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EFFECT OF COARSE WOODY DEBRIS MANIPULATION ON SORICID AND HERPETOFAUNAL COMMUNITIES IN UPLAND PINE STANDS OF THE SOUTHEASTERN COASTAL PLAIN

by

JUSTIN CHARLES DAVIS

(Under the Direction of Steven B. Castleberry)

ABSTRACT

The majority of studies investigating the importance of coarse woody debris (CWD) to forest- floor vertebrates have taken place in the Pacific Northwest and southern Appalachian Mountains, while comparative studies in the southeastern Coastal Plain are lacking. My study was a continuation of a long-term project investigating the importance of CWD as a habitat component for shrew and herpetofaunal communities within managed pine stands in the southeastern Coastal Plain. Results suggest that addition of CWD can increase abundance of southeastern and southern short-tailed shrews. However, downed wood does not appear to be a critical habitat component for amphibians and reptiles. Rising petroleum costs and advances in wood utilization technology have resulted in an emerging biofuels market with potential to decrease CWD volumes left in forests following timber harvests. Therefore, forest managers must understand the value of CWD as an ecosystem component to maintain economically productive forests while conserving biological diversity.

INDEX WORDS: Amphibians, coarse woody debris, herpetofauna, upland pine, reptiles, southeastern Coastal Plain, shrews

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CHAPTER 1

INTRODUCTION AND LITERATURE REVIEW

INTRODUCTION

Coarse woody debris (CWD) is defined as any standing or fallen dead wood, including large branches (≥10 cm in diameter), downed boles, stumps, and decomposing root systems (Harmon et al. 1986; Hunter 1990; McCay et al. 2002; Spies and Cline 1988). CWD inputs within a forest occur when living trees are killed by natural events such as fire, wind, lightning, insects, disease, ice storms, or competition, or by anthropogenic actions such as prescribed burning or harvesting activities (Hunter 1990; Van Lear 1993). Several studies suggest that CWD constitutes a valuable resource for myriad plant and animal species (Bull 2002; Carey and Johnson 1995; Hunter 1990; Loeb 1996; McCay et al. 2002; Whiles and Grubaugh 1996), and that the presence of structural forest features, such as snags and fallen trees, promotes biological diversity (Hansen et al. 1991; Hunter 1990).

Suggested benefits of CWD are numerous. CWD provides small mammals and herpetofauna protective cover from predation and unfavorable climatic conditions (Loeb 1996; Whiles and Grubaugh 1996). Small mammals and herpetofauna also use the microhabitat within and under downed logs for nesting sites (Loeb 1996; Whiles and Grubaugh 1996). Further, small mammals are known to utilize CWD for travel routes and orientation (Loeb 1996; Barry and Francq 1980). Several bat species roost under the loose bark often associated with standing dead trees (snags), and squirrels commonly nest in snag cavities (Loeb 1996). Birds use snags for breeding and roosting activities (primarily via cavities), but also utilize downed CWD for perching, foraging and communicating (Lohr et al. 2002). Finally, decaying CWD provides seed germination sites and food production in the form of fungi and invertebrates, both of which are consumed by a variety of vertebrate species (Boddy 1983; Hunter 1990; Maser and Trappe 1984; Sharitz 1996; Spies et al. 1988).

Most research pertaining specifically to forest-floor vertebrate use of CWD is largely observational, and has taken place under uncontrolled conditions (Harmon et al. 1986; McCay et al. 2002; Whiles and Grubaugh 1996). Additionally, most studies investigating use of CWD by vertebrates have taken place in the Pacific Northwest where old-growth forests and relatively large amounts of CWD are common, while similar studies focused on the southeastern Coastal Plain are lacking (Harmon et al. 1986; McCay et al. 2002; McMinn and Crossley 1996; Whiles and Grubaugh 1996). Average volume of CWD in old-growth forests in the Pacific Northwest can reach 500 m^3, compared with 17.5 m^3 in natural pine stands of South Carolina (McMinn and Hardt 1996; Spies and Cline 1988). Low CWD accumulation in the Southeast is a product of short rotation lengths, silvicultural practices (prescribed burning and mechanical site preparation), and accelerated decomposition due to high humidity (McCay et al. 2002; Sharitz et al. 1992).

While the presence and distribution of CWD is considered important for small mammal, herpetofauna, and avian communities, decay stage may be the factor which determines its real value and use (Bowman et al. 2000; Carey and Johnson 1995; Lohr et al. 2002; McGee et al. 1999; Moseley et al. 2008; Whiles and Grubaugh 1996). For example, intact (hard) logs are more often used for cover and runways, while highly decayed (soft) logs provide a substrate which can be burrowed into or under by small mammals and herpetofauna (Bull 2002; Hayes and Cross 1987). Furthermore, logs in advanced stages of decay retain more moisture than less decayed logs (Boddy 1983; Maser et al. 1984). Moisture is especially important for sustaining amphibian species that are subject to desiccation during dry climatic conditions (Duellman and Trueb 1994; Jaeger 1980; Stebbins and Cohen 1997). Similarly, soricids incur high rates of evaporative water loss because of their high metabolic rates and high surface area to volume ratio (Chew 1951; Churchfield 1990).

Southeastern forests managed primarily for wood production lack the structural diversity and forest floor heterogeneity found in naturally managed and uneven–aged stands (Butts and McComb 2000; Harmon et al. 1986; Hunter 1990; Spies and Cline 1988; Sulkava and Huhta 1998; Van Lear 1993). Moreover, recent advances in timber harvest and wood utilization technology result in a decrease in CWD input during and following harvest operations, due to more efficient use of all parts of the tree and mechanical disruption of existing CWD (Butts and McComb 2000; Carey and Johnson 1995; Hunter 1990; Maser et al. 1988). Recent technology combined with a potentially increasing market for innovative wood-based products and biofuel (e.g. wood pellets, oriented strand board, and cellulosic ethanol) could result in less CWD left following harvest operations (Bies 2006; Hewett et al. 1981).

OBJECTIVES

My study was a continuation of a long-term project investigating the importance of CWD as a habitat component in managed upland pine stands in the southeastern Coastal Plain, representing years 11 and 12 since project initiation in 1997. Specifically, the study was designed to assess the response of soricid and herpetofaunal communities to the removal and addition of CWD (both downed and standing) as compared to control stands. Response variables of interest for shrews were abundance, age structure, and condition (as determined by body weight). For herpetofauna, differences in abundance, species richness, and species diversity among treatments were assessed. I examined CWD decay states within treatments to determine its potential significance as compared to earlier findings within the scope of the long-term project. Additionally, I measured spatially derived topographic variables (slope, elevation, aspect, and distance to nearest stream) within treatment plots to determine their potential influence on herpetofauna and soricid captures.

Coarse Woody Debris and Herpetofauna

Worldwide, amphibian and reptile numbers are declining, and have been doing so over the past 2 decades due to a number of factors including habitat loss and degradation, invasive species, disease and parasitism, pollution, unsustainable harvest and collection, and global warming (Gibbons et al. 2000; Lannoo 2005). In the United States, the Southeast harbors the highest diversity of herpetofauna, providing habitat for more than 50% of North American amphibian and reptile species (Goin and Goin 1971; Tuberville et al. 2005). In the region, herpetofauna species diversity is highest in the Atlantic and Gulf Coastal Plains (Gibbons and Buhlmann 2001).

CWD in the form of logging slash, downed logs, snags, stumps, and decomposing root systems increases habitat heterogeneity and structural diversity, which may help sustain herpetofauna abundance, as well as increase species diversity (Hansen et al. 1991; Harmon et al. 1986). Nesting sites, display arenas (for attracting mates), prey availability, predator avoidance, moisture reservoirs, and thermoregulation sites have all been cited as benefits of CWD to a wide array of herpetofauna species (Whiles and Grubaugh 1996). Although manipulative experiments discerning the relationship between herpetofauna and CWD are lacking, observational data indicate that representatives of all major groups of amphibians and reptiles utilize CWD (Whiles and Grubaugh 1996).

Structure and microhabitat created by downed logs is used by herpetofauna for a variety of reproductive functions (Whiles and Grubaugh 1996). Green anoles (*Anolis carolinensis*), eastern fence lizards (*Sceloporus undulatus*), and skinks (*Eumeces fasciata, E. inexpectatus,* and *E. laticeps*) use downed logs and snags for courtship displays, egg deposition, and aestivation (Cooper and Vitt 1994; Martof et al. 1980; Reilly et al. 2007; Vitt and Cooper 1986; Whiles and Grubaugh 1996). Many plethodontid salamanders and some ambystomid salamanders (e.g. marbled salamander, *Ambystoma opacum*) use microhabitat under and within downed logs as nesting sites for laying eggs (Bury et al. 1991; Graeter et al. 2008; Maser and Trappe 1984; Welsh and Lind 1991; Whiles and Grubaugh 1996). Butts and McComb (2000) found Ensatina (*Ensatina eschscholtzii*) and clouded salamander (*Aneides ferreus*) abundances increased with CWD volume in managed forests in Oregon. Downed logs maintain stable temperature and moisture regimes, providing buffers from temperature extremes and yielding optimal incubation sites and protection from desiccation (Boddy 1983; Graham 1925; Jaeger 1980; Maser and Trappe 1984; Maser et al. 1988; Whiles and Grubaugh 1996).

Coarse woody debris provides an invertebrate prey base for many amphibian and reptile species in the form of insect larvae, termites, centipedes, and earthworms, among many others (Graham 1925; Whiles and Grubaugh 1996). These invertebrates are consumed by small terrestrial snakes, as well as toads, frogs, lizards, and salamanders, which may in turn be fed upon by larger snake species and other higher order vertebrate taxa (Conant and Collins 1998).

CWD not only propagates invertebrate assemblages, but concurrently produces prey while providing cover from potential predators and exposure to potentially unfavorable ambient conditions (Whiles and Grubaugh 1996). In addition to invertebrates, small mammals associated with CWD such as shrews, voles, and mice are consumed by larger snake species such as rat snakes, racers, coachwhips, kingsnakes, pine snakes, and pit vipers, which are also known to utilize logs, rotting stumps, and root systems (Gibbons and Dorcas 2005; Martof et al. 1980; Loeb 1999; Maser and Trappe 1984; Tallmon and Mills 1994).

Because few reptile or amphibian species are viewed as economically valuable, there has been comparatively little incentive to understand their ecology in the context of land management (Gibbons 1983; Whiles and Grubaugh 1996). However, there is increasing evidence that reptiles and amphibians are important components of wildlife communities, serving roles as both predators and prey, and constitute a large portion of ecosystem biomass (Gibbons and Buhlmann 2001; Harmon et al. 1986; Maser and Trappe 1984; Tuberville et al. 2005; Whiles and Grubaugh 1996). Several studies, along with anecdotal observations, indicate that most amphibian and reptile species benefit from the presence of CWD in some capacity (Harmon et al. 1986; Maser and Trappe 1984; Whiles and Grubaugh 1996). Therefore, manipulative experiments designed to elucidate the characteristics of CWD (i.e. volume, structure, decay stage) critical to herpetofauna communities provide a valuable opportunity to further our understanding of this important habitat component within southeastern forests.

Coarse Woody Debris and Soricids

There are 35 shrew species in North America (Reid 2006). Of the 9 species found in the Southeast, 3 species, the least shrew (*Cryptotis parva*), southeastern shrew (*Sorex longirostris*), and southern short-tailed shrew (*Blarina carolinensis*), inhabit the Coastal Plain physiogeographic region. Shrews are often linked with CWD as a preferred microhabitat feature. In reviewing the pertinent literature, Loeb (1996) found that nearly all descriptions of shrew habitat included presence of logs. Benefits of CWD to shrews include nest sites, stable microclimate and moisture regimes, invertebrate production, and protection from predators (Loeb 1999; Maser et al. 1988; Maser and Trappe 1984).

The majority of studies investigating soricid and CWD relationships have taken place in the Pacific Northwest or Appalachian Mountains. For example, Trowbridge's shrew (*S. trowbridgiii*) captures in Oregon and Washington were positively correlated with log presence (Carey and Johnson 1995). In the central and southern Appalachians, smoky (*S. fumeus*), masked (*S. cinereus*), and northern short-tailed shrews (*B. brevicauda*) are positively associated with downed logs (Brannon 2000; Ford et al. 1997; McComb and Rumsey 1982). In the Upper Coastal Plain of Virginia, microhabitat assessments showed that *B. brevicauda* and pygmy shrews (*S. hoyi*) were positively associated with mean diameter of downed wood (Bellows et al. 2001). Although few studies have focused on soricid use of CWD in the Coastal Plain region of the Southeast, Loeb (1999) found higher *B. carolinensis* abundance in plots containing CWD compared to plots lacking CWD in upland pine stands in the South Carolina Coastal Plain.

While these studies suggest that shrews benefit from CWD, others have failed to show significant relationships (Bowman et al. 2000; Getz 1961; McCay et al. 1998; McCay and Komoroski 2004; Mengak and Guynn 2003). For example, Bowman et al. (2000) found no correlation between *B. brevicauda* captures and abundance or decay state of downed logs. During years 1-5 of the long-term project with which this study is associated, McCay and Komoroski (2004) found no difference in *B. carolinensis* and *S. longirostris* capture rates between control plots and plots in which CWD was removed. In the Piedmont Plateau of western South Carolina, Mengak and Guynn (2003) observed no relationship between shrew captures and CWD. Getz (1961) determined moisture to be the most important factor in dictating *B. brevicauda* and *S. cinereus* distributions in southern Michigan, and that food availability was the second most important factor. Getz (1961) postulated that the primary importance of CWD is increased humidity.

Shrews have a high surface area to volume ratio, which increases their susceptibility to evaporative water loss and desiccation (Chew 1951; Churchfield 1990; Reid 2006). Furthermore, the high metabolic rates of shrews require almost constant food consumption (Churchfield 1990). It therefore follows that habitat features such as CWD that increase moisture and humidity at the forest floor level, while promoting invertebrate production should increase shrew abundance. Invertebrate abundance has been shown to be higher in moist environments (Graham 1925; Churchfield 1990). Further, highly decayed logs have an increased moisture holding capacity (Boddy 1983). These characteristics along with the added benefit of nesting sites and cover from predators further support the notion that CWD within a forest stand is a beneficial habitat component for soricid communities. Therefore, long term studies that document the response of shrew communities to CWD as decay stages advance over time may contribute to our understanding of the real value of this habitat feature within forested ecosystems.

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CHAPTER 2

EFFECT OF COARSE WOODY DEBRIS MANIPULATION ON SORICID COMMUNITIES IN UPLAND PINE STANDS OF THE SOUTHEASTERN COASTAL PLAIN[1]

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ABSTRACT

Shrew abundance has been linked with coarse woody debris (CWD) presence, especially downed logs, in multiple regions in the U.S. We investigated the importance of CWD to shrew communities in managed upland pine stands in the southeastern United States Coastal Plain. Using a randomized complete block design, 1 of the following treatments was assigned to 12, 9.3-ha plots: removal (n = 3; all downed CWD ≥10 cm in diameter and ≥ 60 cm long removed), downed addition (n = 3; 5-fold increase in volume of down CWD), snag (n = 3; 10-fold increase in volume of standing dead CWD), and control (n = 3; unmanipulated). Shrews were captured seasonally using drift-fence pitfall trapping arrays within treatment plots. Topographic variables (slope, elevation, aspect, and distance to nearest stream) were measured and included as treatment covariates. We captured 264 southern short-tailed (*Blarina carolinensis*), 136 southeastern (*Sorex longirostris*), and 43 least (*Cryptotis parva*) shrews over 7 seasons from January 2007 - August 2008. *B. carolinensis* captures were higher in downed addition compared to removal plots and *S. longirostris* captures were higher in downed addition and snag compared to removal plots. *C. parva* captures did not differ among treatments. *S. longirostris* captures were influenced by slope. My results suggest that presence of CWD benefits 2 of the 3 species of shrew common in the southeastern Coastal Plain. Forest managers should consider the benefits of CWD to faunal communities in management decisions.

Key words: *Blarina*; Coarse woody debris; *Cryptotis*; Decay state; Shrew; *Sorex*; Topographic variables; upland pine

Acknowledgement of the ecological significance of habitat components such as coarse woody debris (CWD) in the conservation of biodiversity has increased in recent years (Harmon et al. 1986; McMinn and Crossley 1996; McMinn and Hardt 1996). Presence of structural forest features, such as snags and fallen trees, may promote biological diversity (Hansen et al. 1991; Hunter 1990; Sharitz et al. 1992). However, intensive plantation forestry practices can greatly reduce CWD inputs within forest ecosystems (Spies and Cline 1988; Spies et al. 1988). Further, recent advances in wood utilization technology and an emerging biofuels market (i.e. wood pellets and cellulosic ethanol) have the potential to further decrease the amount of woody material left in forests following timber harvests (Bies 2006; Butts and McComb 2000; Carey and Johnson 1995; Hewett et al. 1981; Hunter 1990; Maser and Trappe 1984; Maser et al. 1988). Maintaining CWD while managing for timber products can be challenging, especially in the southeastern United States where shorter rotation lengths, high humidity, and silvicultural practices result in lower CWD accumulations (McCay et al. 2002; McMinn and Hardt 1996; Sharitz et al. 1992).

Shrew abundance has been linked with CWD presence, especially downed logs, in multiple regions in the U.S. (Loeb 1996; Maidens et al. 1998; McCay et al. 1998). In Oregon and Washington, captures of Trowbridge's shrew (*Sorex trowbridgii*) were positively correlated with log presence (Carey and Johnson 1995; McComb and Rumsey 1982). In the southern and central Appalachians, smoky (*S. fumeus*), masked (*S. cinereus*), and northern short-tailed shrew (*Blarina brevicauda*) abundances were positively associated with downed logs (Brannon 2000; Ford et al. 1997; McComb and Rumsey 1982). In the South Carolina Coastal Plain, Loeb (1999) captured more southern short-tailed shrews (*B. carolinensis*) in upland pine stands containing CWD inputs from tornado damage compared to those in which CWD was removed.

While the above studies support the theory that shrews benefit from CWD, others have failed to show significant relationships. For example, in Appalachian forests of New Brunswick, Canada, Bowman et al. (2000) found no correlation between *B. brevicauda* captures and abundance or decay state of downed logs. In a study at the Savannah River Site in the Upper Coastal Plain of South Carolina, *B. carolinesis* and southeastern shrew (*S. longirostris*) captures did not differ between control plots and plots in which CWD was removed (McCay and Komoroski 2004). Similarly, Mengak and Guynn (2003) observed no relationship between *B. carolinensis* captures and CWD in western South Carolina.

Lack of consensus on the relationship between CWD and shrew communities, especially in the southeastern Coastal Plain, demonstrates the need for continued investigation. In addition to volume, understanding additional characteristics of CWD, such as decay state, may be the key to interpreting its importance as a habitat component. Long-term studies that document responses as decay stages advance may clarify the role of CWD in structuring shrew communities.

My study was a continuation of a 2-phase long-term project investigating the importance of CWD as a habitat component in managed upland pine stands in the southeastern Coastal Plain, representing years 6 and 7 of Phase II. Specifically, the study was designed to assess the response of shrew communities to removal and addition of CWD (both downed and standing) as compared to control stands. Additionally, I measured spatially derived topographic variables (slope, elevation, aspect, and distance to nearest stream) within treatment plots to determine their potential influence on shrew captures and to isolate treatment effects. Capture trends from previous years of this project, coupled with shrew natural history and physiology characteristics, were used to predict shrew response to treatments.

I predicted shrew abundance would be positively related to CWD addition treatments, and negatively related to CWD removal treatments. I also predicted higher mean body mass for shrews captured in downed addition plots due to an assumed higher invertebrate prey base (Hanula et al. 2006; Jabin et al. 2004). I predicted a greater percentage of shrews of younger age to be present in downed addition plots, as a result of an assumed higher reproduction rate within those plots, as well as younger and more aggressive individuals outcompeting older individuals for the higher quality habitat that increased CWD may provide (Rychlik 1998). I also expected shrew captures in removal plots to be positively related to rain events, assuming the lack of moisture-holding logs on the ground would restrict movements during dry climatic conditions.

METHODS

Study area.—Study plots were located on the Savannah River Site (SRS), a 78,000-ha National Environmental Research Park administered by the Department of Energy (DOE). The SRS is located in the Upper Coastal Plain and Sandhills physiographic region of South Carolina (White 2005). This region is characterized by sandy soils and gently sloping hills dominated by pines with scattered hardwoods (Kilgo and Blake 2005). The climate of the SRS is humid subtropical with mean annual temperature and rainfall of 18°C and 122.5 cm, respectively (Blake et al. 2005).

When the DOE acquired the SRS in 1951, old-field habitats dominated the site as a result of past land-use activities driven by food crop, cotton, and naval store production (White 2005). Today, the majority of the site has been reforested by the USDA Forest Service (Imm and McLeod 2005; White 2005). Approximately 68% of the SRS is composed of upland pine stands, including loblolly (*Pinus taeda*), slash (*P. elliottii*), and longleaf (*P. palustris*) pines. While much of the land is managed for timber production, nearly two-thirds of the pine forests on SRS are 40-70 years old (Imm and McLeod 2005).

Study plots were located in 3 loblolly pine stands planted between 1950 and 1953. Although the overstory in these stands was dominated by loblolly pine, slash and longleaf pine also were present. Understory vegetation was dominated by sassafrass (*Sassafrass albidum*), black cherry (*Prunus serotina*), lespedeza (*Lespedeza* spp.), blackberry (*Rubus* spp.), and poison oak (*Toxicodendron pubescens*).

Study design.—Study design was a randomized complete block design, with each of 4 treatments randomly assigned in each of the 3 forest stands (blocks). Blocks were chosen based on the following criteria: approximately 45-year-old loblolly pine plantations (at project initiation in 1996); ≥ 76m from the nearest wetland, road, or power line; and large enough to accommodate 4 9.3-ha square plots. Treatments were 1) control, where downed CWD was not manipulated; 2) snag, where standing CWD volume was increased 10-fold; 3) removal, where all downed CWD ≥10 cm in diameter and ≥ 60 cm in length was removed; and 4) downed, where volume of downed CWD was increased five-fold. The control and removal treatment plots were initiated in 1996 (Phase I), while the downed and snag treatment plots were implemented in 2001 (Phase II). Annual removal of CWD was performed in the removal treatment plots. Each treatment plot consisted of a 6-ha core trapping area, surrounded by a 3.3-ha buffer zone subject to the same treatment to minimize edge effect. All 12 plots were thinned in 2001 to a live pine basal area of between 13.8 – 20.8 m^2/ha, and were prescribed burned in summer 2004.

Data collection.—Downed woody debris measurements were taken both throughout the duration of my study (2007-08), as well as 2 years prior (2005-06) to measure change in decay state over time. Inventories were conducted in randomly selected subplots (50 x 50 m) within the inner 4 ha of each treatment plot. Within each subplot, all logs with at least 50% of their measurable length within the subplot were measured, and logs ≥ 10 cm in diameter at the midpoint and ≥ 60 cm in length were included in the inventory. Logs were classified into 1 of 5 decay categories based on the Maser et al. (1979) decay scale, where stage 1 logs were sound, with intact bark; stage 2 logs had mostly sound wood with some bark starting to flake; stage 3 logs had broken branches and were missing bark; stage 4 logs were soft and blocky in texture; and stage 5 logs were powdery in texture and partly buried. Although log volumes were estimated assuming logs were round in circumference, which may overestimate the true volume of downed wood, measurements were consistent among treatments and years.

Shrew sampling was conducted using pitfall drift fence arrays. Drift fences consisted of aluminum flashing buried approximately 15 cm below ground, with 19-l plastic buckets buried flush to the ground against each fence. Each plot contained one cross-shaped array with four 30-m arms extending out from the center of the plot in each of the cardinal directions, and four Y-shaped arrays with three 15-m arms located in each corner of the 6-ha core sampling area (Fig. 2.1). Pitfall traps (buckets) were maintained with 2.5 to 5 cm of soil in the bottom to provide captured animals cover from temperature extremes and desiccation during trapping periods. The bottoms of buckets were perforated, allowing drainage of excess water after heavy rains.

Shrews were sampled in all plots for 14 days each season from January 2007 – August 2008, for a total of 7 sampling seasons, during which traps were checked daily between 0700 and 1700. Shrews were identified to species, weighed for mass (g), and measured for snout-vent and tail lengths (mm). Live shrews were euthanized via cervical dislocation and frozen for subsequent dissection. All individuals were collected under University of Georgia Institutional

Animal Care and Use permit number A2007-10033-0. Rainfall data was collected from a weather station approximately 4 km from study plots.

I quantified spatial variables with the potential to influence shrew abundance within treatment plots. Trap array locations were recorded using a Trimble GeoExplorer 3 handheld GPS unit and differentially corrected using data from a Continually Operating Reference Station (CORS) in Columbia County, Georgia (63 km from our study site). Array locations for Y-shaped arrays were taken at the middle bucket and array locations for each arm of the cross-shaped array were taken at the middle of each array arm. Y-shaped arrays and each arm of the cross-shaped array were considered separate arrays, resulting in 8 arrays with capture data per plot. Buffers were created around array locations with a unique buffer size for each shrew species based on average home range sizes from the published literature (Fig. 2.1). Buffer radii were 55.3 and 45.4 m for *B. carolinensis* (McCay 2001) and *C. parva* (Choate and Fleharty 1973), respectively. Because no home range estimates were available in the literature for *S. longirostris*, a buffer radius of 42 m was used based on home range estimates for *S. cinereus* (Buckner 1966), a similarly sized species with a similar life history (McCay et al. 2004; Reid 2006; Whitaker and Hamilton 1998). Mean elevation, degree of slope and aspect were calculated for each buffered area using the Zonal Statistics tool in the Spatial Analyst extension of ArcGIS 9.2 (Environmental Systems Research Institute). All pixels (and their associated values) with ≥50% of their area within the buffer boundary were used for calculations. Distance to the nearest stream was calculated from the actual array location point (center point of the buffer). All stream orders were considered, although intermittent streams may not have contained water during the study.

Shrews were dissected to determine age and gender, and were designated as age class 1, 2, 3, or 4 based on relative tooth attrition using methods outlined in Pearson (1945) for *B. carolinensis* and Rudd (1955) for *S. longirostris*. Rudd's 7-class system was modified for this study, resulting in four age classes: age class 1 (corresponded to Rudd's age classes 1 and 2); age class 2 (corresponded to Rudd's age classes 3 and 4); age class 3 (corresponded to Rudd's age classes 5 and 6); and age class 4 (corresponded to Rudd's age class 7). For *B. carolinensis*, age class 1 individuals were 0-24 weeks old, 2 were 24-40 weeks old, 3 were 40-64 weeks old, and 4 were older than 64 weeks. For *S. longirostris*, age class 1 included individuals 0-18 weeks old, 2 included individuals 18-36 weeks old, 3 included individuals 36-54 weeks old, and 4 included individuals 54 weeks or older. Age structure analysis was not conducted for *C. parva* due to low sample size ($n = 43$) and lack of accurate aging methodology for the species.

Statistical analysis.—Shrew captures at each Y-shaped array and each arm of cross-shaped arrays were standardized as number of captures/m of fencing for each species. Treatment differences were examined using 2-way analysis of covariance (ANCOVA) with slope, elevation, aspect, and distance to nearest stream as covariates. Two-way ANOVA was used to determine differences in mean body mass for each species among treatments and seasons. *B. carolinensis* and *S. longirostris* weights were blocked by gender and age, and *C. parva* weights were blocked only by gender. All data were tested for normality using Shapiro-Wilks test. Non-normal data were ranked and ANOVA or ANCOVA was performed on ranks. Significant results were further analyzed using adjusted least square means pairwise comparisons. I used forest stand as the block factor for all ANCOVA and ANOVA analyses (except body mass) and SAS 9.1 (SAS Institute 2008) to perform all statistical analyses.

Spearman correlation coefficient was used to examine the effect of precipitation on daily shrew captures within each treatment. Total number of each species captured/sample night/treatment was compared with amount of precipitation 24 hours prior to the morning traps were checked.

Age class frequencies for *B. carolinensis* and *S. longirostris* were analyzed using log-likelihood ratio G-tests to determine if age class ratio differed among treatments. Small sample sizes yielded expected values <5. G-tests will underestimate *P*-values under these circumstances. Therefore, randomization (Monte Carlo simulation) tests based on 1 million replicates were used to generate *P*-value estimates for significance testing (Sokal and Rohlf 1995; Zar 1999). A significant difference in age class ratios among treatments was interpreted as a skewed age class distribution.

RESULTS

Mean volumes (± SE) of downed CWD in 2007 were 59.4 m^3/ha (± 7.9) for downed plots, 34.7 m^3/ha (± 6.3) for snag plots, and 12.7 m^3/ha (± 1.9) for control plots (Fig. 2.2). Downed CWD volume for removal plots was not measured in 2007, but was 0.29 m^3/ha (± 0.14) and 0.24 m^3/ha (± 0.15) in 2005-06, respectively. Mean decay state of logs in downed, control, and snag plots in 2007 were 3.1, 3.3, 3.0, respectively (Fig. 2.3).

A total of 443 shrews was captured over 7 sampling seasons from January 2007 - August 2008. All 3 shrew species that occur on SRS were captured, and additional insectivore captures included 4 eastern moles (*Scalopus aquaticus*). *B. carolinensis* was captured most frequently, representing 59.6% of all captures, while *S. longirostris* and *C. parva* captures made up 30.7%

and 9.7%, respectively. Throughout the study, mortality rates for *B. carolinesis, S. longirostris*, and *C. parva* captured in pitfall traps were 73.1, 87.5, and 60.5%, respectively.

No covariate had an effect on *B. carolinensis* or *C. parva* captures ($P > 0.05$). Slope was a significant covariate for *S. longirostris* captures ($F = 6.05$, $d.f. = 1,86$, $P = 0.0159$). *B. carolinensis* captures were higher in downed addition than removal plots ($F = 3.26$, $d.f. = 3,85$, $P = 0.0253$; Table 2.1). *S. longirostris* captures were higher in downed addition and snag compared to removal plots ($F = 6.76$, $d.f. = 3,86$, $P = 0.0004$). *C. parva* captures did not differ among treatments ($P > 0.05$).

Mean body mass (g) of 252 *B. carolinensis*, 116 *S. longirostris*, and 37 *C. parva* did not differ among treatments for any species ($P > 0.05$; Table 2.2). *B. carolinensis* weighed more in fall and winter than spring ($F = 22.24$, $d.f. = 3, 218$, $P < 0.0001$; Table 2.3). Mean mass of *S. longirostris* was higher in winter than spring ($F = 4.14$, $d.f. = 3, 95$, $P = 0.0083$). *C. parva* weights did not differ among seasons ($P > 0.05$).

S. longirostris and *C. parva* captures were not correlated with precipitation ($P > 0.05$) in the previous 24 hours. *B. carolinensis* captures in removal plots were positively correlated with precipitation ($r = 0.22632$, $P = 0.0250$). Age class frequencies for 255 *B. carolinensis* ($G^2 = 13.96$, $d.f. = 9$, $P = 0.1236$) and 132 *S. longirostris* ($G^2 = 4.78$, $d.f. = 9$, $P = 0.853$) examined were similar among treatments.

DISCUSSION

Capture rates of *B. carolinensis* and *S. longirostris* were higher in downed addition compared to removal plots, supporting my hypothesis. These results differed from those reported by McCay and Komoroski (2004) during Phase I (1997-2001) of the long-term project at SRS, who found no difference in *B. carolinensis* and *S. longirostris* captures among treatments. However, they are consistent with years 2-4 of Phase II (2003-05) in which *S. longirostris* captures were lower in removal compared to downed addition and control plots (Owens 2006). Higher *B. carolinensis* and *S. longirostris* captures in downed addition plots may be attributed to increased prey production associated with CWD. Spiders, centipedes, adult and larval beetles and moths, and earthworms made up 43.3% of the stomach contents of 45 *B. carolinensis* from the SRS (Whitaker et al. 1994) and 70.1% of the diets of 90 *S. longirostris* from Indiana (French 1984). Jabin et al. (2004) recorded higher densities of spiders, centipedes, and adult and larval beetles within closer proximity to CWD. Beetles and moths (adults and larvae) also are known to utilize decaying wood (Hanula 1996; Whitaker et al. 1994), and some earthworm species may also benefit from CWD (Hendrix 1996). Mean decay stage of CWD in downed addition plots in 2007 was 3.1. Progression of CWD through decay stages 2 and 3 begin to support mycorrhizal fungi underneath the bark. Slippage of the bark relocates the fungi, and invertebrate grazers associated with it, alongside the log where it meets the ground (Maser and Trappe 1984). The new 'microhabitat' created at the log/litter interface is favored by snails and slugs, which have been found to comprise 18.5% of *B. carolinensis* diets on the SRS (Whitaker et al. 1994).

Higher abundance of *B. carolinensis* and *S. longirostris* in downed addition plots may also be a product of the role CWD plays as a source of cover. Owls are the primary predator of shrews and are known to take both *B. carolinensis* and *S. longirostris* (Churchfield 1990; French 1980; Genoways and Choate 1998). The overhang area created where logs meet the forest floor can provide protective cover for small mammals (Maser et al. 1979). For example, Hayes and Cross (1987) found that vole captures were positively correlated with greater log overhang.

Shrews may, therefore, utilize log overhang for quiet and inconspicuous travel routes. In addition to predator avoidance, shrews often construct nests beneath downed logs because they maintain a stable microclimate and retain adequate moisture (Boddy 1983; Churchfield 1990; Graham 1925; Loeb 1999). A rapid metabolic rate and high surface area to volume ratio subjects shrews to increased rates of evaporative water loss (Chew 1951; Churchfield 1990). Use of a nest can reduce the resting metabolic rate of shrews by up to 30% (Genoud 1988), thus conserving energy stores and decreasing exposure to predation while foraging.

Higher abundance of *S. longirostris* in snag plots compared to removal plots may be due to increased density of decaying root systems. The nutrient deprivation of roots as a result of tree girdling can lead to enhanced decomposition of root systems (Högberg et al. 2001). McCay (2000) found that voids created by decomposing root systems provided the most suitable daytime refugia for cotton mice (*Peromyscus gossypinus*) in a southeastern pine forest, due to the lack of alternative refuge sites such as rock outcrops. Because *S. longirostris* is considered epigeal in its habits (McCay et al. 2004), it may depend more on the decomposition of root systems for use as travel routes and foraging areas than the more fossorial *B. carolinensis*.

C. parva capture results were consistent with those of Moseley et al. (2008), but were contrary to my hypothesis. Interspecific competition among shrews can be reduced through morphological and behavioral adaptations (Brannon 2000). *B. carolinensis* and *C. parva* both have morphological characteristics (i.e. short tail and reduced pinnae) suggestive of a fossorial lifestyle. However, *C. parva* has been described as being largely epigeal in its habits (McCay et al. 2004), and is more commonly associated with grassy or old-field habitats (Bellows et al. 2001; Davis and Joeris 1945; Reid 2006; Whitaker 1974; Whitaker and Hamilton 1998). It is possible that *C. parva* was outcompeted by the larger *B. carolinensis* and forced to adapt to more open habitats at the edge of forested ecosystems. These habitat types would naturally have lower CWD volumes. Further, while most shrew species are solitary, *C. parva* is more gregarious and is known to nest communally (Churchfield 1990; Whitaker 1974). Communal nesting may be an adaptation to reproduction in habitats with fewer nesting sites in the form of downed logs. Thus, the competitive exclusion of *C. parva* from forested to early successional habitats may have forced the species to evolve not to rely on CWD as a critical habitat component.

I expected higher shrew body mass in downed addition plots, as a result of increased invertebrate prey associated with higher volumes of decaying wood. However, my results failed to show this relationship. Moderate to severe drought conditions persisted at my study site during all sample periods except for winter and spring 2007. Insect activity levels are known to respond positively to increases in environmental moisture (Graham 1925). Therefore, shrew weights compared among treatments during my study may not reflect those that occur during years receiving average rainfall.

I recorded heaviest mean body weights of *B. carolinensis* and *S. longirostris* in fall and winter. These results are consistent with Hartman et al. (2001), who also found higher mean body weights of *B. carolinensis* at the SRS during winter than spring. However, most shrew species attain maximum and minimum body weights during the summer and winter, respectively. Increase in weight likely is due to physiological changes triggered by the onset of the breeding season (Churchfield 1990). Winter weight loss in shrews, known as the Dehnel effect, is a well documented phenomenon, but has only been observed in *Sorex* species in northern and eastern Europe (Churchfield 1990), and thus may not occur in regions with milder climates such as the SRS.

Previous results from the long-term study demonstrated fewer young *B. carolinensis* in removal than control plots during Phase I (1997-2001; McCay and Komoroski 2004) and no difference in age structure among treatments in years 2-4 of Phase II (Moseley et al. 2008). My results were consistent with those from years 2-4 of Phase II. While these results refuted my hypothesis, shrew social patterns driven by breeding activity may provide an explanation. Dispersing juvenile shrews (generally males) do not disperse great distances from their natal area, but rather occupy the first vacant home range available (Churchfield 1990). Most available home ranges are those of overwintered males who have abandoned them in search of females. As available home ranges are utilized by juveniles born early in the breeding season, subsequent juveniles (generally those born later in the breeding season) are forced to wander much further to find an unoccupied home range. Additionally, following breeding many older males are unable to reestablish their home range in the presence of more fit juveniles and are forced to move elsewhere. Excessive wandering by late-born juveniles and older males depletes valuable resources and increases vulnerability to predation (Churchfield 1990). Therefore, the availability of unoccupied home ranges within a given habitat limits the number of shrews in younger age classes, while competitive exclusion of older individuals limits the number of shrews in older age classes.

I found a positive relationship between *B. carolinensis* captures and rainfall. This phenomenon has also been documented for other shrew species. Ford et al. (2002) observed positive correlations between previous 24-hour rainfall and *S. cinereus*, *S. fumeus*, and *B. brevicauda* captures in the Allegheny Mountains of West Virginia. Owens (2006) also noted a similar correlation for *B. carolinensis* and *S. longirostris* during years 3-4 of Phase II of the long-term project at SRS. Increased *B. carolinensis* activity associated with precipitation is likely driven by physiological requirements related to water balance and food intake. High metabolic rates subject shrews to increased rates of evaporative water loss and require almost constant feeding (Chew 1951; Churchfield 1990; Reid 2006). Higher moisture levels are known to positively influence insect abundance (Churchfield 1990; Graham 1925). Increased shrew activity following rain events provides cool, moist conditions under which to forage for more active and abundant prey.

Moisture levels have been shown to influence the distribution of shrew communities (Getz 1961). I examined variation in slope, aspect, and relative elevation as covariates because they have all been shown to affect soil moisture (Famiglietti et al. 1998). These factors can influence soil moisture not only in areas with high levels of topographic variability, but also on sites with little relief. Famiglietti et al. (1998) found that aspect and elevation significantly affected soil moisture levels on a landscape with ≤ 10.6 m of total relief. Further, Ford et al. (1997) found slope was positively related to *S. fumeus* presence and negatively related to *S. hoyi* presence in the southern Appalachians. Elevation relief on my study plots averaged 29.8 m, and slope was a significant covariate for *S. longirostris* captures. By accounting for topographic variability within treatment plots, differences observed in shrew captures among treatments can more confidently be attributed to treatment effects.

The decay state of CWD within treatment plots during this study averaged 3.1 on a scale of 5.0. Continuation of this study until decay state of CWD reaches its maximum level should be considered to fully understand the influence of all aspects of CWD within forested ecosystems. Logs of various decay stages offer shrews functionally different benefits (Hayes and Cross 1987). There is a natural progression of invertebrate species that utilize CWD at different stages of decay (Graham 1925; Hanula 1996; Maser and Trappe 1984) providing all shrew species with preferred prey items at some point in the decomposition process. Getz (1961), considering a number of environmental factors (vegetation type, cover, temperature, food, moisture, and interspecific competition), concluded that the most important factor influencing the local distribution of *S. cinereus* and *B. brevicauda* was moisture. Further, Getz (1961) noted that within situations having favorable moisture conditions, food was the most important factor influencing *B. brevicauda* distribution. Wood in advanced stages of decay has a higher moisture holding capacity (Boddy 1983), and invertebrate communities respond positively to increased moisture levels (Graham 1925). Therefore, management strategies that retain CWD and allow it to reach advanced decay states may yield the habitat component that provides the 2 most important factors contributing to shrew habitation and proliferation.

CONCLUSION

B. carolinensis and *S. longirostris* abundance were positively influenced by the addition of downed CWD. These 2 species are considered habitat generalists, but are most often found in forested ecosystems (French 1980; Genoways and Choate 1998; McCay 2001). *C. parva* is more common in old-field habitats (Bellows et al. 2001; Davis and Joeris 1945; Reid 2006; Whitaker 1974; Whitaker and Hamilton 1998) and was not influenced by the addition of downed CWD. Two of the most important habitat requirements for shrews are moisture and food availability (Churchfield 1990; Getz 1961), both of which have been shown to increase with CWD inputs (Boddy 1983; Graham 1925; Maser and Trappe 1984). Pine plantation management regimes and extensive silvilcultural practices can result in decreased CWD inputs, reduced structural heterogeneity, and decreased biodiversity (Hansen et al. 1991; Hunter 1990; Sharitz et al. 1992; Spies et al. 1988). Technological advances and increasing energy demands can create markets for once unusable forest components such as logging refuse, snags, and stumps (Bies 2006; Butts and McComb 2000; Carey and Johnson 1995; Hewett et al. 1981; Hunter 1990; Maser and Trappe 1984; Maser et al. 1988). For example, Westbrook et al (2007) found that chipping logging residues and understory stems produced competitively priced quality energy chips, while reducing site preparation costs and increasing plantable area. While we do not fully understand the value of CWD to shrew communities, the findings in this study suggest that 2 of the 3 species of shrew common in the southeastern Coastal Plain are benefited by its presence within ecosystems. Therefore, in the interest of conserving biodiversity and maintaining ecosystem health, forest managers should consider the benefits CWD provides to faunal communities and the potential impact removing it may have on the biotic community.

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Table 2.1. Mean shrew captures/m of drift fence (SE) in snag, control, removal, and downed treatment plots in upland loblolly pine (*Pinus taeda*) stands in Barnwell County, South Carolina. Means with different letters indicate significant differences among treatments.

	Treatment			
Species	Snag (n = 24)	Control (n = 24)	Removal (n = 24)	Downed (n = 24)
Blarina carolinensis	0.069 (0.012)ab	0.058 (0.013)ab	0.051 (0.011)b	0.105 (0.012)a
Cryptotis parva	0.004 (0.005)	0.015 (0.005)	0.014 (0.005)	0.012 (0.005)
Sorex longirostris	0.046 (0.009)a	0.026 (0.010)ab	0.013 (0.009)b	0.057 (0.009)a

Table 2.2. Mean (SE) body mass (g) of shrews captured in snag, control, removal, and downed treatment plots in upland loblolly pine (*Pinus taeda*) stands in Barnwell County, South Carolina.

	Treatment			
Species	Snag (n = 3)	Control (n = 3)	Removal (n = 3)	Downed (n = 3)
Blarina carolinensis	6.7 (0.2)	6.9 (0.2)	6.5 (0.2)	6.6 (0.1)
Cryptotis parva	3.7 (0.3)	3.5 (0.2)	3.6 (0.1)	3.8 (0.3)
Sorex longirostris	2.8 (0.1)	2.7 (0.1)	3.0 (0.2)	2.8 (0.1)

Table 2.3. Mean (SE) body mass (g) of shrews captured during spring, summer, fall, and winter in upland loblolly pine (*Pinus taeda*) stands in Barnwell County, South Carolina. Means with different letters indicate significant differences among seasons.

	Season			
Species	Spring	Summer	Fall	Winter
Blarina carolinensis	6.3 (0.1)a	7.3 (0.2)bc	7.5 (0.5)c	7.9 (0.2)c
Cryptotis parva	3.4 (0.1)	3.6 (0.2)	3.8 (0.4)	4.1 (0.4)
Sorex longirostris	2.8 (0.1)a	2.8 (0.1)ab	3.5 (0.0)ab	3.1 (0.2)b

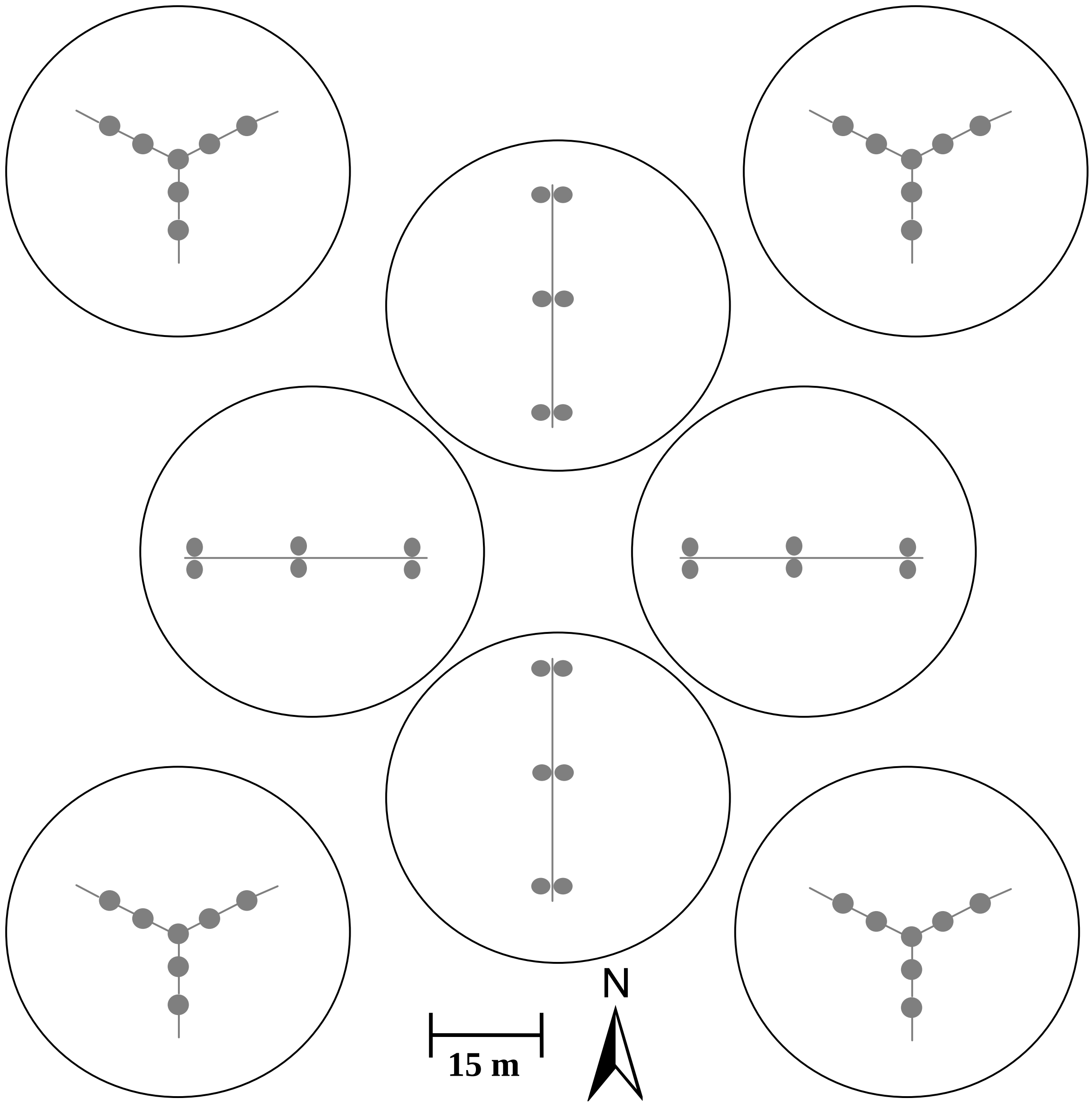

Figure 2.1. Arrangement of drift fence arrays and bucket traps used for sampling shrews on 6-ha core area of a 9.3-ha treatment plot in an upland loblolly pine (*Pinus taeda*) stand at the Savannah River Site, Barnwell County, South Carolina. Also shown are buffers within which topographic variables were measured for covariate analyses.

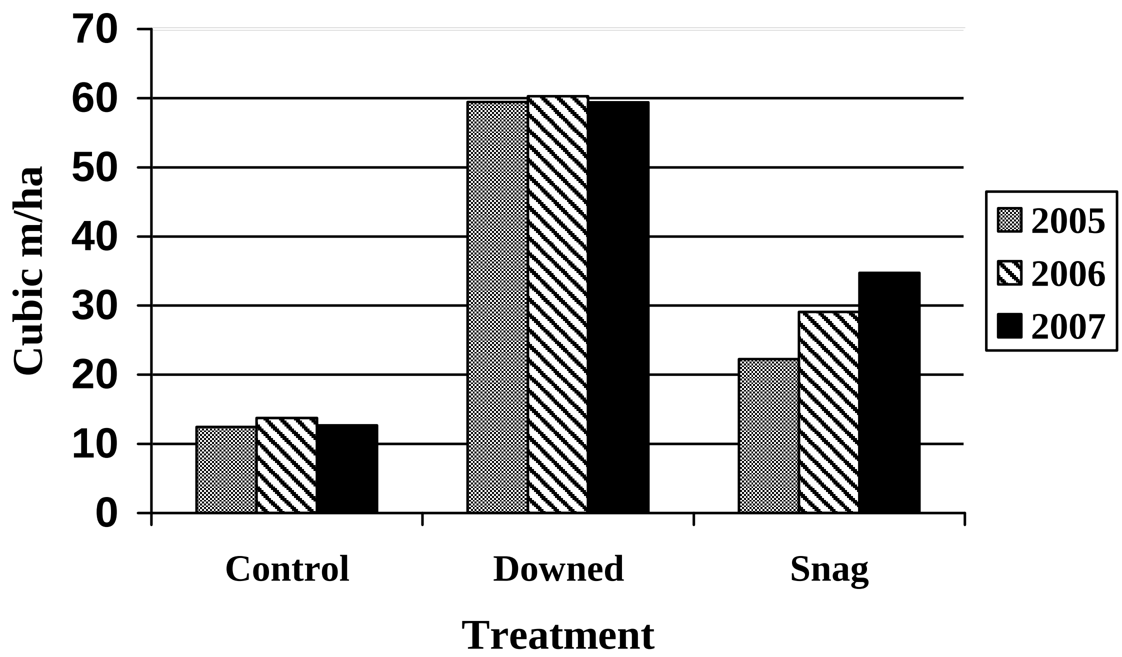

Figure 2.2. Mean volume of down woody debris accumulations over 3 years in control (n = 3), downed addition (n = 3), and snag (n = 3) treatment plots in loblolly pine (*Pinus taeda*) stands on the Savannah River Site, Barnwell County, South Carolina.

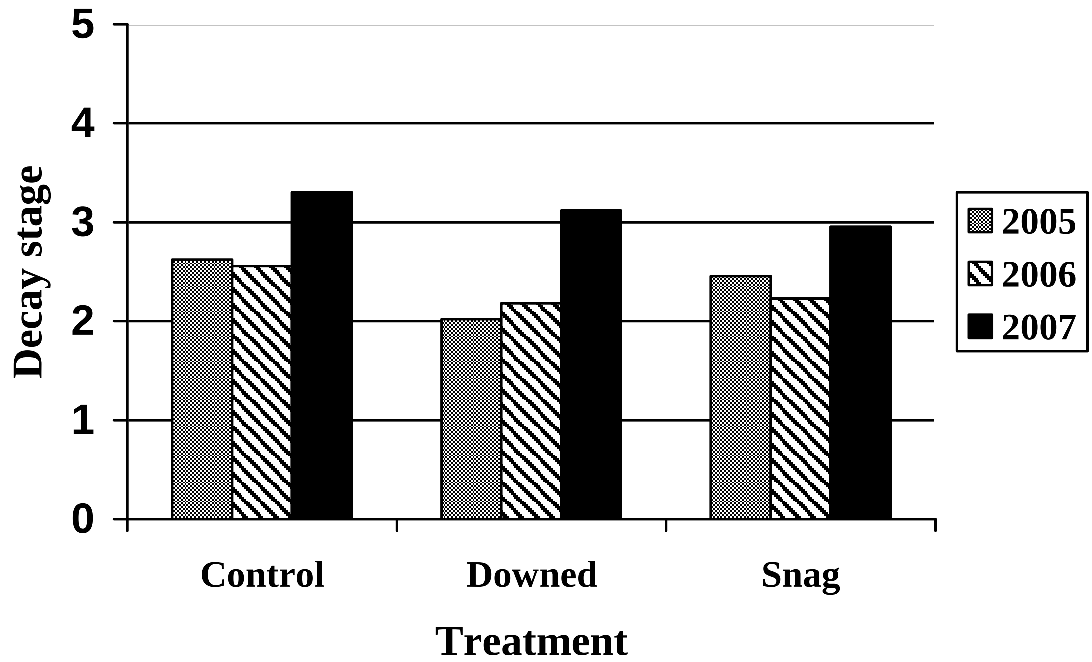

Figure 2.3. Mean decay stage of downed wood, based on Maser et al. (1979), over 3 years in control (n = 3), downed addition (n = 3), and snag (n = 3) treatment plots in loblolly pine (*Pinus taeda*) stands on the Savannah River Site, Barnwell County, South Carolina.

CHAPTER 3

EFFECT OF COARSE WOODY DEBRIS MANIPULATION ON HERPETOFAUNAL COMMUNITIES IN UPLAND PINE STANDS OF THE SOUTHEASTERN COASTAL PLAIN[1]

[1]Davis, J.C., S.B. Castleberry, and J.C. Kilgo. To be submitted to Forest Ecology and Management.

ABSTRACT

Coarse woody debris (CWD) is thought to benefit herpetofauna in a variety of ways including prey production and protection from moisture and temperature extremes. We investigated the importance of CWD to amphibian and reptile communities in managed upland pine stands in the southeastern United States Coastal Plain. Using a randomized complete block design, 1 of the following treatments was assigned to 9.3-ha plots: removal (n = 3; all downed CWD ≥10 cm in diameter and ≥ 60 cm long removed), downed addition (n = 3; 5-fold increase in volume of down CWD), snag (n = 3; 10-fold increase in volume of standing dead CWD), and control (n = 3; unmanipulated). Herpetofauna were captured seasonally using drift-fence pitfall trapping arrays within treatment plots. Topographic variables (slope, elevation, aspect, and distance to nearest stream) were measured and included as treatment covariates. We captured 355 amphibians and 668 reptiles over 7 seasons from January 2007 - August 2008. Abundance, species richness, and species diversity were similar among treatments for anurans, salamanders, and lizards. Snake abundance, species richness, and diversity were higher in removal than downed addition plots. Anuran abundance increased as distance to nearest stream decreased. The majority of species captured during this study are adept at burrowing into the sandy soils of the region. Lack of reliance on CWD may be the result of herpetofaunal adaptation to the longleaf pine (*Pinus palustris*) ecosystem that historically dominated the upland areas of my study area.

Key words: Amphibian; Coarse woody debris; Coastal plain; Herpetofauna; Longleaf; *Pinus palustris*; Reptile; Topographic variables

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1. Introduction

The importance of coarse woody debris (CWD) as an ecosystem component is a topic of increasing concern. Most studies examining the value of CWD were conducted in the Appalachian Mountains or old-growth forests of the Pacific Northwest, while similar studies in the southeastern Coastal Plain are lacking (Harmon et al., 1986; McCay et al., 2002; McMinn and Crossley, 1996; Whiles and Grubaugh, 1996). Average volumes of CWD in the Pacific Northwest approach 500 m^3/ha, while unmanaged forests in South Carolina contain roughly 17.5 m^3/ha (McMinn and Hardt, 1996; Spies and Cline, 1988). Low CWD volume in southeastern forests is a product of short rotation lengths, high humidity (which increases decay rate), and silvicultural practices (McCay et al., 2002; Sharitz et al., 1992). Recent advances in wood utilization technology and an emerging biofuels market (i.e., wood chips, wood pellets, and cellulosic ethanol) have the potential to further decrease the amount of woody material left in forests following timber harvests (Bies, 2006; Butts and McComb, 2000; Carey and Johnson, 1995; Hewett et al. 1981; Hunter, 1990; Maser and Trappe, 1984; Maser et al., 1988). For example, Westbrook et al (2007) found that chipping logging residues and understory stems produced competitively priced quality energy chips, while reducing site preparation costs and increasing plantable area.

Presence of CWD within forested ecosystems provides multiple benefits to herpetofauna communities. CWD sustains stable temperature and moisture regimes, providing incubation sites and protection from desiccation (Boddy, 1983; Graham, 1925; Jaeger, 1980; Maser and Trappe, 1984; Maser et al., 1988; Whiles and Grubaugh, 1996). CWD provides an invertebrate prey base for many amphibian and reptile species in the form of larvae, termites, centipedes, earthworms, and other arthropods which are consumed by small terrestrial snakes, toads, frogs, lizards, and salamanders (Conant and Collins, 1998; Graham, 1925; Whiles and Grubaugh, 1996). Lizards use downed logs and snags for courtship displays and, along with snakes, egg deposition and aestivation sites (Cooper and Vitt, 1994; Martof et al., 1980; Vitt and Cooper, 1986; Whiles and Grubaugh, 1996). CWD also provides habitat for small mammals including voles, shrews, and mice, providing prey for larger snake species such as rat snakes, racers, coachwhips, kingsnakes, pine snakes, and pit vipers (Gibbons and Dorcas, 2005; Martof et al., 1980; Loeb, 1999; Maser and Trappe, 1984; Tallmon and Mills, 1994).

2. Objectives

My study was a continuation of a 2-phase long-term project investigating the importance of CWD as a habitat component in managed upland pine stands in the southeastern Coastal Plain, representing years 6 and 7 of Phase II. Specifically, the study was designed to assess the response of herpetofauna communities to the removal and addition of CWD (both downed and standing) as compared to control stands. I examined the potential significance of CWD decay states within treatments in the context of earlier findings within the scope of the long-term project.

Capture trends from previous years of the long-term project indicate that amphibian and reptile communities do not rely strongly on the presence of CWD (Owens et al., 2008). However, snake communities did show a positive response to removal treatments during years 2-4 of Phase II (2003-05). Owens et al. (2008) suggested the lack of overall response was due to the burrowing capabilities of southeastern herpetofauna as an adaptation to low levels of CWD. While this may be true, the progression of decomposition yields CWD capable of holding more moisture and with a more diverse and abundant invertebrate community (Boddy, 1983; Graham, 1925; Maser and Trappe, 1984). Therefore, the change in decay state of wood during my study compared to previous years had the potential to impact herpetofauna use of CWD within treatment plots. However, given the consistency of results from previous years of the long-term project, I did not expect anuran, salamander, or lizard abundance, species richness, or diversity to differ among treatments. I also did not expect snake capture results to change from the findings of the most recent trapping effort (2003-2005).

Amphibians are more sensitive to fluctuations in environmental moisture than reptiles (Bell and Donnelly, 2006; Spight, 1968; Spotila and Berman, 1976) with activity peaks often associated with precipitation events (Jaeger, 1980; Semlitsch, 1985; Timm et al., 2007; Vasconcelos and Calhoun, 2004). I therefore predicted amphibian captures to positively correlate with rainfall, but did not expect to see a relationship between reptile activity and rain events. I also expected the correlation between amphibians and rainfall to be strongest where CWD had been removed, assuming the absence of moisture-holding logs restricts movement to precipitation events.

Variations in topographic features such as slope, aspect, and relative elevation affect soil moisture (Famiglietti et al., 1998). These variables can influence soil moisture not only in areas with high levels of topographic variability, but also on sites with low levels of relative relief. Famiglietti et al. (1998) found that aspect and elevation significantly affected soil moisture levels on a landscape with ≤ 10.6 m of total relief. Evaporative water loss has been recognized as an important factor in the adaptation of amphibians and reptiles to their surrounding environment (Spotila and Berman, 1976). Accordingly, I quantified slope, elevation, aspect, and distance to the nearest stream and incorporated those metrics into analyses to account for topographic variables that may affect site moisture.

3. Study area

Study plots were located on the Savannah River Site (SRS), a 78,000-ha National Environmental Research Park administered by the Department of Energy (DOE). The SRS is located in the Upper Coastal Plain and Sandhills physiographic region of South Carolina (White, 2005). This region is characterized by sandy soils and gently sloping hills dominated by pines with scattered hardwoods (Kilgo and Blake, 2005). The climate of the SRS is humid subtropical with mean annual temperature and rainfall of 18°C and 122.5 cm, respectively (Blake et al., 2005).

When the DOE acquired the SRS in 1951, old-field habitats dominated the site as a result of past land-use activities driven by crop and cotton production, as well as timber harvests (White, 2005). Today, the majority of the site has been reforested by the U. S. Forest Service (Imm and McLeod, 2005; White, 2005). Approximately 68% of the SRS is composed of upland pine stands, including loblolly (*Pinus taeda*), slash (*P. elliottii*), and longleaf (*P. palustris*) pines. While much of the land is managed for timber production, nearly two-thirds of the pine forests on SRS are 40-70 years old (Imm and McLeod, 2005).

Study plots were located in three loblolly pine stands planted between 1950 and 1953. Although the overstory in these stands was dominated by loblolly pine, slash and longleaf pine also were present. Understory vegetation was dominated by sassafrass (*Sassafrass albidum*), black cherry (*Prunus serotina*), lespedeza (*Lespedeza* spp.), blackberry (*Rubus* spp.), and poison oak (*Toxicodendron pubescens*).

4. Methods

Study design was a randomized complete block, with each of 4 treatments randomly assigned within each of the 3 forest stands (blocks). Blocks were chosen based on the following criteria: approximately 45-year-old loblolly pine plantations (at project initiation in 1996); ≥ 76 m from the nearest wetland, road, or power line; and large enough to accommodate four 9.3-ha square plots. Treatments were 1) control, where downed CWD was not manipulated; 2) snag, where standing CWD volume was increased ten-fold; 3) removal, where all downed CWD ≥10 cm in diameter and ≥ 60 cm in length was removed; and 4) downed, where volume of downed CWD was increased five-fold. The control and removal treatment plots were initiated in 1996 (Phase I), while the downed and snag treatment plots were implemented in 2001 (PhaseII). Annual removal of CWD ≥ 10 cm in diameter and ≥ 60 cm long was performed in the removal treatment plots. Each treatment plot consisted of a 6-ha core trapping area, surrounded by a 3.3-ha buffer zone subject to the same treatment to minimize edge effect. All 12 plots were thinned in 2001 to a live pine basal area of between 13.8 – 20.8 m^2/ha, and were prescribed burned in summer 2004.

Downed woody debris measurements were taken from 2005-08 to measure change in decay state during and 2 years prior to my study. Inventories were conducted in randomly selected subplots (50 x 50 m) within the inner 4 ha of each treatment plot. Within each subplot, all logs with at least 50% of their measurable length within the subplot were measured, and logs ≥ 10 cm in diameter at the midpoint and ≥ 60 cm in length were included in the inventory. Logs were classified into 1 of 5 decay categories based on the Maser et al. (1979) decay scale, where stage 1 logs were sound, with intact bark; stage 2 logs had mostly sound wood with some bark starting to flake; stage 3 logs had broken branches and were missing bark; stage 4 logs were soft and blocky in texture; and stage 5 logs were powdery in texture and partly buried. Log volumes were estimated assuming logs were round in circumference, which may overestimate the true volume of downed wood.

Herpetofauna sampling was conducted using pitfall drift fence arrays. Drift fences consisted of aluminum flashing buried approximately 15 cm below ground, with 19-liter plastic buckets buried flush to the ground against each fence. Each plot contained one cross-shaped array with four 30-m arms extending out from the center of the plot in each of the cardinal directions, and four Y-shaped arrays with three 15-m arms located in each corner of the 6-ha core sampling area (Fig. 3.1). Pitfall traps (buckets) were maintained with 2.5 to 5 cm of soil in the bottom to provide captured animals cover from temperature extremes and desiccation during trapping periods. The bottoms of buckets were perforated, allowing drainage of excess water after heavy rains.

Herpetofauna were sampled in all plots for 14 days each season from January 2007 – August 2008, for a total of 7 sampling seasons, during which traps were checked daily between 0700 and 1700. All captured individuals were identified to species, measured for snout-vent length (mm), and age (juvenile versus adult) was determined if possible. The array of capture and direction of movement (for cross-shaped arrays only) were noted for each animal captured (direction of movements for winter 2007 and the first five days of the spring 2007 sample seasons were not recorded). Rainfall data were collected from a weather station approximately 4 km from study plots.

I quantified topographic variables with the potential to influence herpetofauna abundance by affecting site moisture. Trap array locations were recorded using a Trimble GeoExplorer 3 handheld GPS unit and differentially corrected using data from a Continually Operating

Reference Station (CORS) in Columbia County, Georgia (63 km from the study site). Array locations for Y-shaped arrays were taken at the middle bucket and array locations for each arm of the cross-shaped array were taken at the middle of each array arm. Y-shaped arrays and each arm of the cross-shaped array were considered separate arrays, yielding a total of 8 arrays with capture data per plot. Buffers with a radius of 15.5 m were created around array locations (Fig. 3.1) using home range estimates for Fowler's toad (*Bufo woodhousii fowleri*) reported by Clarke (1974). No reliable home range estimates are available for southern toad (*Bufo terrestris*), the most commonly captured anuran in my study. Thus, we assumed a home range size similar to Fowler's toads because of their similar habitat requirements and life histories (Buhlmann et al., 2005). The buffer size chosen encompasses home range size of most amphibians and reptiles captured in the study for which home range sizes are known. Mean elevation, degree of slope, and aspect were calculated for each buffered area using the Zonal Statistics tool in the Spatial Analyst extension of ArcGIS 9.2 (Environmental Systems Research Institute). All pixels (and their associated values) with ≥50% of their area within the buffer boundary were used for calculations. Distance to the nearest stream was calculated from the actual array location point (center point of the buffer). All stream orders were considered although intermittent streams may not have contained water during this study.

5. Statistical Analysis

Amphibian and reptile abundance, species diversity, and species richness were calculated for each Y-shaped array and each arm of cross-shaped arrays to allow inclusion of topographic variables as covariates in analyses. Species diversity was calculated using the Shannon-Weiner species diversity index (H'; Pielou, 1977). Values were standardized by dividing by length of fencing within each array. Treatment differences were examined using 2-way analysis of covariance (ANCOVA) with slope, elevation, aspect, and distance to nearest stream as covariates. All data were tested for normality using Shapiro-Wilks test. Non-normal data were ranked and ANCOVA was performed on ranks. Significant results were further analyzed using adjusted least square means pairwise comparisons. I used forest stand as the block for all ANCOVA analyses and SAS 9.1 (SAS Institute 2008) to perform all statistical analyses.

Spearman correlation coefficient was used to examine the effect of precipitation on amphibian and reptile captures within each treatment. Total number of each species captured per sample night was compared with the amount of precipitation 24- and 48-hours prior to capture.

Directional data were analyzed to determine if a significant portion of amphibians captured were migratory, as migration events might skew treatment responses. Because anurans made up 78% of all amphibian captures, movement was analyzed for this taxonomic group separately. Movement direction was assumed to be associated with the drift fence side of capture. For example, if an individual was captured on the west side of a drift fence arm, direction of movement was assumed to be east. A chi-square test was used to test for equal captures among drift fence sides. No significant difference in the number of captures per drift fence side was assumed to suggest movement direction was a random occurrence and not associated with migratory patterns. As migratory movements are generally seasonal, captures for both years were combined for each season. Total seasonal captures, as well as seasonal captures per block were analyzed, as the relative proximity of each block to potential breeding sites could influence animal movement among blocks. Small sample sizes yielded expected values <5 and chi-square tests underestimate *P*-values under these circumstances. Therefore, Fisher's exact test was used to generate *P*-values for significance testing (Sokal and Rohlf, 1995; Zar, 1999).

6. Results

Mean volume (± SE) of downed CWD in 2007 was 59.4 m^3/ha (± 7.9) in downed plots, 34.7 m^3/ha (± 6.3) in snag plots, and 12.7 m^3/ha (± 1.9) in control plots (Fig. 3.2). Downed CWD volume for removal plots was not measured in 2007, but was 0.29 m^3/ha (± 0.14) and 0.24 m^3/ha (± 0.15) in 2005 and 2006, respectively. Mean decay states of logs in downed, control, and snag plots in 2007 were 3.1, 3.3, 3, respectively (Fig. 3.3).

Capture totals over 7 trapping seasons from January 2007 - August 2008 were 355 amphibians and 668 reptiles representing 11 and 25 species, respectively. Toads made up the majority (64.8%) of amphibian captures, while salamanders and frogs represented 22% and 13.2%, respectively (Table 3.1). Lizards made up the majority of reptile captures (77.2%) with snakes making up the remainder (22.8%) (Table 3.2).

Abundance, species richness, and species diversity were similar among treatments for anurans, salamanders, and lizards ($P > 0.05$). Snake abundance ($F_{3,86} = 3.07$, $P = 0.0321$), species richness ($F_{3,86} =2.91$, $P = 0.0392$), and species diversity ($F_{3,86} =3.39$, $P = 0.0216$) were higher in removal than downed addition plots (Table 3.3). Anuran abundance was influenced by block ($F_{2,86} = 3.62$, $P = 0.031$) and distance to nearest stream ($F_{1,86} = 6.71$, $P = 0.0112$; Fig. 3.4).

Daily captures for anurans and salamanders were positively correlated with rainfall 24 hours prior to capture in all treatment plots with strongest correlations occurring in removal plots (Table 3.4). Daily captures for anurans and salamanders also were positively correlated with rainfall 48 hours prior to capture in control and downed addition plots, respectively ($r = 0.2143$, $P = 0.0341$; $r = 0.2443$, $P = 0.0154$; Table 3.4). Lizard captures were negatively correlated with rainfall 48 hours prior to capture in control and snag plots ($r = -0.2587$, $P = 0.0101$; $r = -0.3916$, $P < 0.0001$). Amphibian and anuran movement direction did not differ ($P > 0.05$) for total seasonal captures or seasonal captures per block, suggesting that directional movement within each treatment plot was not biased by seasonal migratory movements.

7. Discussion

The lack of a treatment response by amphibians in my study was consistent with trends from the previous 10 years (encompassing both Phases I and II) of the long-term project (McCay et al., 2002; Owens et al., 2008). Ornate chorus frogs (*Pseudacris ornata*), eastern spadefoot toads (*Scaphiopus holbrookii*), *A. talpoideum*, and *B. terrestris* together comprised 71% of amphibian captures. These species are fossorial in nature and prefer sandy soils to accommodate burrowing activities (Brown and Means, 1984; Carr, 1940; Martof et al., 1980; Semlitsch, 1983). While it has been shown that invertebrate abundance can be positively influenced by the presence of CWD (Jabin et al., 2004), sampling within study plots 5 years following initiation of removal treatments showed no significant effect of CWD removal on arthropod abundance (Hanula et al., 2006). Further, invertebrate communities in southeastern pine stands can be highly abundant on the forest floor (Hanula and Wade, 2003). Therefore, given adequate prey and sandy soils, amphibians in the region may have adapted to avoid desiccation and predation by burrowing rather than relying solely on CWD for cover and invertebrate production.

Snakes were the only group of herpetofauna for which I demonstrated a treatment response. Snake abundance was higher in removal than in downed addition plots, consistent with results from years 2-4 (2003-2005) of Phase II (Owens et al., 2008). I also found higher snake diversity and richness in removal compared to downed addition plots, unlike Owens et al. (2008) who reported no difference in diversity among treatments and lower species richness in snag plots. Small, litter-dwelling terrestrial snakes, including the smooth earth snake (*Virginia* *valeriae*), southern ringneck snake (*Diadophis punctatus*), scarlet kingsnake (*Lampropeltis triangulum*), brown snake (*Storeria dekayi*), red-bellied snake (*S. occipitomaculata*), scarlet snake (*Cemophora coccinea*), and southeastern crowned snake (*Tantilla coronata*) constituted almost 82% of all snakes captured during my study. These species commonly forage for soft-bodied arthropods, earthworms, slugs, and spiders under leaf and pine litter, but also can be found in decaying logs and stumps and under rocks (Gibbons and Dorcas, 2005; Martof et al., 1980). *T. coronata* was the most frequently captured snake species during my study, representing 49.3% of all snake captures. *T. coronata* is considered the most common species of small snake in the region and is widely distributed across the area historically dominated by longleaf pine forests (Todd and Andrews, 2008). *T. coronata* and other small litter-dwelling snakes with similar life histories may prefer managed stands that mimic the open understory of longleaf pine stands to which they may be historically adapted (Todd and Andrews, 2008) while avoiding stands with structurally diverse midstories, such as those found in downed addition, control, and snag plots. Furthermore, it is possible that snakes in removal plots range farther in search of invertebrate prey and adequate nesting sites while snakes in downed addition plots centralize their movements around downed CWD for foraging and nesting activities. Increases in movement rate and distance in removal plots would increase the possibility of capture, thus explaining higher abundance and diversity levels seen in those plots compared to downed addition plots.

The inadequacy of pitfall traps in capturing all snake species present within plots may have influenced snake capture results. For example, while 19-l buckets are capable of capturing the young-of-year (YOY) for some large-bodied snake species (e.g. racers, rat snakes, kingsnakes, coachwhips, and pitvipers), adults easily escape (Todd et al., 2007). Todd et al.

(2007) determined that drift fences using a combination of funnel traps and 19-l plastic buckets yielded the greatest number of individual captures. Therefore, snake species richness, diversity, and abundance estimates based solely on pitfall trap captures may underestimate true values.

Contrary to results from years 2-4 (2003-05) of Phase II (Owens et al., 2008), I did not find lower reptile richness and diversity in snag plots. Structural diversity within snag plots has increased over time as a result of snags falling due to high winds and snag senescence (Figure 3.2). Increased CWD volume and structure in snag plots between 2006-08 may have breached a critical threshold value, resulting in a positive reptile response to additional foraging, display, and basking sites. Further, position (standing versus downed) of CWD can affect wood-eating insect communities (Hanula, 1996). Therefore, the falling of snags could have provided reptiles with additional invertebrate prey during 2007-08 that was unavailable in previous years when snags were standing.

My results suggest that reptile communities in the southeastern Coastal Plain are not strongly affected by soil moisture. None of the topographic variables included as covariates in the analysis influenced treatment response for any reptile group. Furthermore, reptile captures were negatively correlated with rainfall. The evolution of scales in the epidermis of reptiles reduces water loss (Zug et al., 2001), and likely explains their independence from moisture-regulating habitat variables and weather conditions. Conversely, the positive correlation I observed between amphibian captures and precipitation is a well documented phenomenon driven by physiological requirements (Carr, 1940; Semlitsch, 1981; Spight, 1968; Spotila and Berman, 1976). Further, correlation coefficient (*r*) values suggest that amphibian movement is more strongly dictated by rain events in removal plots, possibly due to the lack of moisture-holding CWD on the forest floor. By accounting for topographic variability within treatment plots, any differences observed in amphibian and reptile captures among treatments are likely real differences and not an artifact of topography.

8. Conclusion

My results suggest that amphibians and reptiles in the southeastern Coastal Plain do not rely heavily on CWD as a habitat component. Lack of response may be a result of adaptation to the longleaf pine ecosystem that historically dominated the upland areas of my study area (White, 2005). The frequent fires that maintain longleaf pine forests, coupled with increased decomposition rates due to high humidity, yield ecosystems low in structural diversity (McCay et al., 2002; Sharitz et al., 1992). However, the sandy soils of the region provided herpetofauna an avenue for adapting to the paucity of forest floor structure. The majority of species captured during this study are adept at burrowing into the sandy forest floor below the litter layer (Carr, 1940; Pearson, 1955; Pearson, 1957; Semlitsch, 1981; Semlitsch, 1983). A thick layer of pine and leaf litter provides invertebrate prey, protection from predators, and a buffer from temperature and moisture extremes (Geiger et al., 1995; Hanula and Wade, 2003). Utilizing a combination of existing burrows, ground litter, decomposing root systems, and rotting logs and stumps, amphibian and reptile communities in the Southeast appear able to thermoregulate and mitigate moisture loss, as well as survive periodic ground fires, without relying completely on CWD (Gibbons and Dorcas, 2005; Martof et al., 1980).

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Table 3.1. Total number of amphibians captured using drift fence pitfall arrays in control (n = 3), downed (n = 3), removal (n = 3), and snag (n = 3) treatment plots in upland loblolly pine (*Pinus taeda*) stands in Barnwell County, South Carolina, 2007-08.

	Treatment				
Species	Control	Downed	Removal	Snag	Total
Ambystoma tigrinum (Tiger salamander)	0	0	1	0	1
Ambystoma talpoideum (Mole salamander)	15	14	14	26	69
Bufo terrestris (Southern toad)	10	22	30	20	82
Gastrophryne carolinensis (Eastern narrowmouth toad)	28	18	18	24	88
Hyla gratiosa (Barking tree frog)	0	0	1	0	1
Hyla sp. (unidentifiable hylid species)	1	0	2	0	3
Plethodon chlorobryonis (Atlantic coast slimy salamander)	3	0	2	1	6
Pseudacris ornata (Ornate chorus frog)	3	4	13	22	42
Pseudotriton ruber (Southern red salamander)	0	0	2	0	2
Rana sp. (unidentifiable ranid species)	1	0	0	0	1
Schaphiopus holbrookii (Eastern spadefoot toad)	16	14	10	20	60
Total	77	72	93	113	355

Table 3.2. Total number of reptiles captured using drift fence pitfall arrays in control (n = 3), downed (n = 3), removal (n = 3), and snag (n = 3) treatment plots in upland loblolly pine (*Pinus taeda*) stands in Barnwell County, South Carolina, 2007-08.

	Treatment				
Species	Control	Downed	Removal	Snag	Total
Agkistrodon contortrix (Copperhead)	0	1	0	0	1
Anolis carolinensis (Green anole)	21	30	28	10	89
Cemophora coccinea (Scarlet snake)	1	3	7	2	13
Cnemidophorus sexlineatus (Six-lined racerunner)	12	3	16	13	44
Coluber constrictor (Black racer)	2	0	3	0	5
Diadophis punctatus (Southern ringneck snake)	1	0	0	0	1
Elaphe guttata (Corn snake)	1	1	0	1	3
Elaphe obsoleta (Gray rat snake)	1	0	2	0	3
Eumeces fasciatus (Five-lined skink)	13	15	4	30	62
Eumeces inexpectatus (Southeastern five-lined skink)	4	10	3	3	20
Eumeces laticeps (Broadhead skink)	10	17	5	21	53
Eumeces sp. (unidentifiable skink species)	2	1	1	2	6
Heterodon platyrhinos (Eastern hognose snake)	1	0	0	0	1
Heterodon simus (Southern hognose snake)	2	1	2	1	6
Lampropeltis triangulum (Scarlet kingsnake)	0	0	1	0	1
Nerodia fasciata (Banded water snake)	0	0	2	2	4
Nerodia floridana (Florida green water snake)	0	0	1	0	1
Scincella lateralis (Ground skink)	11	5	11	6	33
Sceloporus undulatus (Eastern fence lizard)	30	59	67	53	209
Sistrurus miliarius (Pygmy rattlesnake)	1	0	0	0	1
Storeria dekayi (Brown snake)	1	0	0	0	1
Storeria occipitomaculata (Red-bellied snake)	3	1	1	4	9
Tantilla coronata (Southeastern crowned snake)	21	9	32	13	75
Thamnophis sauritus (Eastern ribbon snake)	1	0	0	0	1
Thamnophis sirtalis (Common garter snake)	0	0	1	1	2
Virginia valeriae (Smooth earth snake)	5	9	8	2	24
Total	144	165	195	164	668

Table 3.3. Mean (SE) number of captures, species richness, and species diversity per m of drift fence for all herpetofauna and taxonomic groups captured in control, downed, removal, and snag treatment plots from January 2007 – August 2008 in upland loblolly pine (*Pinus taeda*) stands in Barnwell County, South Carolina. Different letters indicate significant differences among treatments.

	Treatment			
	Control	Downed	Removal	Snag
Abundance	(*n* = 24)	(*n* = 24)	(*n* = 24)	(*n* =24)
Herpetofauna	0.23(0.03)	0.27(0.03)	0.33(0.03)	0.31(0.03)
Amphibians	0.07(0.02)	0.09(0.02)	0.11(0.01)	0.13(0.02)
Anurans	0.06(0.01)	0.07(0.01)	0.09(0.01)	0.09(0.01)
Salamanders	0.02(0.01)	0.02(0.01)	0.02(0.01)	0.03(0.01)
Reptiles	0.15(0.02)	0.18(0.02)	0.22(0.02)	0.18(0.02)
Lizards	0.01(0.12)	0.15(0.02)	0.15(0.02)	0.15(0.02)
Snakes	0.04(0.01)ab	0.03(0.01)b	0.07(0.01)a	0.03(0.01)ab
Richness				
Herpetofauna	0.16(0.02)	0.15(0.01)	0.19(0.01)	0.18(0.02)
Amphibians	0.05(0.01)	0.05(0.01)	0.08(0.01)	0.07(0.01)
Anurans	0.04(0.01)	0.04(0.01)	0.06(0.01)	0.06(0.01)
Salamanders	0.01(0.003)	0.01(0.003)	0.01(0.003)	0.01(0.003)
Reptiles	0.11(0.01)	0.09(0.01)	0.12(0.01)	0.10(0.01)
Lizards	0.07(0.01)	0.07(0.01)	0.07(0.01)	0.08(0.01)
Snakes	0.04(0.01)ab	0.02(0.01)b	0.04(0.01)a	0.02(0.01)ab
Diversity				
Herpetofauna	0.04(0.003)	0.04(0.003)	0.05(0.003)	0.05(0.003)
Amphibians	0.01(0.003)	0.02(0.003)	0.02(0.003	0.02(0.003)
Anurans	0.01(0.003)	0.01(0.003)	0.02(0.003)	0.02(0.003)
Salamanders	0.001(0.001)	-0.0002(0.001)	0.001(0.001)	0.001(0.001)
Reptiles	0.03(0.004)	0.03(0.003)	0.03(0.003)	0.03(0.003)
Lizards	0.02(0.003)	0.02(0.003)	0.02(0.003)	0.02(0.003)
Snakes	0.01(0.002)ab	0.003(0.002)b	0.01(0.002)a	0.004(0.002)ab

Table 3.4. Correlation coefficients (*r*) and *P*-values from Spearman correlation tests between the amount of precipitation 24- and 48-hours prior to capture in control (n = 3), downed (n = 3), removal (n = 3), and snag (n = 3) treatment plots for daily captures of all herpetofauana, amphibians, anurans, salamanders, reptiles, lizards, and snakes. Herpetofauna were captured using drift fences pitfall traps from January 2007 – August 2008 in upland loblolly pine stands at the Savannah River Site in Barnwell County, South Carolina.

	Control		Downed		Removal		Snag	
	r	*P*-value	*r*	*P*-value	*r*	*P*-value	*r*	*P*-value
24-hours prior to capture								
Herpetofauna	0.037	0.719	0.084	0.409	0.147	0.150	0.207	0.040*
Amphibians	0.370	<0.0001*	0.388	<0.0001*	0.452	<0.0001*	0.433	<0.0001*
Anurans	0.309	0.002*	0.331	< 0.0001*	0.518	<0.0001*	0.453	<0.0001*
Salamanders	0.267	0.008*	0.213	0.035*	0.366	0.0002*	0.281	0.005*
Reptiles	-0.199	0.049*	-0.106	0.297	-0.160	0.116	-0.105	0.305
Lizards	-0.175	0.085	-0.070	0.491	-0.120	0.240	-0.087	0.395
Snakes	-0.119	0.242	-0.147	0.149	-0.090	0.380	0.020	0.845
48-hours prior to capture								
Herpetofauna	-0.171	0.093	-0.120	0.240	-0.314	0.002*	-0.128	0.208
Amphibians	0.188	0.064	0.180	0.076	0.015	0.884	0.142	0.163
Anurans	0.214	0.034*	0.049	0.630	-0.017	0.866	-0.073	0.478
Salamanders	-0.027	0.789	0.244	0.015*	-0.047	0.642	0.137	0.178
Reptiles	-0.256	0.011*	-0.228	0.024*	-0.391	<0.0001*	-0.198	0.051
Lizards	-0.259	0.010*	-0.184	0.070	-0.392	<0.0001*	-0.174	0.086
Snakes	-0.112	0.273	-0.174	0.087	-0.137	0.179	-0.169	0.097

*indicates a significant correlation at $\alpha = 0.05$.

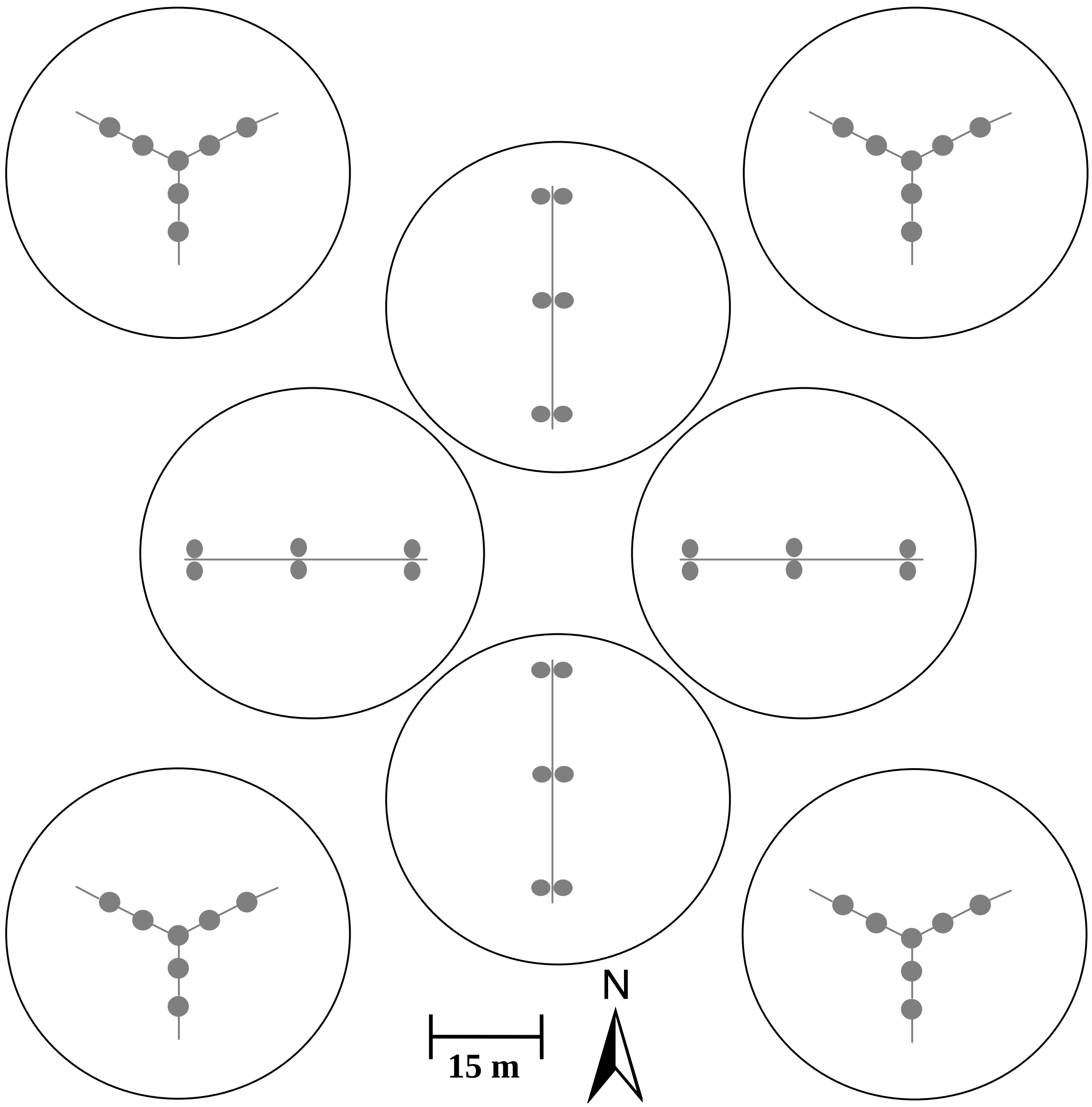

Figure 3.1. Arrangement of drift fence arrays and bucket traps used for sampling herpetofauna on 6-ha core area of a 9.3-ha treatment plot in an upland loblolly pine (*Pinus taeda*) stand at the Savannah River Site, Barnwell County, South Carolina. Also shown are buffers within which topographic variables were measured for covariate analyses.

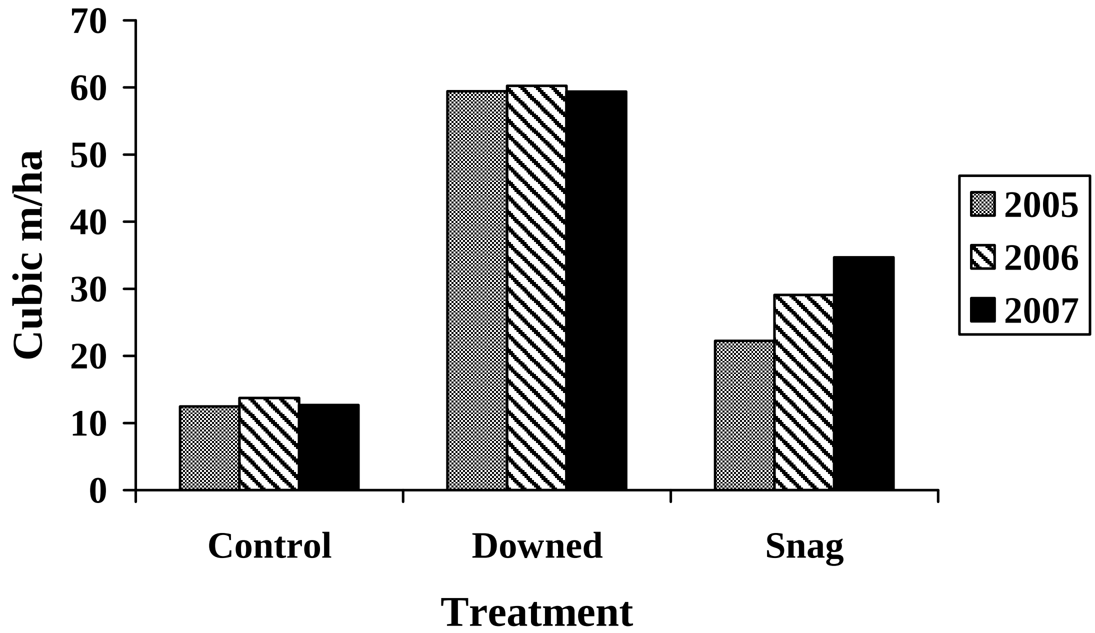

Figure 3.2. Mean volume of down woody debris accumulations over three years in control (*n* = 3), downed addition (*n* = 3), removal (*n* = 3), and snag (*n* = 3) treatment plots in loblolly pine (*Pinus taeda*) stands on the Savannah River Site, Barnwell County, South Carolina.

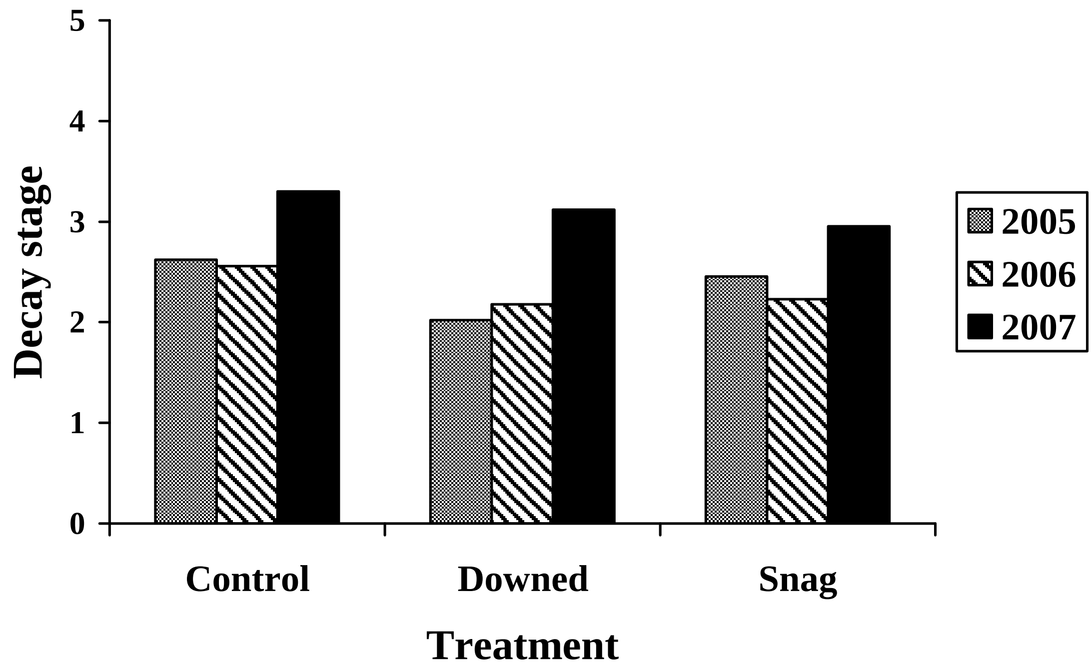

Figure 3.3. Mean decay stage of downed wood, based on Maser et al. (1979), over three years in control (n = 3), downed addition (n = 3), and snag (n = 3) treatment plots in loblolly pine (*Pinus taeda*) stands on the Savannah River Site, Barnwell County, South Carolina

CHAPTER 4

CONCLUSIONS AND MANAGEMENT RECOMMENDATIONS

The southeastern United States produces more timber than any other country in the world, and timber market models forecast that production in the South will increase by >50% between 1995-2040 (Wear and Gries 2002). Pine plantation management generally results in uniform, even-aged stands lacking structural diversity (Hansen et al. 1991). Coarse woody debris (CWD) is thought to promote biological diversity by maintaining the structural heterogeneity that naturally occurs in unmanaged forests (Hansen et al. 1991; Hunter 1990). With 15% of southeastern forests in pine plantations (Conner and Hartsell 2002), the potential for declines in biodiversity is an issue of increasing concern. The objective of my study was to continue a long-term investigation of the importance of CWD as a habitat component for shrew and herpetofauna communities within managed pine stands in the southeastern Coastal Plain.

In general, shrew abundance was higher in plots with increased volumes of downed CWD. Adequate moisture and prey availability have been proposed as the 2 most important factors governing the distribution of shrews (Getz 1961). Wood in advanced stages of decay has a higher moisture holding capacity (Boddy 1983), and invertebrate communities respond positively to increased moisture levels (Graham 1925). Increases in these 2 critical CWD functions manifested through the decomposition process may explain why my study demonstrated differences among treatments while results from previous years were relatively inconclusive. CWD within treatment plots during early years of the long-term project may not have reached a level of decay that functions to improve habitat quality or prey availability. Therefore, management strategies that retain CWD and allow it to reach advanced decay states may yield the habitat component that provides the 2 most important factors contributing to shrew habitation and proliferation.

Herpetofauna capture results indicated that downed wood, at least in the current stage of decay, is not a critical habitat component for amphibians and reptiles. In fact, snake abundance, diversity, and richness were positively influenced by the removal of CWD. The southeastern Coastal Plain was historically dominated by longleaf pine forests characterized by an open canopy and grassy understory perpetuated by frequent ground fires (White 2005; Landers et al. 1995). Many of the species captured exhibit burrowing behavior (Brown and Means 1984; Carr 1940; Gibbons and Dorcas 2005; Martof et al. 1980; Semlitsch 1983), which may be an adaptation to an ecosystem with low levels of CWD and periodic removal of ground cover through fire. While results suggest that retention of CWD into the middle stages of decay (mean decay stage of CWD in treatment plots was 3.1 on a 5.0 scale) does not affect herpetofauna communities, the effect of CWD in later stages of decomposition is unknown. Coarse woody debris in various stages of decay offers functionally different benefits (Hayes and Cross 1987). Therefore, continuation of the long-term project until decay state of CWD reaches its maximum level should be considered to fully understand its importance and provide managers with complete data upon which to base management decisions.

Sustainable forest management in the United States is evolving from a production-oriented system to one encompassing a more holistic view of the impacts of resource use on ecosystem health and biodiversity (Sharitz et al. 1992). Rising petroleum costs and recent advances in wood utilization technology have spurred research into alternative energy products.

The result is an emerging biofuels market (i.e. wood pellets and cellulosic ethanol) that has the potential to further decrease the amount of woody material left in forests following timber harvests (Bies 2006; Butts and McComb 2000; Carey and Johnson 1995; Hewett et al. 1981; Hunter 1990; Maser and Trappe 1984; Maser et al. 1988). Therefore, forest managers must understand the value of CWD as an ecosystem component to maintain economically productive forests while conserving biological diversity.

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