Trends in Snag Populations in Northern Arizona Mixed-Conifer and Ponderosa Pine Forests, 1997-2012

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Abstract

We monitored snag populations in drought-stressed mixed-conifer and ponderosa pine (*Pinus ponderosa*) forests, northern Arizona, at 5-yr intervals from 1997-2012. Snag density increased from 1997-2007 in both forest types, with accelerated change due to drought-related tree mortality during the period 2002-2007. Snag density declined non-significantly from 2007-2012, but median snag density in 2012 remained significantly greater than median density in 1997. Differences in composition of snag populations also persisted through 2012. Our results suggest that the drought-mediated spike in tree mortality observed from 2002-2007 did not persist from 2007-2012 and snag populations may be converging toward previous levels in terms of both snag density and composition of snag populations.

Keywords: climate change, drought, snag creation, snag dynamics, spatial variability

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Introduction

Snags (standing dead trees) provide important resources in forest systems (Harmon and others 1986; Bull and others 1997; McComb and Lindenmayer 1999; Laudenslayer and others 2002), and managers have long been interested in snag numbers and dynamics (Thomas and others 1979; Morrison and others 1986; Mellen and others 2002; Hutto 2006; Marcot and others 2010). We have been monitoring snag populations at 5-yr intervals in mixed-conifer and ponderosa pine (*Pinus ponderosa*) forests in north-central Arizona since 1997. Previous reports from this study summarized results through 2007, and they documented significant increases in snag density and changes in species composition and size- and decay-class distributions of snag populations in both forest types during this period (Ganey and Vojta 2005; 2012). Changes in snag populations were most pronounced from 2002–2007 (Ganey and Vojta 2012), corresponding with a large spike in tree mortality during this period (Ganey and Vojta 2011). This spike appeared to be mediated by a period of severe, global climate-change type drought (defined here as a drought accompanied by warmer temperatures [Breshears and others 2005]) in this area (Weiss and others 2009; Williams and others 2010), and closely followed an extreme climate year in 2002 (Salzer and Kipfmueller 2005; Koepke and others 2010). Whether the observed increase in tree mortality was caused by that extreme climate year or reflected the cumulative effects of longer-term drought remains unknown, but it has potential implications in terms of future tree mortality patterns and corresponding changes in snag populations.

Here, we expand on previous reports to document trends in snag populations through 2012. We summarize changes in snag density from 1997 to 2012, emphasizing the period from 2007 to 2012. We describe changes in composition (species, size-class, and decay-class distributions) of snag populations and summarize patterns in snag recruitment. The addition of data from the 2012 sample provides an opportunity to determine whether or not the accelerated tree mortality and consequent changes in density and composition of snag populations documented during the 2002–2007 period continued after that time.

Study Area

The study area encompassed approximately 73,000 ha within the Coconino and Kaibab National Forests, north-central Arizona (Figure 1). Within this area, study plots were randomly located in mixed-conifer and ponderosa pine forests (plot selection described in Ganey 1999). Mixed-conifer forests were dominated numerically by ponderosa pine, white fir (*Abies concolor* Lindl. *ex* Hildebr.), and Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco), which together accounted for approximately 90 percent of total trees in this forest type (as sampled in 2004; Ganey and Vojta 2011). Other species included Gambel oak (*Quercus gambelii* Nutt.), quaking aspen (*Populus tremuloides* Michaux), and limber pine (*P. flexilis* James), in that order of frequency. Ponderosa pine accounted for over 90 percent of trees in ponderosa pine forest (Ganey and Vojta 2011). Gambel oak also was relatively common (approximately 8 percent of total trees by frequency), and alligator juniper (*Juniperus deppeana* Steud), Douglas-fir, quaking aspen, limber pine, pinyon pine (*P. edulis*), and other species of juniper were present in small numbers in some stands.

The study plots were distributed across a wide range of topographic conditions and soil types. They covered the entire elevational range of these forest types within this area, from the transition zone between pinyon–juniper woodland and ponderosa pine at lower elevations to the ecotone between mixed-conifer and Engelmann spruce (*Picea engelmanni*

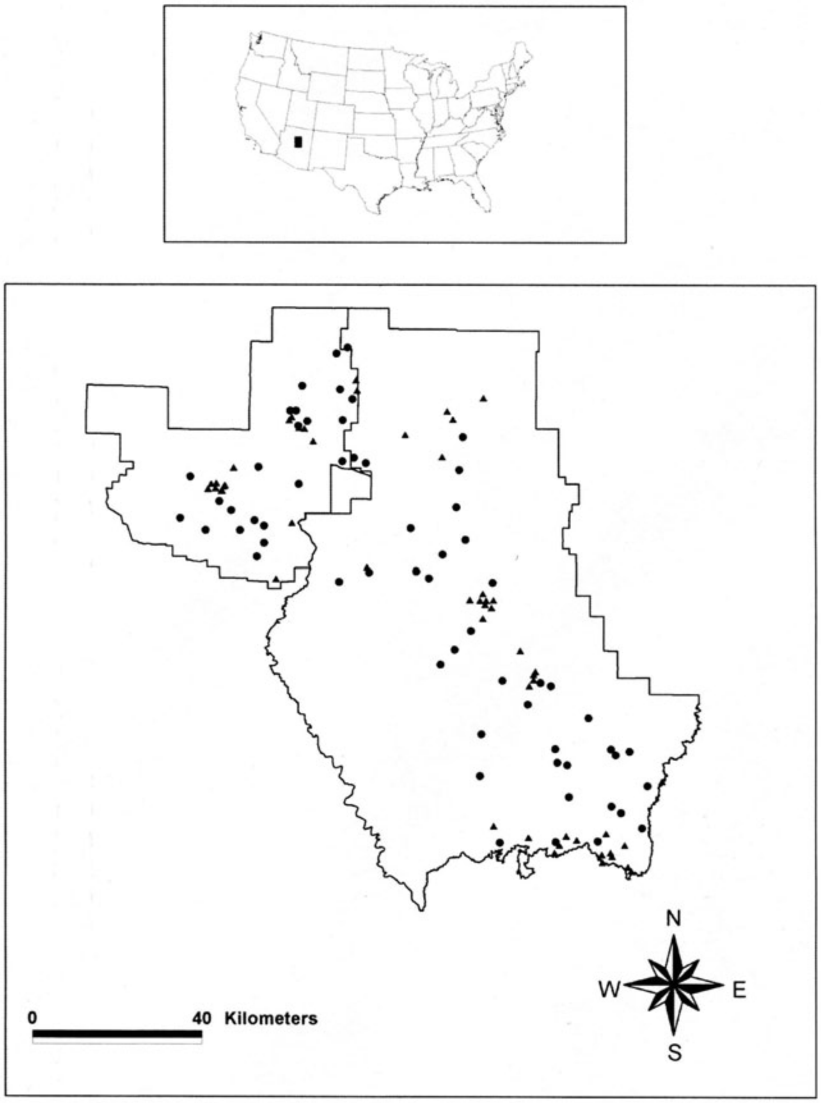

Figure 1. Location of the study area (black box, top) in northern Arizona, USA, and locations of snag-monitoring plots within the study area (bottom). Plots were located in the Kaibab (left) and Coconino (right) National Forests, north-central Arizona. Plots in ponderosa pine forest (*n* = 60) are indicated by circles, plots in mixed-conifer forest (*n* = 53) by triangles.

Parry *ex* Engelm.)–corkbark fir (*Abies lasiocarpa* var. *arizonica* [Merriam] Lemmon) forests at higher elevations (Brown and others 1980). Plots also included both commercial forest lands and administratively reserved lands such as wilderness and other roadless areas. Consequently, plots represented a wide range of forest structural conditions. For example, as of 2004, density of trees ≥20 cm in diameter at breast height (dbh) ranged from 78 to 489 (median = 266.7) trees ha^{-1} in mixed-conifer forest and from 11 to 689 (median = 227.8) trees ha^{-1} in ponderosa pine forest, and basal area ranged from 7 to 52 (median = 25.2) and from 1 to 44 (median = 19.7) $m^2 ha^{-1}$ in mixed-conifer and ponderosa pine forest, respectively (Ganey and Vojta 2011).

Methods

We sampled snags in 113 plots (1 ha each in area, *n* = 53 and 60 plots in mixed-conifer and ponderosa pine forest, respectively) randomly established in 1997 (see Ganey 1999 for details on plot selection). We sampled all snags ≥2 m in height and ≥20 cm in diameter at breast height (dbh) in 1997, 2002, and 2007. We did not sample smaller-diameter snags based on the assumption that these were less important to cavity-nesting birds (Balda 1975;

Scott 1978; Cunningham and others 1980; Li and Martin 1991; Conway and Martin 1993) and/or roosting bats (Rabe and others 1998; Bernardos and others 2004; Solvesky and Chambers 2009). All snags were uniquely marked with numbered metal tags, allowing us to distinguish existing snags from new snags when re-sampling plots (with some exceptions; see below). For all snags, we recorded snag species, dbh (nearest cm), and decay class (1 to 5, with larger numbers indicating later decay stages; after Raphael and White 1984).

We focused analyses within rather than between forest types, because we were interested primarily in trends over time within forest type. We relied on nonparametric statistical tests (Conover 1980), because distributions for many snag parameters deviated greatly from normality. We present medians and ranges as estimates of central tendency and variability, because many parameters were so highly skewed that a few plots greatly influenced mean estimates.

We included all plots in summarizing snag numbers and composition of snag populations because our objective was to quantify these parameters across the landscape, including recently disturbed areas. We excluded plots that experienced severe wildfire when summarizing recruitment patterns, however, because intense wildfire melted the metal tags used to mark snags (Ganey and Vojta 2005), making it difficult to determine whether dead trees represented recently created or pre-existing snags.

We summarized data on snag density and composition of snag populations in 1997, 2002, 2007, and 2012. We compared snag density across years using the Friedman test (Conover 1980). Where that test indicated overall differences across years, we compared sequential pairs of years (1997 vs. 2002, 2002 vs. 2007, and 2007 vs. 2012) using Wilcoxon signed-ranks test (essentially a two-sample equivalent of the Friedman test [Conover 1980]), to determine which years differed. We set the *P*-value for significance at <0.05 in all omnibus tests, and used a Bonferroni correction (Conover 1980) for all multiple comparisons, setting the significance level at 0.05/number of comparisons.

We compared species composition and size- and decay-class distributions of snag populations across years using chi-square tests (Conover 1980). We also evaluated differences in composition of populations of snags recruited in three time intervals, 1997 to 2002, 2002 to 2007, and 2007 to 2012. We compared species composition and decay-class distribution of newly recruited snags between sequential pairs of years using chi-square tests, and compared diameter distributions between sequential time periods using Kolmogorov-Smirnov tests (Conover 1980).

For each 5-yr interval between samples, we estimated the percentage of snags existing at the beginning of the interval that fell before the end of the interval. We compared proportions of falling snags between sequential time intervals (1997 to 2002, 2002 to 2007, and 2007 to 2012) using the test for comparing two proportions described by Zar (1984; essentially an analogue of a chi-square test).

Results

No plots underwent commercial timber harvest between 1997 and 2012. Three plots in ponderosa pine forest underwent thinning of smaller trees during this period. Ten plots experienced prescribed fire or low-severity wildfire during this period (one in mixed-conifer and nine in ponderosa pine forest), and seven plots experienced high severity wildfire (four in mixed-conifer and three in ponderosa pine forest).

Changes in Snag Density

Snag density differed across years in both mixed-conifer and ponderosa pine forest (Figure 2; Friedman test, $P < 0.001$ in both forest types). Snag density increased significantly in mixed-conifer forest from 1997 to 2002 and again from 2002 to 2007 (Wilcoxon tests, both P-values <0.001), then declined non-significantly from 2007 to 2012 ($P = 0.080$). Despite this recent decline, median snag density in 2012 (47 snags ha^{-1}) remained almost 1.7 times greater than median snag density in 1997 (28 snags ha^{-1}) in mixed-conifer forest. Spatial variation in snag density was pronounced, with density ranging from slightly >0 to over 300 snags ha^{-1} in mixed-conifer plots (Figure 2).

Snag density in ponderosa pine forest did not differ between 1997 and 2002 ($P = 0.103$), increased significantly from 2002 to 2007 ($P < 0.001$), and declined non-significantly (after adjusting for multiple comparisons; $P = 0.021$) from 2007 to 2012. Despite this decline, median snag density in 2012 (10 snags ha^{-1}) was double the median snag density in 1997 (5 snags ha^{-1}). As in mixed-conifer forest, spatial variation among plots was pronounced in ponderosa pine forest, with snag density ranging from zero to almost 200 snags ha^{-1} (Figure 2).

Wildfire resulted in large increases in snag numbers in some plots, but did not explain much of the overall increase in snag density across time because it affected relatively few plots (Figure 3). Burned plots dampened the observed decline in snag numbers from 2007-2012, however. This decline would have been greater in both forest types without the influence of burned plots, and much greater in mixed-conifer forest.

Most of the net change in snag numbers over time was driven by creation of new snags. Recruitment of new snags increased significantly from 2002 to 2007 in both forest types (Wilcoxon tests, $P < 0.001$ in both forest types), then declined significantly from 2007 to 2012 ($P < 0.001$ in mixed-conifer forest, $P = 0.005$ in ponderosa pine forest forest,

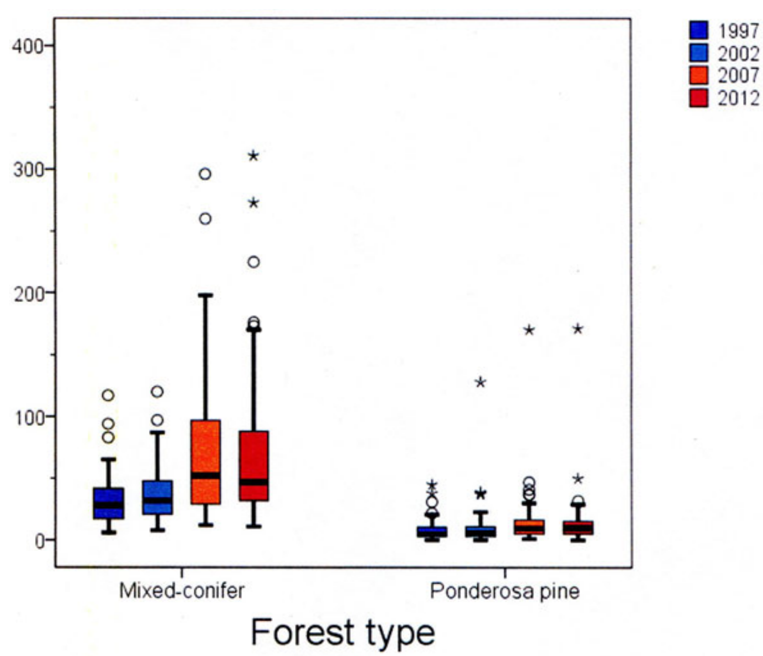

Figure 2. Trends in density (snags ha^{-1}) of snags in northern Arizona mixed-conifer and ponderosa pine forest, 1997 to 2012. Based on sampling at 5-yr intervals within 1-ha plots (n = 53 and 60 plots in mixed-conifer and ponderosa pine forest, respectively). Boxes indicate the interquartile range (25th to 75th percentile), the line within the box indicates the median, and the external lines show the range in the data excluding outliers (circles, defined as observations more than 1.5 box lengths outside the box) and extremes (asterisks, defined as observations more than 3 box lengths outside the box).

Figure 4), paralleling changes in snag density. As with snag density, snag recruitment varied extensively across plots as well as years, ranging from zero to 264 new snags ha^{-1} in mixed-conifer forest and from zero to 169 new snags ha^{-1} in ponderosa pine forest (Figure 4).

Changes in snag longevity did not drive increases in snag numbers. In fact, proportions of existing snags falling during a time interval increased significantly across most time periods in both forest types. In mixed-conifer forest, 20.5 percent of snags standing in 1997 (n = 1,771) fell by 2002, 27.3 percent of snags standing in 2002 (n = 1,963) fell by 2007, and 28.3 percent of snags standing in 2007 (n = 3,914) fell by 2012. These proportions differed significantly between time periods (test comparing two proportions, 1997-2002 vs. 2002-2007, P <0.001; 2002-2007 vs. 2007-2012, P = 0.027). Corresponding proportions

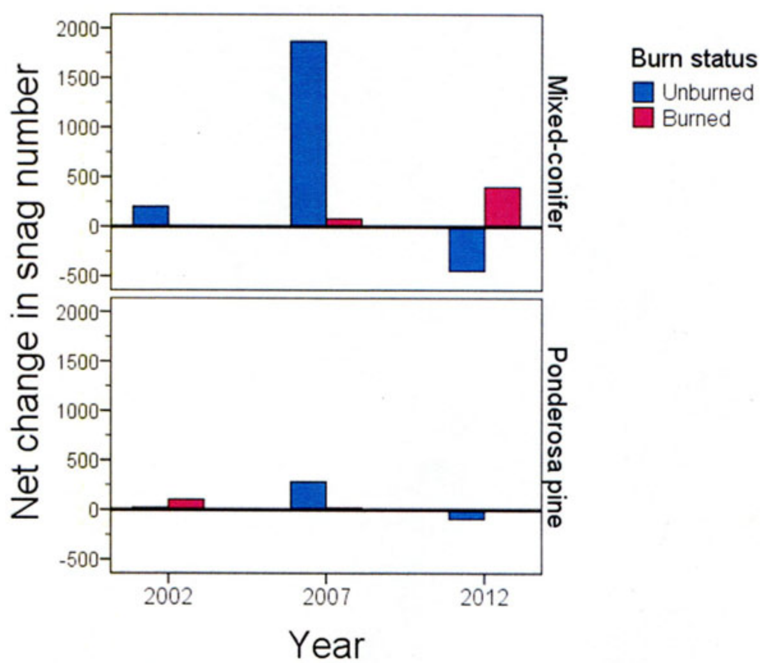

Figure 3. Change in snag numbers aggregated across plots within mixed-conifer (n = 53 plots) and ponderosa pine (n = 60 plots) forest, northern Arizona. Numbers reflect net change in snag numbers relative to five years earlier. Numbers of plots that burned were cumulative across time periods. In mixed-conifer forest, one plot had burned by 2002, two by 2007, and five by 2007. In ponderosa pine forest, four plots had burned by 2002, six by 2007, and 12 by 2012.

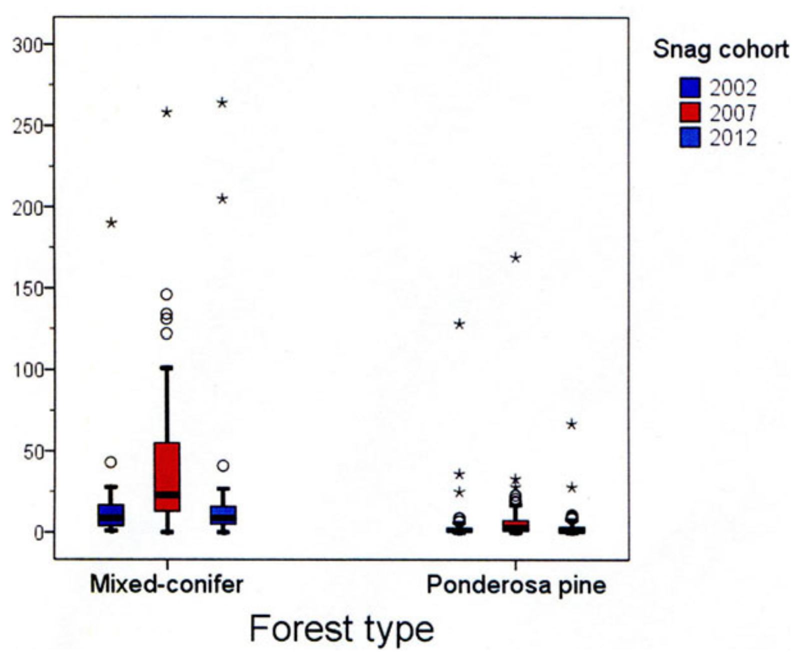

Figure 4. Number of new snags ha^{-1} recruited by time period in mixed-conifer and ponderosa pine forests, northern Arizona (n = 53 and 60 plots in mixed-conifer and ponderosa pine forest, respectively). Snag cohorts 2002, 2007, and 2012 represented trees that died between 1997 and 2002, 2002 and 2007, and 2007 and 2012, respectively. Boxes indicate the interquartile range (25th to 75th percentile), the line within the box indicates the median, and the external lines show the range in the data excluding outliers (circles, defined as observations more than 1.5 box lengths outside the box) and extremes (asterisks, defined as observations more than 3 box lengths outside the box).

of snags falling in ponderosa pine forest were 34.2 percent, 43.5 percent, and 39.6 percent, respectively (n = 475, 604, and 906, respectively). These proportions differed significantly between 1997-2002 and 2002-2007 (P = 0.002), but not between 2002-2007 and 2007-2012 (P = 0.130).

Changes in Composition of Snag Populations

Species composition of snag populations changed significantly from 1997 to 2012 in both forest types (chi-square tests, P <0.001; Figure 5). In mixed-conifer forest, species composition did not differ between 1997 and 2002 (P >0.05), but differed significantly between 2002 and 2007 and between 2007 and 2012 (both P-values <0.001). The proportion of white fir snags more than doubled from 2002 to 2007 in mixed-conifer forest, with corresponding decreases in proportions of ponderosa pine and Gambel oak. By 2012 species composition of snags in mixed-conifer forest appeared to be shifting back towards composition prior to 2007, although proportions of white fir snags and Gambel oak snags remained relatively higher and lower, respectively, than during earlier periods (Figure 5). These changes were driven primarily by recruitment patterns, which differed significantly among snag cohorts (chi-square tests, P <0.001; Figure 6). In particular, a large pulse in recruitment of white fir snags occurred between 2002 and 2007. This pulse weakened slightly between 2007 and 2012 (Figure 6).

Snag populations in ponderosa pine forest were dominated by ponderosa pine in all years, but proportions of ponderosa pine snags declined slightly from 1997 to 2012, and proportions of Gambel oak snags increased slightly (Figure 5**.** Again, changes in species composition appeared to be driven primarily by recruitment patterns, which differed significantly among snag cohorts (chi-square tests, P <0.001). Proportions of ponderosa pine snags declined and proportions of Gambel oak snags increased in successive cohorts (Figure 6).

Size-class distributions differed significantly across years in both forest types (chi-square tests, P <0.001 in mixed-conifer forest, and P = 0.029 in ponderosa pine forest). The

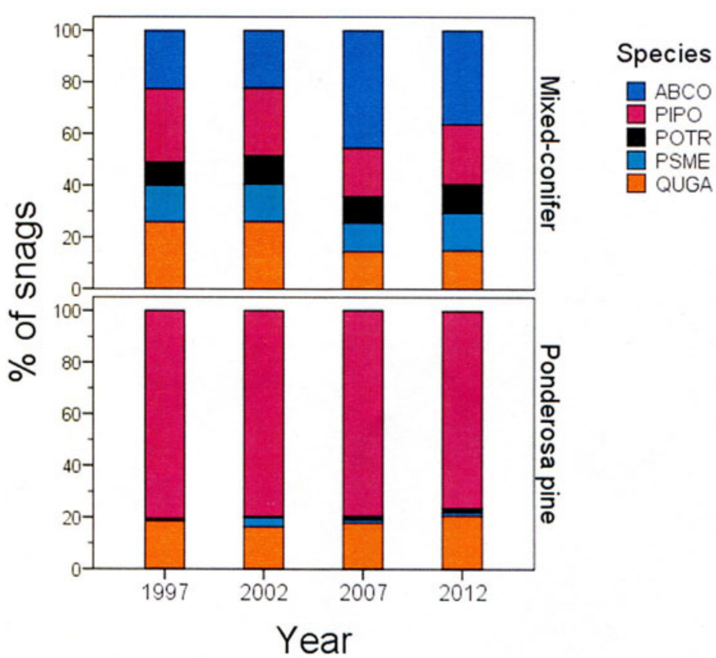

Figure 5. Species composition (percent) of snag populations in northern Arizona mixed-conifer and ponderosa pine forest in 1997, 2002, 2007, and 2012. Species acronyms: ABCO = white fir, PIPO = ponderosa pine, POTR = quaking aspen, PSME = Douglas-fir, and QUGA = Gambel oak. These species comprised 98 percent of all snags sampled in mixed-conifer forest and 96.3 percent of all snags sampled in ponderosa pine forest.

greatest change occurred in the smallest size classes in both forest types (Figure 7) and was driven primarily by recruitment patterns. Diameter distributions of snags recruited from 2002 to 2012 were skewed toward smaller diameter snags (Figure 8). Consequently, proportions of snags collectively increased in the smaller three size classes and decreased in the larger three size classes in both forest types. By 2012, diameter distributions appeared to be shifting back towards distributions observed prior to 2007 in ponderosa pine forest, but not in mixed-conifer forest.

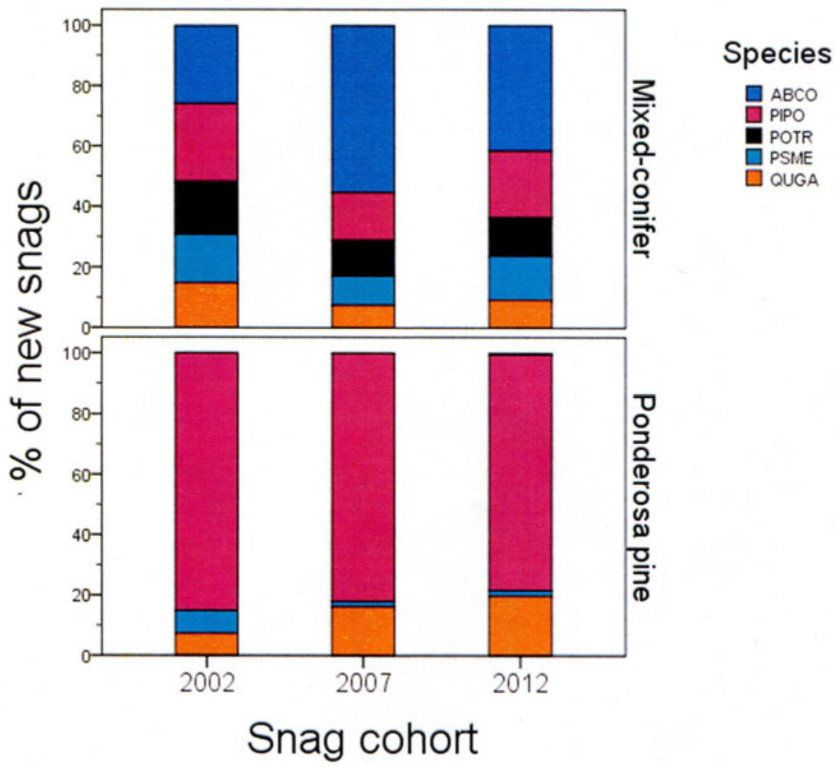

Figure 6. Species composition (percent) of newly recruited snags in northern Arizona mixed-conifer and ponderosa pine forest by time period. Snag cohorts 2002, 2007, and 2012 represented trees that died between 1997 and 2002, 2002 and 2007, and 2007 and 2012, respectively. Species acronyms: ABCO = white fir, PIPO = ponderosa pine, POTR = quaking aspen, PSME = Douglas-fir, and QUGA = Gambel oak. These species comprised >95 percent of all snags in all cohorts in mixed-conifer forest, and >94 percent of all snags in all cohorts in ponderosa pine forest.

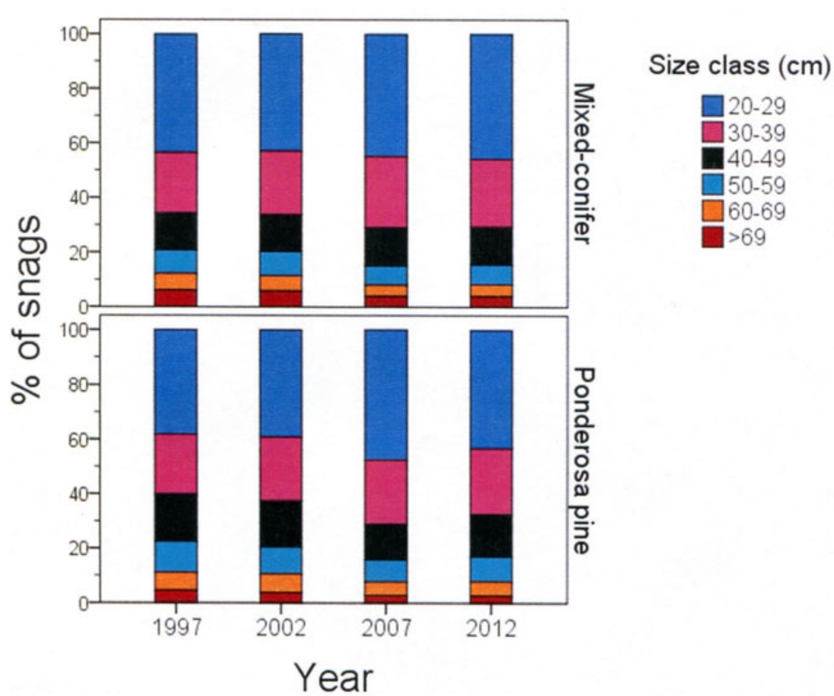

Figure 7. Size-class distribution (percent) of snags aggregated across plots in northern Arizona mixed-conifer and ponderosa pine forest in 1997, 2002, 2007, and 2012.

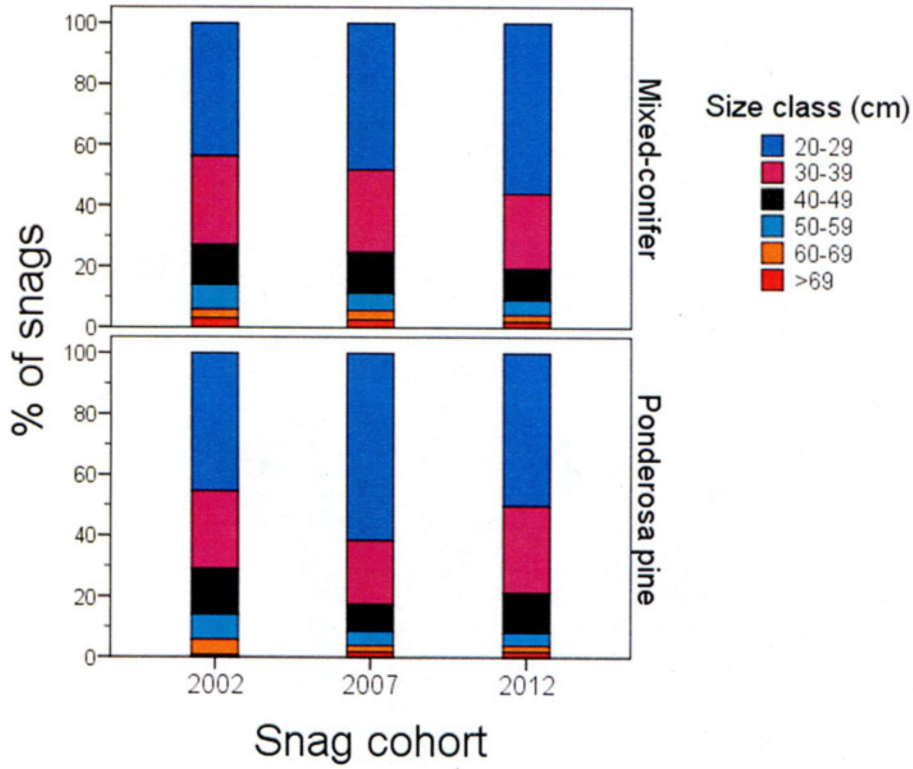

Figure 8. Percent of snags recruited by diameter class in northern Arizona mixed-conifer and ponderosa pine forests by time period. Snag cohorts 2002, 2007, and 2012 represented trees that died between 1997 and 2002, 2002 and 2007, and 2007 and 2012, respectively.

Because most snags enter the population in decay classes 1 or 2 (Ganey and Vojta 2005), the influx of large numbers of new snags shifted decay-class composition in both forest types from a distribution dominated by snags in older decay classes (classes 4 and 5) in 1997 to a distribution dominated by snags in younger decay classes (classes 1–3) in 2012 (chi-square tests, $P < 0.001$ in both forest types; Figure 8). Much of this change occurred between 2002 and 2007, however, and distributions appeared to be shifting back toward dominance by older decay classes by 2012 (Figure 9).

Discussion

Previous results from this study documented pronounced changes in snag density and composition of snag populations between 1997 and 2007 (Ganey and Vojta 2005; 2012). Much of that change occurred between 2002 and 2007, driven by increased tree mortality during that period. The proximate cause for much of that mortality was damage due to a suite of forest insects, mediated by drought conditions. Ganey and Vojta (2011) hypothesized that much of the observed tree mortality was due to the effects of an extreme climate year in 2002, but the relative influence of that extreme year versus cumulative effects of longer term drought was unknown.

Our results indicate that the accelerated, and possibly unsustainable (Ganey and Vojta 2011), tree mortality observed from 2002-2007 did not persist during the period from 2007-2012. Creation of new snags was far lower during the latter period, returning approximately to levels observed from 1997–2002 (Figure 4). This lends support to the hypothesis that the spike in tree mortality from 2002-2007 was attributable to the extreme climate year in 2002. That year featured low water availability and elevated temperatures, a combination of features that has been demonstrated to result in rapid increases in tree mortality (Adams

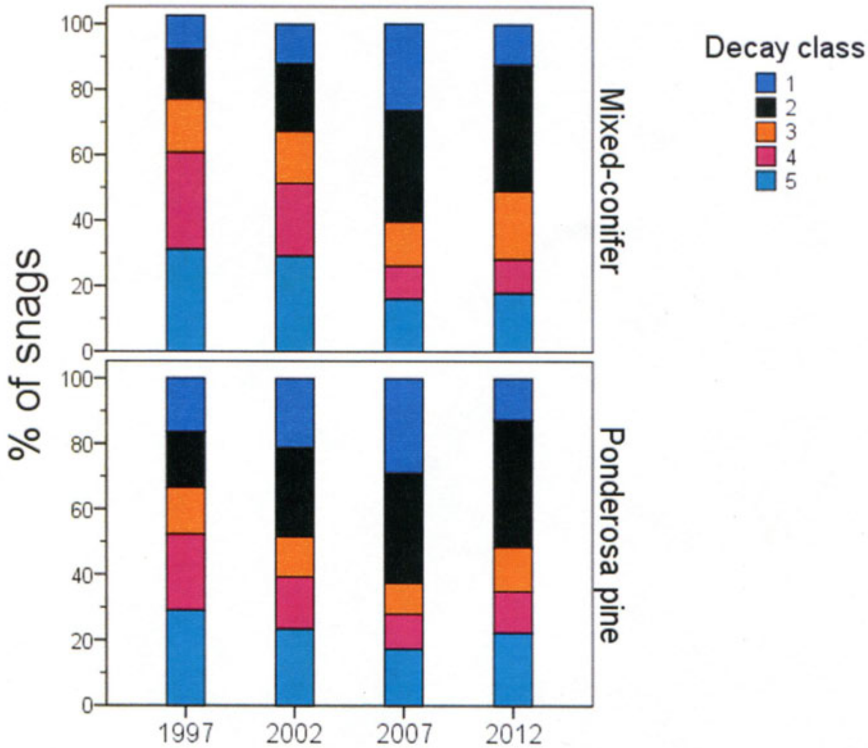

Figure 9. Decay-class composition (after Raphael and White 1984) of snags in northern Arizona mixed-conifer and ponderosa pine forests in 1997, 2002, 2007, and 2012. Snags in decay class 1 retained needles (or leaves), twigs, and intact limbs. Decay-class 2 snags retained twigs and intact limbs but lacked needles (or leaves). Class 3 snags had limbs that were largely intact, but lacked twigs. Class 4 snags had limbs that were mostly broken, and class 5 snags lacked large limbs entirely. In both forest types, decay-class distributions shifted from dominance by snags in later decay classes (classes 4 and 5) in 1997 to snags in early decay classes (classes 1 and 2) in 2012.

and others 2009; Breshears and others 2009; Allen and others 2010; Koepke and others 2010; Williams and others 2012).

Future patterns in snag dynamics will depend on climate patterns and the interaction between those patterns and various disturbance factors (Dale and others 2001; McKenzie and others 2004; Parker and others 2006; Adams and others 2009; Breshears and others 2009; Negron and others 2009; Allen and others 2010). Most climate models predict that the southwestern United States will become increasingly warmer and drier (Seager and others 2007; Seager and Vecchi 2010; Williams and others 2012). If long-term drought becomes more common, continued increases in tree mortality are likely (Breshears and others 2005; 2009; Negron and others 2009; Allen and others 2010; Williams and others 2010; 2012). Our observations suggest that those increases can spike suddenly when extremely warm and dry years occur during those drought periods (Adams and others 2009; Koepke and others 2010; Allen and others 2010), but that tree mortality can return to previous levels fairly quickly following such periods. This highlights the often episodic nature of snag creation and illustrates the difficulty involved in either modeling snag dynamics or managing for particular levels of snag retention in an era of changing climate.

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