled all treatments in two to three replicate clearcuts in North Carolina and one to three replicate clearcuts in Georgia simultaneously. We identified captured herpetofauna to species and released individuals 10 - 15 m from the drift fence array. Sampling procedures were approved by the North Carolina State University Institutional Animal Care and Use Committee (11-022-O), the Georgia Department of Natural Resources (scientific collection permit number 29-WJH-13-156), and the North Carolina Wildlife Resources Commission (scientific collection permit number 11SC00534).

Quantifying DWD

In 2011, we measured DWD using a line-intersect sampling (LIS) technique (Van Wagner 1968). We established 7.32-m transects from the plot center point at 0º, 120º, and 240º azimuths for sampling coarse woody debris (CWD), DWD ≥ 7.62 cm in diameter for a length of at least 0.914 m (Woodall and Monleon 2008), and 3.13-m transects for sampling fine woody debris (FWD), debris smaller than CWD. Although the LIS method worked well for scattered DWD, few DWD piles fell within our LIS plots. Therefore, we used a visual encounter method to census piled DWD. In North Carolina, we visually located and measured the length, width, height, and visually estimated the packing ratio (i.e., density of wood, 0% - 100%) of total woody debris of all hardwood and large pine piles. We defined hardwood piles as two or more hardwood trees retained on the forest floor as a result of retention treatments (i.e., retention of 15%, 30% or all woody biomass), whereas we defined large pine piles as any pine residue pile over 1.6 m in height or 3.8 m in length and < 50% soil. Pines were stripped of their limbs at harvest, and pine piles were created when pine limbs were pushed together during shearing. In Georgia, we visually located and measured the width, height, and packing ratio of all windrows and spot piles. Several windrows were created in each treatment unit. Windrows often were the length of the entire treatment unit, so we measured lengths using post-harvest aerial imagery (Google Maps, Mountain View, California) in ArcGIS (ESRI, Redland, California, USA). Georgia windrows and spot piles contained primarily pine debris. For both states, we summed volume of piled DWD estimated from the visual encounter method and the volume of scattered debris estimated using the LIS method to generate total debris volume (m^3/ha) for each treatment unit. Each site contained two to three logging decks for harvesting the six treatment units. Large butts of trees from the entire clearcut (i.e., all treatment units within a site) were left at the logging decks, but were not included in the final debris volume estimates for individual treatment units. We used a GPS point at each pile location using a Garmin Rino GPS (Olathe, Kansas, USA) and used ArcGIS to ensure no piles were double-counted across methods. We measured distance of each drift fence array to the nearest debris pile with ArcGIS. In North Carolina, where DWD piles were distributed throughout the site instead of in windrows or spot piles, we estimated the location of each DWD pile with a GPS unit and used ArcGIS to estimate the volume of piled DWD within 50 m of each drift fence array. Further, North Carolina sites had parallel drainage ditches to lower the water table and improve pine growth, and we measured the distance of each drift fence array to the nearest ditch using ArcGIS. Each of the Georgia sites had un-harvested wet depressions and we measured the distance of each drift fence array to nearest wet depression using ArcGIS.

Quantifying Vegetation

In 2013, when vegetation structure was greatest, we estimated vertical vegetation structure and groundcover composition at each drift fence array along three 10-m transects. We established each transect starting at the center of the array and radiating outwards directly between two drift fence arms. Each transect had ten sampling points at 1-m increments. We counted the number of times the vegetation touched the pole at each decimeter on a 2-m tall, 4.8-cm diameter pole at each point. We used the average number of times vegetation touched the pole per point across the 30 points as an index of vegetative structure at each fence (Moorman and Guynn 2001). We recorded all groundcover types (bare ground, CWD, FWD, herbaceous vegetation, litter, and woody vegetation) that touched the bottom of the pole at each point. Then, we calculated the % cover of each type at each drift fence by dividing the number of points with each ground cover type by 30. Groundcover could be >100% because more than one vegetation type may have been present at a point.

Weather Covariates

We collected hourly rainfall, temperature, and relative humidity data with an on-site Hobo U23 Pro v2 temperature logger (Hobo, Cape Cod, Massachusetts, USA) and Rainwise rain gauge (Trenton, Maine, USA) in North Carolina in 2012 and 2013. The on-site weather station was ≤ six kilometers from all sites. For North Carolina weather data in 2011 and Georgia weather data in all three years, we accessed the online National Oceanic and Atmospheric Administration (NOAA) National Climatic Data Center and downloaded hourly temperature and relative humidity data and daily rainfall data from the nearest weather station to each of the sites. We calculated daily mean temperature and daily mean relative humidity from the hourly weather data. We treated rain as a binomial covariate (i.e., rain within the previous 24 hours = 1, no rain within previous 24 hours = 0).

Statistical Analyses

We used separate randomized complete block design analysis of variances (ANOVA) for each state to test for differences among treatments in DWD volume, groundcover, and vertical vegetation structure. We used a two-way ANOVA to determine differences between states and among years in daily mean temperature using state and year as class variables. We used separate randomized complete block design ANOVAs for each state and year to test for differences among treatments in combined species richness, amphibian species richness, and reptile species richness with site as the blocking factor and treatment as a class variable. We used Tukey's Studentized Range criteria to separate treatment means when models were significant at the $\alpha = 0.05$ level. We conducted all analyses using statistical software program R (version 3.0.2; R Core Team 2012).

Additionally, for species with > 200 captures/year/state we tested for relationships between abundance and DWD volume, vertical vegetation structure, and groundcover composition using hierarchical N-mixture modeling (Royle 2004, Kery et al. 2005, Fiske and Chandler 2011). We conducted separate N-mixture model analyses for each state because site preparation activities varied greatly and conducted separate modeling procedures for each year because of vegetation growth and DWD decay over time. We used drift fence array as the experimental unit because the smallest-scale habitat data was collected at the drift fence array. We sometimes had to close buckets due to excessive rainfall or the presence of fire ants (*Solenopsis invicta*), an invasive species that depredates captured herpetofauna (Todd et al. 2008). Therefore, we calculated effort as the number of traps open per night per drift fence (i.e., 0 - 4). We standardized covariates, except rain and effort, before analysis by subtracting the mean and dividing by the standard deviation. We tested all covariates for collinearity by examining the correlation coefficient and omitted one covariate from analysis if two covariates had a correlation coefficient > 0.6. We used the estimate of retained DWD volume in each unit as a continuous predictor variable instead of treatment. For this analysis, focusing on the DWD volume retained, vertical vegetation structure, and groundcover composition allowed us to better understand the mechanisms affecting amphibian response to woody biomass harvests. Scattered DWD was pushed into piles by V-shearing and windrowing, which occurred before amphibian and reptile sampling. Therefore, we did not distinguish between CLUS and DISP treatments.

Detection of herpetofauna can vary based on large-scale habitat characteristics (e.g., aspect and elevation), small-scale habitat characteristics (e.g., volume of DWD within 50 m of the drift fence), environmental conditions (e.g., rain and temperature), and among species (Petranka et al. 1994, MacKenzie et al. 2002, Bailey et al. 2004, Otto and Roloff 2011). Therefore, we used the pcount function in package unmarked (Fiske and Chandler 2011) to fit hierarchical N-mixture models that incorporate detection probabilities to estimate abundances of species with > 200 captures/year/state. Function pcount uses spatially and temporally replicated count data to fit the latent N-mixture model and allows abundance estimation without identification of individuals (Royle 2004, Kery et al. 2005). We ran separate single-season N-mixture models for each year, state, and species, and identified significant predictors of abundance at the drift fence arrays. We included Julian date, daily rain occurrence (binomial), daily mean temperature, daily mean relative humidity, and daily effort as observation-level covariates (covariates that can affect detection only). We included volume of DWD in the entire treatment unit, distance of drift fence to nearest DWD pile, distance of drift fence to nearest ditch or wetland, and volume of piled DWD within 50 m of drift fence array (North Carolina only) as site-level covariates (covariates that affect detection and abundance). We added % bare ground, CWD, FWD, litter, herbaceous vegetation, and woody vegetation and vertical vegetation structure at the drift fence array as site-level covariates in the 2013 models (Table 1). For each species, we ran the global N-mixture detection model (i.e., all site- and observation-level covariates on detection) with constant abundance and determined covariates that were significant predictors of detection. We added the significant predictors of detection to the global N-mixture abundance model (i.e., significant predictors on detection and all site-level covariates on abundance). We ran Poisson and negative binomial mixture distributions for each model and used the global model with the lowest Akaike's Information Criteria (AIC) score (Akaike 1973) to identify significant predictors of abundance. We considered a covariate as a predictor if the beta value had a 95% confidence interval that did not include zero (Shake et al. 2011).

RESULTS

In North Carolina, piled hardwood comprised the majority of DWD in all treatments except NOBHG (Table 2). In Georgia, piled pine comprised the majority of DWD in all treatments (Table 3). Overall, treatments were similar in vertical vegetation structure and groundcover in North Carolina (Table 2) and Georgia (Table 3). Herbaceous vegetation comprised the majority of groundcover followed by bare ground in both states. Temperature data for 2011 was incomplete so we did not include 2011 in the ANOVA to compare temperature between states and among years. Daily mean temperature was different between states ($F_{1,311}$ = 41.12, P < 0.01) and years ($F_{1,311}$ = 56.87, P < 0.01) with Georgia nearly 7º C warmer than North Carolina, and 2012 approximately 8º C warmer than 2013.

Over three years and in both states we had 171 reptile captures representing 21 species and 7,995 amphibian captures representing 18 species during 43,350 trap nights. In North Carolina, we captured 14 reptile species (Table 4) and 12 amphibian species (Table 5). In 2011, combined species richness ($F_{5,15}$ = 0.57, P = 0.72), amphibian species richness ($F_{5,15}$ = 1.26, P = 0.33), and reptile species richness ($F_{5,15}$ = 0.61, P = 0.69) were similar among treatments. In 2012, combined species richness ($F_{5,15}$ = 2.67, P = 0.06) and amphibian species richness ($F_{5,15}$ = 1.71, P = 0.19) were similar among treatments; however, reptile species richness differed among treatments ($F_{5,15}$ = 3.83, P = 0.02) with greater richness (mean ± standard error) in 15DISP (7.25 ± 0.48) and 15CLUS (6.75 ± 0.41) than 30DISP (4.75 ± 0.25; P = 0.02 and P = 0.04, respectively). In 2013, amphibian species richness was similar among treatments ($F_{5,15}$ = 1.34, P = 0.30). Reptile species richness differed among treatments ($F_{5,15}$ = 2.84, P = 0.05) with greater richness in 30DISP (1.75 ± 0.48) than 30CLUS (0.25 ± 0.25, P = 0.05). Combined species richness differed among treatments ($F_{5,15}$ = 3.24, P = 0.04) and was greater in 30DISP (5.50 ± 0.29) than 30CLUS (3.00 ± 0.71; P = 0.04).

In Georgia, we captured 13 reptile species (Table 6) and 11 amphibian species (Table 7). In 2011, we sampled only two replicates in Georgia and did not compare species richness among treatments. In 2012, combined species richness ($F_{5,15}$ = 2.50, P = 0.08) and reptile species richness ($F_{5,15}$ = 1.15, P = 0.38) were similar among treatments. Amphibian species richness differed among treatments ($F_{5,15}$ = 3.00, P = 0.05) with greater richness in NOBIOHARV (3.50 ± 0.29) than 15DISP (2.25 ± 0.25; P = 0.02). In 2013, combined species richness ($F_{5,15}$ = 0.30, P = 0.91), amphibian species richness ($F_{5,15}$ = 0.57, P = 0.72), and reptile species richness ($F_{5,15}$ = 0.30, P = 0.91) were similar among treatments.

Individual species with >200 captures/year/state included the southern toad (*Anaxyrus terrestris*; n = 4,817) and the eastern narrowmouth toad (*Gastrophryne carolinensis*; n = 2,115), which comprised 60% and 27% of total amphibian captures, respectively. Julian date and temperature were correlated and we removed Julian date from the North Carolina 2012 and 2013 analyses and all Georgia analyses. We used Julian date instead of temperature and relative humidity for the 2011 North Carolina models because the temperature data compiled from the NOAA website was incomplete and the on-site weather station was not installed until 2012. Detection of both species depended on weather covariates (Tables 8-11). No covariates were predictors of southern toad abundance in North Carolina (Table 8) or

Georgia (Table 9). No covariates were predictors of eastern narrowmouth toad abundance in North Carolina (Table 10). Eastern narrowmouth toad abundance was positively related to volume of retained DWD in the treatment unit ($\beta = 0.29$ (log-scale), $z = 1.96$, $P = 0.05$) and negatively related to distance to the nearest wet depression ($\beta = -0.30$ (log-scale), $z = 2.09$, $P = 0.04$) in 2012 in Georgia (Figure 1; Table 11).

DISCUSSION

DWD was reduced by approximately 81% in treatment plots with no BHGs (Fritts et al. in review) with minimal detectable effects on reptile or amphibian richness or southern toad abundance, indicating most herpetofauna species in the southeastern United States likely are not affected by current levels of woody biomass harvesting. The lack of overall herpetofaunal response to woody biomass harvests is consistent with other studies involving woody debris manipulations in pine forests of the southeastern United States (Owens et al. 2008, Davis et al. 2010, Matthews et al. 2010, Homyack et al. 2013). For example, total herpetofauna captures, species diversity, and abundance of southern toads were not affected by removal of nearly 85% of DWD 2.5 years after establishment of pine plantations in eastern North Carolina (Homyack et al. 2013). Similarly, removal of nearly all DWD in mature pine stands in South Carolina had no effect on amphibian, anuran, and lizard species diversity and southern toad and eastern narrowmouth toad abundance for up to 12 years post-reduction (Owens et al. 2008, Davis et al. 2010).

Eastern narrowmouth toads may have been more reliant on retained DWD volume in 2012 in Georgia than in North Carolina because of different site preparation activities and warmer mean daily temperatures. Site preparation in Georgia resulted in relatively few and large DWD windrows and spot piles while DWD in North Carolina was more evenly distributed throughout the treatment units. Eastern narrowmouth toads are smaller, more fossorial, and have moister skin than southern toads (Collins and Collins 1993, Dorcas and Gibbons 2008); therefore, this species may be more prone to desiccation and dependent on the wet and humid microclimates that DWD cover provides. Previous research has determined that the two most important environmental variables that influence eastern narrowmouth toad presence are cover and moisture (Carter 1934, Robertson and Tyson 1950, Anderson 1954, Martof 1955). Therefore, retaining relatively small piles distributed throughout the treatment units with fewer areas of bare ground may be better than windrowing for terrestrial and fossorial amphibians that have high desiccation risk, such as eastern narrowmouth toads. Additionally, Georgia sites typically were hotter than North Carolina sites, which may have further increased desiccation risk of eastern narrowmouth toads and possibly made them more reliant on retained DWD. However, eastern narrowmouth toads did not respond to DWD extractions in North Carolina or for two years in Georgia, similar to previous research that detected no differences in captures of eastern narrowmouth toads when nearly all woody biomass was experimentally removed in southeastern United States pine forests (Owens et al. 2008). Eastern narrowmouth toads use cover other than DWD, including vegetation (Carr 1940), which may explain the lack of dependency of eastern narrowmouth toad abundance on DWD volume in Georgia as vegetation became well established in 2013.

Southern toads possess adaptations, including burrowing, using cover other than DWD, and not relying on DWD as an important foraging substrate, that contribute to occupancy of disturbed areas with low volumes of DWD (Neill 1950, Ashton and Ashton 1998, Moseley et al. 2005). Additionally, the southern toad is a habitat generalist able to tolerate habitat alteration and high levels of water loss (Bartlett and Bartlett 1999, Hillyard 1999), which may further explain their lack of dependency on DWD. In fact, southern toads select clearcuts over mature forest (Graeter et al. 2008). However, juvenile southern toads are smaller and had greater mortality in clearcuts with experimental debris removal than in unharvested forests in South Carolina, indicating that clearcuts may be poor quality habitat (Todd and Rothermel 2006). Further, several studies have demonstrated the persistence of southern toads in areas with DWD experimentally removed over areas with CWD experimentally added (Todd and Rothermel 2006, Owens et al. 2008, Davis et al 2010, Homyack et al. 2013, Fritts et al. Ch. 3). In fact, southern toads select areas with no CWD during nocturnal activity and do not select CWD for diurnal refuge when other cover sources are available (Fritts et al. Ch. 3).

Our large-scale, experimental study adds to a growing body of research suggesting amphibians and reptiles in the southeastern United States may have a weaker relationship with DWD than in other regions, where DWD volumes in forests are greater. Southeastern United States Coastal Plain forests historically were influenced by frequent fires, both lightning-caused and anthropogenic, so ground-dwelling wildlife in the region likely evolved tolerance of these disturbances and the environmental conditions they promoted (Russell et al. 2004). Further, fires coupled with the rapid decay rate of DWD in the region may have resulted in relatively low DWD volumes (Moorman et al. 1999). For example, CWD volumes in unmanaged and managed pine forests in the southeastern United States have been estimated to be approximately 18 m^3/ha and 6 m^3/ha, respectively (McMinn and Hardt 1996), while CWD volumes in the northwestern United States have been estimated to be 77 m^3/ha in lodgepole pine (*Pinus contorta*) stands (Herrero et al. 2014) and > 800 m^3/ha in coastal Oregon Douglas fir (*Pseudotsuga menziesi*) forests (Spies and Cline 1988).

Clearcut harvesting likely is the dominant disturbance event in intensively managed loblolly pine plantations and largely drives changes in amphibian populations, regardless of the volume of woody biomass retained as DWD. Although reptile species have variable responses to clearcutting (Enge and Marion 1986, Phelps and Lancia 1995, Goldstein et al. 2005, Todd and Andrews 2008), amphibian populations often decline following canopy removal (Dupuis et al. 1995, Knapp et al. 2003). Amphibian population declines following forest harvesting can be attributed to evacuation and reduced reproductive success (Enge and Marion 1986, Semlitsch et al. 2008). Although we did not compare pre-harvest to post-harvest herpetofauna communities, other studies in mature southeastern United State Coastal Plain loblolly pine plantations resulted in greater capture success of salamanders and the captures being less dominated by a single species (Owens et al. 2008, Davis et al. 2010).

BHGs may not be necessary to sustain amphibian and reptile communities at current levels of woody biomass harvests. Considerable volumes of woody biomass (20.65 ± 1.45 m^3/ha) were retained in NOBHG treatment units, likely because current harvest inefficiencies due to mechanical and financial limitations prevented complete harvest of logging residues (Fritts et al. in review). In fact, DWD volumes in all treatment units, including NOBHG treatments, exceeded by at least three-fold the recommendations set by the Forest Guild for Piedmont and Coastal Plain physiographic regions of the southeastern United States (Perschel et al. 2012). The minimum volume of DWD retained in any treatment unit was 16.28 m^3/ha (7.81 tons/ha), which may be greater than the threshold needed to sustain amphibian diversity and southern toad abundance. However, future technological advances in harvest machinery or increases in woody biomass prices could result in increases in recovered debris, thereby leading to lower levels of retained DWD, reinforcing the need for BHGs to sustain amphibian populations. If technological advances result in lower volumes of DWD than what occurred on our study sites, it may be beneficial to retain DWD piles distributed throughout the sites for amphibians with high desiccation risk, such as eastern narrowmouth toads. We suggest future research should examine fine-scale use of DWD by herpetofauna to better understand the mechanisms responsible for potential responses to woody biomass harvesting.

ACKNOWLEDGMENTS

Funding was provided by the United States Department of Agriculture National Institute of Food and Agriculture Managed Ecosystems Program, the Department of the Interior Southeast Climate Science Center, the Biofuels Center of North Carolina, the North Carolina Bioenergy Initiative, Weyerhaeuser Company, Plum Creek, Georgia Pacific, and the National Council for Air and Stream Improvement. Special thanks to M. Adams, J. Bowser, S. Bruno,

J. Eells, E. Evans, J. Kiser, J. Kraus, B. Moore, K. Moore, J. O'Connor, R. Skinner, C. Tyson, and N. Weaver for field data collection. We thank J. Zajdel, B. Fritts, R. Evans, P. Taillie, A. Prince, M. Lashley, M. Elfelt, B. Hardin, and countless other volunteers for installing drift fences and building funnel traps.

LITERATURE CITED

Akaike, H. 1973. Information theory as an extension of the maximum likelihood principle. Pages 267-281 In: B. N. Petrov and F. Dsake (editors). Second International Symposium on Information Theory. Akademiai Kiado, Budapest, Hungary.

Anderson, P. K. 1954. Studies in the ecology of the narrow-mouthed toad, *Microhyla carolinensis carolinensis*. Tulane Studies in Zoology 2: 15-46.

Ashton, R. E., Jr., P. S. Ashton. 1988. Handbook of reptiles and amphibians of Florida: Part Three: The amphibians. Windward Publishing, Miami, Florida, USA.

Bailey, L. L., K. H. Pollock, T. R. Simons. 2004. Estimating detection probability parameters for plethodon salamanders using the robust capture-recapture design. Journal of Wildlife Management 68: 1-13.

Bartlett, R. D., P. P. Bartlett. 1999. A field guide to Florida reptiles and amphibians. Gulf Publishing Company, Houston, Texas, USA.

Brooks, R. T., T. D. Kyker-Snowman. 2008. Forest floor temperature and relative humidity following timber harvesting in southern New England, USA. Forest Ecology and Management 254: 65-73.

Burgdorf, S. J., D. C. Rudolph, R. N. Conner, D. Saenz, R. R. Schaefer. 2005. A successful trap design for capturing large terrestrial snakes. Herpetological Review 36: 421-424.

Butts, S. R., W. C. McComb. 2000. Associations of forest-floor vertebrates with coarse woody debris in managed forests of western Oregon. Journal of Wildlife Management 64: 95-104.

Carr, A. F., Jr. 1940. A contribution to the Herpetology of Florida. University of Florida biological Publications Science Series, Volume 3, Number 1. University of Florida Press, Gainesville, USA.

Carter, H. A. 1934. Georgia records of *Gastrophryne carolinensis* (Holbrook). Copeia 1934: 138.

Collins, J. T., S. L. Collins. 1993. Amphibians and reptiles in Kansas. Issue 13 of Public education series. University of Kansas, Museum of Natural History, Lawrence, USA.

Cook, J., J. Beyea. 2000. Bioenergy in the United States: progress and possibilities. Biomass and Bioenergy 18: 441-455.

Conant, R., J. T. Collins. 1998. Reptiles and Amphibians: Eastern/Central North America. Houghton Mifflin Co., Boston, Massachusetts, USA.

Davis, J. C., S. B. Castleberry, J. C. Kilgo. 2010. Influence of coarse woody debris on herpetofaunal communities in upland pine stands of the southeastern Coastal Plain. Forest Ecology and Management 259: 1111-1117.

Dorcas, M. E., J. W. Gibbons. 2008. Frogs and toads of the Southeast. University of

Georgia Press, Athens, USA.

Dupuis, L. A., J. N. Smith, F. Bunnell. 1995. Relation of terrestrial-breeding amphibian abundance to tree-stand age. Conservation Biology 9: 64-653.

Energy Information Administration (EIA). 2013. Annual Energy Outlook 2014. United States Department of Energy, Energy Information Administration, Washington, DC, USA.

Eneg, K. M., W. R. Marion. 1986. Effects of clearcutting and site preparation on herpetofauna of a north Florida flatwoods. Forest Ecology and Management 14: 177-192.

Evans, A. M., M. J. Kelty. 2010. Ecology of dead wood in the Northeast. Forest Guild. Sante Fe, New Mexico, USA.

Evans, J. M., R. J. Fletcher, Jr., J. R. R. Alavalapati, A. L. Smith, D. Geller, P. Lal, D. Vasudev, M. Acevedo, F. Calabria, T. Upadhyay. 2013. Forestry bioenergy in the Southeast United State: implications for wildlife habitat and biodiversity. National Wildlife Federation, Merrifield, Virginia, USA.

Fiske, I., R. B. Chandler. 2011. Unmarked: An R package for fitting hierarchical models of wildlife occurrence and abundance. Journal of Statistical Software 43: 1-23.

Fritts, S. F., C. E. Moorman, D. W. Hazel, B. D. Jackson. in review. Biomass harvesting guidelines affect downed woody debris retention.

Gibbons, J. W., K. A. Buhlmann. 2001. Reptiles and amphibians. Pages 372–390 in J. G. Dickson (editor). Wildlife of Southern Forests: Habitat and Management. Hancock House Publishers, Blaine, Washington, USA.

Goh, C. S., M. Junginger, M. Cocchi, D. Marchal, D. Thran, C. Hennig, J. Heinimo, L. Nikolaisen, P. Schouwenberg, D. Bradley, R. Hess, J. Jacobson, L. Ovard, M. Deutmeyer. 2013. Wood pellet market and trade: a global perspective. Bioproducts, Biofuels, Biorefineries 7: 24-42.

Goldstein, M. I., R. N. Wilkins, T. E lacher, Jr. 2005. Spatiotemporal responses of reptiles and amphibians to timber harvest treatments. Journal of Wildlife Management 69: 525-539.

Graeter, G. J., B. B. Rothermel, J. W. Gibbons. 2008. Habitat selection and movement of pond-breeding amphibians in experimentally fragmented pine forest. Journal of Wildlife Management 72: 473-482.

Graham, S. A. 1925. The felled tree trunk as an ecological unit. Ecology 6: 397-416.

Greenberg, C. H., D. G. Neary, L. D. Harris. 1994. Effect of high-intensity wildfire and silvicultural treatments on reptile communities in sand-pine scrub. Conservation Biology 8: 1047-1057.

Greenberg, C. H., T. Waldrop. 2008. Short-term response of reptiles and amphibians to prescribed fire and mechanical fuel reduction in a southern Appalachian upland hardwood forest. Forest Ecology and Management 255: 2883-2893.

Hanson, C., L. Yonavjak, C. Clarke, S. Minnemeyer, L. Boisrobert, L, A. Leach, A. K. Schleeweis. 2010. Southern forests for the future. World Resources Institute. Washington, DC, USA.

Harmon, M. E., J. F. Franklin, F. J. Swanson, P. Sollins, S. V. Gregory, J. D. Lattin, N. H. Anderson, S. P. Cline, N. G. Aumen, J. R. Sedell, G. W. Lienkaemper, K. Cromack Jr., K. W. Cummins. 1986. Ecology of coarse woody debris in temperate ecosystems. Advanced Ecological Research 15: 133-302.

Harmon, M. E., and C. Hua. 1991. Coarse Woody Debris Dynamics in Two Old-Growth Ecosystems. Bioscience 41: 604-610.

Hassinger, J. D. 1989. Dead wood for wildlife. Pages 171-184 In: Proceedings of the 1989 Penn State Forest Resources Issues Conference. April 4 – 6, 1989, University Park, Pennsylvania, USA.

Herrero, C., O. Krankina, V. J. Monleon, F. Bravo. 2014. Amount and distribution of coarse woody debris in pine ecosystems of northwestern Spain, Russia, and the United States. Journal of Biogeosciences and Forestry 7: 53-60.

Hillring, B. 2006. World trade in forest products and wood fuel. Biomass and Bioenergy 30: 815-825.

Hillyard, S. D. 1999. Behavioral, molecular and integrative mechanisms of amphibian osmoregulation. Journal of Experimental Zoology 283: 662-674.

Homyack, J. A., A. Aardweg, T. A. Gorman, D. R. Chalcraft. 2013. Initial effects of woody biomass removal and intercropping of switchgrass (*Panicum virgatum*) on herpetofauna in eastern North Carolina. Wildlife Society Bulletin 37: 327-335.

Homyack, J. A., C. A. Haas, W. A. Hopkins. 2011. Energetics of surface-active terrestrial salamanders in experimentally harvested forest. Journal of Wildlife Management 75: 1267-1278.

Homyack, J. A., C. A. Haas. 2009. Long-term effects of experimental forest harvesting on abundance and reproductive demography of terrestrial salamanders. Biological Conservation 142: 110-121.

Jaeger, R. G. 1980. Microhabitats of a terrestrial forest salamander. Copeia 2: 265-268.

Janowiak, M. K., C. R. Webster. 2010. Promoting Ecological Sustainability in Woody Biomass Harvesting. Journal of Forestry 108: 16-23.

Jørgensen, C. B. 1997. 200 years of amphibian water economy: from Robert Townsend to the present. Biological Reviews 72: 153-237.

Kery, M., J. A. Royle, and H. Schmid. 2005. Modeling avian abundance from replicated counts using binomial mixture models. Ecological Applications 15:1450-1461.

Kluber, M. R., D. H. Olson, K. J. Puettmann. 2009. Downed wood microclimates and their potential impact on Plethodontid salamander habitat in the Oregon coast range. Northwest Science 83: 25-34.

Kentucky Division of Forestry (KYDOF). 2011. Recommendations for the harvesting of woody biomass. Kentucky Division of Forestry. Frankfurt, USA.

Knapp, S. M., C. A. Haas, D. N. Harpole, R. L Kirkpatrick. 2003. Initial effects of clearcutting and alternative silvicultural practices on terrestrial salamander abundance. Conservation Biology 17: 752-762.

Lattimore, B., C. T. Smith, B. D. Titus, I. Stupak, and G. Egnell. 2009. Environmental factors in woodfuel production: Opportunities, risks, and criteria and indicators for sustainable practices. Biomass and Bioenergy 33:1321-1342.

MacKenzie, D. I., J. D. Nichols, G. B. Lachman, S. Droege, J. A. Royle, C. A. Langtimm. 2002. Estimating site occupancy rates when detection probabilities are less than one. Ecology 83: 2248-2255.

Mantau U., U. Saal, F. Steierer, H. Verkerk, J. Oldenburger, N. Leek, K. Prins. 2010. EUwood – Real potential for changes in growth and use of EU forests. Final report. Hamburg, Germany.

Martof, B. S. 1955. Observations on the life history and ecology of the amphibians of the Athens area, Georgia. Copeia 1955: 166-170.

Matthews, C. E., C. E. Moorman, C. H. Greenberg, T. A. Waldrop. 2010. Response of reptiles and amphibians to repeated fuel reduction treatments. Journal of Wildlife Management 74: 1301-1310.

McLeod, R. F., J. E. Gates. 1998. Response of herpetofaunal communities to forest cutting and burning at Chesapeake Farms, Maryland. American Midland Naturalist 139: 164-177.

McMinn, J. W., R. A. Hardt. 1996. Accumulation of coarse woody debris in southern forests. Pages 1-9 In: J. W. McMinn and D. A. Crossley, Jr. (editors). Biodiversity of Coarse Woody Debris in Southern Forests. Proceedings of the Workshop on Coarse Woody Debris in Southern Forests: Effects on Biodiversity. United States Department of Agriculture Forest Service General Technical Report. SE-94.

Minnesota Forest Resources Council (MFRC). 2007. Biomass Harvest Guidelines. Minnesota Forest Resources Council. St. Paul, USA.

Moorman, C. E., D. C. Guynn Jr. 2001. Effects of group-selection opening size on breeding bird habitat use in a bottomland forest. Ecological Applications 11: 1680-1691.

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

Moseley, K. R., S. B. Castleberry, J. L. Hanula, W. M. Ford. 2005. Diet of southern toads (Bufo terrestris) in loblolly pine (*Pinus taeda*) stands subject to coarse woody debris manipulations. American Midland Naturalist 153: 327-337.

Neill, W. T. 1950. Reptiles and amphibians in urban areas of Georgia. Herpetologica 6: 113-116.

Otto, R. V., G. J. Roloff. 2011. Using multiple methods to assess detection probabilities of forest-floor wildlife. Journal of Wildlife Management 75: 423-431.

Otto, C. R., A. J. Kroll, H. C. McKenny. 2013. Amphibian response to downed wood retention in managed forests: A prospectus for future biomass harvests in North American. Forest Ecology and Management 304: 275-285.

Owens, A. K., K. R. Moseley, T. S. McCay, S. B. Castleberry, J. C. Kilgo, W. M. Ford. 2008. Amphibian and reptile community response to coarse woody debris manipulations in upland loblolly pine (*Pinus taeda*) forests. Forest Ecology and Management 256: 2078-2083.

Pennsylvania Department of Conservation and Natural Resources (PADCNR). 2008. Harvesting woody biomass for energy. Pennsylvania Department of Conservation and Natural Resources. Harrisburg, USA.

Perlack, R. D., L. L. Wright, A. F. Turhollow, R. L. Graham, B. J. Stokes, D. C. Erback.

2005. Biomass as feedstock for a bioenergy and bioproducts industry: The technical feasibility of a billion-ton annual supply. United States Department of Energy. Oak Ridge National Laboratory. Oak Ridge, Tennessee, USA.

Perschel, B., A. Evans, M. DeBonis. 2012. Forest biomass retention and harvesting guidelines for the Southeast. Forest Guild Southeast Biomass Working Group. Forest Guild. Sante Fe, New Mexico, USA.

Petranka, J. W., M. P. Brannon, M. E. Hopey, C. K. Smith. 1994. Effects of timber harvesting on low elevation populations of southern Appalachian salamanders. Forest Ecology and Management 67: 135-147.

Phelps, J. P., R. A. Lancia. 1995. Effects of a clear-cut on the herpetofauna of a South Carolina bottomland swamp. Brimleyana 22:31-45.

Ranius, T., L. Fahrig. 2006. Targets for maintenance of dead wood for biodiversity conservation based on extinction thresholds. Scandinavian Journal of Forest Research 21: 201-208.

R Core Team. 2012. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

REN21. 2013. Renewables 2013 Global Status Report. Renewables Energy Policy Network for the 21st Century. Paris, France.

Riffell, S., J. Verschuyl, D. Miller, T. B. Wigley. 2011. Biofuel harvests, coarse woody debris, and biodiversity – A meta-analysis. Forest Ecology and Management 21: 878-887.

Rittenhouse, T. A., E. B. Harper, L. R. Rehard, R. D. Semlitsch. 2008. The role of microhabitats in the desiccation and survival of anurans in recently harvested oak-hickory forest. Copeia 2008: 807-814.

Robertson, T. S., E. L. Tyson. 1950. Herpetological notes from eastern North Carolina. Journal of the Elisha Mitchell Scientific Society 66: 130-147.

Röser D., A. Asikainen, K. Raulund-Rasmussen, I. Stupak, editors. 2008. Sustainable use of forest biomass for energy. A synthesis with focus on the Nordic and Baltic Region. Managing forest ecosystems. Volume 12. Springer, Dordrecht, the Netherlands.

Royle, J. A. 2004. N-Mixture models for estimating population size from spatially replicated counts. Biometrics 60: 108–105.

Rudolphi, J., Gustafsson, L. 2005. Effects of forest-fuel harvesting on the amount of deadwood on clear-cuts. Scandinavian Journal of Forest Research 20: 235-242.

Russell, K. B., T. B. Wigley, W. M. Bauhman, H. H. Hanlin, W. M. Ford. 2004. Responses of southeastern amphibians and reptiles to forest management: a review. Pages 319-334 In: H. M. Rauscher and K. Johnsen (editors). Southern Forest Science: Past, Present, and Future. Southern Research Station, Asheville, North Carolina, USA.

Semlitsch, R. D., C. A. Conner, D. J. Hocking. 2008. Effects of timber harvesting on pond-breeding amphibian persistence: Testing the evacuation hypothesis. Ecological Applications 18: 283-289.

Shake, C. S., C. E. Moorman, M. R. Burchell. 2011. Cropland edge, forest succession, and landscape affect shrubland bird nest predation. Journal of Wildlife Management 75: 825–835.

Spies, T. A., S. P. Cline. 1988. Coarse woody debris in forests and plantations of coastal Oregon. Pages 5-24 In: C. Maser and R. F. Tarrant (editors). From the Forest to the Trees: A Story of Fallen Trees. United States Department of Agriculture, Forest Service, Pacific Northwest Research Station and United States Department of the Interior Bureau of Land Management, Portland, Oregon, USA.

Todd, B. D., B. B. Rothermel. 2006. Assessing quality of clearcut habitats for amphibians: effects on abundances versus vital rates in the southern toad (*Bufo terrestris*). Biological Conservation 133: 178-185.

Todd, B. D., B. B. Rothermel, R. N. Reed. 2008. Habitat alteration increases invasive fire ant abundance to the detriment of amphibians and reptiles. Biological Invasions 10: 539-546.

Todd, B. D., K. M. Andrews. 2008. Response of a reptile guild to forest harvesting. Conservation Biology 22: 753-761.

Tuberville, T. D., J. D. Willson, M. E. Dorcas, J. W. Gibbons. 2005. Herpetofaunal species richness of southeastern national parks. Southeastern Naturalist 4: 537-569.

United States Department of Agriculture (USDA). 2010. A USDA regional roadmap to meeting the biofuels goals of the renewable fuels standard by 2022. United States Department of Agriculture, Washington, DC, USA.

Valigura, R. A., M. G. Messina. 1994. Modification of Texas clearcut environments with loblolly pine shelterwoods. Journal of Environmental Management 40: 283-295.

Van Wagner, C. E. 1968. The line intersect method in forest fuel sampling. Forest Science 14:20-26.

Vitt, L. J., J. P. Caldwell. 2009. Spacing, movements, and orientation. Pages 217-238 In: L. J. Vitt and J. P. Caldwell (editors). Herpetology: An Introductory Biology of Amphibians and Reptiles. Academic Press, San Diego, California, USA.

Welsh, H. H. J., L. M. Ollivier. 1998. Stream amphibians as indicators of ecosystem stress: A case study from California's redwoods. Ecological Applications 8: 1118-1132.

Whiles, M. R., J. W. Grubaugh. 1996. Importance of coarse woody debris to southern forest herpetofauna. Pages 94-100 In: J. W. McMinn and D. A. Crossley, Jr. (editors). Biodiversity and Coarse Woody Debris in Southern Forests. Proceedings of the Workshop on Coarse Woody Debris in Southern Forests: Effects on Biodiversity. United States Department of Agriculture Forest Service General Technical Report SE-94.

Woodall, C.W., Monleon, V.J. 2008. Sampling protocol, estimation, and analysis procedures for the down woody debris indicator of the FIA program. NRS-GTR-22.United States Department of Agriculture Forest Service. Newtown Square, Pennsylvania, USA.

Table 1. Covariates measured in six woody biomass harvest treatments in clearcut loblolly pine (*Pinus taeda*) plantations in Beaufort County, North Carolina (n = 4) and Glynn (n = 3) and Chatham Counties (n = 1), Georgia in 2011-2013. Covariates were used in determining predictors of southern toad (*Anaxyrus terrestris*) and eastern narrowmouth toad (*Gastrophryne carolinensis*) abundance using hierarchical N-mixture models.

Covariate Level	Covariate	Definition
Observation	DATE	Julian date
	EFFORT	number of buckets open per night/drift fence
	RAIN	binomial (rain occurred in previous 24 hours = 1, if not =0)
	RELHU	daily mean relative humidity
	TEMP	daily mean temperature
Site	BG	% bare ground groundcover
	CWD	% coarse woody debris groundcover
	DISTD	distance (m) of drift fence array to ditch or wetland
	DISTP	distance (m) of drift fence array to debris pile
	FWD	% fine woody debris groundcover
	HE	% herbaceous groundcover
	LL	% litter groundcover
	VDF50	volume (m^3/ha) of piled DWD within 50 m of the drift fence array
	VEST	vertical vegetation structure at the drift fence array
	VT	volume (m^3/ha) DWD in the treatment unit
	WP	% woody groundcover

Table 2. Mean ± standard error of habitat covariates estimated in six woody biomass harvest treatments in loblolly pine (*Pinus taeda*) plantations in Beaufort County, North Carolina (n = 4) in 2011-2013. Treatments were:1) no Biomass Harvesting Guidelines (NOBHG); 2) 15% DWD retention in piles (15CLUS); 3)15% DWD retention distributed evenly throughout the treatment unit (15DISP); 4) 30% DWD retention in piles (30CLUS); 5) 30% DWD retention distributed evenly throughout the treatment unit (30DISP); 6) and no woody biomass harvesting (NOBIOHARV). Covariates are defined in Table 1. Different letters indicate significantly different means at the α = 0.05 level.

	Treatment							
	NOBHG	15DISP	15CLUS	30DISP	30CLUS	NOBIOHARV	$F_{5, 65}$	P_{trt}
VT	20.65±1.45a	40.80±13.11a	37.76±9.42a	55.75±12.49a	55.17±12.49a	108.20±20.05b	9.83	<0.01
DISTP	40.63±7.38	36.91±11.46	33.24±5.85	24.68±6.23	19.97±4.57	25.14±5.67	1.17	0.34
DISTD	70.48±4.52a	98.70±6.35b	60.65±8.41a	76.39±4.29ab	67.31±6.53a	58.71±7.59a	6.42	<0.01
VDF50	16.73±8.25a	40.01±10.07a	11.65±4.06a	19.86±5.80a	90.28±28.10b	12.26±4.10a	5.51	<0.01
CWD%	1.92±0.96	3.08±0.91	1.82±1.05	4.89±1.67	2.40±1.05	3.56±1.59	0.93	0.47
FWD%	30.33±4.07	29.83±4.93	29.55±4.53	27.44±3.81	21.60±5.36	31.89±3.93	0.75	0.59
BG%	48.33±3.87	50.75±4.47	43.36±5.77	44.33±5.26	40.90±5.05	41.22±6.04	0.83	0.54
WP%	12.33±2.62	15.83±5.54	6.00±3.16	21.89±9.28	13.10±4.89	14.22±5.86	1.42	0.23
HE%	70.58±4.23	61.50±4.41	71.18±5.05	66.33±7.98	58.40±8.46	52.56±5.25	1.63	0.17
LL%	24.08±4.01	23.67±4.70	25.64±6.37	18.56±5.23	14.40±4.97	32.44±4.51	1.57	0.18
VEST	8.83±0.70	6.89±0.59	8.28±1.63	5.56±1.07	7.59±1.22	7.29±0.86	1.4	0.24

Table 3. Mean ± standard error of habitat covariates estimated in six woody biomass harvest treatments in loblolly pine (*Pinus taeda*) plantations in in Glynn County (n = 3) and Chatham County (n = 1), Georgia in 2011-2013. Treatments were: 1) no Biomass Harvesting Guidelines (NOBHG); 2) 15% DWD retention in piles (15CLUS); 3)15% DWD retention distributed evenly throughout the treatment unit (15DISP); 4) 30% DWD retention in piles (30CLUS); 5) 30% DWD retention distributed evenly throughout the treatment unit (30DISP); 6) and no woody biomass harvesting (NOBIOHARV). Covariates are defined in Table 1. Different letters indicate significantly different means at the α = 0.05 level.

	Treatment							
	NOBHG	15DISP	15CLUS	30DISP	30CLUS	NOBIOHARV	$F_{5, 65}$	P_{trt}
VT	319.13±40.80a	368.81±41.73ab	296.17±41.16a	299.65±40.45a	359.98±46.69a	373.39±24.85b	2.6	0.03
DISTP	14.23±1.83	17.82±2.98	14.82±1.74	19.94±3.44	17.77±3.35	13.53±2.02	1.18	0.33
CWD%	4.25±1.54	4.33±0.73	5.25±1.01	5.00±1.12	3.75±0.66	3.58±0.72	0.45	0.81
FWD%	8.08±1.94	8.92±2.62	10.08±2.40	7.25±1.70	6.67±1.50	4.42±1.15	0.99	0.43
bG%	54.00±5.34	52.33±5.67	47.00±5.87	50.33±2.81	43.25±3.77	56.33±3.80	1.17	0.33
WP%	15.25±4.52	10.08±4.35	7.42±2.30	7.75±3.07	8.08±2.34	7.75±2.09	0.89	0.49
HE%	106.33±3.97	101.83±11.28	103.67±8.75	108.33±4.56	98.25±6.19	89.42±4.27	0.96	0.45
LL%	4.08±1.12a	2.92±0.97ab	3.92±2.15ab	2.25±1.07a	9.58±2.94b	3.08±1.44a	3.57	<0.01
VEST	5.34±0.19	5.08±0.39	5.55±0.31	6.27±0.71	5.24±0.31	5.67±0.33	1.09	0.37

Table 4. Reptile captures (mean ± standard error) in six woody biomass harvest treatments in loblolly pine (*Pinus taeda*) plantations in Beaufort County, North Carolina (n = 4) in 2011-2013. Treatments were: 1) no Biomass Harvesting Guidelines (NOBHG); 2) 15% DWD retention in piles (15CLUS); 3)15% DWD retention distributed evenly throughout the treatment unit (15DISP); 4) 30% DWD retention in piles (30CLUS); 5) 30% DWD retention distributed evenly throughout the treatment unit (30DISP); 6) and no woody biomass harvesting (NOBIOHARV).

	Treatment					
Species	NOBHGS	15CLUS	15DISP	30CLUS	30DISP	NOBIOHARV
Agkistrodon contortrix	0.00±0.00	0.00±0.00	0.25±0.25	0.00±0.00	0.00±0.00	0.00±0.00
Anolis carolinensis	0.50±0.29	0.00±0.00	0.25±0.25	0.00±0.00	0.50±0.50	0.25±0.25
Carphophis amoenus	0.75±0.25	0.25±0.25	1.00±0.41	0.75±0.48	1.25±0.48	0.25±0.25
Chelydra serpentina	0.00±0.00	0.00±0.00	0.00±0.00	0.00±0.00	0.25±0.25	0.00±0.00
Clemmys guttata	0.00±0.00	0.00±0.00	0.00±0.00	0.25±0.25	0.00±0.00	0.00±0.00
Coluber constrictor	1.00±0.71	1.25±0.48	0.50±0.29	0.75±0.48	0.25±0.25	0.50±0.50
Diadophis punctatus	0.25±0.25	0.25±0.25	0.25±0.25	0.00±0.00	0.25±0.25	0.50±0.29
Eumeces inexpectatus	0.00±0.00	0.00±0.00	0.25±0.25	0.00±0.00	0.00±0.00	0.00±0.00
Heterodon platirhinos	0.00±0.00	0.50±0.29	1.00±1.00	0.00±0.00	0.75±0.48	0.00±0.00
Scincella lateralis	1.50±0.29	1.50±0.65	2.75±0.75	1.50±0.50	1.00±0.41	1.50±0.29
Pantherophis obsoleta	0.00±0.00	0.50±0.50	0.00±0.00	0.00±0.00	0.00±0.00	0.25±0.25
Plestiodon fasciatus	0.00±0.00	0.25±0.25	0.50±0.29	0.25±0.25	0.00±0.00	0.25±0.25
Terrapene carolina	0.00±0.00	0.00±0.00	0.25±0.25	0.00±0.00	0.00±0.00	0.25±0.25
Trachemys scripta	0.00±0.00	0.25±0.25	0.00±0.00	0.00±0.00	0.00±0.00	0.00±0.00

Table 5. Amphibian captures (mean ± standard error) in six woody biomass harvest treatments in loblolly pine (*Pinus taeda*) plantations in Beaufort County, North Carolina (n = 4) in 2011-2013. Treatments were: 1) no Biomass Harvesting Guidelines (NOBHG); 2) 15% DWD retention in piles (15CLUS); 3)15% DWD retention distributed evenly throughout the treatment unit (15DISP); 4) 30% DWD retention in piles (30CLUS); 5) 30% DWD retention distributed evenly throughout the treatment unit (30DISP); 6) and no woody biomass harvesting (NOBIOHARV).

	Treatment					
Species	NOBHGS	15CLUS	15DISP	30CLUS	30DISP	NO BIOHARV
Acris gryllus	0.00±0.00	0.00±0.00	0.25±0.25	0.00±0.00	0.00±0.00	0.25±0.25
Anaxyrus quercicus	5.25±3.20	3.25±2.63	3.75±2.25	1.00±0.58	4.00±3.67	0.00±0.00
Anaxyrus terrestris	180.75±54.06	136.25±24.97	207.25±56.31	203.75±78.71	145.5±41.55	146±29.87
Gastrophryne carolinensis	74.75±18.64	53.75±18.25	45.00±9.62	49.75±5.22	46.25±16.71	57.25±13.49
Hyla chrysoscelis	0.75±0.48	1.25±0.95	1.25±1.25	0.00±0.00	0.00±0.00	1.75±1.75
Hyla femoralis	2.00±0.41	1.75±1.44	1.50±0.65	1.25±0.95	0.75±0.48	1.00±0.71
Lithobates catesbiana	0.00±0.00	0.00±0.00	0.00±0.00	0.00±0.00	0.00±0.00	0.25±0.25
Lithobates clamitans	1.25±0.63	0.5±0.29	0.25±0.25	0.25±0.25	0.75±0.48	1.50±1.50
Lithobates sphenocephalus	6.00±1.22	4.50±0.65	5.50±1.76	18.5±12.4	4.75±2.17	4.25±2.17
Plethodon chlorobryonis	1.75±1.03	1.00±0.71	2.25±0.25	1.00±0.58	1.00±0.41	1.50±0.65
Pseudacris brimleyi	0.25±0.25	1.50±0.29	0.75±0.48	0.50±0.29	0.50±0.29	0.00±0.00
Pseudacris crucifer	1.00±1.00	0.00±0.00	0.00±0.00	0.25±0.25	0.00±0.00	0.00±0.00

Table 6. Reptile captures (mean ± standard error) in six woody biomass harvest treatments in loblolly pine (*Pinus taeda*) plantations in Glynn County (n = 3) and Chatham County (n = 1), Georgia in 2011-2013. Treatments were: 1) no Biomass Harvesting Guidelines (NOBHG); 2) 15% DWD retention in piles (15CLUS); 3)15% DWD retention distributed evenly throughout the treatment unit (15DISP); 4) 30% DWD retention in piles (30CLUS); 5) 30% DWD retention distributed evenly throughout the treatment unit (30DISP); 6) and no woody biomass harvesting (NOBIOHARV).

	Treatment					
Species	NOBHGS	15CLUS	15DISP	30CLUS	30DISP	NOBIOHARV
Agkistrodon piscivorus	0.00±0.00	0.25±0.25	0.00±0.00	0.00±0.00	0.00±0.00	0.00±0.00
Anolis carolinensis	0.00±0.00	0.00±0.00	0.25±0.25	0.25±0.25	1.25±1.25	0.00±0.00
Aspidoscelis sexlineatus	0.50±0.29	0.25±0.25	0.25±0.25	1.25±1.25	0.00±0.00	0.25±0.25
Coluber constrictor	0.00±0.00	0.00±0.00	0.00±0.00	0.00±0.00	0.00±0.00	0.25±0.25
Eumeces inexpectatus	0.00±0.00	1.00±0.71	0.25±0.25	1.00±1.00	0.00±0.00	0.00±0.00
Farancia abacura	0.25±0.25	0.00±0.00	0.00±0.00	0.00±0.00	0.00±0.00	0.25±0.25
Kinosternum subrubrum	0.00±0.00	0.25±0.25	0.00±0.00	0.00±0.00	1.00±0.71	0.00±0.00
Plestiodon fasciatus	0.00±0.00	0.25±0.25	0.00±0.00	0.00±0.00	0.00±0.00	0.00±0.00
Regina regida	0.00±0.00	0.75±0.75	0.25±0.25	0.00±0.00	0.25±0.25	0.25±0.25
Sceloporus undulatus	0.00±0.00	0.00±0.00	0.00±0.00	0.00±0.00	0.00±0.00	0.00±0.00
Scincella lateralis	0.75±0.48	0.00±0.00	0.50±0.29	1.00±0.58	0.50±0.29	0.25±0.25
Storeria occipitomaculata	0.25±0.25	0.00±0.00	0.00±0.00	0.00±0.00	0.25±0.25	0.00±0.00
Terrapene carolina	0.00±0.00	0.00±0.00	0.00±0.00	0.50±0.29	0.00±0.00	0.00±0.00

Table 7. Amphibian captures (mean ± standard error) in six woody biomass harvest treatments in loblolly pine (*Pinus taeda*) plantations in Glynn County (n = 3) and Chatham County (n = 1), Georgia in 2011-2013. Treatments were: 1) no Biomass Harvesting Guidelines (NOBHG); 2) 15% DWD retention in piles (15CLUS); 3)15% DWD retention distributed evenly throughout the treatment unit (15DISP); 4) 30% DWD retention in piles (30CLUS); 5) 30% DWD retention distributed evenly throughout the treatment unit (30DISP); 6) and no woody biomass harvesting (NOBIOHARV).

	Treatment					
Species	NOBHGS	15CLUS	15DISP	30CLUS	30DISP	NOBIOHARV
Acris crepitans	0.25±0.25	0.25±0.25	0.00±0.00	1.25±1.25	0.25±0.25	0.25±0.25
Acris gryllus	1.75±1.44	0.00±0.00	1.25±1.25	0.00±0.00	1.00±0.58	1.50±1.19
Anaxyrus quercicus	11.25±10.92	0.75±0.75	0.25±0.25	0.00±0.00	0.25±0.25	10.75±10.09
Anaxyrus terrestris	36.25±11.13	45.00±16.18	25.25±8.88	24.00±4.97	27.25±6.79	37.00±11.97
Gastrophryne carolinensis	22.00±14.59	37.75±11.10	34.25±10.75	27.75±9.2	23.75±5.66	34.00±15.68
Hyla femoralis	0.25±0.25	0.25±0.25	0.00±0.00	0.50±0.29	0.25±0.25	0.25±0.25
Lithobates sphenocephalus	15.25±3.04	12.00±3.85	11.25±4.77	35.50±28.63	35.75±10.08	23.00±9.19
Pseudacris nigrita	0.50±0.29	0.50±0.29	0.25±0.25	0.75±0.75	0.00±0.00	0.00±0.00
Pseudacris ocularis	3.50±1.76	1.00±0.41	2.00±1.15	2.75±2.14	1.50±1.19	3.75±3.42
Lithobates clamitans	0.00±0.00	0.00±0.00	1.25±1.25	0.00±0.00	0.25±0.25	0.25±0.25
Scaphiopus holbrookii	0.00±0.00	0.00±0.00	0.00±0.00	0.00±0.00	0.00±0.00	0.25±0.25

Table 8. Beta (β) estimates, standard errors (SE), z-values, and P-values of covariates on detection (logit-scale) and abundance (log-scale) in global model to determine predictors of southern toad (*Anaxyrus terrestris*) abundance at drift fence arrays with various volumes of downed woody debris, groundcover, and vertical vegetation structure. Southern toads were captured in loblolly pine (*Pinus taeda*) plantations in Beaufort County, North Carolina (n = 4) in 2011-2013. Covariates are defined in Table 1.

	Detection (logit-scale)					Abundance (log-scale)				
Year		β	SE	z	P(>|z|)		β	SE	z	P(>|z|)
2011	Intercept	-5.87	0.14	-41.1	<0.01	Intercept	4.81	0.16	29.51	<0.01
	DATE	-0.15	0.03	-5.14	<0.01	DISTD	-0.10	0.08	-1.38	0.17
	RAIN	0.28	0.07	4.14	<0.01	DISTP	-0.10	0.12	-0.82	0.41
						VDF50	0.14	0.13	1.12	0.26
						VT	-0.23	0.14	-1.73	0.08
2012	Intercept	-5.8	0.11	-53.73	<0.01	Intercept	5.09	0.12	43.04	<0.01
	EFFORT	0.39	0.03	11.54	<0.01	DISTD	0.11	0.08	1.33	0.18
	RAIN	0.69	0.05	14.08	<0.01	DISTP	0.05	0.09	0.62	0.53
	RELHU	0.08	0.02	3.33	<0.01	VDF50	0.14	0.09	1.49	0.14
	TEMP	-0.08	0.02	-3.44	<0.01	VT	0.02	0.1	0.15	0.88

Table 8 (continued).

Year	Detection (logit-scale)					Abundance (log-scale)				
		β	SE	z	P(>|z|)		β	SE	z	P(>|z|)
2013	Intercept	-6.32	0.25	-25.5	<0.01	Intercept	4.20	0.27	15.75	<0.01
	EFFORT	0.05	0.02	2.07	0.04	BG	-0.10	0.19	-0.54	0.59
	FWD	-0.32	0.20	-1.59	0.11	CWD	0.05	0.15	0.31	0.76
	LL	-0.39	0.24	-1.62	0.10	DISTD	0.10	0.14	0.76	0.45
	RAIN	0.89	0.09	10.27	<0.01	DISTP	0.07	0.18	0.38	0.70
	RELHU	0.66	0.05	12.17	<0.01	FWD	-0.35	0.24	-1.44	0.15
	TEMP	-0.36	0.05	-7.01	<0.01	HE	0.21	0.17	1.23	0.22
	VEST	0.76	0.35	2.19	0.03	LL	0.02	0.30	0.07	0.94
						VDF50	0.03	0.19	0.15	0.88
						VEST	0.02	0.40	0.04	0.97
						VT	-0.28	0.19	-1.47	0.14
						WP	-0.31	0.22	-1.40	0.16

Table 9. Beta (β) estimates, standard errors (SE), z-values, and P-values of covariates on detection (logit-scale) and abundance (log-scale) in global model to determine predictors of southern toad (*Anaxyrus terrestris*) abundance at drift fence arrays with various volumes of downed woody debris, groundcover, and vertical vegetation structure. Southern toads were captured in loblolly pine (*Pinus taeda*) plantations in Glynn County (n = 3) and Chatham County (n = 1), Georgia in 2011-2013. Covariates are defined in Table 1.

	Detection (logit-scale)					Abundance (log-scale)				
Year		β	SE	z	P(>|z|)		β	SE	z	P(>|z|)
2011	Intercept	-4.54	0.80	-5.66	<0.01	Intercept	2.90	0.79	3.66	<0.01
	DISTD	0.36	0.70	0.51	0.61	DISTD	-0.19	0.66	-0.28	0.78
	DISTP	-0.02	0.58	-0.03	0.98	DISTP	0.19	0.58	0.32	0.75
	RH	0.45	0.07	5.95	<0.01	VT	-0.11	0.10	-1.14	0.25
	TEMP	0.64	0.11	5.92	<0.01					
2012	Intercept	-4.65	0.32	-14.46	<0.01	Intercept	3.50	0.32	10.82	<0.01
	TEMP	0.35	0.06	6.13	<0.01	DISTD	0.14	0.11	1.31	0.19
	VT	0.16	0.39	0.41	0.69	DISTP	-0.03	0.13	-0.25	0.80
						VT	-0.38	0.40	-0.96	0.34

Table 9 (continued).

		Detection (logit-scale)					Abundance (log-scale)			
Year		β	SE	z	P(>|z|)		β	SE	z	P(>|z|)
2013	Intercept	-8.16	1.57	-5.18	<0.01	Intercept	1.70	0.48	3.59	<0.01
	DISTD	0.17	0.26	0.65	0.51	BG	0.17	0.19	0.88	0.38
	FWD	-0.63	0.30	-2.10	0.04	CWD	0.08	0.17	0.44	0.66
	LL	-0.47	0.86	-0.55	0.58	DISTD	0.39	0.28	1.39	0.16
	RAIN	0.04	0.02	2.17	0.03	DISTP	-0.21	0.19	-1.09	0.27
						FWD	0.26	0.31	0.83	0.41
						HE	0.04	0.19	0.22	0.82
						LL	-0.01	0.83	-0.01	0.99
						VEST	-0.22	0.22	-1.03	0.30
						VT	-0.29	0.22	-1.31	0.19
						WV	0.09	0.20	0.45	0.65

Table 10. Beta (β) estimates, standard errors (SE), z-values, and P-values of covariates on detection (logit-scale) and abundance (log-scale) in global model to determine predictors of eastern narrowmouth toad (*Gastrophryne carolinensis*) abundance at drift fence arrays with various volumes of downed woody debris, groundcover, and vertical vegetation structure. Eastern narrowmouth toads were captured in loblolly pine (*Pinus taeda*) plantation in Beaufort County, North Carolina (n = 4) in 2011-2013. Covariates are defined in Table 1.

	Detection (logit-scale)					Abundance (log-scale)				
Year		β	SE	z	P(>|z|)		β	SE	z	P(>|z|)
2011	Intercept	-7.47	1.21	-6.19	<0.01	Intercept	3.95	1.21	3.26	<0.01
	DATE	0.51	0.11	4.85	<0.01	DISTD	0.17	2.25	0.08	0.94
	DTD	0.23	2.24	0.10	0.92	DTDP	0.01	0.16	0.08	0.94
						VDF50	-0.41	0.23	-1.76	0.08
						VT	0.35	0.21	1.66	0.10
2012	Intercept	-7.20	0.26	-27.98	<0.01	Intercept	5.11	0.25	20.17	<0.01
	DISTP	-0.14	0.53	-0.25	0.80	DISTD	0.07	0.08	0.87	0.38
	EFFORT	0.15	0.05	3.20	<0.01	DISTP	-0.06	0.54	-0.11	0.91
	RAIN	1.17	0.09	13.61	<0.01	VDF50	-0.05	0.40	-0.13	0.90
	TEMP	0.31	0.04	7.82	<0.01	VT	0.09	0.11	0.86	0.39
	VDF50	-0.26	0.40	-0.65	0.52					

Table 10 (continued).

Year	Detection (logit-scale)					Abundance (log-scale)				
		β	SE	z	P(>|z|)		Est	SE	z	P(>|z|)
2013	Intercept	-6.85	0.44	-15.63	<0.01	Intercept	3.97	1.21	3.29	<0.01
	CWD	0.19	0.34	0.56	0.58	BG	0.12	0.17	0.71	0.48
	EFFORT	-0.13	0.02	-5.73	<0.01	CWD	0.06	0.35	0.17	0.86
	FWD	0.15	0.54	0.27	0.79	DISTD	0.06	0.16	0.36	0.72
	HE	-0.09	0.56	-0.16	0.87	DISTP	0.12	0.14	0.85	0.40
	RAIN	-0.56	0.13	-4.44	<0.01	FWD	0.16	0.55	0.28	0.78
	RELHU	0.75	0.08	9.35	<0.01	HE	-0.25	0.57	-0.44	0.66
	TEMP	0.91	0.09	9.84	<0.01	HORST	-0.08	0.14	-0.58	0.56
	VDF50	-0.27	0.52	-0.51	0.61	LL	-0.19	0.16	-1.20	0.23
	VT	0.22	0.50	0.44	0.66	VDF50	-0.16	0.53	-0.31	0.76
						VT	0.31	0.52	0.59	0.55
						WP	-0.15	0.19	-0.76	0.45

Table 11. Beta estimates (β), standard errors (SE), z-values, and P-values of covariates on detection (logit-scale) and abundance (log-scale) in global model to determine predictors of eastern narrowmouth toad (*Gastrophryne carolinensis*) abundance at drift fence arrays with various volumes of downed woody debris, groundcover, and vertical vegetation structure. Eastern narrowmouth toads were captured in loblolly pine (*Pinus taeda*) plantations in Glynn County (n = 3) and Chatham County (n = 1), Georgia in 2011-2013. Covariates are defined in Table 1.

		Detection (logit-scale)					Abundance (log-scale)			
Year		β	SE	z	P(>|z|)		β	SE	Z	P(>|z|)
2011	Intercept	-8.11	1.67	-4.86	<0.01	Intercept	6.57	1.82	3.61	<0.01
	RAIN	1.18	0.15	8.09	<0.01	DISTD	0.00	0.00	0.74	0.46
	TEMP	1.15	0.19	5.99	<0.01	DTDP	-0.03	0.02	-1.30	0.19
	VT	0.00	0.00	-0.03	0.97	VT	0.00	0.00	-0.58	0.56
2012	Intercept	-5.90	0.32	-18.28	<0.01	Intercept	4.43	0.33	13.33	<0.01
	TEMP	-0.41	0.06	-6.58	<0.01	DISTD	-0.30	0.14	-2.09	0.04
	DISTP	-0.38	0.46	-0.82	0.41	DISTP	0.34	0.48	0.72	0.47
						VT	0.29	0.15	1.96	0.05

Table 11 (continued).

		Detection (logit-scale)				Abundance (log-scale)				
Year		β	SE	z	P(>|z|)		β	SE	Z	P(>|z|)
2013	Intercept	-8.96	1.03	-8.70	<0.01	Intercept	2.57	0.25	10.12	<0.01
	RAIN	0.06	0.01	4.33	<0.01	BG	0.01	0.10	0.13	0.90
	VT	-0.30	0.38	-0.77	0.44	CWD	0.10	0.10	1.03	0.30
						DISTD	0.16	0.10	1.66	0.10
						DISTP	-0.05	0.12	-0.43	0.67
						FWD	-0.16	0.11	-1.38	0.17
						HE	0.11	0.09	1.23	0.22
						LL	-0.05	0.10	-0.52	0.60
						VEST	0.02	0.10	0.24	0.81
						VT	0.12	0.39	0.30	0.76
						WV	0.04	0.10	0.43	0.67

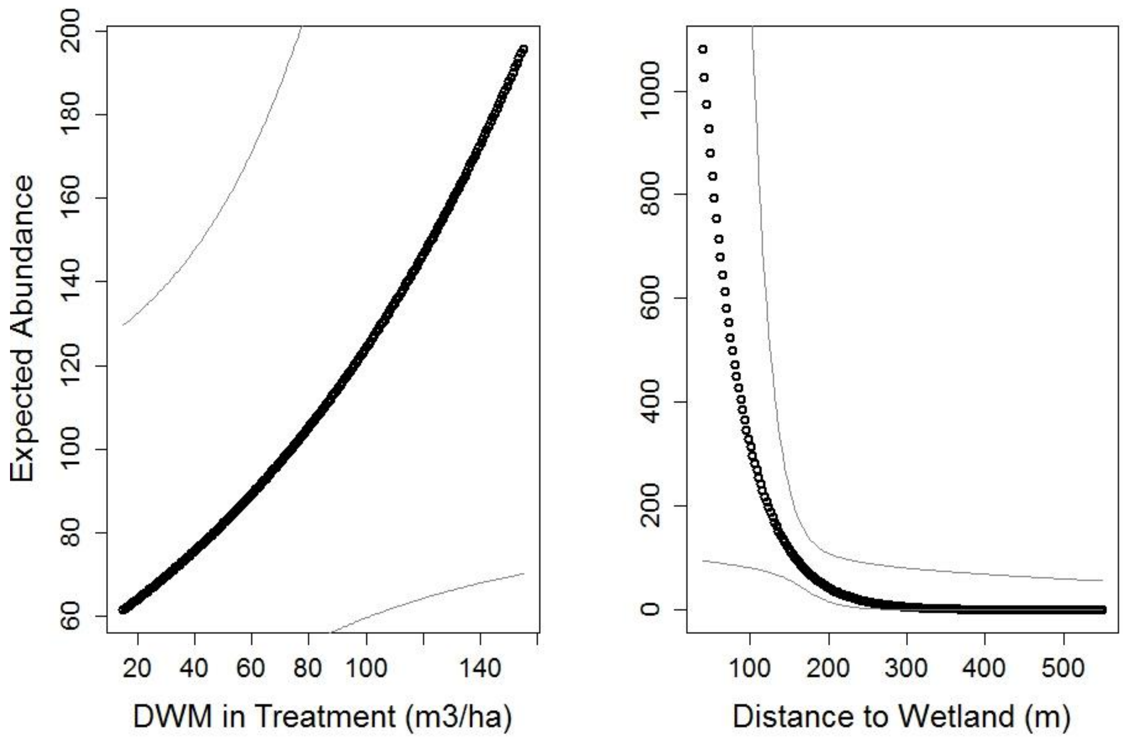

Figure 1. Relationship between eastern narrowmouth toad (*Gastrophryne carolinensis*) abundance (expected abundance and 95% confidence interval) and volume of retained downed woody debris (DWD) and distance to wet depression area in loblolly pine (*Pinus taeda*) plantations in Glynn County (n = 3) and Chatham County (n = 1), Georgia in 2012.

CHAPTER 3

Quantifying Multi-scale Habitat Use of Woody Biomass by Southern Toads Using Two Approaches

ABSTRACT

Use of woody biomass from forests is increasing worldwide as the demand for cleaner, renewable energy expands; however, harvesting woody biomass could degrade habitat for wildlife, especially ground-dwelling species dependent upon downed woody debris (DWD) for food and cover. We tested potential impacts of woody biomass harvests on amphibians using the southern toad (*Anaxyrus terrestris*) as an indicator species. We investigated toad use of DWD in pine plantations of the southeastern United States from May-August 2013 using: 1) a controlled enclosure experiment; and 2) a field-based radio-telemetry study. In the enclosure experiment, southern toads equipped with a Passive Implant Transponder tag selected among four 4-m^2 treatments, each surrounded by an antenna to record toad movements: 1) 100% of the area covered by CWD (100CWD); 2) approximately 25% of the volume of 100CWD dispersed evenly throughout the treatment (25CWD); 3) approximately 50% of the volume of 100CWD in one large pile (50CWD); and 4) no CWD (0CWD). We estimated daily mean temperature in the enclosure using six thermochron recorders in each treatment and compiled rainfall data from an onsite weather station. We used compositional analysis to determine if toads within the enclosure selected treatments and analysis of variance to determine if rain and temperature were predictors of treatment selection. We used mixed-effects logistic regression to determine if diurnal refuge site selection by female radio-marked toads differed from available habitat. Toads within the enclosure ($n = 47$) did not use treatments randomly during nocturnal hours and ranking of treatments from most to least selected was 0CWD, 100CWD, 25CWD, 50CWD. The proportion of time toads spent in 100CWD showed an interaction between rain and daily mean temperature ($P = 0.04$); when no rain occurred during the enclosure trial, toads spent a greater proportion of time ($P = 0.03$) in 100CWD as temperature increased. Radio-marked toads ($n = 37$) avoided grass ($P < 0.01$), bare ground ($P < 0.01$), and leaf litter ($P = 0.04$) as diurnal refuge sites, but no relationship with DWD existed. Piled CWD in recently harvested pine plantations may provide refuge for amphibians during periods of hot, dry weather conditions or when other cover sources are limited.

KEYWORDS: amphibians, *Anaxyrus terrestris*, down woody debris, southeastern United States, southern toad, telemetry, woody biomass harvest

INTRODUCTION

Demand for woody biomass from forests is driven by domestic and international policy requiring renewable energy and biofuels to help mitigate climate change (Perlack et al. 2005, Mantau et al. 2010). Biomass is the largest domestic source of renewable energy in the United States, accounting for approximately 3% of total U.S. primary energy consumption and nearly 5% of production (EIA 2013). Approximately 75% of biomass consumed in the United States is derived from forestlands (Perlack et al. 2005). Woody biomass harvests extract downed woody debris (DWD), which may include treetops, limbs, slash, and felled small trees, during traditional timber harvests (Rudolphi and Gustafsson 2005). Additionally, forests in the U.S. supply the European Union with woody biomass, where woody biomass is a critical source of renewable-based energy and currently accounts for slightly > 50%, on average by country, of the gross inland energy consumption from renewable energy sources (Mantau et al. 2010). In 2012, nearly 40% of the world's trade in wood pellets (i.e., a form of woody biomass burned for energy) was exported from North America to Europe, an increase of 50% from 2011 (REN21 2013).

Despite the benefits of replacing non-renewable energy with woody biomass, reductions of DWD following woody biomass harvests could alter ecosystem services and impact wildlife populations (Perschel et al. 2012, Evans et al. 2013). Coarse woody debris (CWD), debris ≥ 7.62 cm in diameter for a length of at least 0.914 m (Woodall and Monleon 2008), is important for energy flow and nutrient cycles, atmospheric carbon balance, and wildlife habitat (Harmon et al. 1986, Currie and Nadelhoffer 2002, Ranius and Fahrig 2006). Wildlife use CWD for nesting, feeding, travel, and refugia from extreme temperatures (Hassinger 1989, Lanham and Guynn 1996, Loeb 1996, Whiles and Grubaugh 1996), and the abundance of this structural component can predict the abundance of some species (Carey and Johnson 1995, Rosenvald et al. 2011). Fine woody debris (FWD) (debris smaller than CWD) also is an important component of forest ecosystems. Branches and foliage comprise only a small proportion of total tree biomass, but contain up to half of the nitrogen, phosphorous, calcium, potassium, and magnesium immobilized in trees (Alban et al. 1978).

The sustainability of woody biomass harvests is of particular importance in the southeastern United States, where growth of forestry bioenergy production facilities is most rapid

(Mendell and Lang 2012). The Southeast region is considered the "wood basket" of the United States and currently is the largest exporter of wood pellets in the world (Hanson et al. 2010, Evans et al. 2013, Goh et al. 2013). However, current combined woody biomass and roundwood harvests in the southeastern United States decrease retained DWD by approximately 81% compared to roundwood harvests alone (Fritts et al. in review). Further, the United States Department of Agriculture (USDA) predicted that ≈ 50% of second generation biofuels needed to meet the United States biofuel mandates will originate from the region by 2020 (USDA 2010), which may further reduce DWD available for ecosystem services.

Amphibians are critical components of forest ecosystems due to their high abundance and their roles as apex predators (Burton and Likens 1975a, Bruce and Christiansen 1976, Gibbons et. al 2006). Yet, over 30% of global amphibian species are threatened with extinction and more than 43% are experiencing population declines (Stuart 2004). The southeastern United States is a global center for amphibian diversity (IUCN 2013). Within the southeastern United States, amphibian species diversity is greatest in the Coastal Plain physiographic province (Gibbons and Buhlman 2001).

Amphibians are appropriate environmental indicators of the ecological sustainability of woody biomass harvests because their physiology makes them particularly susceptible to forest disturbances and manipulations that alter forest-floor temperature and moisture regimes. As poikilothermic ectotherms, temperature affects nearly all physiological processes of amphibians (Gatten et al. 1992, Rome et al. 1992), and the physiological state can affect behavior and growth. Therefore, habitat selection can have direct physiological and functional consequences (Huey 1991). Metabolism increases with temperature in ectotherms and can account for the majority of the overall energy budget (Burton and Likens 1975b, Pough 1980, Spotila and Standora 1985, Gatten et al. 1992). Greater temperatures that result in increased metabolic rates could cause changes in surface activities such as foraging and mating, which in turn may affect reproductive demography and abundance (Homyack et al. 2011). Further, amphibians risk desiccation because their skin does not prevent evaporative water loss (Jørgensen 1997). Thermoregulation coupled with the risk of desiccation may make DWD particularly important to amphibians. Amphibians often are associated with DWD, which provides micro-habitats in a range of temperature and moisture regimes, with the temperature under and inside woody refugia often lower than ambient (Graham 1925, Hansen et al. 1991, Whiles and Grubaugh 1996, Butts and McComb 2000, Gibbons and Buhlman 2001, Kluber et al. 2009). Other than protection from desiccation and thermoregulation, amphibians use DWD for reproduction and as a feeding substrate (Hassinger 1989, Whiles and Grubaugh 1996). Therefore, amphibian populations may be sensitive to reductions of DWD associated with harvesting woody biomass.

Amphibians often respond negatively to timber harvests (Petranka et al. 1994, McLeod and Gates 1998, Knapp et al. 2003), and it has been suggested to retain DWD following timber harvests to mitigate negative effects (deMaynadier and Hunter 1995, McKinney et al. 2006). However, woody biomass harvests remove DWD in conjunction with overstory removal. Overstory removal leads to changes in temperature and moisture regimes on the forest floor

(Liechty et al. 1992, Valigura and Messina 1994, Chen et al. 1999, Harpole and Haas 1999, Zheng et al. 2000); hence, desiccation risk for amphibians often is greater in clearcuts than in older forest stands (Rothermel and Luhring 2005). Retaining woody biomass during timber harvests could mitigate harsh environmental conditions, buffer climate extremes, and lessen potential negative effects such as reducing mortality by providing suitable microclimates (Rittenhouse et al. 2008, Scheffers et al. 2014).

We used the southern toad (*Anaxyrus terrestris*) as an indicator of amphibian response to woody biomass harvests. Southern toads are less susceptible to forest canopy removal than other amphibians and are among the most abundant species in regenerating clearcuts (Fritts et al. Ch. 2, Homyack et al. 2013). They are capable of storing and reabsorbing large quantities of water in their bladders and able to tolerate higher temperatures and desiccation risks than other amphibians (Thorson and Svihla 1943, Stebbins and Cohen 1995, Hillyard 1999, Zug 2001). Therefore, southern toads have been used as a conservative metric for examining the potential impacts of silviculture on amphibians (Todd and Rothermel 2006). We employed two methodologies to assess simultaneously woody biomass (i.e., DWD) use by southern toads in southeastern United States Coastal Plain loblolly pine (*Pinus taeda*) plantations following clearcut and woody biomass harvests: 1) a manipulative enclosure experiment that examined nocturnal habitat selection and diurnal refuge site selection among treatments with varying volumes and spatial allocations of DWD; and, 2) a radio-telemetry study that investigated diurnal microhabitat selection in relation to DWD. We also examined the relationships between environmental variables (i.e., rainfall and temperature) and selection of CWD to assess how climatic conditions may affect the importance of DWD to amphibians. Our results can be used to inform strategies that promote amphibian biodiversity following woody biomass harvests and to determine potential changes in amphibian habitat selection under varying harvesting regimes and future climate change predictions.

METHODS

Study Area

We conducted the study in a regenerating loblolly pine stand in the southeastern Coastal Plain Physiographic Region in Beaufort County, North Carolina. The project was part of a large-scale study to evaluate wildlife response to woody biomass harvesting. The radio telemetry study was conducted on a 75.43-ha site that contained six woody biomass treatments each with varying volumes of retained DWD (Fritts et al. in review). The enclosure was located in the same regenerating stand, but outside of the treatment boundaries. The stand was clearcut for roundwood and woody biomass and had treatment implementation in winter 2010–2011. The site received two post-harvest herbicide applications with Chopper© (BASF Corporation, Research Triangle Park, North Carolina): a broadcast application year one post-harvest and a banded application year two post-harvest. Following clearcut and woody biomass harvests but before our study, the site was V-sheared to reduce stumps. The site was bedded and replanted fall 2011-winter 2012 (i.e., year two post-harvest and before sampling) at a density of ≈ 1,100 trees/ha. Ditches occurred throughout the sites to provide water drainage for pine seedlings.

Data Collection

Enclosure Trials. We compared habitat selection of southern toads among varying volumes and spatial distributions of CWD using an enclosure experiment from May-August 2013. In 2012, we constructed an 8-m^2 enclosure ≥ 50 m from the nearest ditch or road to allow the CWD to weather for one year prior to conducting the study. Enclosure construction occurred two years post-harvest and V-shearing and sampling occurred one year post-bedding and planting. Each treatment contained approximately equal areas covered by bare ground, bedded soil, and seedling pine trees. The outside barrier of the enclosure consisted of 1-m tall silt fencing buried ≥ 20 cm into the ground to avoid subterranean movement of individual toads. Woody biomass harvests typically glean CWD as opposed to smaller diameter DWD; therefore, we exclusively used CWD in our enclosure experiment. We created the treatments by clearing all vegetation, litter, and woody biomass from inside the enclosure and placing large pine harvest residue into the enclosed treatments. In 2013, CWD was progressing from decay stage two into decay stage three and was beginning to fade in color (Maser et al. 1979). Four 4-m^2 treatments were allocated randomly on the bare ground within the four quadrants of the enclosure: 1) 100% of the area covered by CWD (1.10 m^3, 100CWD); 2) ≈ 50% of the area covered with CWD in one large pile (0.60m^3, 50CWD; 3) ≈ 50% of the area covered with CWD and ≈ 25% of the volume of 100CWD dispersed evenly throughout the treatment (0.25m^3, 25CWD); and 4) no CWD (0CWD). We removed vegetation from the enclosure as needed.

We conducted habitat selection trials using an antenna system that detected Passive Integrated Transponder (PIT; Oregon RFID, Portland, Oregon, USA) tags. We created antennas using 10 AWG THHN wire and surrounded each treatment on two sides within the enclosure with one loop of wire to create one antenna per treatment (Figure 1). Where wires intersected the silt fence, we pulled wires through the barrier so that toads traveling near the fence would not continuously activate the antennas and drain the power source. We connected each antenna to a standard remote tuning board (Oregon RFID, Portland, Oregon, USA) and connected tuning boards to a multi-antenna HDX data logger (Oregon RFID, Portland, Oregon, USA) using Twinax cables (Oregon RFID, Portland, Oregon, USA). The system was powered with a 12-volt marine battery.

We captured adult male and female toads by road-cruising within an 8-km radius of the enclosure. We placed individuals in a 1-m diameter x 50-cm deep plastic container with soil and deionized water for 24 to 72 hours, depending on the number of toads captured and needed for each trial, to ensure toads started the experiment hydrated. We fed crickets (*Acheta domesticus*) to toads *ad libitum*. The plastic container remained outdoors so that toads were exposed to natural temperatures and photoperiods. Before using toads in a trial, we glued a 12-mm PIT tag (Oregon RFID, Portland, Oregon, USA) onto the lower back of the toad using medical-grade cyanoacrylate gel adhesive. We released ≤ five toads, depending on capture success, at the center of the enclosure between 2030 and 2130 for 48-hour trials. Our experimental density of ≤ five toads within one 4-m^2 treatment is lower than natural post-metamorphic densities of southern toads along pond margins (Beck and

Congdon 1999). Occasionally, we did not capture five toads and therefore released two to four toads inside the enclosure. Antenna wires were placed ≈ 10 cm within each treatment, and we tuned antennas so that PIT tags within ≈ 10 cm of an antenna would activate a transfer of data from the PIT tag to the data logger (i.e., a "hit"). Therefore, a hit was possible only when a toad was within the treatment unit and not between treatment boundaries. Antennas scanned for PIT tags in sequence (i.e., Treatment 1 (100CWD), Treatment 2 (50CWD), Treatment 3 (25CWD), Treatment 4 (0CWD)), and each antenna scanned every 2.3 seconds. For each hit, the date, timestamp, toad's PIT tag number, and antenna number (i.e., treatment plot location of a toad) were stored on the data logger. We downloaded data from the data logger every other day. Following enclosure trials, recovered toads were released near their capture site. All procedures were approved by the North Carolina State University Institutional Animal Care and Use Committee (11-022-O) and the North Carolina Wildlife Resources Commission (scientific collection permit number 11SC00534).

In addition to toad locations, we collected temperature data every 30 minutes using two Thermocron I-buttons (Maxim, San Jose, California, USA) at three locations in each treatment plot: in the center; on the interior edge; and halfway between the center and interior edge (Figure 1). We placed I-buttons inside 10-cm-tall segments of PVC piping hammered 4 cm into the ground to prevent loss. In 100CWD and 50CWD, we positioned two I-buttons under the center of the DWD pile, two under DWD on the edge of the pile, and two in the open to collect ambient temperatures. In 25CWD, we positioned two I-button containers, each with two I-buttons, under separate logs and the third container in the open. In 0CWD, all I-buttons were in the open because no CWD was present. We used the temperature data to determine the mean temperature during each 48-hour toad trial. We estimated daily rainfall using a rain gauge (Rainwise, Trenton, Maine, USA) located ≈ six km from the enclosure.

Radio Telemetry. In 2013, we captured toads using 18 'Y'-shaped drift fence arrays with 7.6-m arms within the clearcut. Drift fences were constructed of silt-fence material with a 19-L plastic bucket buried flush with the ground at each end. We drilled three small holes in the bottom of each bucket for water drainage and placed sponges in each trap that were wetted daily. We captured and housed toads as described above.

We attached Holohil BD-2 transmitters (0.90 g, Holohil Systems, Ltd., Carp, Ontario, Canada) to female toads weighing a ≥ 20 g using a belt attachment. The belt and transmitter weighed ≈ 1.1 g. We used only female toads because males are smaller than females and to ensure the belt and transmitter were ≤ 10% of the toad's body weight (Berteaux et al. 1994). We threaded flexible silicone tubing with a 1.5-mm inside diameter (Harvard Apparatus, Holliston, Massachusetts, USA) through a loop soldered onto each radio to make the belts. We threaded medium-weight, nylon fishing line through the silicone, placed the belt with the attached radio transmitter around the toad directly anterior to the hind legs with the transmitter resting on the dorsal side of the toad and tied a knot in the fishing line. We ensured that all fishing line was covered by the silicone tubing to limit skin irritation.

We released toads at random locations within the clearcut ≥ 50 m from the nearest road, ditch or edge between 2030 and 2130. We located each toad daily between 0700 and 1800 with a handheld three-element yagi antenna by homing (White and Garrott 1990) until we visually located individuals or their burrow or cover. We estimated groundcover at each toad location and at a paired location at a random direction 10 m from each toad location. We centered a 0.5-m^2 PVC frame on the toad and random locations and estimated % groundcover of CWD, litter (FWD and leaf litter), grass, forbs, woody vegetation, bare ground, and water (i.e., puddled rainwater). We checked the belts every five days to ensure proper fitting. At the end of the radio-telemetry project, captured toads were released near their capture site.

Statistical Analysis

Enclosure Trials. Toads are active nocturnally throughout the summer (Lannoo 2005); therefore, we analyzed nocturnal habitat selection and diurnal refuge site selection separately. We determined nocturnal habitat selection by calculating the proportion of time each toad spent in each treatment during nocturnal hours. We assumed toads remained within a single treatment when more than one consecutive hit (i.e., an entrance hit and an exit hit) existed on the same antenna. We assumed the first hit was when the toad entered the treatment and each subsequent hit was when toad approached the antenna, but did not cross the antenna to exit the treatment. We assumed toads exited a treatment when the next hit was recorded on a different antenna (i.e., the toad entered a different treatment). When only one hit was recorded on an antenna, we assumed the toad approached the treatment, but did not enter the treatment; therefore we did not use the associated hit in analysis. We calculated time spent in each treatment by subtracting the time of the earliest antenna hit from the time of the latest antenna hit for each consecutive series of hits of the same antenna. For example, if antenna two was hit at 0220, 0230, and 0240, the time spent in treatment two for that particular series of consecutive hits was assumed to be 20 minutes (i.e., 0240 – 0220). We summed the total time spent in each treatment for each toad over the entire trial and calculated the proportion of time each toad spent in each treatment.

We used package adehabitat (Calenge 2006) in statistical program R (R Core Team 2012) to conduct a parametric compositional analyses to determine if the proportion of time spent in each treatment during nocturnal hours differed from treatment availability. We used a matrix of proportions of time each toad spent in each treatment during nocturnal hours with each column representing a treatment and each row representing a toad (i.e., the toad is the unit of replication) versus a matrix of proportions of time available for each treatment (i.e., time available for each treatment = 25%).

We tested for differences in daily mean temperature among treatments using a one-way Analysis of Variance (ANOVA). We used four univariate ANOVA tests to determine if mean temperature on the enclosure ground and rain were predictors of nocturnal habitat selection. The dependent variable for each ANOVA was a vector of proportion of time spent in one treatment for each toad during nocturnal hours (Aebischer et al. 1993, Aitchison 1986). The independent variables were rain occurrence (1 = rain occurred during the trial and 0 = rain did not occur during the trial) and a vector of mean temperatures on the enclosure ground in all treatments during each trial (i.e., mean temperature for each 48-hour period, both day and night, estimated over the 24 I-buttons). We included an interaction term between rain and temperature. We considered a covariate as a predictor of nocturnal habitat selection if the associated P-value was ≤ 0.05.

Antennas did not receive hits during diurnal hours, so we determined the treatment that toads used for diurnal refuge by identifying the antenna with the last hit of the night (i.e., we assumed the toad was no longer active and seeking refuge for the day). We calculated the proportion of the days toads spent in each treatment by counting the number of days toads spent in each treatment and dividing by the total number of days toads spent in all treatments.

Radio Telemetry. We analyzed diurnal refuge site locations using a global mixed effects logistic regression model with a binomial response (i.e., 1 = toad location, 0 = random location). We used estimated % groundcover of each cover type (i.e., CWD, litter, grass, forb, woody vegetation, bare ground, and standing water) as independent continuous covariates. We used the individual toad as a random effect to allow for correlation among observations of the same toad. We used variance inflation factors to assess collinearity among independent variables and dropped a covariate from the analysis if the factor was >3 (Zuur et al. 2009). We did not include data from toads that had only one observation. We ran the global model and considered a covariate as a predictors of microhabitat selection if the covariate had an associated P-value ≤ 0.05.

RESULTS

Enclosure Trials

We captured 47 toads (31 females and 16 males) for enclosure trials. Eleven toads either escaped the enclosure or were depredated and were unable to be recovered. Toads did not use treatments randomly during nocturnal hours ($n = 47$, $\Lambda = 0.49$, $P < 0.01$; Table 1). The rankings of nocturnal habitat selection from most selected to least was 0CWD, 100CWD, 25CWD, 50CWD. Daily mean temperatures (º Celsius) across the six I-buttons were (mean ± SE) 27.66º C ± 0.33º C in 0CWD, 26.91º C ± 0.30º C in 25CWD, 27.26º C ± 0.51º C in 50CWD, and 25.13º C ± 0.23º C in 100CWD. Daily mean temperatures differed among treatments ($F_{3,96} = 6.27$, $P < 0.01$) and were 2.13º C greater in 0CWD than 100CWD ($P < 0.01$), and 1.47º C greater in 0CWD than 50CWD ($P = 0.02$). Daily mean temperatures were 1.29º C greater in 0CWD than 25CWD ($P = 0.06$), but the difference was only marginally significant.

Twenty-two toad trials were omitted from the nocturnal habitat selection analysis because of missing I-button data. Mean trial temperature did not predict the proportion of time a toad ($n = 25$) spent in 0CWD ($t = 0.04$, $P = 0.30$), 25CWD ($t = -0.35$, $P = 0.73$) or 50CWD ($t = 0.08$, $P = 0.93$). Rain did not predict the proportion of time a toad spent in 0CWD ($t = -0.89$, $P = 0.38$), 25CWD ($t = 0.79$, $P = 0.44$) or 50CWD ($t = 0.05$, $P = 0.96$). There was an interaction between rain and mean trial temperature ($t = 4.86$, $P = 0.04$) for the proportion of time a toad spent in 100CWD (Figure 2). When rain occurred during the trial (i.e., rain = 1), mean trial temperature was not a predictor of the proportion of time toads spent in 100CWD ($\beta = 0.01$, t

= 0.23, P = 0.82). When rain did not occur during the trial (i.e., rain = 0), toads spent a greater proportion of time in 100CWD as mean trial temperature increased (β = 0.18, t = 2.60, P = 0.03; Figure 2). The proportion of diurnal refuge sites located within 100CWD was 75%, while the proportion of diurnal refuge sites located within each of the other three treatments was 8%.

Radio Telemetry

We attached radio transmitters to 40 individual female toads. Three toads were omitted because we located the toad only one time before either the belt slipped (n = 1) or the toad died (n = 2). We acquired between two and 27 locations (mean = 7.10; standard error = 1.22) per toad before the toad was depredated, died of other causes, or moved into a ditch and was unable to be recovered. Toads sometimes traveled into ditches (0.02%) that were covered with vegetation and we could not estimate groundcover because of the inability to visually locate the toad. We did not include forb groundcover in the models because of collinearity. Toads (n = 37) avoided grass (β = -0.02, z = -3.40, P < 0.01), bare ground (β = -0.06, z = -6.07, P < 0.01), and leaf litter/sawdust (β = -0.01, z = 0.01, P = 0.05) as diurnal refuge sites, but there was no relationship with CWD cover (β = 0.02, z = 1.59, P = 0.11, Figure 3). Toad diurnal refuge locations included in or under a piece of CWD (13.25%), in or under piled CWD/FWD (21.69%), in or under FWD/sawdust (13.25%), under live vegetation (30.12%), in a ditch (13.25%), burrowed into soil (3.61%), and in the open/not in a refuge (4.82%).

DISCUSSION

Woody biomass harvests likely have minimal long-term impacts on southern toad populations because toads may use sources of cover other than DWD. Southern toads in the enclosure selected CWD during nocturnal hours and often used CWD for diurnal refuge, but toads may switch to other cover sources when available. Piled CWD has been demonstrated to lower desiccation risk in other anurans (Seebacher and Alford 2002, Rittenhouse et al. 2008), but radio-marked toads did not select CWD for diurnal refuge sites in our study. Although radio-marked toads often were located under or near CWD, toads also used several other sources of cover for diurnal refuge including woody vegetation, herbaceous groundcover, and FWD, indicating their ability to use what cover was available. Conversely, southern toads within the enclosure selected 100CWD for diurnal refuge seemingly because other cover sources were not present (i.e., vegetation was removed manually).

Southern toads exhibited thermoregulatory behavior based on temperature and precipitation, presumably to decrease desiccation risk or to maximize their energy budget. Amphibians experience rapid water loss through their permeable skin at high temperatures or low humidity (Vitt and Caldwell 2009). Behavioral hypothermia in toads reduces water loss (Malvin and Wood 1991), and southern toads are known to exhibit thermoregulatory behavior based on hydration state (Forster 2013). Similar to American toads (*Anaxyrus americanus*), which seek lower temperatures that decrease evaporative water loss when dehydrating (Tracy et al. 1993), southern toads selected 100CWD when rain did not occur and temperature increased. Further, increases in temperature directly affect amphibian metabolic rates and energy expenditure (Pough 1980, Rome et al. 1992, Homyack et al. 2010), which may encourage ecological trade-offs that influence an individual's fitness, such as growth (Congdon et al. 1982, Sears 2005, DuRant et al. 2007). For example, newly metamorphosed western toads (*Bufo boreas*) maximize energy ingestion, linear growth, and weight increase at 27º C and are able to behaviorally thermoregulate to keep their body at that temperature (Lillywhite et al. 1973). Further, exposure of anurans to greater air temperatures not only increases growth, but also decreases maturation times, which can affect reproductive output (Hadfield 1966, Smith 1976, Lillywhite 1970). Male green and golden bell frogs (*Litoria aurea*) exposed to three temperatures, 15º C, 22 ºC, and 28º C, experienced growth and sexual maturation only at 28º C, indicating the importance of temperature on amphibian reproduction and abundance (Browne and Edwards 2003).

Amphibians in the southeastern United States may have adapted to low levels of DWD because frequent fires coupled with rapid decay rates of DWD in this region maintained DWD volumes relatively low compared to other regions of the United States (Harmon et al. 1986, Sharitz et al. 1992, Moorman et al. 1999). Therefore, amphibians in the region do not respond strongly to CWD, but instead may be linked to burrow and leaf litter availability (Rothermel and Luhring 2005, Owens et al. 2008, Davis et al. 2010). For example, southern toads, eastern narrowmouth toads (*Gastrophryne carolinensis*), and spadefoot toads (*Scaphiophus holbrookii*) avoid heat and desiccation by burrowing into soil, using pre-existing burrows, tunnels, and root systems, and using cover other than woody debris (Wright 1932, Ashton and Ashton 1998, Russell et al. 1999).

Retaining piles of CWD scattered throughout the landscape to provide cover during hot and dry weather before vegetation becomes established may limit impacts of woody biomass harvesting on amphibians, particularly those with high desiccation risk. Clearcut harvesting can increase soil and air temperatures (Liechty et al. 1992, Valigura and Messina 1994), and retaining DWD following timber harvests may mitigate negative effects on amphibians (deMaynadier and Hunter 1995, McKinney et al. 2006). Thus, retaining DWD immediately following overstory removal (i.e., before vegetation cover is available) may be beneficial to many amphibians. Microhabitats have the potential to buffer climate extremes and likely reduce mortality during extreme climate events (Scheffers et al. 2014). Yet, the size of CWD piles and minimum volume of CWD needed to sustain amphibian populations is not known. Further, climate change is expected to lead to a warmer and drier southeastern United States (IPCC 2013, U.S. Global Change Research Program 2014). Therefore, retaining DWD may be more critical if future climate predictions are accurate.

Woody biomass harvests on our study sites retained ≥ 16.28 m^3/ha of DWD (Fritts et al. in review), which exceeded by over three-fold the sustainability recommendations set by the Forest Guild (Perschel et al. 2012). However, as markets for wood-based energy expand and technological advances in conversion and harvest machinery occur, there may be less DWD, particularly CWD, retained following woody biomass harvests (Haynes 2003, Perlack et al. 2005, Janowiak and Webster 2010). We recommend research that quantifies the minimum volume of CWD needed to sustain amphibian populations in forests subjected to woody biomass harvests. This information in conjunction with results from mechanistic studies like ours can be used to guide development of BHGs that meet their sustainability objectives.

ACKNOWLEDGMENTS

Funding was provided by the U.S. Department of Agriculture National Institute of Food and Agriculture Managed Ecosystems Program, the Department of Interior Southeast Climate Science Center, the Biofuels Center of North Carolina, the North Carolina Bioenergy Initiative, Weyerhaeuser Company, and the National Council of Air and Stream Improvement. We thank J. Bowser, E. Evans, B. Moore, J. Eells, J. Kiser, J. Bowser, and J. O'Connor for field data collection and J. Fritts for helping with enclosure operation and maintenance.

LITERATURE CITED

Aebischer, N. J., P. A. Robertson, R. E. Kenward. 1993. Compositional analysis of habitat use from animal radio-tracking data. Ecology 74: 1313-1325.

Aitchison, J. 1986. The statistical analysis of compositional data. Chapman and Hall, London, New York, USA.

Alban, D. H., D. A. Perald, B. E. Schlaegel. 1978. Biomass and nutrient distribution in aspen, pine, and spruce stands on the same soil type in Minnesota. Canadian Journal of Forest Research 8: 290-299.

Ashton, R. E., Jr., P. S. Ashton. 1988. Handbook of reptiles and amphibians of Florida: Part Three: The amphibians. Windward Publishing, Miami, Florida, USA.

Beck, C. W., J. D. Congdon. 1999. Effects of individual variation in age and size at metamorphosis on growth and survivorship of southern toad (*Bufo terrestris*) metamorphosis. Canadian Journal of Zoology 77: 944-951.

Berteaux, D., R. Duhamel, J. M. Bergenson. 1994. Can radio collars affect dominance relationships in *Microtus*? Canadian Journal of Zoology 72:785–789.

Browne, R. K., D. L. Edwards. 2003. The effect of temperature on the growth and development of the endangered green and golden bell frog (*Litoria aurea*). Journal of Thermal Biology 28: 295-299.

Bruce, K. J., J. L. Christiansen. 1976. The food and food habits of Blanchard's cricket frog, *Acris crepitans blanchardi* (Amphibia, Anura, Hylidae), in Iowa. Journal of Herpetology 10: 63-74.

Burton, T. M., G. E. Likens. 1975. Salamander populations and biomass in the Hubbard Brook Experimental Forest, New Hampshire. Copeia 1975: 541-546.

Burton, T. M. G. E. Likens. 1975. Energy flow and nutrient cycling in the salamander populations in the Hubbard Brook Experimental Forest, New Hampshire. Ecology 56: 1068-1080.

Butts, S. R., W. C. McComb. 2000. Associations of forest-floor vertebrates with coarse woody debris in managed forests of western Oregon. Journal of Wildlife Management 64:95-104.

Calenge, C. 2006. The package adehabitat for the R software: a tool for the analysis of space and habitat use by animals. Ecological Modelling 197: 516-519

Carey A. B., M. L. Johnson. 1995. Small mammals in managed, naturally young, and old-growth forests. Ecological Applications 5: 336-352.

Chen, J., S. C. Saunders, T. R. Crow, R. J. Naiman, K. D. Brosofske, G. D. Mroz, B. L. Brookshire, and J. F. Franklin. 1999. Microclimate in forest ecosystem and landscape ecology. Bioscience 49: 288-297.

Congdon, J. D., A. E. Dunham, D.W. Tinkle. 1982. Energy budgets and life histories of reptiles. Pages 233-271 In: Gans, C. (editor). Biology of the Reptilia. Academic Press, New York, USA.

Currie, W. S., K. J. Nadelhoffer. 2002. The imprint of land-use history: Patterns of carbon and nitrogen in downed woody debris at the Harvard Forest. Ecosystems 5: 446-60.

Davis, J. C., S. B. Castleberry, J. C. Kilgo. 2010. Influence of coarse woody debris on herpetofaunal communities in upland pine stands of the southeastern Coastal Plain. Forest Ecology and Management 259: 1111-1117.

deMaynadier, P .G., M. L. Hunter, Jr. 1995. The relationship between forest management and amphibian ecology: a review of the North American literature. Environmental Review 3: 230-261.

DuRant, S. E., W. A. Hopkins, L. E. Talent. 2007. Energy acquisition and allocation in an ectothermic predator exposed to a common environmental stressor. Comparative Biochemistry and Physiology Part C 145: 442-448.

Energy Information Administration (EIA). 2013. Annual Energy Outlook 2014. United States Department of Energy, Energy Information Administration, Washington, DC, USA.

Evans, J. M., R. J. Fletcher, Jr., J. R. R. Alavalapati, A. L. Smith, D. Geller, P. Lal, D. Vasudev, M. Acevedo, F. Calabria, T. Upadhyay. 2013. Forestry bioenergy in the Southeast

United State: implications for wildlife habitat and biodiversity. National Wildlife Federation, Merrifield, Virginia, USA.

Forster, C. E. 2013. Dehydration in southern toads (*Anaxyrus terrestris*): Metabolic costs and effects of temperature selection. Scripps Senior Thesis. Paper 161.

Fritts, S. R., C. E. Moorman, D. W. Hazel, B. D. Jackson. in review. Biomass harvesting guidelines affect downed woody debris retention.

Gatten, R. E., K. Miller, R. J. Full. 1992. Energetics at rest, during locomotion. Pages 314-377 In M. E. Feder and W. W. Burggren (editors). Environmental Physiology of the Amphibians. University of Chicago Presss, Chicago, Illinois, USA.

Gibbons, J. W., K. A. Buhlmann. 2001. Reptiles and amphibians. Pages 372–390 In J.G. Dickson (editor). Wildlife of Southern Forests: Habitat and Management. Hancock House Publishers, Blaine, Washington, USA.

Gibbons, J. W., C. T. Winne, D. E. Scott, J. D. Willson, X. Glaudas, K. M. Andrews, B. D. Todd, L. A. Fedewa, L. Wilkinson, R. N. Tsalliagos, S. J. Harper, J. I. Greene, T. D. Tuberville, B. S. Metts, M. E. Dorcas, J. P. Nestor, C. A. Young, T.Akre, R. B. Reed, K. A. Buhlmann, J. Norman, D. A. Croshaw, C. Hagen, B. B. Rothermel. 2006. Remarkable amphibian biomass and abundance in an isolated wetland: implications for wetland conservation. Conservation Biology 20: 1457-1465.

Goh, C. S., M. Junginger, M. Cocchi, D. Marchal, D. Thran, C. Hennig, J. Heinimo, L. Nikolaisen, P. Schouwenberg, D. Bradley, R. Hess, J. Jacobson, L. Ovard, M. Deutmeyer. 2013. Wood pellet market and trade: a global perspective. Bioproducts, Biofuels, Biorefineries 7: 24-42.

Graham, S. A. 1925. The felled tree trunk as an ecological unit. Ecology 6:397-416.

Hadfield, S. 1966. Observations on body temperature and activity in the toad *Bufo woodhousei fowleri*. Copeia: 581-582.

Hansen, A. J., T. A. Spies, F. J. Swanson, J. L. Ohmann. 1991. Conserving biodiversity in managed forests. Bioscience 41: 382-392.

Hanson, C., L. Yonavjak, C. Clarke, S. Minnemeyer, L. Boisrobert, A. Leach, K. Schleeweis. 2010. Southern forests for the future. World Resources Institute, Washington, DC, USA.

Harmon, M. E., J. F. Franklin, F. J. Swanson, P. Sollins, S. V. Gregory, J. D. Lattin, N. H. Anderson, S. P. Cline, N. G. Aumen, J. R. Sedell, G. W. Lienkaemper, K. Cromack Jr., K. W. Cummins. 1986. Ecology of coarse woody debris in temperate ecosystems. Advanced Ecological Research. 15: 133-302.

Harpole, D. N., C. A. Haas. 1999. Effects of seven silvicultural treatments on terrestrial salamanders. Forest Ecology and Management 114: 349-356.

Hassinger, J. D. 1989. Dead wood for wildlife. Pages 171-184 in Proceedings of the 1989

Penn State Forest Resources Issues Conference. April 4 – 6 1989. University Park, Pennsylvania, USA.

Haynes, R. W. 2003. An analysis of the timber situation in the United States: 1952 to 2050. United States Department of Agriculture Forest Service, PNW-GTR-560. Pacific Northwest Research Station, USA.

Hillyard, S. D. 1999. Behavioral, molecular and integrative mechanisms of amphibian osmoregulation. Journal of Experimental Zoology 283: 662-674.

Homyack, J. A., A. Aardweg, T. A. Gorman, D. R. Chalcraft. 2013. Initial effects of woody biomass removal and intercropping of switchgrass (*Panicum virgatum*) on herpetofauna in eastern North Carolina. Wildlife Society Bulletin 37: 327-335.

Homyack, J. A., C. A. Haas, W. A. Hopkins. 2011. Energetics of surface-active terrestrial salamanders in experimentally harvested forest. Journal of Wildlife Management 75: 1267-1278.

Homyack, J. A., C. A. Haas, W. A. Hopkins. 2010. Influence of temperature and body mass on standard metabolic rate of eastern red-backed salamanders (*Plethodon cinereus*). Journal of Thermal Biology 35: 143-146.

Huey, R. B. 1991. Physiological consequences of habitat selection. The American Naturalist 137: 91-115.

Intergovernmental Panel on Climate Change (IPCC). 2013. Climate Change 2013: The Physical Science Basis. In: Stocker, T.F., D. Qin, G.-K. Plattner, M. Tignor, S.K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex and P. M. Midgley (editors). Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, New York, USA.

International Union for Conservation of Nature (IUCN). 2013. The IUCN Red List of Threatened Species: Amphibians on the IUCN Red List, Geographic Patterns. Version 2013.2.

Janowiak, M. K., Webster, C. R. 2010. Promoting ecological sustainability in woody biomass harvesting. Journal of Forestry 108: 16-23.

Jørgensen, C. B. 1997. 200 years of amphibian water economy: from Robert Townsend to the present. Biological Reviews 72: 153-237.

Kluber, M. R., D. H. Olson, K. J. Puettmann. 2009. Downed wood microclimates and their potential impact on Plethodontid salamander habitat in the Oregon coast range. Northwest Science 83: 25-34.

Knapp, S. M., C. A. Haas, D. N. Harpole, R. L. Kirkpatrick. 2003. Initial effects of clearcutting and alternative silvicultural practices on terrestrial salamander abundance. Conservation Biology 17:752–762.

Lanham, J. D., D. C. Guynn Jr. 1996. Influence of coarse woody debris on birds in southern forests. Pages 101-107 In: J. W. McMinn and D. A. Crossley Jr. (editors). Biodiversity and coarse woody debris in southern forests: proceedings of the workshop on coarse woody debris in southern forests: effects on biodiversity. United State Department of Agriculture Forest Service General Technical Report SE-94, Southern Research Station, Asheville, North Carolina, USA.

Lannoo, M. (editor). 2005. Amphibian Declines: The conservation status of United States Species. University of California Press, Berkeley and Los Angeles, California, USA.

Liechty, H. W., M. J. Helmes, D. D. Reed, G. D. Mroz. 1992. Changes in microclimate after stand conversion in two northern hardwood stands. Forest Ecology and Management 50: 253-264.

Lillywhite, H. B. 1970. Behavioral temperature regulation in bullfrog, *Rana catesbeiana*. Copeia 1970: 158-168.

Lillywhite, H. B. 1973. Role of thermoregulation in growth energetics of toad, *Bufo boreas*. Ecology 54: 375-383.

Loeb, S. C. 1996. The role of coarse woody debris in the ecology of southeastern mammals. Pages 108-118 In J. W. McMinn and D. A. Crossley Jr. (editors). Biodiversity and coarse woody debris in southern forests: proceedings of the workshop on coarse woody debris in southern forests: effects on biodiversity. United State Department of Agriculture Forest

Service. General Technical Report SE-94, Southern Research Station, Asheville, North Carolina, USA.

Malvin, G. M., S. C. Wood. 1991. Behavioral thermoregulation of the toad, *Bufo marinus*: Effects of air humidity. Journal of Experimental Zoology 258: 322-326.

Mantau U., U. Saal, F. Steierer, H. Verkerk, J. Oldenburger, N. Leek, K. Prins. 2010. EUwood - Real potential for changes in growth and use of EU forests. Final report. Hamburg, Germany.

Maser, C., R. G. Anderson, K. Cromack Jr., J.T. Williams, R. E. Martin. 1979. Dead and down woody debris. Pages 78-95 In: J.W. Thomas (editor). Wildlife habitats in managed forests: the Blue Mountains of Oregon and Washington. United States Department of Agriculture Forest Service Agricultural Handbook No. 553.

McKinney, H. C., W. S. Keeton, and T. M. Donovan. 2006. Effects of structural complexity enhancement on eastern red-backed salamander (*Plethodon cinereus*) populations in northern hardwood forests. Forest Ecology and Management 230: 186-196.

McLeod, R. F., J. E. Gates. 1998. Response of herpetofaunal communities to forest cutting and burning at Chesapeake Farms, Maryland. American Midland Naturalist 139:164-177.

Mendell, B. C., A. H. Lang. 2012. Wood for bioenergy: forests as a resource for biomass and biofuels. Forest History Society.

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

Owens, A. K., K. R. Moseley, T. S. McCay, S. B. Castleberry, J. C. Kilgo, W. M. Ford. 2008. Amphibian and reptile community response to coarse woody debris manipulations in upland loblolly pine (*Pinus taeda*) forests. Forest Ecology and Management 256: 2078-2083.

Perlack, R. D., L. L. Wright, A. F., Turhollow, R. L. Graham, B. J. Stokes, Erback D. C. 2005. Biomass as feedstock for a bioenergy and bioproducts industry: the technical feasibility of a billion-ton annual supply. Oakridge National Laboratory, Oak Ridge, Tennessee, USA.

Perschel, B., A. Evans, M. DeBonis. 2012. Forest biomass retention and harvesting guidelines for the Southeast. Forest Guild Southeast Biomass Working Group. Forest Guild. Sante Fe, New Mexico, USA.

Petranka, J. W., S. S. Murray. 2001. Effectiveness of removal sampling for determining salamander density and biomass: a case study in an Appalachian streamside community. Journal of Herpetology 35: 36-44.

Pough, F. H. 1980. The advantages of ectothermy for tetrapods. The American Naturalist 115: 92-112.

Ranius, T., and L. Fahrig. 2006. Targets for maintenance of dead wood for biodiversity conservation based on extinction thresholds. Scandinavian Journal of Forest Research 21: 201-208.

REN21. 2013. Renewables 2013 Global Status Report. Renewables Energy Policy Network for the 21st Century. Paris, France.

R Core Team. 2012. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Rittenhouse, T. A., E. B. Harper, L. R. Rehard, R. D. Semlitsch. 2008. The role of microhabitats in the desiccation and survival of anurans in recently harvested oak-hickory forest. Copeia 2008: 807-814.

Rome, L. C., E. D. Stevens, H. B. John-Alder. 1992. The influence of temperature and thermal acclimation on physiological function. In M. E. Feder and W. W. Burggren (editors). Environmental Physiology of the Amphibians. University of Chicago Press, Chicago, Illinois, USA.

Rosenvald R., A. Lohmus, A. Kraut, L. Remm. 2011. Bird communities in hemiboreal old-growth forests: The roles of food supply, stand structure, and site type. Forest Ecology and Management 262: 1541-1550.

Rothermel, B. B., T. M. Luhring. 2005. Burrow availability and desiccation risk of mole salamanders (*Amystoma talpoideum*) in harvested versus unharvested forest stands. Journal of Herpetology 39: 619-626.

Rudolphi, J., L. Gustafsson. 2005. Effects of forest-fuel harvesting on the amount of deadwood on clear-cuts. Scandinavian Journal of Forest Research 20: 235-242.

Russell, K. R., D. H. Van Lear, D. C. Guynn Jr. 1999. Prescribed fire effects on herpetofauna: review and management implications. Wildlife Society Bulletin 27: 374-384.

Scheffers, B. R., D. P. Edwards, A. Diesmos, S. E. Williams, T. A. Evans. 2014. Microhabitats reduce animal's exposure to climate extremes. Global Change Biology 20: 495-503.

Sears, M. W. 2005. Resting metabolic expenditure as a potential source of variation in growth rates of the sagebrush lizard. Comparative Biochemistry and Physiology Part A 140: 171-172.

Seebacher, F., R. A. Alford. 2002. Shelter microhabitats determine body temperature and dehydration rates of a terrestrial amphibian (*Bufo marinus*). Journal of Herpetology 36: 69-75.

Sharitz, R. R., L. R. Boring, D. H. Van Lear, J .E. Pinder III. 1992. Integrating ecological concepts with natural resource management of southern forests. Ecological Applications 2: 226-237.

Smith, G. C. 1976. Ecological energetics of three species of ectothermic vertebrates. Ecology 57: 252-264.

Spotila, J. R., E. A. Standora. 1985. Energy budgets of ectothermic vertebrates. American Zoologist 25: 95-125.

Stebbins, R. C., N. W. Cohen. 1995. A Natural History of Amphibians. Princeton University Press, Princeton, New Jersey, USA.

Stuart, S. N., J. S. Chanson, N. A. Cox, B. E. Young, A. S. Rodrigues, D. L. Fischman, R. W. Waller. 2004. Status and trends of amphibian declines and extinctions worldwide. Science 306: 1783-1786.

Thorson, T., A. Svihla. 1943. Correlation of the habitats of amphibians with their ability to survive the loss of body water. Ecology 24: 374-381.

Todd, B. D., B. B. Rothermel. 2006. Assessing quality of clearcut habitats for amphibians: effects on abundances versus vital rates in the southern toad (*Bufo terrestris*). Biological Conservation 133: 178-185.

Tracy, C. R., Christian, K. A., O-Connor, M. P. Tracy, C. R. 1993. Behavioral thermoregulation by *Bufo americanus*: the importance of the hydric environment. Herpetologica 49: 375-382.

United States Department of Agriculture (USDA). 2010. A USDA regional roadmap to meeting the biofuels goals of the renewable fuels standard by 2022. United States Department of Agriculture, Washington, DC, USA.

United States Global Change Research Program. 2014. Climate change impacts in the United States. United State National Climate Assessment, Washington, DC, USA.

Valigura, R. A., M. G. Messina. 1994. Modification of Texas clearcut environments with loblolly pine shelterwoods. Journal of Environmental Management 40: 283-295.

Whiles, M. R., J. W. Grubaugh. 1996. Importance of coarse woody debris to southern forest herpetofauna. Pages 94-100 In: J. W. McMinn and D. A. Crossley, Jr. (editors). Biodiversity and Coarse Woody Debris in Southern Forests. Proceedings of the Workshop on Coarse Woody Debris in Southern Forests: Effects on Biodiversity. United States Department of Agriculture Forest Service General Technical Report SE-94.

White, G. C., R. A. Garrott. 1990. Analysis of wildlife radio-tracking data. Academic Press, San Diego, California, USA.

Woodall, C. W., V. J. Monleon. 2008. Sampling protocol, estimation, and analysis procedures for the down woody debriss indicator of the FIA program. NRS-GTR-22.United States Department of Agriculture Forest Service. Newtown Square, Pennsylvania, USA.

Wright, A. H. 1932. Life histories of the frogs of the Okefinokee Swamp, Georgia. North American Salientia (Anura) Number 2. Macmillan Press, New York, USA.

Vitt, L. J., J. P. Caldwell. 2009. Thermoregulation, Performance, and Energetics. Pages 191-216 In: L. J. Vitt and J. P. Caldwell (editors). Herpetology: An Introductory Biology of Amphibians and Reptiles. Academic Press, San Diego, California, USA.

Zheng, D., J. Chen, B. Song, M. Xu, P. Sneed. R. Jensen. 2000. Effects of silvicultural treatments on summer forest microclimate in southeastern Missouri Ozarks. Climate Research 15: 45-59.

Zug, G. R. 2001. Herpetology: An Introductory Biology of Amphibians and Reptiles. Academic Press, San Diego, California, USA.

Zuur, A. F., E. N. Ieno, N. J. Walker, A. A. Saveliev, G. M. Smith. 1999. Mixed effects models and extensions in ecology with R. Springer. New York, USA.

Table 1. Ranking matrices, sample size (n), lambda (Λ), and probability values based on comparing southern toad (*Anaxyrus terrestris*) proportional nocturnal habitat use within an enclosure with four treatments with varying volumes and spatial allocations of downed woody debris: 1) bare ground with 100% of the area covered by CWD (1.10 m^3, 100CWD); 2) bare ground with ≈ 50% of the area covered with CWD in one large pile (0.60m^3, 50CWD); 3) bare ground with ≈ 50% of the area covered with CWD and 25% of the volume of 100CWD dispersed evenly throughout the treatment (0.25m^3, 25CWD); and 4) bare ground with no CWD (0CWD) in a loblolly pine (*Pinus taeda*) plantation that was three years post-clearcut in Beaufort County, North Carolina (May-August 2013). A triple sign represents significant deviation from random at P ≤ 0.05.

	Treatment						
	0CWD	25CWD	50CWD	100CWD	n	Λ	P
0CWD	0	+++	+++	+	47	0.49	< 0.01
25CWD	---	0	+++	-			
50CWD	---	---	0	---			
100CWD	-	+	+++	0			

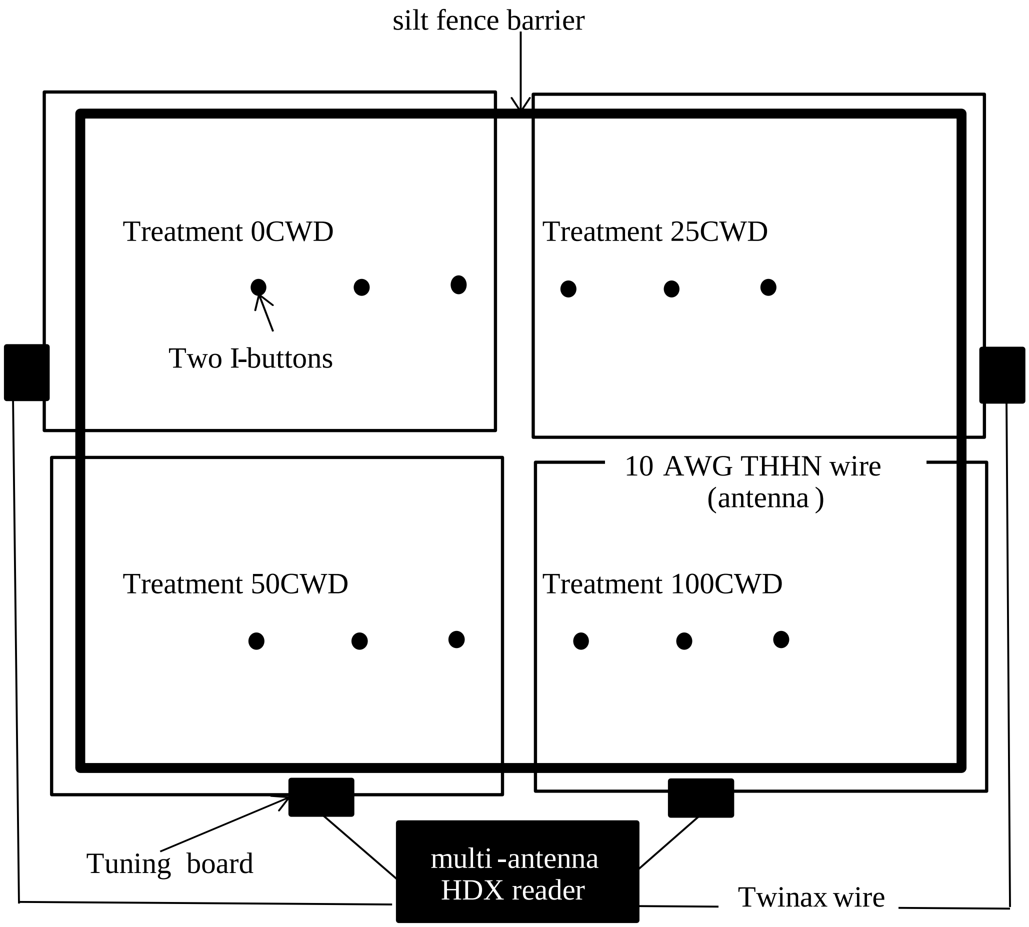

Figure 1. Schematic of enclosure used to assess southern toad (*Anaxyrus terrestris*) habitat selection in four 4-m^2 treatments with varying volumes and spatial allocations of downed woody debris: 1) bare ground with 100% of the area covered by CWD (1.10 m^3, 100CWD); 2) bare ground with ≈ 50% of the area covered with CWD in one large pile (0.60m^3, 50CWD); 3) bare ground with ≈ 50% of the area covered with CWD and 25% of the volume of 100CWD dispersed evenly throughout the treatment (0.25m^3, 25CWD); and 4) no CWD (0CWD) in a loblolly pine (*Pinus taeda*) plantation in Beaufort County, North Carolina (not drawn to scale).

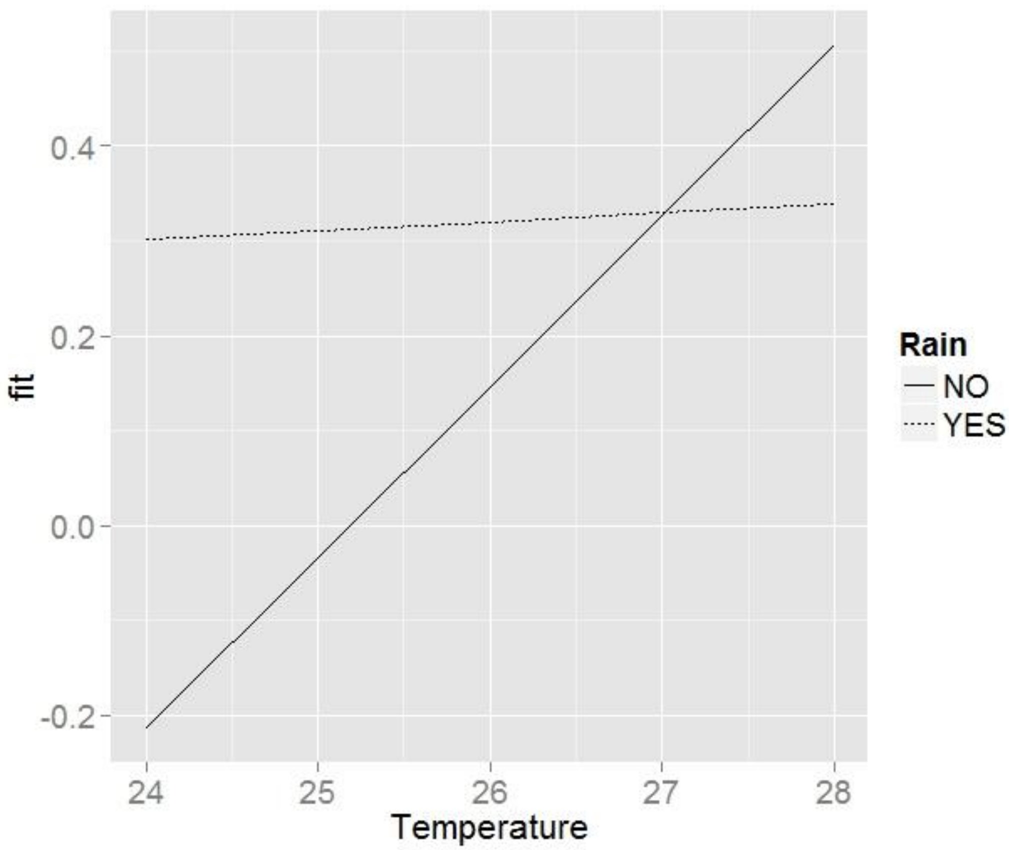

Figure 2. The interaction between rain (binomial) and temperature (º C) in treatment 100CWD resulting from an analysis of variance (fit) of southern toad (*Anaxyrus terrestris*; n = 47) proportional nocturnal habitat use within an enclosure with four treatments with varying volumes and spatial allocations of downed woody debris: 1) bare ground with 100% of the area covered by CWD (1.10 m^3, 100CWD); 2) bare ground with ≈ 50% of the area covered with CWD in one large pile (0.60m^3, 50CWD); 3) bare ground with ≈ 50% of the area covered with CWD and 25% of the volume of 100CWD dispersed evenly throughout the treatment (0.25m^3, 25CWD); and 4) bare ground with no CWD (0CWD) in a loblolly pine (*Pinus taeda*) plantation three year post-clearcut in Beaufort County, North Carolina (May–August 2013).

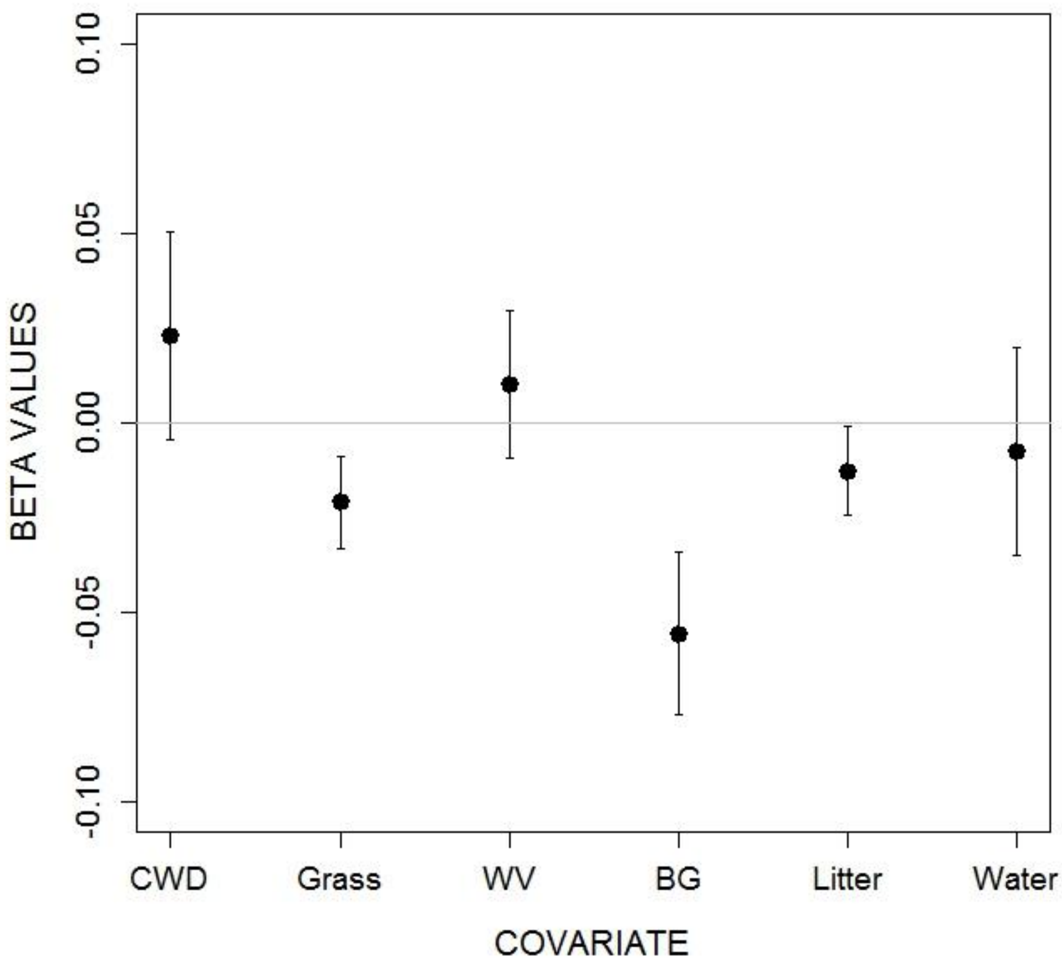

Figure 3. Beta (β) value estimates and 95% confidence intervals of parameters from a global mixed effects logistic regression model of telemetered southern toad (*Anaxyrus terrestris;* n = 37) diurnal refuge selection in a loblolly pine (*Pinus taeda*) plantation three years post-clearcut in Beaufort County, North Carolina (May-August 2013). We considered a covariate as significant if $P \leq 0.05$. Covariates included CWD = coarse woody debris ($P = 0.11$), grass ($P < 0.01$), WV = woody vegetation ($P = 0.32$), BG = bare ground ($P < 0.01$), Litter = fine woody debris and leaf litter ($P = 0.04$), and water ($P = 0.60$).

CHAPTER 4

Shrew Response to Variable Woody Debris Retention Following Clearcut Harvests

ABSTRACT

Shrews could be sensitive to downed woody debris (DWD) removal because their high moisture and metabolic requirements may result in dependency on the microhabitats that DWD provides. However, DWD in the form of woody biomass is a major feedstock for bioenergy worldwide and extraction of woody biomass is expected to increase as domestic and international policies that promote bioenergy development are implemented. To ensure the ecological sustainability of woody biomass harvesting, Biomass Harvesting Guidelines (BHGs) have been developed that suggest retaining DWD. However, the success of BHGs at reaching sustainability goals has not been tested in an operational setting. Our goals were to determine: 1) if woody biomass harvests negatively affect shrew relative abundance; 2) given affects exist, if BHGs mitigate potential negative consequences of woody biomass harvesting; and 3) the relationships between shrew capture success and DWD and vegetation. We captured shrews using drift fence arrays in six woody biomass harvesting treatments in pine plantation clearcuts in North Carolina (n = 4) and Georgia (n = 4), USA from April to August 2011-2013. Treatments included: 1) woody biomass harvest with no BHGs; 2) 15% retention with woody biomass dispersed; 3) 15% retention with woody biomass clustered; 4) 30% retention with woody biomass dispersed; 5) 30% retention with woody biomass clustered; and 6) no woody biomass harvest. We compared relative abundance of shrews among treatments using an analysis of variance. We used general linear regression models to determine if DWD volume and vegetation affected shrew capture success for species with > 100 captures/state/year. We had 1,252 shrew captures representing three species, *Cryptotis parva, Blarina carolinensis,* and *Sorex longirostris*, in 43,350 trap nights. We did not detect consistent differences of shrew relative abundance among treatments; however, relative abundance increased over the span of the study as vegetation became well-established. In 2013, total shrew capture success was negatively related to volume of DWD within 50 m of the drift fence array ($P = 0.05$) and positively related to bare groundcover ($P < 0.01$) in North Carolina and negatively related to herbaceous groundcover ($P < 0.01$) and leaf litter groundcover ($P = 0.02$) and positively related to woody vegetation groundcover ($P < 0.01$) and vertical vegetation structure ($P = 0.03$) in Georgia. Our results suggest that shrews are more associated with vegetation than DWD in the southeastern United States Coastal Plain and that woody biomass harvests may not affect shrew populations in the region.

KEYWORDS: biomass harvesting guidelines, downed woody debris, intensively managed pine plantation, shrew, *Sorex*, woody biomass

INTRODUCTION

Shrews are key components of forest food webs and have been used as indicators of the ecological effects of intensive forestry practices (Hamilton 1941, Van Zyll de Jong 1983, Carey and Harrington 2001, Ford and Rodrigue 2001, Matthews et al. 2009). Shrews have high nutritional and moisture requirements, and therefore may be sensitive to forestry practices that change forest floor microhabitats (Chew 1951, Getz 1961, Churchfield 1990, Matthews et al. 2009). Specifically, shrew presence and abundance have been linked with canopy cover, leaf litter depth and cover, and downed woody debris availability (Cary and Johnson 1995, Lee 1995, Butts and McComb 2000, Hartling and Silva 2004, Greenberg et al. 2007).

Downed woody debris (DWD), particularly coarse woody debris (CWD), debris ≥ 7.62 cm in diameter for a length of at least 0.914 m (Woodall and Monleon 2008), influences shrew presence and abundance. For example, the state of decay of woody debris cover and the amount of log cover influence the presence of shrew species (French 1980, Cromer et al. 2007). Population sizes of *Sorex trowbridgii* and *S. monticolus* in the Pacific Northwest of the United States are positively associated with abundance of coarse woody debris (Carey and Johnson 1995, Butts and McComb 2000), while in the southeastern Coastal Plain of the United States, some species of shrews are positively related to the presence of CWD and others species are not (McCay and Komoroski 2004, Cromer et al. 2007, Moseley et al. 2008, Davis et al. 2010). For example, capture success of southern short-tailed shrews (*Blarina carolinensis*) and southeastern shrews (*S. longirostris*) often are greater in areas with greater volumes of retained downed CWD in southeastern United States pine forests; however, capture success of least shrews (*Cryptotis parva*) may not be associated with DWD (Loeb 1999, Moseley et al. 2008, Davis et al. 2010).

Aside from being an important structural and habitat component in forest ecosystems, wood from forests (i.e., woody biomass) is a major feedstock of bioenergy worldwide and demand for woody biomass is increasing (Perlack et al. 2005, Hillring 2006, Mantau et al. 2010). The southeastern United States is the largest exporter of wood pellets and is experiencing perhaps the most rapid growth of forest bioenergy production facilities in the world (Mendell and Lang 2012, Evans et al. 2013, Goh et al. 2013). Domestic and foreign policies that drive bioenergy production may increase the demand for forest bioenergy, which could involve levels of woody biomass extraction that threaten the long-term functioning and sustainability of forests in the southeastern United States (Evans et al. 2013). Further, demand is expected to continue to increase as renewable energy mandates are implemented in the European Union, which imports wood pellets produced from southeastern United States forests (Goh et al. 2013). In fact, pellet production may increase by 87% in 2014 over the 2012 production level in the United States alone (Forisk Consulting 2013). Woody biomass also is a feedstock of second generation biofuels, and the United States Department of Agriculture (USDA) predicts that ≈ 50% of second generation biofuels needed to meet United States biofuel mandates will originate from the Southeast region by 2020 (USDA 2010).

Concerns exist about the potential ecological consequences of harvesting woody biomass, particularly in regards to decreasing volumes of DWD, and have prompted government and private organizations to develop Biomass Harvesting Guidelines (BHGs; e.g., MFRC 2007, Röser et al 2008, PADCNR 2008, KYDOF 2011, Perschel et al. 2012). Woody biomass harvests glean forest harvest residues, including treetops, limbs, slash, and felled small trees, during traditional timber harvests. Although woody biomass has been harvested for energy production for decades (Stuart 1981, Van Hook et al. 1982, Watson et al. 1986, Puttock 1987), current levels of extraction have resulted in decreases of DWD by ≤ 81% in southeastern United States pine plantations (Fritts et al. in review). Yet, DWD is an integral part of the ecosystem and provides habitat for a variety of wildlife (Harmon et al. 1986, Lattimore et al. 2009, Evans and Kelty 2010, Janowiak and Webster 2010, Riffell et al. 2011). Therefore, BHGs typically focus on a target volume of woody biomass to be retained on the forest floor as DWD to maintain biological diversity and site productivity and have been created under the idea that "more" DWD is better than "less" (Harmon and Hua 1991, Ranius and Fahrig 2006). BHGs often recommend retaining volumes of both CWD and fine woody debris (FWD), debris smaller than CWD, to meet sustainability goals. However, suggested volumes, sizes, and spatial arrangements of DWD vary among BHG documents. Thus, research that determines the response of wildlife, particularly species that are associated with DWD such as shrews, to woody biomass harvesting and experimental BHGs is needed.

Further, shrews may be sensitive to silvicultural practices that decrease canopy cover (Greenberg et al. 2007, Matthews et al. 2009), and woody biomass harvests often occur simultaneously with canopy removal from clearcut harvesting. Forest harvesting can lead to increases in air temperature, more variable moisture regimes, and less litter on the forest floor (Valigura and Messina 1994, Ash 1995, Zheng et al. 2000). DWD retains moisture and provides microhabitats in a range of temperature and moisture regimes with the temperature under and inside of logs often lower than ambient (Graham 1925, Jaeger 1980, Kluber et al. 2009). The high metabolic rate of shrews leads to increased evaporative water loss and potential desiccation (Churchfield 1990, Ochocińska and Taylor 2005). Hence, shrews may be dependent on DWD because they are sensitive to changes in environmental moisture

(Getz 1961). Therefore, retaining woody biomass during clearcut timber harvests may be particularly important for shrews because DWD may mitigate harsh environmental conditions resulting from the clearcut harvest by providing suitable microclimates.

Lack of consensus on the associations between shrews and DWD in the southeastern United States coupled with an absence of research on operational-scale research on woody biomass harvesting warrant investigation of shrew response to variations in DWD retention following woody biomass harvests. Our objective was to use an operational-scale experiment to determine if shrew relative abundance varied among woody biomass harvesting treatments. Additionally, we used the operational-scale experiment coupled with fine-scale measures of DWD and vegetation to: 1) determine if shrew relative abundance is related to DWD volume, vertical vegetation structure, and groundcover; 2) given an effect is observed, determine if BHGs mitigate changes in shrew relative abundance following biomass harvest and; 3) identify the threshold volume of DWD that best sustains shrew populations. Because DWD decays relatively quickly in the Southeast (Moorman et al. 1999), the first 3-4 years post-harvest provide the best time to detect shrew responses.

METHODS

Study Area and Design

Our study was conducted on eight sites (i.e., replicate clearcuts) in the Coastal Plain Physiographic Region of the southeastern United States: four in Beaufort County, North Carolina; three in Glynn County, Georgia; and one in Chatham County, Georgia, USA. All study sites were intensively managed loblolly pine (*Pinus taeda*) plantations. North Carolina sites were managed for sawtimber production, had two commercial thinning entries before the final harvest, and were 32-39 years old at the time of clearcut harvest. Georgia sites were managed for chip-and-saw and pulpwood production and were between 25-33 years of age at the time of final harvest. Three Georgia sites had one commercial thinning entry and one site had two commercial thinning entries before clearcut harvest. North Carolina soil series were predominately loam and silt loam. Georgia soil series were predominantly loam, clay loam, and fine sandy loam.

Clearcut harvests with treatment implementations were completed in 2010-2011. North Carolina sites (mean ± SE) were 70.5 ± 6.1 ha and Georgia sites were 64.4 ± 3.1 ha. Each site was divided into six randomly-assigned treatments that were 11.7 ± 0.5 ha (range = 8.4–16.3 ha) in North Carolina and 10.7 ± 0.4 ha (range = 7.6–14.3 ha) in Georgia: 1) clearcut with a traditional woody biomass harvest and no BHGs implemented (NOBHG); 2) clearcut with 15% retention with woody biomass evenly dispersed throughout the treatment unit (15DISP); 3) clearcut with 15% retention with woody biomass in large piles throughout the treatment unit (15CLUS); 4) clearcut with 30% retention with woody biomass evenly dispersed throughout the treatment unit (30DISP); 5) clearcut with 30% retention with woody biomass in large piles throughout the treatment unit (30CLUS); and 6) clearcut with no woody biomass harvest (i.e., clearcut only; NOBIOHARV), which served as a reference. Retention targets focused on hardwoods not merchantable as roundwood from the retention areas in the four BHG treatments or from the entire treatment area in NOBIOHARV. For NOBHG treatments, we instructed loggers to follow normal operating procedures for a typical woody biomass harvest. For the NOBIOHARV treatments, we asked loggers to leave all woody biomass not harvested as roundwood on the ground as DWD, although sawtimber and pulpwood were harvested as normal. The four BHG treatments were implemented using retention areas that represented either 15% or 30% of the total treatment plot area using ArcGIS (ESRI, Redlands, California, USA). We located retention areas using a handheld Garmin Rino GPS (Olathe, Kansas, USA) and flagged boundaries. During clearcut and treatment implementation, all standing pines merchantable as roundwood were cut and brought to the logging deck with a skidder and grapple, and most hardwoods from the retention areas were left intact and redistributed throughout the treatment unit with a skidder. Each retention area was clearcut after the other 85% or 70% of the non-retention treatment area was harvested. In the non-retention areas, woody biomass was recovered and chipped at the logging decks during harvest. In the retention areas, loggers retained woody biomass to be either spread evenly throughout the dispersed (DISP) treatments or placed in grapple-sized piles randomly throughout the clustered (CLUS) treatments.

North Carolina sites were V-sheared in 2011 to reduce stumps and fell any remaining stems following harvest and before shrew sampling. Sites were bedded to provide a raised growing surface for seedlings and planted fall-winter 2011-2012 (i.e., after one year of shrew sampling) at a density of ≈ 1,100 trees/ha. Shearing moved DWD into the 3-m space between pine beds, and rearranged DWD into long, linear rows. North Carolina sites received two post-harvest herbicide applications with Chopper© (BASF, Raleigh, North

Carolina, USA) with Red River Supreme surfactant–a broadcast application year one post-harvest and a banded application year two post-harvest.

Two of the Georgia sites were clearcut with treatments implemented in winter 2010-2011 and were available for sampling in 2011. The remaining two Georgia sites were clearcut in summer and fall 2011 and were available for sampling in 2012. Following harvest of the Georgia sites, most debris was windrowed or spot piled in relatively few large piles in each treatment unit; few individual stems and no smaller DWD piles were distributed throughout the treatment units. Two of the Georgia sites were double-bedded in summer 2011 and one site was double-bedded in fall 2011 and all three were planted in winter 2012 at a density of ≈ 1,495 trees/ha. The same three sites received a banded herbicide application of Arsenal© (BASF, Raleigh, North Carolina, USA) and Sulfometuron in spring 2012. The fourth Georgia site was bedded, planted at a density of ≈ 726 trees/ha, and received a broadcast herbicide treatment with Chopper© (BASF, Raleigh, North Carolina, USA) in spring 2012.

Shrew Sampling

We sampled shrews from mid-April to early-August in 2011-2013. In North Carolina, we sampled for one year after harvesting and V-shearing (2011) and for two years after bedding and planting (2012 and 2013). In 2012 and 2013, we sampled all eight sites, but we were unable to sample two of the Georgia sites in 2011 because they were not yet harvested. Therefore, in Georgia, we sampled two sites one year after harvesting and windrowing and spot piling (2011) and all four sites for two years after bedding and planting (2012 and 2013). We established three 'Y'-shaped drift fence arrays with 7.6-m arms in each treatment unit.

Drift fences were > 30 m apart and were constructed from silt-fence material with a 19-L plastic bucket buried flush with the ground at each end. The center of one randomly selected drift fence in each treatment unit had a three-sided funnel trap modified from Burgdorf et al. (2005), and the other two drift fences in each treatment unit had 19-L buckets buried flush with the ground. We drilled three small holes in the bottom of each bucket for water drainage and placed sponges in each trap that were wetted daily. We used bucket lids raised ≈ 25 cm to provide shade. We checked open traps daily for ten consecutive days in 2011 and five consecutive days in 2012 and 2013. Traps remained closed for two to three consecutive days between sampling periods. We sampled two to three replicate clearcuts in North Carolina and one to three replicate clearcuts in Georgia simultaneously. We identified each captured shrew to species and measured body length, tail length, and weight. Live shrews were released ≈ 10 m from the drift fence array. All sampling and handling procedures were approved by the North Carolina State University Institutional Animal Care and Use Committee (11-022-O), the Georgia Department of Natural Resources (scientific collection permit number 29-WJH-13-156), and the North Carolina Wildlife Resources Commission (scientific collection permit number 11SC00534).

Quantifying DWD

In 2011, we measured the volume of DWD in each treatment unit using a line-intersect sampling (LIS) technique (Van Wagner 1968). We established 7.32-m transects radiating from the plot center point at 0°, 120°, and 240° azimuths for sampling CWD and 3.13-m transects radiating from the plot center for sampling FWD. Because few piles fell within our

LIS plots, we also used a visual encounter method to census piled DWD. In North Carolina, we visually located and measured the length, width, height, and packing ratio (i.e., density of wood versus air or soil; 0–100%) of total woody debris of all hardwood and large pine piles. Most large hardwoods were retained for BHG implementation while large pine trees were harvested as roundwood (Fritts et al. in review). We defined hardwood piles as ≥ two hardwood trees retained on the forest floor as a result of retention treatments (i.e., retention of 15%, 30% or all woody biomass), whereas we defined large pine piles as any pine residue pile > 1.6 m in height or > 3.8 m in length and estimated as having < 50% soil. Pines were stripped of their limbs at harvest, and pine piles were created when pine limbs were pushed together during shearing. In Georgia, we visually located and measured the width, height, and packing ratio of all windrows and spot piles. Windrows often were the length of the entire treatment unit, so we measured the lengths using aerial photographs in ArcGIS. Georgia windrows and spot piles contained primarily pine debris. For both states, we summed the volume of piled DWD estimated from the visual encounter method and the volume of scattered DWD estimated using the LIS method to generate total DWD volume (m^3/ha) for each treatment unit. Each site contained two to three logging decks for harvesting the six treatment units. Large butts of trees from the entire clearcut (i.e., all treatment units) were retained at the logging deck, so we did not include the deck debris in the final debris volume estimates. We recorded pile locations using a Garmin Rino GPS (Olathe, Kansas, USA) and used ArcGIS to ensure no piles were double-counted. We measured the distance of each drift fence array to the nearest DWD pile. In North Carolina, where DWD piles were distributed throughout the site instead of in windrows or spot piles, we used ArcGIS to estimate the volume of piled DWD within 50 m of each drift fence array. Additionally, we measured the distance of each drift fence array to the nearest parallel drainage ditch, which were present to lower the water table and improve pine growth. In Georgia, we measured the distance of each drift fence array to the nearest unharvested wetland depression.

Quantifying Vegetation

In 2013, when vegetation cover was greatest, we estimated vertical vegetation structure and groundcover along three 10-m transects at each drift fence array. We established transects starting at the array center and radiating outwards directly between two drift fence arms. We sampled vegetation at ten points with 1-m increments on each transect. We used the average number of times vegetation touched a 2-m tall, 4.8-cm diameter pole at each point across the 30 points as an index of vertical vegetative structure at each fence (Moorman and Guynn 2001). We recorded all groundcover types (bare ground, CWD, FWD, herbaceous, litter, and woody) that touched the bottom of the pole at each point. Then, we calculated the % cover of each ground cover type at each drift fence by dividing the number of points with each ground cover type by 30. Groundcover could be >100% because more than one vegetation type may have been present at one point.

Statistical Analyses

We calculated the number of captures per 100 trap nights (hereafter, relative abundance) of all shrew species combined (i.e., total shrews) and each individual shrew species for each treatment unit by dividing the number of shrews captured by the number of traps (i.e., buckets and funnel traps) open in the treatment unit during the entire sampling season and multiplying by 100. We tested for differences in total shrew relative abundance and the relative abundance of individual shrew species among treatments in each state and each year using randomized complete block design analysis of variance (ANOVA) with relative abundance as the dependent variable, replicated as the blocking factor, and treatment as a class predictor. We used Tukey's Studentized Range criteria to separate treatment means when models were significant at the $\alpha = 0.05$ level. We conducted all analyses using statistical software program R (version 3.0.2; R Core Team 2012).

Additionally, to determine fine-scale shrew population response to woody biomass harvesting, we identified predictors of total number of shrew captures and number of captures of each shrew species for which captures were > 100 in one year in one state. We analyzed shrew captures at each drift fence array as the experimental unit to determine how DWD, vertical vegetation structure, and ground cover at this scale affected shrew capture success. We used separate models for each state, year, and species. We used generalized linear models with number of shrew captures per drift fence (i.e., combined shrew captures and captures of each individual species) as each response variable and volume of DWD in the treatment unit, distance of drift fence array to nearest debris pile, distance of drift fence array to the nearest ditch (North Carolina) or unharvested wetland (Georgia), volume of piled DWD within 50 m of the drift fence array (North Carolina only), and effort as independent explanatory variables. In 2013, we included vertical vegetation structure and % cover of bare ground, CWD, FWD, litter, herbaceous vegetation, and woody vegetation as independent explanatory variables (Table 1). We assessed collinearity using variation inflation factors and dropped one covariate if the factor was > 3.0. We tested each response variable for normality using the Shapiro-Wilks test. When response variables were not normally distributed, we applied a Poisson GLM. We assumed overdispersion when the residual deviance divided by the residual degrees of freedom was > 1 and corrected the standard errors using a quasi-GLM model when overdispersion was detected. We selected the best models at predicting shrew captures by dropping one explanatory variable, in turn, and each time applying an analysis of deviance test between the previous and current models (Zuur et al. 2009). We validated each selected model by plotting the residuals over the predicted values to assess homogeneity, making a histogram of the residuals to verify normality, plotting the residuals against each explanatory variable to verify independence, and using Cook's distance to assess the model for influential observations (Cook 1979, Fox 2002, Zuur et al. 2009). We set $\alpha = 0.05$.

RESULTS

In North Carolina, piled hardwood comprised the majority of DWD in all treatments except NOBHG (Table 2). In Georgia, piled pine comprised the majority of DWD in all treatments (Table 3). In 2013, treatments were similar in vertical vegetation structure and groundcover in North Carolina (Table 2) and Georgia (Table 3). Herbaceous vegetation comprised the majority of groundcover followed by bare ground in both states.

We had 1,252 shrew captures in 43,350 trap nights in three years in both states. Trap-related shrew mortality was ≈ 56%. We captured three species: 606 *Cryptotis parva*, 605 *Sorex longirostris*, and 41 *Blarina carolinensis* (Table 4). We did not compare the relative abundance of shrews among treatments in North Carolina in 2011 or *B. carolinensis* in any year in North Carolina because of low capture success (Table 5). In North Carolina in 2012, the relative abundances of all shrew species combined ($F_{5,63}$ = 1.52, P = 0.20) and *C. parva* ($F_{5,63}$ = 0.67, P = 0.64) were similar among treatments. The relative abundance of *S. longirostris* differed among treatments in North Carolina in 2012 ($F_{5,63}$ = 2.49, P = 0.04) with nearly three times greater relative abundance in NOBIOHARV than 15DISP (Table 5). In North Carolina in 2013, the relative abundances of all shrew species combined ($F_{5,63}$ = 0.93, P = 0.46), *C. parva* ($F_{5,63}$ = 1.45, P = 0.22), and *S. longirostris* ($F_{5,63}$ = 0.90, P = 0.49) were similar among treatments (Table 5).

In Georgia, no shrews were captured in 2011, only 2 shrews were captured in 2012, and few *B. carolinensis* were captured in 2013 (Table 4); therefore, we investigated shrew response to treatments in Georgia only for 2013 captures. The relative abundance of all shrew species combined ($F_{5,63}$ = 2.09, P = 0.08), *C. parva* ($F_{5,63}$ = 2.03, P = 0.09), and *S. longirostris* ($F_{5,63}$ = 1.99, P = 0.09) were similar among treatments in 2013 in Georgia (Table 6).

For North Carolina captures, we determined if total shrew captures in 2012 and 2013, *C. parva* captures n 2013, and *S. longirostris* captures in 2012 and 2013 were related to DWD volume, vertical vegetation structure, and groundcover at the drift fence array (Table 4). No covariates were collinear. No response variables were normally distributed and all were overdispersed; therefore, we used a Poisson distribution and quasi-GLM model for all data sets. We identified three observations as outliers and omitted the observations from the linear regressions. In 2012, *S. longirostris* capture success was positively related to effort ($P = 0.01$). In 2013, total shrew capture success was positively related to bare ground ($\beta = 0.22$, $t = 0.07$, $P < 0.01$) and negatively related to volume of DWD within 50 m of the drift fence array ($\beta = -0.18$, $t = -1.99$, $P = 0.05$; Figure 1), and *C. parva* capture success was positively related to bare ground ($\beta = 0.21$, $t = 0.09$, $P = 0.02$; Figure 2). No covariates were predictors of total shrew capture success in 2012 or *S. longirostris* capture success in 2013.

For Georgia captures, we determined if total shrew capture success in 2013 and *C. parva* capture success in 2013 was related to DWD volume, vertical vegetation structure, and groundcover at the drift fence array (Table 4). No covariates were collinear. Response variables were not normally distributed and all were overdispersed; therefore, we used a Poisson distribution and quasi-GLM model for all data sets. We identified two observations as outliers and omitted the observations from the linear regressions. In 2013, total shrew capture success was negatively related to herbaceous groundcover ($\beta = -0.93$, $t = -2.87$, $P < 0.01$) and leaf litter groundcover ($\beta = -3.45$, $t = -2.45$, $P = 0.02$) and positively related to woody vegetation groundcover ($\beta = 2.13$, $t = 3.76$, $P < 0.01$), vertical vegetation structure ($\beta = 0.18$, $t = 2.20$, $P = 0.03$), distance to nearest unharvested wet depression ($\beta = -0.002$, $t = -2.82$, $P < 0.01$), and effort ($\beta = 0.01$, $t = 5.48$, $P < 0.01$; Figure 3). *C. parva* capture success was positively related to woody vegetation groundcover ($\beta = 2.04$, $t = 2.91$, $P = 0.01$), herbaceous groundcover (β = -1.03, t = 2.70, P = 0.01), and distance to nearest unharvested wet depression (β = -0.003, t = -2.82, P < 0.01; Figure 4).

DISCUSSION

Our results suggest that shrew populations are more associated with vegetation composition and structure than to DWD availability in southeastern United States pine plantations and that shrews are not sensitive to current levels of woody biomass harvests. Hence, BHGs may not be needed to sustain shrew populations. However, considerable volumes of woody biomass was retained in all treatment units. The minimum volume of DWD retained in a treatment unit was 16.28 m^3/ha (7.81 tons/ha) (Fritts et al. in review), which exceeds by over three-fold the Forest Guild's volume recommendations of 2.24 tons/ha in pine forests of the Piedmont and Coastal Plain physiographic regions of the U.S. (Perschel et al. 2012). Therefore, the current levels of woody biomass harvesting in the region may be above the threshold needed to sustain shrew populations if a threshold exists.

S. longirostris may be more sensitive to woody biomass harvests than *B. carolinenesis* and *C. parva*, but our results add to the inconsistencies regarding the associations of *S. longirostris* with DWD and vegetation (McCay and Komoroski 2004, Moseley et al. 2008, Davis et al. 2010). Although some studies have revealed positive relationships between *S. longirostris* and decayed wood (French 1980, Cromer et al. 2007, Davis et al. 2010), others have found no relationship (McCay and Komoroski 2004). Further, a study by Moseley et al. (2008) resulted in no differences in the relative abundance of *S. longirostris* among treatments with experimental DWD manipulations during two years of the study, but the relative abundance was lower in DWD removal treatments for one year of the study. Our study indicated a treatment effect on the relative abundance of *S. longirostris*, but the effect was weak and only in one state during one year. *S. longirostris* was absent in Georgia during the first two years of our study, but appeared as the vegetation became better established in 2013. Similarly, other studies have demonstrated associations of *S. longirostris* with early to mid-successional disturbed woodlands with a dense understory and leaf litter (Rose et al. 1990, Erdle and Pagels 1995).

Shrew relative abundance increased each year as vegetation became well-established. Although it is possible that increases in shrew relative abundance were because DWD decayed throughout the study, our results suggest that shrews were responding to the increase in vegetative structure. Our results are similar to two previous studies that documented no correlation between *C. parva* capture success and DWD (Moseley 2008, Davis 2010). In fact, *C. parva* is associated with early successional vegetation, particularly fields with dense grasses and forbs (Hamilton 1934, Davis and Joeris 1945, Schmidly 1983, Bellows et al. 2001), and it has been suggested that competitive exclusion of *C. parva* from forested stands by *B. carolinensis* has forced *C. parva* to inhabit and adapt to open habitats at the forest edge, which have lower DWD volumes (Davis et al. 2010). Further, previous research has demonstrated that *B. carolinensis* is more affiliated with vegetation. Specifically *B. carolinensis* has a greater positive association with canopy height and biomass of woody vines than DWD in the southeastern Coastal Plain of the United States (Wolfe and Lohoefener 1983, Mengak and Guynn 2003)

Shrews may have been more reliant on groundcover in Georgia than in North Carolina because site preparation in Georgia resulted in relatively few and large DWD windrows and spot piles and vaster areas of bare ground between windrows, while DWD in North Carolina was more evenly distributed throughout the treatment units. Although total shrew capture success and the capture success of *C. parva* increased with bare ground cover in North Carolina, vast areas of bare ground were absent. Instead, small patches of loose bare soil were remnants of prior bedding and planting activities. Additionally, Georgia sites had greater mean temperatures than North Carolina sites, which may have further increased desiccation risk, resulting in a greater association of shrews with vegetative cover.

Shrews in the southeastern Coastal Plain may not be as dependent on DWD as in other regions of the United States. Although shrew abundance has been linked to DWD presence in the southeastern United States and elsewhere (Carey and Johnson 1995, Loeb 1996, Maidens et al. 1998, McCay et al. 1998, Butts and McComb 2000), the only relationship we identified between the capture success of shrews and DWD volume was negative. Specifically, some studies have demonstrated positive relationships between DWD volume and shrews in the Pacific Northwest and central Appalachians (McComb and Rumsey 1982, Carey and Johnson 1995, Ford et al. 1997), but research conducted in the southern Appalachians limited shrew response to reductions of DWD (Matthews et al. 2009, Raybuck et al. 2012). Further, studies in the southeastern U.S. Coastal Plain have elicited various responses of *B. carolinensis, C. parva,* and *S. longirostris* to experimental DWD removal (Loeb 1999, McCay and Komoroski 2004, Moseley et al. 2008, Davis et al. 2010). Coastal

Plain forests historically were influenced by frequent fires; therefore, ground-dwelling wildlife in the region evolved tolerance of these frequent disturbances and the environmental conditions they promoted (Russell et al. 2004). Further, DWD decays rapidly in the southeastern United States (Moorman et al. 1999). Therefore, DWD volumes may have remained relatively lower than in other regions. For example, CWD volumes in unmanaged and managed pine forests in the southeastern United States have been estimated to be ≈ 18 m^3/ha and six m^3/ha, respectively (McMinn and Hardt 1996), while CWD volumes in lodgepole pine (*Pinus contorta*) stands without recorded disturbances in the northwestern United States has been estimated at 77 m^3/ha (Herrero et al. 2014).

BHGs may not be necessary to sustain shrew populations at current levels of woody biomass harvesting. Current harvest technologies or market inefficiencies may prevent complete harvest of logging residues. However, future technological advances in harvest machinery or rises in wood chip prices could result in increases in harvested debris, thereby leading to lower levels of retained DWD, reinforcing the need for BHGs to sustain shrew populations. Although we did not determine a minimum threshold of DWD needed to maintain shrew relative abundance at their current levels, the threshold may be lower than the volume of DWD retained on our study sites.

ACKNOWLEDGMENTS

Funding was provided by the U.S. Department of Agriculture National Institute of Food and Agriculture Managed Ecosystems Program, the Department of Interior Southeast Climate Science Center, the Biofuels Center of North Carolina, the North Carolina Bioenergy

Initiative, the National Council for Air and Stream Improvement, Weyerhaeuser Company, Plum Creek, and Georgia Pacific. Special thanks to M. Adams, J. Bowser, S. Bruno, J. Eells, E. Evans, J. Kiser, J. Kraus, B. Moore, K. Moore, J. O'Connor, R. Skinner, N. Weaver for field data collection. We thank M. Elfelt, R. Evans, B. Fritts, C. Fritts, J. Fritts, B. Hardin, M. Lashley, A. Prince, J. Swingen, P. Taillie, J. Zajdel and countless other volunteers for installing drift fences and building funnel traps.

LITERATURE CITED

Ash, A. N. 1995. Effects of clear-cutting on litter parameters in the Southern Blue Ridge Mountains. Castanea 60: 89-97.

Bellows, A. S., J. F. Pagels, J. C. Mitchell. 2011. Macrohabitats and microhabitat affinities of small mammals in a fragmented landscape on the Upper Coastal Plain of Virginia. American Midland Naturalist 146: 345-360.

Burgdorf, S. J., D. C. Rudolph, R. N. Conner, D. Saenz, and R. R. Schaefer. 2005. A successful trap design for capturing large terrestrial snakes. Herpetological Review 36:421-424.

Butts, S. R., W. C. McComb. 2000. Associations of forest-floor vertebrates with coarse woody debris in managed forests of western Oregon. Journal of Wildlife Management 64:95-104.

Carey, A. B., C. A. Harrington. 2001. Small mammals in young forests: implications for management for sustainability. Forest Ecology and Management 154: 289-309.

Carey, A. B., M. L. Johnson. 1995. Small mammals in managed, naturally young, and old-growth forests. Ecological Applications 5: 336-352.

Chew, R. M. 1951. The water exchanges of some small mammals. Ecological Monographs 21: 215-225.

Churchfield, S. 1990. The Natural History of Shrews. Cornell University Press. Ithaca, New York, USA.

Cook, R. D. 1979. Influential Observations in Linear Regression. Journal of the American Statistical Association 74: 169–174.

Cromer, R. B., C. A. Gresham, M. Goddard, J. D. Lanham, H. G. Hanlin. 2007. Association between two bottomland hardwood forest shrew species and hurricane-generated woody debris. Southeastern Naturalist 6: 235-246.

Davis, J. C., S. B. Castleberry, J. C. Kilgo. 2010. Influence of coarse woody debris on the soricid community in southeastern coastal plain pine stands. Journal of Mammalogy 91: 993-999.

Davis, W. B., L. Joeris. 1945. Notes on the life history of the little short-tailed shrew. Journal of Mammalogy 25: 370-403.

Erdle, S. Y., J. F. Pagels. 1995. Observations on *Sorex longirostris* (Mammalia: Soricidae) and associates in eastern portions of the historical Great Dismal Swamp. Banisteria 6: 17-23.

Evans, A. M., M. J. Kelty. 2010. Ecology of dead wood in the Northeast. Forest Guild. Santa Fe, New Mexico, USA.

Evans, J. M., R. J. Fletcher, Jr., J. R. R. Alavalapati, A. L. Smith, D. Geller, P. Lal, D. Vasudev, M. Acevedo, F. Calabria, T. Upadhyay. 2013. Forestry bioenergy in the Southeast

United State: implications for wildlife habitat and biodiversity. National Wildlife Federation, Merrifield, Virginia, USA.

Ford, W. M., J. Laerm, K. G. Barker. 1997. Soricid response to forest stand age in southern Appalachian cove hardwood communities. Forest Ecology and Management 91: 175-181.

Ford, W. M., J. L Rodrigue. 2001. Soricid abundance in partial overstory removal harvests and riparian areas in an industrial forest landscape of the central Appalachians. Forest Ecology and Management 152: 159-168.

Forisk Consulting. 2013. Forisk News, November 8, 2013. Athens, Georgia, USA.

Fox, J. 2002. An R and S-Plus companion to applied regression. Sage Publications. Thousand Oaks, California, USA.

French, T. W. 1980. Natural history of the southeastern shrew, *Sorex longirostris* Bachman. American Midland Naturalist 104: 13-31.

Fritts, S. F., C. E. Moorman, D. W. Hazel, and B. D. Jackson. in review. Biomass harvesting guidelines affect downed woody debris retention.

Getz, L. L. 1961. Factors influencing the local distribution of shrews. American Midland Naturalist 66: 67-88.

Goh, C. S., M. Junginger, M. Cocchi, D. Marchal, D. Thran, C. Hannig, J. Heinimo, L. Nikolaisen, P. P. Schouwenberg, D. Bradley, R. Hess, J. Jacobson, L. Ovard, M. Deutmeyer.

2013. Wood pellet market and trade: A global perspective. Biofuels, Bioproducts, and Biorefining 7: 24-42.

Graham, S. A. 1925. The felled tree trunk as an ecological unit. Ecology 6:397-416.

Greenberg, C. H., S. Miller, T. A. Waldrop. 2007. Short-term response of shrews to prescribed fire and mechanical fuel reduction in a Southern Appalachian upland hardwood forest. Forest Ecology and Management 243: 231-236.

Hamilton, W. J., Jr. 1934. Habits of Cryptotis parva in New York. Journal of Mammalogy 15: 154-155.

Hamilton, J. W., Jr. 1941. The food of small forest mammals in eastern United States. Journal of Mammalogy 22: 250-263.

Harmon, M. E., J. F. Franklin, F. J. Swanson, P. Sollins, S. V. Gregory, J. D. Lattin, N. H. Anderson, S. P. Cline, N. G. Aumen, J. R. Sedell, G. W. Lienkaemper, K. Cromack Jr., K. W. Cummins. 1986. Ecology of coarse woody debris in temperate ecosystems. Advanced Ecological Research 15:133-302.

Harmon, M. E., C. Hua. 1991. Coarse Woody Debris Dynamics in Two Old-Growth Ecosystems. Bioscience 41:604-610.

Hartling, L., M. Silva. 2004. Abundance and species richness of shrews within forested habitats on Prince Edward Island. American Midland Naturalist 151: 399-407.

Herrero, C., O. Krankina, V. J. Monleon, F. Bravo. 2014. Amount and distribution of coarse woody debris in pine ecosystems of northwestern Spain, Russia, and the United States. Journal of Biogeosciences and Forestry 7: 53-60.

Hillring, B. 2006. World trade in forest products and wood fuel. Biomass and Bioenergy 30: 815-825.

Jaeger, R. G. 1980. Microhabitats of a terrestrial forest salamander. Copeia 2: 265-268.

Janowiak, M. K., C. R. Webster. 2010. Promoting Ecological Sustainability in Woody Biomass Harvesting. Journal of Forestry 108:16-23.

Kentucky Division of Forestry (KY DOF). 2011. Recommendations for the harvesting of woody biomass. Kentucky Division of Forestry. Frankfurt, USA.

Kluber, M. R., D. H. Olson, K. J. Puettmann. 2009. Downed wood microclimates and their potential impact on Plethodontid salamander habitat in the Oregon coast range. Northwest Science 83: 25-34.

Lattimore, B., C. T. Smith, B. D. Titus, I. Stupak, G. Egnell. 2009. Environmental factors in woodfuel production: Opportunities, risks, and criteria and indicators for sustainable practices. Biomass and Bioenergy 33:1321-1342.

Lee, S. D. 1995. Comparison of population characteristics of three species of shrews and the shrew-mold in habitats with different amounts of coarse woody debris. Acta Theriologica 40: 415-424.

Loeb, S. C. 1996. The role of coarse woody debris in the ecology of southeastern mammals. Pages 108-118 In: McMinn J. W. and D. A. Crossley Jr. (editors). Biodiversity and coarse woody debris in southern forests: proceedings of the workshop on coarse woody debris in southern forests: effects on biodiversity. United State Department of Agriculture Forest Service. General Technical Report SE-94, Southern Research Station, Asheville, North Carolina, USA.

Loeb, S. C. 1999. Responses of small mammals to coarse woody debris in a southeastern pine forest. Journal of Mammalogy 80: 460-471.

Maidens, D. A., M. A. Menzel, and J. Laerm. 1998. Notes on the effect of size and level of decay of coarse woody debris on relative abundance of shrews and salamanders in the southern Appalachian Mountains. Georgia Journal of Science 56: 226-233.

Mantau U., U. Saal, F. Steierer, H. Verkerk, J. Oldenburger, N. Leek, K. Prins. 2010. EUwood – Real potential for changes in growth and use of EU forests. Final report. Hamburg, Germany.

Matthews, C. E., C. E. Moorman, C. H. Greenbery, T. A. Waldrop. 2009. Response of soricid populations to repeated fire and fuel reduction treatments in the southern Appalachian Mountains. Forest Ecology and Management 257: 1939-1944.

McCay, T. S., J. Laerm, M. A. Menzel, W. M. Ford. 1998. Methods used to survey shrews (insectivore: Soricidae) and the importance of forest-floor structure. Brimleyana 25: 110-119.

McCay, T. S., M. J. Komoroski. 2004. Demographic responses of shrews to removal of coarse woody debris in a managed pine forest. Forest Ecology and Management 189: 387-395.

McComb, W. C., R. L. Rumsey. 1982. Response of small mammals to forest clearings created by herbicides in the central Appalachians. Brimleyana 8: 121-134.

McMinn, J. W., R. A. Hardt. 1996. Accumulation of coarse woody debris in southern forests. Pages 1-9 In: J. W. McMinn, D.A. Crossley, Jr. (editors). Biodiversity of Coarse Woody Debris in Southern Forests. Proceedings of the Workshop on Coarse Woody Debris sin Southern Forests: Effects on Biodiversity. United States Department of Agriculture Forest Service General Technical Report. SE-94.

Mendell, B. C., A. H. Lang. 2012. Wood for bioenergy: forests as a resource for biomass and biofuels. Forest History Society.

Mengak, M. T., D. C. Guynn, Jr. 2003. Small mammal microhabitat use on young loblolly pine regeneration areas. Forest Ecology and Management 173: 309-317.

Minnesota Forest Resources Council (MFRC). 2007. Biomass Harvest Guidelines. Minnesota Forest Resources Council. St. Paul, USA.

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

Moorman, C. E., D. C. Guynn Jr. 2001. Effects of group-selection opening size on breeding bird habitat use in a bottomland forest. Ecological Applications 11:1680-1691.

Moseley, K. R., A. K. Owens, S. B. Castleberry, W. M. Ford, J. C. Kilgo, T. S. McCay. 2008. Soricid response to coarse woody debris manipulations in Coastal Plain loblolly pine forests. Forest Ecology and Management 255: 2306-2311.

Ochocińska, D., J. E. Taylor. 2005. Living at the physiological limits: field and maximum metabolic rates of the common shrew (*Sorex araneus*). Physiological and Biochemical Zoology 78: 808-818.

Pennsylvania Department of Conservation and Natural Resources (PADCNR). 2008. Harvesting woody biomass for energy. Pennsylvania Department of Conservation and Natural Resources. Harrisburg, USA.

Perlack, R. D., L. L. Wright, A. F. Turhollow, R. L. Graham, B. J. Stokes, D. C. Erback.

2005. Biomass as feedstock for a bioenergy and bioproducts industry:the technical feasibility of a billion-ton annual supply. United States Department of Energy. Oak Ridge National Laboratory. Oak Ridge, Tennessee, USA.

Perschel, B, A. Evans, M. DeBonis. 2012. Forest biomass retention and harvesting guidelines for the Southeast. Forest Guild Southeast Biomass Working Group. Forest Guild. Sante Fe, New Mexico, USA.

Puttock, G. D. 1987. The economics of collecting and processing whole-tree chips and logging residues for energy. Forest Products Journal 37: 15-20.

R Core Team. 2012. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Ranius, T., L. Fahrig. 2006. Targets for maintenance of dead wood for biodiversity conservation based on extinction thresholds. Scandinavian Journal of Forest Research 21: 201-208.

Raybuck, A. L., C. E. Moorman, C. H. Greenberg, C. S. DePerno, K. Gross, D. M. Simon, G. S. Warburton. 2012. Short-term response of small mammals following oak regeneration silviculture treatments. Forest Ecology and Management 274: 10-16.

Riffell, S., J. Verschuyl, D. Miller, T. B. Wigley. 2011. Biofuel harvests, coarse woody debris, and biodiversity – A meta-analysis. Forest Ecology and Management 21:878-887.

Rose, R. K., R. K. Everton, J. F. Standavich, J. W. Walke. 1990. Small mammals in the Great Dismal Swamp of Virginia and North Carolina. Brimleyana 16: 87-101.

Röser D., A. Asikainen, K. Raulund-Rasmussen, I. Stupak, editors. 2008. Sustainable use of forest biomass for energy. A synthesis with focus on the Nordic and Baltic Region. Managing forest ecosystems. Volume 12. Springer, Dordrecht, the Netherlands.

Russell, K. B., T. B. Wigley, W. M. Bauhman, H. H. Hanlin, W. M. Ford. 2004. Responses of southeastern amphibians and reptiles to forest management: a review. Pages 319-334 In: H. M. Rauscher and K. Johnsen (editors). Southern Forest Science: Past, Present, and Future. Southern Research Station, Asheville, North Carolina, USA.

Schmidly, D. J. 1983. Texas mammals east of the Balcones Fault zone. Texas A & M University Press, College Station, USA.

Stuart, W. B., Porter, C. D., Walbridge, T. A., Oderwald, R. G. 1981. Economics of modifying harvesting systems to recover energy wood. Forest Products Journal 31: 37 – 42.

United States Department of Agriculture (USDA). 2010. A USDA regional roadmap to meeting the biofuels goals of the renewable fuels standard by 2022. United States Department of Agriculture, Washington, DC, USA.

Valigura, R. A., M. G. Messina. 1994. Modification of Texas clearcut environments with loblolly pine shelterwoods. Journal of Environmental Management 40:283-295.

Van Hook, R. I., D. W. Johnson, D. C. West, L. K. Mann. 1982. Environmental effects of harvesting forests for energy. Forest Ecology and Management 4: 79-94.

Van Wagner, C. E. 1968. The line intersect method in forest fuel sampling. Forest Science 14:20-26.

Van Zyll de Jong, C. G. 1983. Handbook of Canadian Mammals, Volume I: Marsupials and Insectivores. National Museum of Natural Sciences, Ottawa, Canada.

Watson, W. F., Stokes, B.J., Savelle, I.W. 1986. Comparisons of two methods of harvesting biomass for energy. Forest Products Journal 36: 63-68.

Woodall, C. W., V. J. Monleon. 2008. Sampling protocol, estimation, and analysis procedures for the down woody debris indicator of the FIA program. NRS-GTR-22.United States Department of Agriculture Forest Service. Newtown Square, Pennsylvania, USA.

Wolfe, J. L, R. Lohoefener. 1983. The small mammal fauna of a longleaf-slash pine forest in southern Mississippi. Journal of the Mississippi Academy of Sciences 28: 37-47.

Zheng, D., J. Chen, B. Song, M. Xu, P. Sneed, R. Jensen. 2000. Effects of silvicultural treatments on summer forest microclimate in southeastern Missouri Ozarks. Climate Research 15: 45-59.

Zuur, A. F., E. N. Ieno, N. J. Walker, A. A. Saveliev, G. M. Smith. 1999. Mixed effects models and extensions in ecology with R. Springer. New York, USA.

Table 1. Covariates measured in six woody biomass harvest treatments in clearcut loblolly pine (*Pinus taeda*) stands in Beaufort County, North Carolina (n = 4) and Glynn (n = 3) and Chatham Counties (n = 1), Georgia in 2011-2013. Covariates were used in determining predictors of *Blarina carolinensis, Cryptotis parva*, and *Sorex longirostris* using linear regression.

Covariate	Definition
EFFORT	number of buckets open per night/drift fence
VT	volume (m^3/ha) DWD in the treatment unit
DISTP	distance (m) of drift fence array to debris pile
DISTD	distance (m) of drift fence array to ditch or unharvested wet depression
VDF50	volume (m^3/ha) of piled DWD within 50 m of the drift fence array (NC only)
CWD	% coarse woody debris groundcover (2013 only)
FWD	% fine woody debris groundcover (2013 only)
BG	% bare ground groundcover (2013 only)
WP	% woody groundcover (2013 only)
HE	% herbaceous groundcover (2013 only)
LL	% litter groundcover (2013 only)
VEST	vertical vegetation structure at the drift fence array (2013 only)

Table 2. Mean ± standard error of habitat covariates estimated in six woody biomass harvest treatments in loblolly pine (*Pinus taeda*) plantations in Beaufort County, North Carolina (n = 4) in 2011-2013. Treatments were: 1) no Biomass Harvesting Guidelines (NOBHG); 2) 15% woody biomass retention in piles (15CLUS); 3)15% woody biomass retention distributed evenly throughout the treatment unit (15DISP); 4) 30% woody biomass retention in piles (30CLUS); 5) 30% woody biomass retention distributed evenly throughout the treatment unit (30DISP); 6) and no woody biomass harvesting (NOBIOHARV). Covariates are defined in Table 1. Different letters indicate significantly different means at the α = 0.05 level.

	Treatment							
	NOBHG	15DISP	15CLUS	30DISP	30CLUS	NOBIOHARV	$F_{5, 65}$	P_{trt}
VT	20.65±1.45a	40.80±13.11a	37.76±9.42a	55.75±12.49a	55.17±12.49a	108.20±20.05b	9.83	<0.01
DISTP	40.63±7.38	36.91±11.46	33.24±5.85	24.68±6.23	19.97±4.57	25.14±5.67	1.17	0.34
DISTD	70.48±4.52a	98.70±6.35b	60.65±8.41a	76.39±4.29ab	67.31±6.53a	58.71±7.59a	6.42	<0.01
VDF50	16.73±8.25a	40.01±10.07a	11.65±4.06a	19.86±5.80a	90.28±28.10b	12.26±4.10a	5.51	<0.01
CWD%	1.92±0.96	3.08±0.91	1.82±1.05	4.89±1.67	2.40±1.05	3.56±1.59	0.93	0.47
FWD%	30.33±4.07	29.83±4.93	29.55±4.53	27.44±3.81	21.60±5.36	31.89±3.93	0.75	0.59
BG%	48.33±3.87	50.75±4.47	43.36±5.77	44.33±5.26	40.90±5.05	41.22±6.04	0.83	0.54
WP%	12.33±2.62	15.83±5.54	6.00±3.16	21.89±9.28	13.10±4.89	14.22±5.86	1.42	0.23
HE%	70.58±4.23	61.50±4.41	71.18±5.05	66.33±7.98	58.40±8.46	52.56±5.25	1.63	0.17
LL%	24.08±4.01	23.67±4.70	25.64±6.37	18.56±5.23	14.40±4.97	32.44±4.51	1.57	0.18
VEST	8.83±0.70	6.89±0.59	8.28±1.63	5.56±1.07	7.59±1.22	7.29±0.86	1.4	0.24

Table 3. Mean ± standard error of habitat covariates estimated in six woody biomass harvest treatments in loblolly pine (*Pinus taeda*) plantations in in Glynn County (n = 3) and Chatham County (n = 1), Georgia in 2011-2013. Treatments were: 1) no Biomass Harvesting Guidelines (NOBHG); 2) 15% woody biomass retention in piles (15CLUS); 3)15% woody biomass retention distributed evenly throughout the treatment unit (15DISP); 4) 30% woody biomass retention in piles (30CLUS); 5) 30% woody biomass retention distributed evenly throughout the treatment unit (30DISP); 6) and no woody biomass harvesting (NOBIOHARV). Covariates are defined in Table 1. Different letters indicate significantly different means at the α = 0.05 level.

	Treatment							
	NOBHG	15DISP	15CLUS	30DISP	30CLUS	NOBIOHARV	$F_{5, 65}$	P_{trt}
VT	319.13±40.80a	368.81±41.73ab	296.17±41.16a	299.65±40.45a	359.98±46.69a	373.39±24.85b	2.6	0.03
DISTP	14.23±1.83	17.82±2.98	14.82±1.74	19.94±3.44	17.77±3.35	13.53±2.02	1.18	0.33
CWD%	4.25±1.54	4.33±0.73	5.25±1.01	5.00±1.12	3.75±0.66	3.58±0.72	0.45	0.81
FWD%	8.08±1.94	8.92±2.62	10.08±2.40	7.25±1.70	6.67±1.50	4.42±1.15	0.99	0.43
BG%	54.00±5.34	52.33±5.67	47.00±5.87	50.33±2.81	43.25±3.77	56.33±3.80	1.17	0.33
WP%	15.25±4.52	10.08±4.35	7.42±2.30	7.75±3.07	8.08±2.34	7.75±2.09	0.89	0.49
HE%	106.33±3.97	101.83±11.28	103.67±8.75	108.33±4.56	98.25±6.19	89.42±4.27	0.96	0.45
LL%	4.08±1.12a	2.92±0.97ab	3.92±2.15ab	2.25±1.07a	9.58±2.94b	3.08±1.44a	3.57	<0.01
VEST	5.34±0.19	5.08±0.39	5.55±0.31	6.27±0.71	5.24±0.31	5.67±0.33	1.09	0.37

Table 4. Captures of southern short-tailed shrews (*Blarina carolinensis*), least shrews (*Cryptotis parva*), southeastern shrews (*Sorex longirostris*), and total shrews in six woody biomass harvest treatments in loblolly pine (*Pinus taeda*) plantations in Beaufort County, North Carolina (n = 4) and Glynn County (n = 3) and Chatham County (n = 1), Georgia in 2011-2013. Treatments were: 1) no Biomass Harvesting Guidelines (NOBHG); 2) 15% woody biomass retention in piles (15CLUS); 3)15% woody biomass retention distributed evenly throughout the treatment unit (15DISP); 4) 30% woody biomass retention in piles (30CLUS); 5) 30% woody biomass retention distributed evenly throughout the treatment unit (30DISP); 6) and no woody biomass harvesting (NOBIOHARV).

	North Carolina			Georgia			
	2011	2012	2013	2011	2012	2013	Total
Blarina carolinensis	10	14	12	0	2	3	41
Cryptotis parva	7	58	329	0	0	212	606
Sorex longirostris	33	235	312	0	0	25	605
Total	50	307	653	0	2	240	1252

Table 5. Captures per 100 trap nights (mean ± SE) of *Blarina carolinensis*, *Cryptotis parva*, and *Sorex longirostris* in clearcut upland loblolly pine (*Pinus taeda*) plantations in Beaufort County, North Carolina (n = 4) in 2011-2013. Treatments were: 1) no Biomass Harvesting Guidelines (NOBHG); 2) 15% woody biomass retention in piles (15CLUS); 3)15% woody biomass retention distributed evenly throughout the treatment unit (15DISP); 4) 30% woody biomass retention in piles (30CLUS); 5) 30% woody biomass retention distributed evenly throughout the treatment unit (30DISP); 6) and no woody biomass harvesting (NOBIOHARV). Treatment means were compared using two-way analysis of variance. Different letters indicate significantly different means at the α = 0.05 level. No p-value indicates a sample size too small to compare.

		Treatment						
Species	Year	NOBHGS	15DISP	15CLUS	30DISP	30CLUS	NOBIO HARV	P_{trt}
B. carolinensis	2011	0.10±0.24	0.09±0.22	0.00±0.00	0.22±0.33	0.06±0.21	0.04±0.14	
	2012	0.06±0.21	0.12±0.28	0.11.±0.26	0.24±0.35	0.06±0.20	0.27±0.40	
	2013	0.27±0.09	0.19±0.11	0.00±0.00	0.18±0.10	0.18±0.10	0.27±017	
C. parva	2011	0.00±0.00	0.06±0.21	0.21±0.31	0.04±0.14	0.00±0.00	0.04±0.14	
	2012	0.55±0.68	0.73±0.95	0.65±0.77	0.68±0.73	1.00±1.87	0.35±0.82	0.64
	2013	1.49±0.92	1.76±1.53	1.51±1.23	1.70±0.89	2.33±2.29	1.14±0.98	0.22
S. longirostris	2011	0.28±0.32	0.51±0.68	0.30±0.39	0.36±0.61	0.16±0.43	0.04±0.14	
	2012	2.31±1.83ab	1.18±0.79a	1.91±1.82ab	3.04±1.58ab	1.91±1.17ab	3.22±2.54b	0.04
	2013	1.80±1.29	1.25±0.88	1.60±1.55	2.00±1.30	1.34±1.08	1.36±0.67	0.49
Total	2011	0.037±0.43	0.51±0.45	0.66±0.73	0.63±0.67	0.22±0.45	0.13±0.31	
	2012	2.92±1.70	2.03±1.46	3.44±2.25	3.97±1.80	2.96±1.66	3.84±3.15	0.20
	2013	10.94±2.34	9.21±1.87	9.33±3.44	11.28±2.66	11.21±3.48	7.77±1.25	0.47

Table 6. Captures per 100 trap nights (mean ± SE) of *Blarina carolinensis*, *Cryptotis parva*, and *Sorex longirostris* in clearcut upland loblolly pine (*Pinus taeda*) plantations in Glynn County (n = 3) and Chatham County (n = 1), Georgia in 2013. Treatments were: 1) no Biomass Harvesting Guidelines (NOBHG); 2) 15% woody biomass retention in piles (15CLUS); 3)15% woody biomass retention distributed evenly throughout the treatment unit (15DISP); 4) 30% woody biomass retention in piles (30CLUS); 5) 30% woody biomass retention distributed evenly throughout the treatment unit (30DISP); 6) and no woody biomass harvesting (NOBIOHARV). Treatment means were compared using two-way analysis of variance. Different letters indicate significantly different means at the α = 0.05 level. No p-value indicates a sample size too small to compare.

	Treatment						
Species	NOBHGS	15DISP	15CLUS	30DISP	30CLUS	NOBIO HARV	P_{trt}
B. carolinensis	0.06±0.22	0.06±0.21	0.14±0.48	0.00±0.00	0.00±0.00	0.00±0.00	
C. parva	0.73±95	0.30±0.48	0.18±0.44	0.06±0.22	0.24±0.63	0.18±0.32	0.09
S. longirostris	2.74±2.21	3.02±2.32	3.31±2.53	2.24±2.21	1.38±1.57	2.67±0.21	0.09
Total	3.53±2.78	3.37±2.57	3.63±2.72	2.31±2.32	1.63±1.75	2.85±2.20	0.08

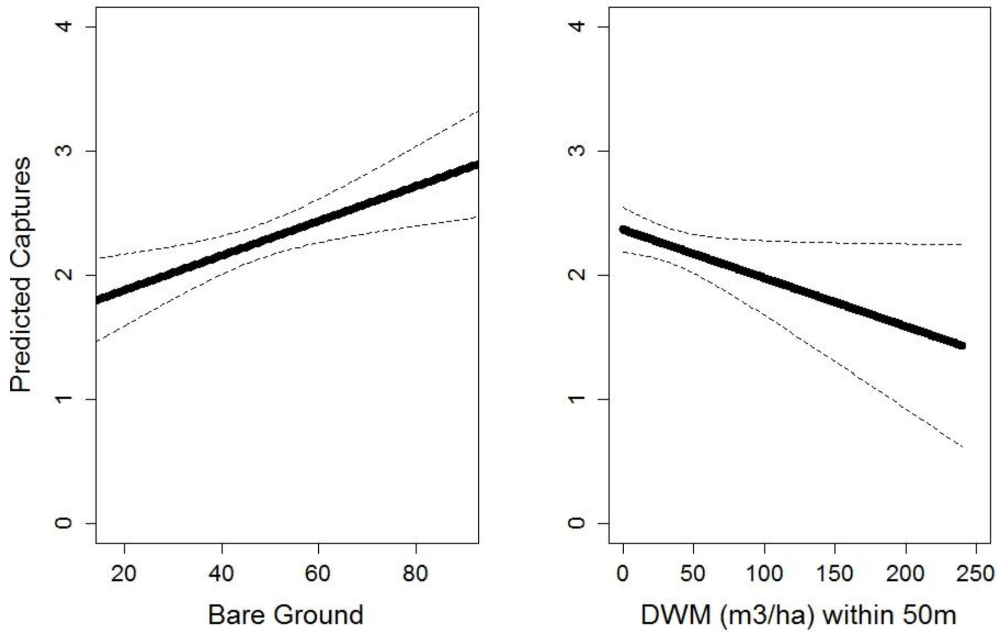

Figure 1. Total captures (predicted captures and 95% confidence interval) of three shrew species, the southeastern shrew (*Sorex longirostris*), the least shrew (*Cryptotis parva*), and the southern short-tailed shrew (*Blarina carolinensis*), were positively correlated to % bare ground at the drift fence array and negatively related to volume of downed woody debris (DWD) within 50 m of the drift fence array in loblolly pine (*Pinus taeda*) plantations in Beaufort County, North Carolina in 2013.

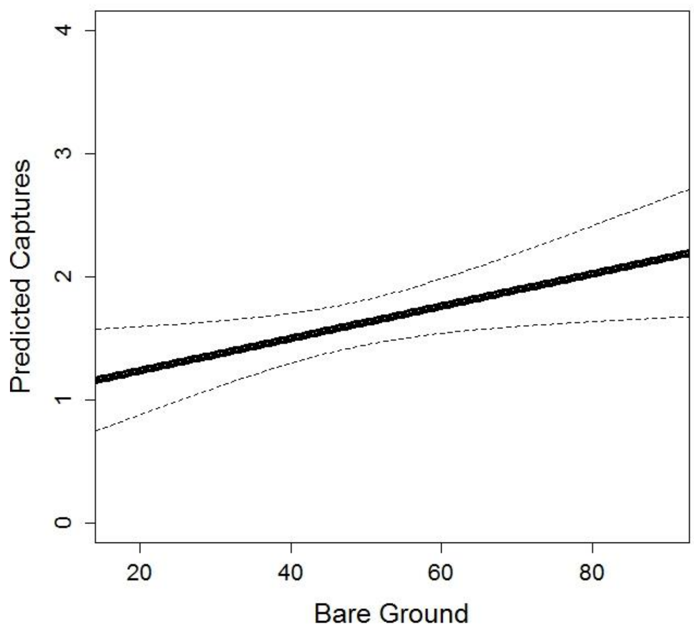

Figure 2. Number of captures (predicted captures and 95% confidence interval) of the least shrew (*Cryptotis parva*) were positively correlated to % bare ground at the drift fence array in loblolly pine (*Pinus taeda*) plantations in Beaufort County, North Carolina in 2013.

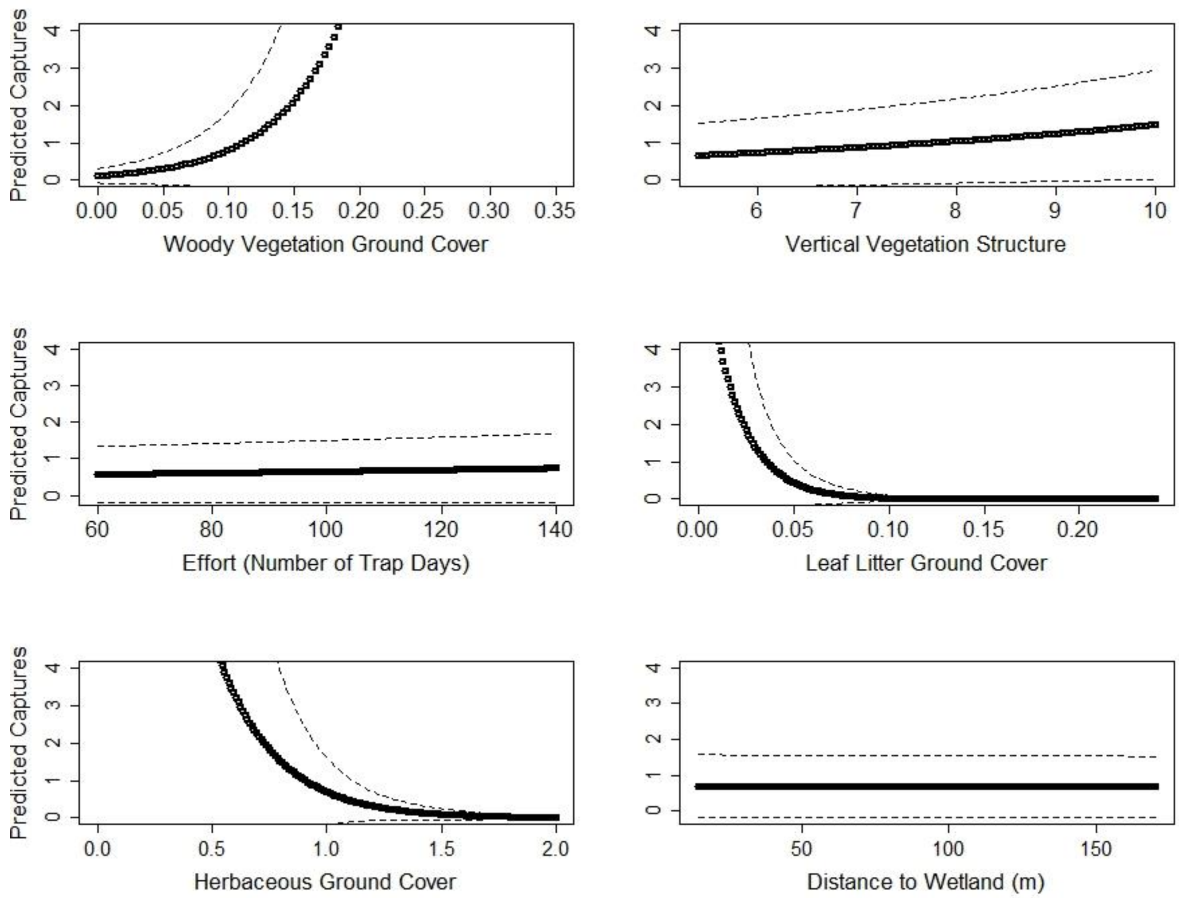

Figure 3. Capture success (predicted captures and 95% confidence interval) of three shrew species combined, the southeastern shrew (*Sorex longirostris*), the least shrew (*Cryptotis parva*), and the southern short-tailed shrew (*Blarina carolinensis*), were positively related to woody vegetation groundcover, vertical vegetation structure, and effort (number of buckets open per sampling season) at the drift fence array and negatively related to leaf litter groundcover, herbaceous groundcover, and distance to nearest wetland at the drift fence array in loblolly pine (*Pinus taeda*) plantations in Glynn County and Chatham County, Georgia in 2013.

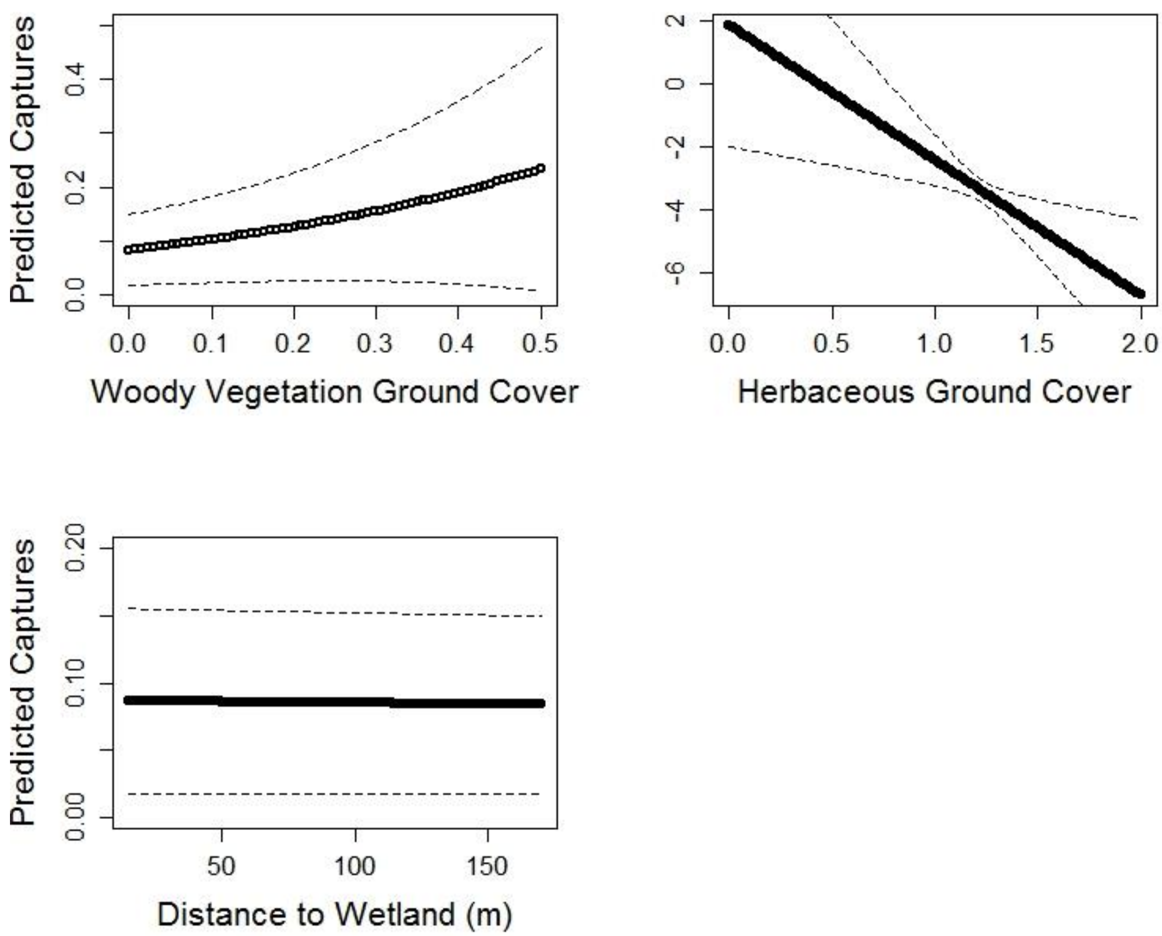

Figure 4. Capture success (predicted captures and 95% confidence interval) of least shrew (*Cryptotis parva*) was positively related to woody vegetation groundcover and negatively related to herbaceous groundcover at the drift fence array and distance to the nearest unharvested wet depression area from the drift fence array in loblolly pine (*Pinus taeda*) plantations in Glynn County and Chatham County, Georgia in 2013.