

wildlife management

Effects of Repeated Burning on Snag Abundance in Shortleaf Pine Woodlands

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Forest managers are restoring and maintaining forest woodlands across substantial areas of the United States, and these efforts typically require the use of frequent prescribed fire. The effects of frequent prescribed fire on important habitat components such as snags remain unknown. We conducted a study to determine how snag densities are affected by repeated prescribed burning in stands being restored to shortleaf pine (*Pinus echinata*) woodlands in the Ouachita Mountains of western Arkansas during summer 2015. We assessed snag densities in woodland forest stands that had 1, 2, 3, and 4–6 burns ($n = 6$ stands in each burn class) after overstory thinning and midstory removal. In woodlands, density of large snags (≥ 15 cm dbh) was not significantly related to the previous number of burns, but density of small snags (5–14.9 cm dbh) was lower in stands with greater numbers of burns. Repeated burning did not appear to result in lower numbers of large snags, which are an important habitat component for numerous wildlife species.

Keywords: Arkansas, fire, Ouachita, *Pinus echinata*, shortleaf pine

Frequent fire historically maintained a mosaic of forests, grasslands, savannas, and open woodlands dominated by pine (*Pinus* spp.) or oak (*Quercus* spp.) throughout the eastern United States (e.g., Abrams 1992, Waldrop et al. 1992, Lorimer 2001). Across the southeastern United States, fires maintained pine woodlands, which provided unique habitat for a number of species, including the red-cockaded woodpecker (*Picoides borealis*). Pine woodlands are characterized as having a relatively open overstory dominated by large diameter, mature pines with little or no mid-story and abundant herbaceous flora in the understory. Herbaceous understory vegetation is maintained and woody stem growth is retarded by frequent (1- to 5-year interval) low-intensity burns. To restore forest landscapes to these more historic, open-forest conditions (Waldrop et al. 1992, Lorimer 2001), land managers are currently implementing woodland and savanna restoration programs throughout North America (e.g., Perry et al. 2009). In response to the decline of the red-cockaded woodpecker and other species associated with pine woodlands, large tracts of forestlands in the southeastern United States are now being managed as fire-adapted woodlands, including $>800,000$ ha of national forestlands (e.g., Bowman et al. 1999). Restoration of these woodlands typically involves thinning of forest overstories, reducing or removing of midstories, and frequent prescribed burning. The effect of these restoration and

maintenance efforts on various taxa and their habitat is important information for forest managers.

Snags provide important roosting, nesting, and foraging substrate for numerous species of wildlife (Scott et al. 1977, Britzke et al. 2003, Perry and Thill 2007), and forest managers understand the importance of retaining snags for wildlife. Thus, land management agencies often have guidelines for providing target densities of snags during harvest entries (e.g., 5 snags/ha >30 cm dbh) (Ouachita National Forest 2005), and providing adequate snag densities is an important aspect of managing forests for wildlife. Nevertheless, various management actions, including prescribed fire, can negatively or positively affect snag availability across the landscape over time.

In stands with no recent history of fire, prescribed burns may initially create abundant snags by killing small trees and tree species that are not fire tolerant. For example, in ponderosa pine (*Pinus ponderosa*) forests of New Mexico, initial burns created many large (>47.5 cm dbh) snags and second burns consumed some of those, but additional burns did not eliminate the remaining snags (Holden et al. 2006). Comparable studies from southeastern forests dominated by shortleaf pine (*Pinus echinata*) are not available, and factors affecting snag dynamics differ in the Southeast compared with those in other regions. For instance, snag decay rates in the Southeast are

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Table 1. Attributes of forest stands where snag density was measured in the Ouachita Mountains of Arkansas, 2015, including number of previous burns, stand age, area, and dates of previous burns.

No. burns	Stand	Age (yr)	Area (ha)	Burn date ¹
1	C849S16	98	23	4/18/2014
1	C1236S11	86	43	3/25/2011
1	C1247S15	105	24	4/12/2010
1	C1265S1	100	13	3/1/2012
1	C1282S33	95	37	1/18/2009
1	C1282S24	95	11	1/25/2009
2	C800S10	74	53	3/19/2011, 4/23/2014
2	C801S12	89	39	2/26/2012, 3/29/2015
2	C1248S4	93	26	4/16/2009, 5/10/2013
2	C1248S2	105	19	4/16/2009, 5/10/2013
2	C1249S20	82	20	4/16/2005, 4/22/2013
2	C1249S2	80	30	4/16/2005, 4/22/2013
3	C806S6	87	30	2/28/2010, 3/3/2011, 4/10/2014
3	C1056S2	87	46	2/22/2007, 3/6/2010, 3/7/2013
3	C1243S26	95	89	3/11/2004, 3/16/2007, 3/23/2011
3	C1257S32	95	39	2002, 1/8/2009, 3/27/2013
3	C1258S8	90	72	1999, 2/27/2004, 3/24/2008
3	C1264S1	102	44	2003, 3/16/2007, 4/1/2012
4	C1305S13	100	37	1998, 3/20/2005, 12/9/2006, 4/13/2010
5	C1251S20	85	89	1998, 2002, 1/30/2008, 3/6/2011, 6/7/2012
5	C1256S8	98	48	2000, 2002, 2/22/2007, 4/16/2009, 5/10/2013
5	C1256S12	101	32	2000, 2002, 2/22/2007, 4/16/2009, 5/10/2013
5	C1306S8	115	40	1998, 2000, 2/22/2004, 11/3/2006, 4/13/2010
6	C1259S10	86	50	1997, 2000, 2003, 6/30/2006, 4/8/2009, 4/22/2013

¹ For some burns, year of burn was available but exact date was not.

typically faster due to higher temperatures, greater humidity, and a longer growing season than in cooler and drier regions (Dickson et al. 1983, Moorman et al. 1999).

Little information exists on the effects of frequent prescribed burning on long-term snag abundance in shortleaf pine-dominated stands being restored to woodlands, and it is not known how the frequent burning necessary to maintain these woodlands affects snag density. Therefore, our objective was to determine whether successive burns over time result in significantly lower snag densities in stands being converted to shortleaf pine-dominated woodlands.

Methods

Study Sites

The study was conducted on the Poteau-Cold Springs and Mena-Oden Ranger Districts of the Ouachita National Forest. This area contains approximately 103,000 ha of forest designated for restoration and maintenance of shortleaf pine woodlands. The Ouachita Mountain ecological subregion extends from central Arkansas into eastern Oklahoma and consists of a series of east-to-west-oriented mountains. Throughout this region, elevation ranges from 100 to 800 m, mean annual precipitation ranges from 112 to 142 cm, mean annual temperature ranges from 16.0 to 17.0° C, and the growing season is 200–240 days (McNab and Avers 1994).

We selected stands that met our criteria (see below) and were in the process of being restored to pine woodlands or stands where restoration was considered complete. Before restoration, stands were typical of late-rotation, upland, mixed shortleaf pine-hardwood, second-growth forests found throughout the western Ouachita Mountains. These stands had shortleaf pine basal areas (BAs) of approximately 23.4 m²/ha (18.3 m²/ha of overstory and 5.1 m²/ha of midstory) and hardwood BAs of 7.5 m²/ha (2.2 m²/ha of overstory and 5.3 m²/ha of midstory) (Guldin et al. 1994). Because the Ouachita Mountains run east to west, pine-dominated stands

are generally limited to flat areas and southerly slopes, whereas hardwood stands dominate north slopes. Stands were mature (>66 years old) and relatively unmanaged, dominated by shortleaf pine, oaks, and hickories (*Carya* spp.), but included numerous other hardwood species such as blackgum (*Nyssa sylvatica*), maples (*Acer* spp.), and sweetgum (*Liquidambar styraciflua*) (Table 1).

Restoration of pine woodlands began with a wildlife stand improvement (WSI) harvest, whereby approximately 36% of the overstory pine and 80% of the overstory hardwood BA was removed, which left residual pine BAs of 13.7–16.1 m²/ha and residual hardwood BAs of 1.4–1.6 m²/ha. Retained trees were mature dominant or codominant overstory trees. Most midstory trees were felled, but a few smaller midstory species such as dogwood (*Cornus florida*) were retained for mast production (for wildlife). All study stands had been subjected to WSI before burning. Study stands ranged in size from 11 to 89 ha (Table 1) and were mostly situated on southerly aspects with slopes <20%. Overstory age of stands ranged from 74 to 115 years (mean ± SE = 93.5 ± 1.90 years) based on US Department of Agriculture Forest Service inventory data.

We assessed 6 stands each in 4 burn classes (24 stands total). Burn classes were (1) 1 previous burn, (2) 2 previous burns, (3) 3 previous burns, and (4) 4–6 previous burns (restored). Some stands ($n = 12$) contained ephemeral stream drainages in which water flow was usually limited to heavy rain events. Unthinned 15- to 50-m-wide buffer strips (greenbelts) were retained around these drainages for water quality protection. Greenbelts were primarily mixed pine-hardwood forest containing mature (≥ 50 years old) trees. No cutting or midstory removal was conducted in greenbelts, but they were burned when the surrounding woodlands were burned. We examined snag densities in greenbelts separately from the rest of stands to compare differences in snag density between areas subjected to WSI and burning with areas that were burned but not subject to WSI (greenbelts).

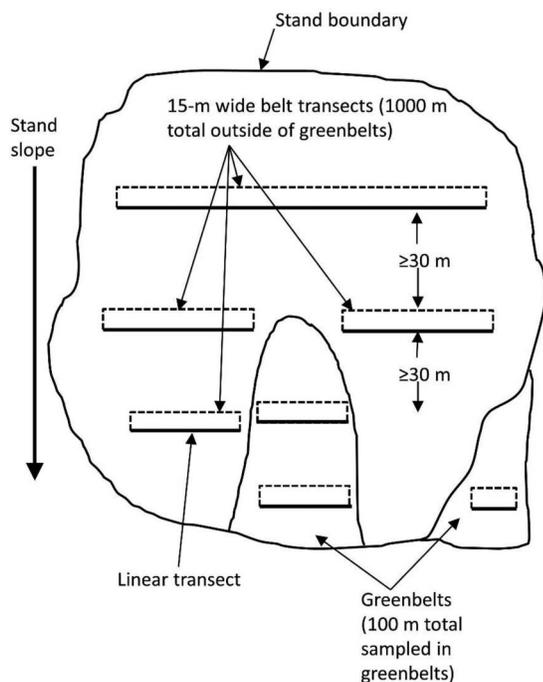


Figure 1. Stand and belt transect arrangement used for sampling snag densities in 24 woodland stands of the Ouachita Mountains of western Arkansas, 2015.

Burning

Most burns were conducted during the dormant season or early green-up (November to April), and most (96%) burns for which the exact burn date was recorded occurred between September 21 and May 10. However, two stands had burns that occurred in June, and the exact dates for 15 burns (24% of all burns) were not recorded other than year the burn occurred. Time elapsed between the last burn and snag measurements in our study was 4–87 months (mean \pm SE = 38.8 ± 4.5 months). For stands burned more than once previously, the interval between successive burns was 1–8 years (3.7 ± 0.3 years).

Snag Sampling

In each stand, multiple belt transects (15-m-wide) that ran perpendicular to the slope and were ≥ 30 m apart were established (Figure 1). The number of transects varied among stands based on the size and shape of individual stands. The total sample area in the thinned portion of each stand was 1.5 ha (1,000 m of linear transect \times 15 m wide). In stands that contained greenbelts, we surveyed an additional 100 m (0.15 ha) of transect in these greenbelts; these areas were compared with similarly burned adjacent woodlands.

From April to September 2015, we tallied all snags ≥ 5 cm dbh and ≥ 2 m tall within transects. Each snag was classified as pine or hardwood, and we measured dbh (cm) and height (± 0.25 m). We classified each snag into 1 of 4 decay classes:

- Class 1: Full height with branches and fine twigs, $\geq 90\%$ of bark remaining.
- Class 2: Some major branches remaining; top missing; $\geq 40\%$ of bark remaining.
- Class 3: No major branches remaining, but limb stubs may still be present; ≥ 2 m tall; more than one-half bole gone or trunk less than one-half of original diameter. Little bark remaining.
- Class 4: Advanced decay; ≥ 2 m tall; no bark left.

Analysis

Total snag density was calculated for each stand by combining all transects. Snag densities (number/ha) were compared with the number of previous burns (1–6) using linear regression. Densities were regressed separately for greenbelts and woodland portions of stands and by two snag diameter classes; small (5–14.9 cm dbh) and large (≥ 15 cm dbh). We selected these two size classes based on sizes of snags used for roosting by bats in the region and for minimum size needed for small cavity-nesting birds. For example, Perry et al. (2016) found 83% of Indiana bats in snags ≥ 15 cm dbh, and Perry and Thill (2007) found 80% of female and 64% of male northern long-eared bats roosting in this larger size class. Harlow and Guynn (1983) reported the minimum size of snags needed by small cavity-nesting birds was 15 cm dbh. Snag densities in adjacent greenbelts were compared with densities found in woodland areas using paired *t*-tests (each woodland paired with its imbedded greenbelts), stratified by the 2 snag diameter classes. We compared median decay class (classes 1–4) among the 4 burn classes (1, 2, 3, and ≥ 4 previous burns) using Kruskal-Wallis tests; we compared decay classes in greenbelts and stands separately.

Results

We recorded 906 snags in woodland portions of 24 stands and 133 snags in greenbelts located within 12 of those stands. Mean \pm SE diameter of snags in woodlands was 18.0 ± 0.4 cm (range, 5.0–72.0 cm) and diameter of snags in greenbelts was 14.5 ± 0.8 cm (range, 5.0–51.4 cm). Snags in woodland portions of stands were primarily hardwood (57%), followed by pine (38%) and cedar (5%) (Figure 2), which was similar to proportions in greenbelts (63, 27, and 9%, respectively).

Total density (all snags ≥ 5 cm) in woodlands was highly variable, ranging from 7.3 to 54.0 snags/ha (mean \pm SE = 25.2 ± 2.9 snags/ha) among stands, as were densities in greenbelts (range, 33.3–166.7 snags/ha; 73.9 ± 11.4 snags/ha). Mean density of all snags in greenbelts was 3 times greater than density in woodlands ($t_{11} = 3.97$, $P = 0.002$). Similarly, densities of large snags in greenbelts (25.0 ± 4.7 snags/ha) and small snags in greenbelts (48.9 ± 10.5 snags/ha) were greater than in woodlands (large = 11.7 ± 2.7 snags/ha, $t_{11} = -2.76$, $P = 0.018$; small = 11.8 ± 2.8 snags/ha, $t_{11} = 3.36$, $P = 0.006$).

In woodland portions of stands, density of large snags was not significantly related to the previous number of burns ($F_{1, 22} = 0.00$, $P = 0.976$) (Figure 3). However, density of small snags was significantly related to the previous number of burns ($F_{1, 22} = 7.03$, $P = 0.015$), with small snag density lower in stands with more burns (Figure 3). The strength of this relationship was moderate ($r^2 = 0.24$; standardized parameter estimate = -0.49), and the resulting predictive model was small snag density (number/ha) = $21.7 - 3.0x$, where x = the previous number of burns. In greenbelt portions of stands, there was not a significant relationship between the number of previous burns and density of large snags ($F_{1, 10} = 0.34$, $P = 0.576$) or small snags ($F_{1, 10} = 0.77$, $P = 0.400$).

The median decay class of snags ranged from 2.25 to 2.94 among the four burn classes in greenbelts and from 2.41 to 2.96 in woodlands. Median decay class did not differ among burn classes in woodlands ($\chi^2 = 5.53$, $df = 3$, $P = 0.137$) or among burn classes in greenbelts ($\chi^2 = 3.78$, $df = 3$, $P = 0.287$).

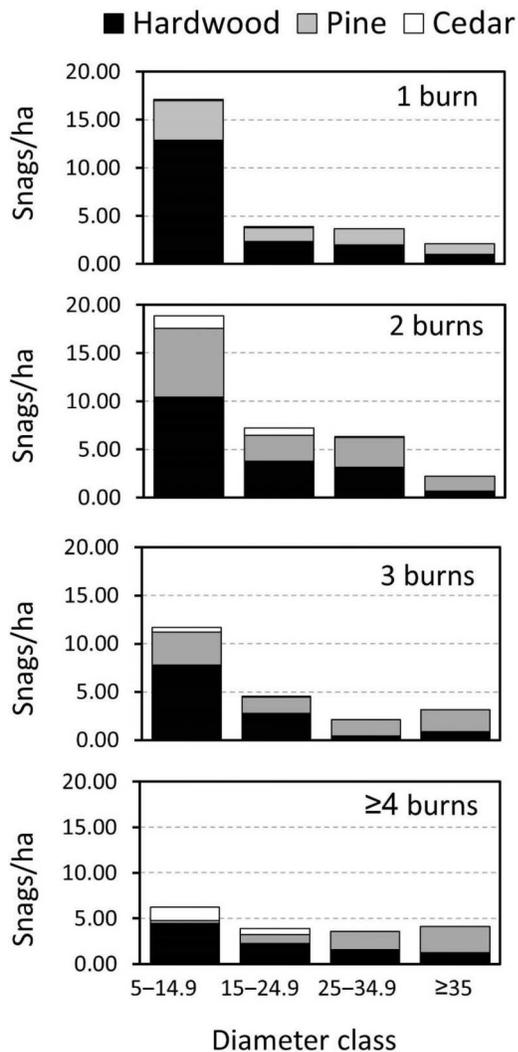


Figure 2. Mean densities of pine, hardwood, and cedar snags (number of snags/ha) of 4 diameter classes in pine woodland stands burned 1 time, 2 times, 3 times, and ≥ 4 times in the Ouachita Mountains of western Arkansas. Each burn class contained 6 different stands.

Discussion

We did not control for fire season, fire intensity, rotation length, or fire weather parameters for fires in this study. However, our goal was to estimate snag abundance based on realistic conditions across an operational landscape that typically includes burns during different times of the year, using fires of varying intensity under varying fire weather parameters. Although season of fire, fire intensity, and fire return interval are three factors commonly used to evaluate the ecological effects of fire (Glitzenstein et al. 1995), the interval between burns and the season in which fires are ignited may not show consistent effects on recruitment and survival of snags in Southern pines (Lloyd et al. 2012). Nevertheless, fall fires may cause the greatest mortality and fires that cause crown scorch late in the growing season may produce more mortality than similar fires early in the growing season (Wade and Johansen 1986, Menges and Deyrup 2001). Only one fire in this study was known to have occurred during fall (November 3 in a stand burned 5 previous times), but that stand had snag densities (small = 8.7, large = 11.3 snags/ha) similar to others in that burn class (Figure 3).

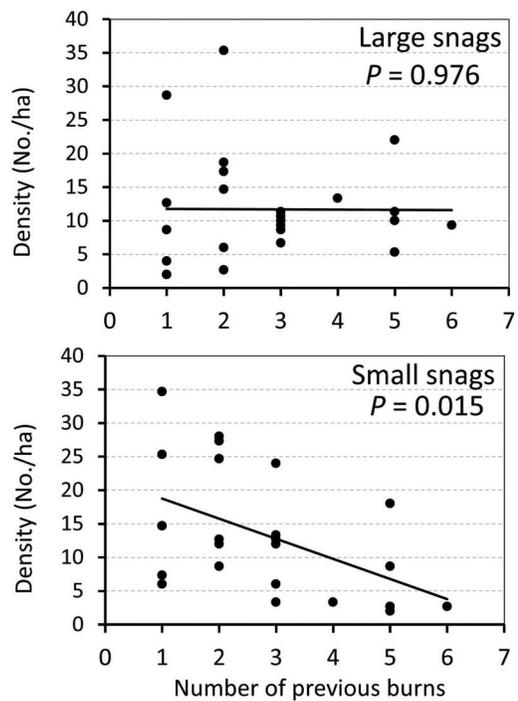


Figure 3. Linear regression of small snag (5–14.9 cm dbh) and large snag (≥ 15 cm dbh) density (number of snags/ha) with number of previous prescribed burns in 24 pine woodland stands of the Ouachita Mountains of western Arkansas.

Interval between burns may also affect tree mortality because longer periods between successive burns result in greater fuel accumulation (Doren et al. 1993). Interval between burns in our study ranged from 1 to 8 years (mean = 3.7 years). Four stands had an interval of 7 or 8 years between successive burns, but the interval between successive burns in most stands was ≤ 4 years. In the 4 stands that had an interval of 7 or 8 years, mean \pm SE densities of large snags in woodland portions of these stands (12.0 ± 2.3 snags/ha) and total snags (25.2 ± 7.0 snags/ha) were similar to averages from all stands combined (large snag density = 11.7 snags/ha and total snags = 25.2 snags/ha). Thus, stands that had occasional longer intervals between burns did not have significantly greater snag densities. Length of time between the last burn and our measurements varied greatly (4–87 months), which could have affected our snag counts because some snags killed by burning may have fallen before we conducted measures. However, there was no significant correlation between the number of months since the last burn and densities of small snags ($P = 0.523$), large snags ($P = 0.163$), total snag density ($P = 0.217$).

With an increasing number of burns comes the likelihood that some burns may be more intense, leading to death in some overstory trees. Even low-intensity, ground-level fire may injure larger trees, creating avenues for pathogens such as fungi to enter (Smith and Sutherland 2006). However, shortleaf pine, which is relatively fire tolerant, may not succumb easily to low-intensity, prescribed fire. Because we found no significant difference in large snag density among stands burned a different number of times, factors other than fire, such as lightning strikes, insects, or disease, may have been the primary contributors to large-snag creation. However, high variability in the density of large snags among stands with the same number of previous burns probably limited our ability to discern significant differences. Nevertheless, we could not determine whether burning

led to death of individual trees, and we were unable to discern the exact cause of death for most snags. Consequently, we can report that an association existed between number of burns and small snag density but not that increased number of burns caused a decline in small snag density. Because we did not collect pretreatment data on snag abundance, it is not known whether pretreatment snag densities differed among stands. Extensive disturbance associated with WSI treatments (e.g., partial overstory harvesting, midstory removal, skidding, and hazard tree removal) probably created or destroyed snags; thus, pretreatment snag densities may not have correlated with posttreatment snag densities.

Snag size is an important factor affecting snag longevity and use by wildlife. Larger diameter snags typically stand for longer periods than small diameter snags (e.g., Morrison and Raphael 1993, Dickson et al. 1995, Cain 1996); therefore, larger snags provide useable habitat for longer periods than small snags. Larger diameter snags are also favored by many wildlife species (e.g., Conner and Saenz 2005, Kalcounis-Rüppell et al. 2005). Because of the importance of large diameter snags, snag retention guidelines in many land management plans call for retaining minimal densities of large diameter snags but often do not reference smaller snags. Nevertheless, small snags do have value, as solitary male bats of many species often roost in small (<10-cm diameter) snags (e.g., Perry and Thill 2007, Perry et al. 2016), and many bird species also use small snags for foraging (e.g., Brawn et al. 1982). We found that large diameter snags were not affected by successive controlled burns, whereas small snag density was lower in stands with more burns.

Management activities such as partial harvest and thinning that reduce overall tree densities in a forest stand typically result in stands with lower snag density because these treatments reduce the total number of trees available to produce snags (Perry and Thill 2013). In this study, WSI treatments used initially in the woodland restoration process reduced overstory tree densities and left few small midstory trees (5–15 cm). These initial low densities of small trees reduced the potential source for new small snags, and repeated burning prevented the establishment of new small trees that would have provided a source for new small snags. Further, repeated burning typically reduces the number of small trees (e.g., <25 cm diameter) in forest stands (Huddle and Pallardy 1996, Hutchinson et al. 2005), and smaller diameter trees succumb more easily to fire (e.g., Wade 1993). Thus, it is not surprising that small snag density was negatively correlated with the number of burns (longer time since WSI) in these stands.

Stands burned only 1 or 2 times had an abundance of smaller snags, which may have resulted from a pulse of fire-intolerant trees killed by the reintroduction of fire. Tree species differ in their susceptibility to mortality from fire, based on differences in the thickness of their bark and differences among species in the maximum temperatures their tissues can withstand (Wade and Johansen 1986). Fire-related mortality of trees of the same species can often be explained by differences in bark thickness (McNab 1977), which can vary among individual trees of the same species based on genetics (e.g., Pederick 1970). Thus, individual trees or clusters of genetically related trees in a stand may be inherently similar in susceptibility to mortality from fire, and the first 1 or 2 burns may have killed the more fire-intolerant individuals, leading to genetically more fire-tolerant stands.

Management Implications

Of ecological concern is providing adequate snag densities through the life of a forest stand for the various wildlife taxa that require snags. However, when land managers are tasked with managing vast areas of forestland, opportunities to manipulate snag densities occur primarily during harvest or thinning entries and entries involving other tasks such as road/pond construction or understory treatments. Consequently, snag retention guidelines typically apply to areas undergoing harvest or thinning. Suggested snag retention densities vary based on wildlife species present, forest type, snag size, and region. If maintaining snags of a particular size or density is a management priority, managers may need to augment snag availability by periodically creating snags in woodlands subjected to frequent burning.

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