

Management Indicator Species of the Kaibab National Forest: an evaluation of population and habitat trends

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Isolated aspen stand. Photo by Heather McRae.



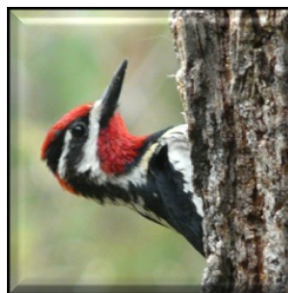
Pygmy nuthatch. Photo by the Smithsonian Inst.



Pumpkin Fire, Kaibab National Forest



Mule deer. Photo by Bill Noble



Red-naped sapsucker. Photo by the Smithsonian Inst.



Northern Goshawk © Tom Munson



Tree encroachment, Kaibab National Forest

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1. Management Indicator Species

INTRODUCTION

This report provides information about management indicator species (MIS) for the Kaibab National Forest (KNF) in northern Arizona. It also includes a description of the status and trends of the major habitat types on the KNF. This report is intended to be used for regulatory, planning, and public education purposes. It is considered a “living document” which allows the KNF to update specific sections of the report as new science becomes available. In calendar year 2007 the MIS portion of the document was updated. The habitat portion was updated in calendar year 2009.

The 1.6 million acre KNF is located on the Colorado Plateau. The Mogollon Rim forms the south boundary while Grand Canyon National Park bisects the center of the forest. Changes in elevation, soil type (basalt in the south vs. limestone in the north), historical land use, and land ownership lend way to a diverse landscape comprised of 3 spatially distinct ranger districts. The Williams (613,459 acres) and Tusayan (331,428 acres) Ranger Districts (RDs) make up the South Zone (SZ), while the North Kaibab (NK) RD (655,078 acres) encompasses land north and south of the Grand Canyon. Elevations on the KNF range from approximately 3,300 feet in Kanab Creek to 10,418 feet at the top of Kendrick Mountain. The Forest is comprised of grasslands and sagebrush in the lower elevations, making the transition to pinyon-juniper and ponderosa pine to mixed-conifer, fir, spruce and aspen with increasing elevation. As a result of such topographical and geological variability there are some distinct differences, particularly between the NKRD and the SZ which make uniform forest-wide trend estimates somewhat problematic in nature in some instances. We have noted these exceptions by district where applicable. For example, diminished aspen recruitment is a concern on the Williams RD where the effects from elk browse are numerous. However, in the case of the NKRD, where the Arizona Game and Fish Department (AZGFD) manages for much lower elk numbers, the resulting pressure on aspen recruitment in the area is negligible. Different management strategies for this large game species affect both the interpretation of trends for elk and aspen in different ways for each district. Other notable differences between the North and South Kaibab are an artifact of previous logging operations. Forests on the NKRD contain trees of greater diameter than those on the Williams RD because they were not commercially logged as heavily during historic times. The NKRD contains the only example of a fundamentally intact mature forest on the KNF. However, the Williams RD contains the only active spotted owl protected activity centers (PACs) on the Forest and habitat changes on the SZ are more likely to impact this MIS. Finally, the spatially disjunct nature of the 3 districts can influence population estimates for wide ranging mammal herds such as elk, deer, and, pronghorn by affecting connectivity and seasonal movement patterns.

Regulatory Background

National Directives

The USDA Forest Service is charged with managing all renewable resources, including wildlife, on National Forest lands. This obligation was enacted by Congress and set forth in the National Forest Management Act (NFMA) of 1976. As a federal law, the NFMA is the primary statute governing the administration of National Forests. The Forest Service first promulgated regulations implementing NFMA in September, 1979, and subsequently revised them in 1982 (known as the 1982 Rule). The 1976 legislation requires the Secretary of Agriculture to assess forest lands, and develop and implement a land and resource management plan for each unit of the National Forest System. These management plans, commonly known as Forest Plans, guide management activities on each National Forest. Therefore, site-specific projects proposed on national forests must comply with the applicable Forest Plan. The 1982 regulations require Forest Plans to manage fish and wildlife habitat so viable populations of existing native and desired non-native vertebrate species are maintained in the planning area (i.e., each individual National Forest). Under the 1982 regulations, a viable population is regarded as one that has the estimated numbers and distribution of reproductive individuals to insure its continued existence, is well distributed in the planning area, and that habitat must be well distributed so that those individuals can interact with others in the planning area.

Because it is impossible to address the thousands of species that occur on National Forests, the use of MIS serves as a barometer for more than the selected species and a surrogate for addressing other species' ecological needs. As directed by NFMA and the 1982 Rule, each Forest Plan identifies and selects certain vertebrate, invertebrate, or plant species present in each National Forest as MIS because "their population changes are believed to indicate the effects of management activities" (36 CFR 219.19(a)(1)). Additionally, the 1982 regulations require that "Population trends of the management indicator species will be monitored and relationships to habitat changes determined" (36 CFR 219.19(a)(6)). Each Forest "selects management indicators that best represent the issues, concerns, and ... recovery of Federally-listed species, provide continued viability of sensitive species, and enhance management of wildlife and fish for commercial, recreational, scientific, subsistence, or aesthetic values or uses" (Forest Service Manual (FSM) 2621.1). The FSM defines management indicators as "Plant and animal species, communities or special habitats, selected for emphasis in planning, and which are monitored during forest plan implementation in order to assess the effects of management activities on their populations and the populations of other species with similar habitat needs which they may represent (FSM 2620.5)." Therefore, important characteristics of MIS are that they have narrow habitat associations, representing ecosystem components important to multiple species, and are capable of being effectively monitored.

Under the 1982 Rule, Forest Service officials have broad discretion to select MIS. The deciding official, using information provided by an interdisciplinary planning team, determines whether the population changes of certain species are "believed to indicate the effects of management activities." Beliefs or opinions about the reliability of such relationships are subject to change because of increased scientific knowledge, and as a

result of implementation and monitoring of Forest Plans. Therefore, officials may periodically need to reevaluate the MIS selected for forest plans and make appropriate adjustments. Furthermore, the 1982 Rule specifies that species are to be selected from various categories “where appropriate,” indicating there is no requirement that all categories of species or habitats be represented.

It is important to note that both the MIS concept and its application have come under considerable criticism (Patton 1987, Landres et al. 1988, Noss 1990, Weaver 1995, Simberloff 1998, Caro and O’Doherty 1999). Reflecting this, efforts have been made to drop the use of MIS entirely. Nonetheless, the 1982 regulations remain in effect.

Kaibab National Forest’s Forest Plan

The KNF’s Forest Plan was signed in 1987 and is currently under revision. In the 1987 Forest Plan, the KNF considers the management of wildlife and their habitat a public issue and management concern. The KNF expects an increase in the desire for consumptive and non-consumptive wildlife benefits. The Forest Plan addresses this rising demand with increased effort to improve and maintain habitats through direct and indirect methods. It concentrates on improving diversity, providing quality old-growth habitats and variable composition of vegetation successional stages, and integrating desired wildlife habitat characteristics in the design of all vegetative treatments, whether they are for habitat improvement or for other purposes. The Forest Plan establishes guidelines for timber treatments to protect areas of sensitive habitats and measures for the recruitment, protection, and management of snag habitats. Direct habitat improvement includes: (1) prescribed burning and wildland fire use; (2) seeding and planting of desirable browse and herbaceous species; (3) water developments; and (4) the creation of wildlife openings. Actions are also included to locate, survey, and inventory riparian and aquatic habitats as well as plant species requiring protection. The increased levels of direct and indirect habitat improvement are expected to improve habitat quality and diversity in both the short and long term. The management direction that is adopted by the KNF within its Forest Plan specifies the following actions:

- Improve wildlife habitats through expanding knowledge of species requirements, development of habitat quality and diversity, and the identification and protection of key habitats.

- Cooperate with the AGFD to achieve management goals and objectives specified in the Arizona Wildlife and Fisheries Comprehensive Plan, and in carrying out the cooperative agreement for the management of the Grand Canyon National Game Preserve. Support the AGFD in meeting its objectives for the state.

- Improve habitats for listed threatened, endangered, or sensitive species of plants and animals and other species as they become threatened or endangered. Work toward recovery and de-listing of species.

- Identify and protect areas that contain threatened, endangered, and sensitive species of plants and animals. Consult with the U.S. Fish and Wildlife Service when activities have the potential to impact species protected under the Endangered Species Act.

Expand agency, conservationist, and citizen understanding and cooperation in wildlife and fishery habitat management, activities, and use.

The KNF Forest Plan has been amended 7 times, with 5 of the amendments providing direct benefits for the management of wildlife habitat. Probably the most significant amendment for wildlife was incorporating the Management Recommendations for the Northern Goshawk in the Southwestern United States put forward by Reynolds et al. (1992). These management recommendations were based on empirical findings on goshawk populations (*Accipiter gentilis*) and nesting and foraging behavior collected through decades of research by dozens of research scientists. The effort also focused on the “goshawk foodweb” which included habitat needs for 14 key goshawk prey species. These data were evaluated by a committee of research scientists and forest managers and modeled using a planning horizon of 1,000 years. Goshawks were designated a sensitive species by the Regional Forester of the Southwest Region in 1982 and were later designated a MIS across the Region.

The original KNF Forest Plan described even-aged harvest strategies for timber management. At that time, old-growth reserves, an important habitat component for goshawk nesting, were in designated blocks selected for their forest structural attributes, but were delineated before subsequent information on goshawk nesting habitat was available. In 1987, the Forest Plan did not include constraints on timber management operations within goshawk foraging areas, which cover the majority of the forest. Even-aged harvest within these areas could be detrimental to habitat needs of many goshawk prey species. To address these issues, the 1996 Forest Plan incorporated uneven-aged forest management using group selection harvests for regeneration, as suggested in the goshawk management recommendations (Reynolds et al. 1992).

The original approach to ecosystem management put forward by Reynolds et al. (1992) is based on providing (1) nesting, post-fledging, and foraging areas for goshawks, and (2) habitat to support abundant populations of 14 primary goshawk prey (Appendix 1). A key aspect of this plan is that silvicultural activities manage for a mixed distribution of tree sizes and ages that creates beneficial forest conditions. The conditions include a grouped distribution of trees and small openings to help sustain desired forest conditions. Managing for this level of heterogeneity with large-sized, old-aged trees across the landscape should also benefit overall forest health, soil productivity, and habitats of other old-growth dependent plants and animals. By including these guidelines, “best science” principles were incorporated into the Forest Plan and realigned timber management operations towards an ecosystem based approach, thereby more closely adhering to the direction of the National Forest Management Act and the 1982 Rule.

Reynolds et al. (1992) developed the goshawk foodweb approach in an a priori manner. Since then, our knowledge of fire history in southwestern forests has greatly expanded, largely due to the efforts of the Ecological Restoration Institute, Northern Arizona University in Flagstaff (www.eri.nau.edu). The harvest prescriptions developed by the Ecological Restoration Institute are based on years of research data on fire behavior,

forest ecology, and site-specific evidence. Their restoration prescriptions lead to very similar forest conditions in terms of basal area and overall structure. In both instances, the resulting forests have far fewer trees per acre than today's forest. Part of the Forest Service mandate is to provide commodities and economic value to the communities while providing for wildlife and their habitats. The recommendations described by Reynolds et al. (1992) achieves something close to restoration while ensuring a potential commodity output of future timber, retaining old-growth characteristics across the landscape, and achieving forest sustainability. While the KNF's Forest Plan still leaves more trees per acre than before European settlement, the resulting forest is expected to create sustainable wildlife habitat.

Management Indicator Species

Overview of Management Indicator Species

In general, a MIS is one whose habitat requirements most reflect those of the habitat/community of concern. The intended use is to be an indicator of habitat quality, track effects of management on the habitat, and predict future conditions. The MIS concept (36 CFR 219.19) was adopted by the Forest Service to serve as a barometer for species viability at the Forest level. As stated before, MIS is defined as any species, groups of species, or species habitat elements selected to monitor to elucidate the affects of resource management on population recovery, maintenance of population viability, or ecosystem diversity (USDA Forest Service 1984). MIS are identified in individual Forest Plans for each National Forest. Within forest plans, MIS generally represent habitats or species thought to be sensitive to management activities. MIS serve multiple functions in forest planning. They can focus management direction developed for project alternatives during planning, provide a means to analyze effects of alternatives on biological diversity, and serve as a reliable feedback mechanism during forest plan implementation. The latter is accomplished by monitoring population trends in relationship to habitat changes resulting from the selected alternative (36 CFR 219.19 (a) (6)).

MIS are selected from one of five categories as mandated in the NFMA and the 1982 Rule (36 CFR 219.19 (a) (1)). The first three categories include species identified on State and Federal lists as endangered or threatened, or are species commonly hunted, fished, or trapped, or non-game species of special interest. These species are chosen not based on how management activities affect them but because they have importance inherent in their presence on the forest for either biological or social reasons. The last two groups identify species that have specific characteristics that forest planners believe make them effective indicators. These groups are species with special habitat needs that may be significantly influenced by planned management programs, or additional plant or animal species selected because their population changes are believed to indicate the effects of management activities on other species of selected biological communities or on water quality (36 CFR 219.19 (a)(1)).

Because the categories of MIS potentially contain thousands of species, agency regulations provide 5 principles which guide the selection of MIS:

Choose MIS to reflect major management issues and challenges. This principle guides selection of species that directly reflect the species correspondence to specific management. It suggests that the management practices are known as well as their affects on the species chosen.

MIS function to facilitate evaluation. Select MIS that are easily monitored, thereby, facilitating evaluation of management issues and challenges and their consequences on ecosystem function, i.e., assist in monitoring the Forest Plan.

Consider MIS chosen on neighboring planning units. Include species that are best monitored at larger spatial scales. This principle encourages cooperation among National Forests and understanding ecosystem at scales beyond forest borders.

Consider whether employing MIS is the best approach to evaluate the management problem. Because MIS are only one of many tools to evaluate management, they are not necessarily the best choice, e.g., measurement of actual habitat or habitat acreage may serve better than monitoring a particular species.

Choose an adequate but limited number of species. MIS are chosen to monitor the effectiveness of the Forest Plan and only those necessary to do so should be included. Because monitoring MIS can be costly and time intensive, it is important to be effective in both the particular species chosen as well as the total number of species in order to collect the best information with the allotted resources. Too many species would reduce a Forest's ability to effectively monitor and evaluate management activities.

Kaibab National Forest's Management Indicator Species

In 1987, the KNF selected 18 MIS species, all of which were maintained during the 1996 Forest Plan amendment (Table 1). Each species was selected to represent a particular habitat or habitat characteristic found on the forest. As indicators, they were selected to represent all wildlife and rare plant species found or associated with habitat or habitat components thought to indicate forest health and effects of management activities.

When the MIS species were selected, the Forest Plan called for even-aged timber management. Therefore, the table divides vegetation types by early and late seral stage. Eventually, as management continues under the revised 1996 Forest Plan, descriptions of "seral stage" and "stand conditions" will no longer apply due to application of uneven-aged management prescriptions. In addition to incorporating the MRNG, the 1996 amendment also included the Mexican spotted owl recovery plan. In areas where federally listed species have been located or are suspected to be, federal standards for the species take precedence.

It is important to note that not all of the species selected in 1986 specifically have value as MIS on the KNF. Some of the selected MIS do not actually occur on the KNF or occur too infrequently to be reliable indicators for the habitats they were selected to represent. Habitats for these species are either limited in frequency or only occur in areas

too limited to maintain a population of the species. Some species have proven to be impractical to monitor and others are poor indicators of management effects on the Forest. However, for species with populations of sufficient size and distribution, or for which significant effort has gone into population monitoring, population trends can be determined or inferred. Future Forest Plan revision will seek to identify a more parsimonious list. However, in order for current management to continue, this report will describe the current MIS.

Table 1. Management indicator species of the Kaibab National Forest, Coconino County, AZ and the habitat or habitat components they represent.

	<u>Management Indicator Species</u>	<u>Habitat or Habitat Component</u>
Insects	Aquatic macroinvertebrates	Riparian
	Cinnamon teal	Late-seral wetlands
	Northern Goshawk	Late-seral ponderosa pine
	Hairy woodpecker	Snags in ponderosa pine, mixed conifer and spruce-fir
	Lincoln's sparrow	Late-seral, high elevation (>7,000') riparian
	Lucy's warbler	Late-seral, low elevation (<7,000') riparian
Birds	Juniper titmouse	Late-seral pinyon-juniper, and snags in pinyon-juniper
	Pygmy nuthatch	Late-seral ponderosa pine
	Mexican Spotted owl	Late-seral mixed conifer and spruce-fir
	Turkey	Late-seral ponderosa pine
	Red-naped sapsucker	Late-seral aspen and snags in aspen
	Yellow-breasted chat	Late-seral, low elevation (<7,000') riparian
	Elk	Early-seral ponderosa pine, mixed conifer, spruce-fir
	Mule deer	Early-seral aspen and pinyon-juniper
Mammals	Pronghorn	Early- and late-seral grassland
	Red squirrel	Late-seral mixed conifer and spruce-fir
	Tassel-eared squirrel	Early-seral ponderosa pine
Plants	Arizona bugbane	Instead of describing a vegetation characteristic, The Forest Plan describes habitat where the plant is found.

Management Indicator Species Population Estimates

Overview

MIS were adopted to provide a means for evaluating the consequences of land management activities on the species, ecosystem diversity, and habitat condition. If a population of MIS is declining, it is assumed that it indicates a decline in its habitat and community. In order to evaluate activities, forests are mandated to monitor habitat and population trends. The 1982 rule specifies that "Population trends of the management

indicator species will be monitored...” and that “Inventories shall include quantitative data making possible the evaluation of diversity in terms of its prior and present conditions. For each planning alternative, the interdisciplinary team shall consider how diversity will be affected... (36 CFR 219.26).”

Habitat monitoring is well established in the literature and the methods standardized, but it often applies to plant species or habitat structure identified as MIS. However, direct relationships between habitat and vertebrate species can be difficult to obtain. Thus, direct sampling of vertebrate species may be necessary to collect the required quantitative data to assess population trends. Monitoring can be complex and expensive endeavor, but can also achieve the “best science” required for management analyses. Monitoring of habitat trends is equally important because changes in habitat conditions and population trends function together as indicators of ecological change. In many cases, making inferences regarding the consequences of management will be difficult without the complementary lines of evidence contained in habitat trend and population trend information.

Because both species information and habitat condition is important, we discuss both in the following species accounts prior to population trend estimates. The first section in the species accounts reviews natural history. This section provides general information about the species life history, with emphasis on habitat and the habitat components to which it is ecologically tied, i.e., space requirements, migration status, food habits and requirements, and predation or social elements affecting how the species functions within its community. The second section reports potential management impacts to habitat or habitat components important to the species. For most species, this section is general in nature; details should be reported in project evaluations (but see part 2, this document, for a more thorough investigation of habitat trends including comparative data sets for each habitat type). Where information was available for the Kaibab National Forest, this section discusses species specific impacts. The most common management treatments assessed are timber removal, grazing, and both prescribed and natural fire. The third section reports species trend estimates from multiple sources when available. Population data exists for some species, primarily game species and landbird species. Avian assessments include the KNF landbird surveys, breeding bird survey routes, and other constant effort surveys within and adjacent to KNF lands. For example, if the primary literature or a University thesis provided estimates of densities from research conducted on or adjacent to the forest, it is included here. This population data is generally either applicable only to local populations, as in the case of most game species surveys, or in aggregate across a bioregion that encompasses the KNF.

The final section of each species account is the population trend estimate. Trends were concluded to be increasing, decreasing or stable. Rather, population trends were extracted and summarized from published literature or reports when available. Estimates are made for the whole forest across districts and habitat type. Although this is counter intuitive as populations can change within habitat types, in response to different perturbations in different ways, or temporally, it is what is mandated by law to implement forest plans and evaluate alternatives in project planning. Where possible, we included

assessments or predictions of how the species is faring at different spatial levels, including within different habitat types, or in response to different treatments in addition to the forest wide estimate. However, existing population data and projected population trends suitable for use at a bioregional scale are not suitable for determination of cause and effect relationships. Confounding variables such as intermixed public and private land ownership patterns, variable land histories and changes in habitat, stochastic variables such as habitat disturbances from fire and climate change, and effects that occur off the National Forest lands make it difficult, if not impossible, to determine the cause of changes in population trend. Some factors, such as survey methodology, are controlled to limit variability. Nevertheless, population trends from breeding bird surveys are derived from aggregating data across many individual survey routes occurring across different National Forest System lands, State-managed lands, and private lands. Changes in habitat or populations that may be occurring differentially between public and private land cannot easily be distinguished in the aggregated population trends. For migratory species, it is even more difficult to isolate possible causal factors related to changes in population trend due to the possibility of effects in distant locations or along the migratory path. Nonetheless, general ecological theory suggests that changes in availability of overall habitat would be expected to change population capacity at the local scale. Natural variability also can be more pronounced for rare species or species at the edge of their range. A substantial portion of KNF MIS are species at the margins of their ranges or considered rare on the landscape. These smaller subpopulations are likely isolated from the larger meta-population and thus are more susceptible to stochastic events, potentially resulting in these subpopulations being dramatically reduced or eliminated from the forest. Because many of these species populations will never be well distributed across federal lands, they cannot be self-sustaining, making the trend analysis a legal exercise rather than a biological one.

Population Trend Assessment Sources

When assessing a species trend estimate, we made every attempt to use as many resources as available that represent “best available science”. For non-game species, we updated old sources with more accurate and reliable data sources. We then followed a 2-step method that first looked for actual density or abundance estimates and secondly followed with large scale population sources. As part of the first step, we searched primary peer-reviewed literature for historical and or current benchmarks. We then supplemented this information with data from theses or dissertations conducted on or adjacent to the KNF. In several instances, biologists from a variety of organizations have allowed us to use unpublished data. These data were assessed for accuracy and collection methods were evaluated for making valid comparisons with other sources. Finally, we used data collected directly from the KNF during our Landbird and MIS species surveys. The second step included assessing population abundance and trend estimates from a variety of sources that collected data in northern Arizona, including the Breeding Bird Survey (BBS), the Christmas Bird Count (CBC), and NatureServe Ranks (NSR). The smallest area at which these sources assess population trends is at the state level (BBS, CBC, NSR) and increase in scale to conservation regions or bioregions (BBS, NSR), and lastly to a global estimate (NSR). It is important to note that trend estimates from these larger sources are often conflicting and serve here to supplement actual data used in step

one. Thus, we interpreted such indices with caution and emphasize the importance of using all available information when assessing populations (Jackson et al. 2002).

For game species, state wildlife agencies already monitor population trends to determine harvest. Thus the forest uses information collected by game agencies as a means for cooperative planning and species management. The importance of state data is recognized in the 1982 planning regulations and states that "... monitoring [population trends] will be done in cooperation with state fish and wildlife agencies, to the extent practicable" (36 CFR 219.19). States generally set their game population objectives to ensure a harvestable surplus. Their population monitoring was used to assess trends in numbers and distribution across the forest.

The following is a description of the sources most relied on for population trend estimates:

Breeding Bird Surveys and Bird Conservation Regions

The Breeding Bird Survey (BBS) is the primary information source on population change and relative abundance for most North American bird species (U.S.D.I. U.S. Geologic Survey 2001, Sauer and Link 2002, Sauer et. al. 2003). Beginning in 1966, BBS started with standardized roadside surveys in the eastern United States and by 1968 were conducted across the contiguous United States and southern Canada. Currently, there are 4,500 randomly established, active roadside survey routes (Sauer et. al. 2003, Sauer et. al. 2004). The greatest density occurs in the Eastern, central Rocky Mountain, and the Pacific-Coast states. In Arizona, routes occur in low densities (Sauer et. al. 2004).

BBS data are challenging to analyze. The precision of abundance estimates varies by route number in a given region and the number of detections per species per route. A species with low abundance may in fact be rare or might be present but poorly sampled due to localized distribution, difficulty in observing the species due to behavior, or because it may be associated with habitats not adequately represented along roadways (Sauer et al. 2003). Species abundance estimates are assessed in terms of trends over time to determine population changes.

Bird conservation planning occurs for a variety of species groups (e.g., neotropical migrants, waterfowl, shorebirds, etc.) and at different spatial scales. The North American Bird Conservation Initiative was developed to provide a framework for conserving North American birds and to integrate the varied, ongoing efforts of national and regional groups. One result was the creation of Bird Conservation Regions (BCRs) across North America. BBS routes were reclassified to provide ecoregion-based analyses of bird population trends instead of those defined by political boundaries (Sauer et.al. 2003). Summarizing data by State can create biologically meaningless results due to the varied habitat types being aggregated. Instead, BCRs include similar habitats across portions of many States and Provinces. While the aggregated BCR area can be enormous relative to a single State or Province, these areas are more biologically meaningful.

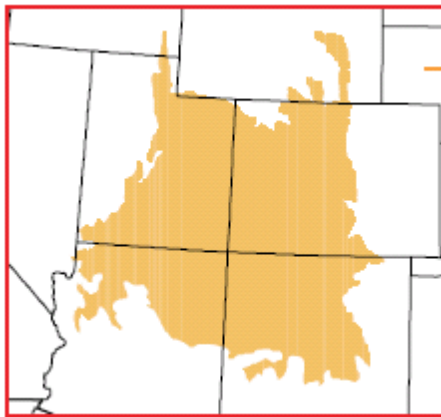
There are 37 BCRs across North America (Sauer et. al. 2005) two of which occur on the KNF: the Northern Kaibab Ranger District is in the Southern Rockies/Colorado Plateau BCR and the Williams and Tusayan Ranger Districts occur in the Sierra Madre Occidental BCR (Table 2). Historically, only species occurring on 14 or more BBS routes were summarized in order to limit the estimate's variance. When the data are available, BBS results are calculated for 3 time periods: 1966-1979, 1980-2005, and 1966-2005 (Sauer et.al. 2005). The starting date, 1966, was adopted for conformity, although considerable variation exists for BBS starting dates across North America. Species analyzed within BCRs can be classified into 12 guilds based on breeding habitat, nest type, nest location, and migration (Table 3).

We conducted 121 analyses using all the available BBS trend estimate and BCR trend estimate engines for the eight non-game avian MIS (Appendix 2). Regional trends were analyzed using "Southern Rockies" and "Arizona" for all three time periods. BCR trends were analyzed using "Southern Rockies/Colorado Plateau", "Sierra Madre Occidental", and "Arizona" for 1966-2005, 1980-2005, and for all applicable guilds (Table 3). Significance is defined as trends with p-values less than 0.1.

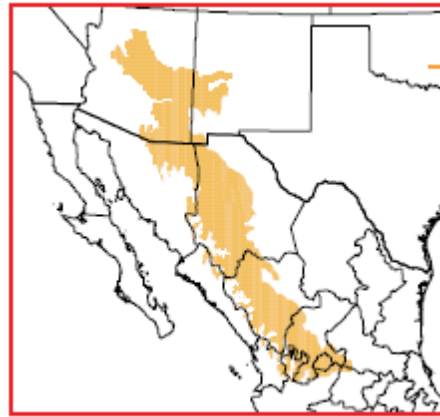
Table 2. Bird Conservation Regions overlapping the Kaibab National Forest, Coconino County, AZ.

Bird Conservation Region	Total # of BBS Routes in BCR¹	Overlapping Kaibab Ranger District(s)
Southern Rockies/Colorado Plateau	215	North Kaibab
Sierra Madre Occidental	35	Williams and Tusayan

¹The range of BBS routes per BCR = 1 to 371; mean = 120.



Map 1. Southern Rockies/Colorado Plateau.
(from US NABCI 2000)



Map 2. Sierra Madre Occidental.
(from US NABCI 2000)

Table 3. Management Indicator Species and associated guild groups, as defined in Sauer et al. (2005), for analysis by Bird Conservation Region using trend estimate engines for the Kaibab National Forest, Coconino County, AZ.

Management Indicator Species	Guilds									
	Breeding Habitats			Nest Type		Migration			Nest Location	
	<u>Woodland</u>	<u>Successional Scrub</u>	<u>Wetland Open Water</u>	<u>Cavity</u>	<u>Open cup</u>	<u>Permanent Resident</u>	<u>Short Distance</u>	<u>Neotropical</u>	<u>Ground/ Low</u>	<u>Mid-Story/ Canopy</u>
Hairy Woodpecker	X			X		X				
Red-naped Sapsucker	X			X			X			
Pygmy Nuthatch	X			X		X				X
Juniper Titmouse		X		X		X				X
Lincoln’s Sparrow		X			X			X	X	
Lucy’s Warbler		X		X			X			X
Yellow-breasted Chat		X			X			X	X	
Cinnamon Teal			X							

Kaibab National Forest Landbird Surveys

The KNF launched forest-wide landbird surveys in 2005 to address MIS monitoring obligations and cover those species not monitored by the AGFD. In addition, this approach resulted in a comprehensive avifauna list that allows us to track nearly all bird species occurring on the KNF. The survey effort was expanded in 2006 and again in 2007. It is intended to be repeated annually.

The surveys use point-transect distance sampling stratified by habitat-type. Similar to the BBS, these surveys are made up of routes with stops occurring every 250 m at which observers record all birds and MIS species and the distance to each individual for a 5 minute period (Appendix 3). Detections and distance to individuals are recorded for MIS only along transect segments between point count stations (hence the name “point-transect surveys”). Whereas point counts are effective in monitoring larger species groups (i.e., everything seen or heard), transects work better when monitoring limited numbers of species (here that equals MIS). Combined, the results allow us to effectively meet our MIS obligations while also monitoring the KNF avifauna community in general. Our pilot effort in 2005 used 2-15 transects per habitat type. We increased the total sample in 2006, but could not achieve the level of statistical robustness that we felt would provide adequate results. To increase survey effectiveness, we dropped some habitat types in 2007 (e.g., aspen (*Populus tremuloides*) and montane grassland) and increased the effort in those habitats most impacted by management. Ultimately, the KNF’s goal is 30 point-transects per habitat type for the ponderosa pine and mixed conifer forests and pinyon-juniper woodlands, allowing for more meaningful analyses. In addition, the KNF uses line-transect distance sampling to monitor red and tassel-eared squirrels via direct detection and squirrel sign. Because data is analyzed using distance sampling for both birds and mammals, estimates of species are statistically reliable and address scientific design issues inherent with many monitoring programs. Randomly selecting sites stratified by habitat cover type from across the KNF allows us to determine forest wide trends by species. Therefore, the KNF strives to provide the “best science available” while meeting its monitoring responsibility.

NatureServe Ranks

The Nature Conservancy and the Natural Heritage Network jointly established NatureServe in 1999 to facilitate the use of biodiversity information in conservation planning and management. NatureServe works in partnership with 85 independent Natural Heritage programs and Conservation Data Centers that gather scientific information on rare species and ecosystems.

The conservation status of a species or community is designated by a number from 1 to 5, preceded by a letter reflecting the appropriate geographic scale of the assessment (G = Global, N = National, and S = Subnational). The numbers have the following meaning:

1. **Critically Imperiled:** At very high risk of extinction due to extreme rarity (often 5 or fewer populations), very steep declines, or other factors.
2. **Imperiled:** At high risk of extinction due to very restricted range, very few populations (often 20 or fewer), steep declines, or other factors.
3. **Vulnerable:** At moderate risk of extinction due to a restricted range, relatively few populations (often 80 or fewer), recent and widespread declines, or other factors.
4. **Apparently Secure:** Uncommon but not rare; some cause for long-term concern due to declines or other factors.

5. **Secure:** Common; widespread and abundant.

For example, G1 would indicate that a species is critically imperiled across its entire range (i.e., globally). In this sense, the species as a whole is regarded as being at very high risk of extinction. A rank of S3 would indicate the species is vulnerable and at moderate risk within a particular state or province, even though it may be more secure elsewhere. Extinct or missing species and ecological communities are designated with either an "X" (presumed extinct or extirpated) if there is no expectation that they still survive, or an "H" (possibly extinct or extirpated) if they are known only from historical records but there is a chance they may still exist. Other variants and qualifiers are used to add information or indicate uncertainty.

Arizona Game and Fish Department Annual Summary Report

The AGFD produces an annual summary regarding the status of game species. These reports include information for the previous five years. Trend lines in the following graphs were created by running polynomial regressions on the data. The counts give estimates of population trends, but they do not provide estimates of actual population numbers.

AGFD survey methods include: Fixed-wing aircraft for pronghorn (some flight time is allotted to deer surveys where habitat conditions favor this method); helicopters for deer and elk; foot, horseback, and vehicular surveys; fixed-route surveys repeated in the same manner and at the same time every year. The latter methods can use call counts and sight indices to aid in calculating relative abundance.

The AGFD Hunt Units overlap portions of the Forest and include adjacent areas outside the Forest boundary (Appendix 4). Trends from the following Hunt Units are included in this report:

Williams Ranger District	Units 7 (SubUnit 7W, in particular), 8, and 10
Tusayan Ranger District	Unit 9
North Kaibab Ranger District	Unit 12 (SubUnit 12A, in particular)

SPECIES ACCOUNTS

Aquatic Macroinvertebrates

Invertebrate is an English word that describes any animal without a spinal column. The group includes 97% of all animal species. That means that all the animals with spinal cords, mammals, reptiles, amphibians, and fish only make up 3% of the known species occupying the earth! Although macroinvertebrates can be small, you can still see many of the species with the naked eye.

Life History:

Aquatic macroinvertebrates live in a variety of riparian habitats where water is present. Together, the group of species provides a vital link in the food chain between primary producers (algae and macrophytes) and fish. Because of their strict habitat requirements, many species are useful indicators of aquatic habitat conditions and changes (Mangum 1986). Our MIS aquatic macroinvertebrates include such common insects as mayflies (Ephemeroptera), stoneflies (Plecoptera), caddisflies (Trichoptera), and two-winged flies (Diptera). Additionally this group includes crustaceans (Crustacea), mollusks (Mollusca), and freshwater earthworms (Oligochaeta; Mangum 1986). Aquatic macroinvertebrates were selected for monitoring the health of late-seral, riparian habitats within the Forest because a diverse and abundant array of these species indicates a functional aquatic habitat. While many macroinvertebrates inhabit still water, mayflies, stoneflies, caddisflies indicate good water quality in stream systems.

Because the force of running water can be powerful, aquatic macroinvertebrates have evolved a variety of anatomical and behavioral adaptations to survive in this challenging ecosystem. Anatomical adaptations such as a flattened body, streamlined shape, suckers, friction pads and hooks, small size, secretions, and ballast (such as caddisfly houses) all help to resist the force of the water. Behavioral adaptations such as living in slow water among vegetation or friction layers on the stream bottom, upstream movement in the water, and upstream dispersal of winged adults allow individuals to escape from the harsher areas of streams (Hynes 1970).

Aquatic insects go through a series of metamorphoses or life stages from hatching to adults. Complete metamorphosis for aquatic insects progresses through egg, larva, pupa, and adult while incomplete progresses through egg, nymph, and adult. Mayflies and stoneflies are examples of species that exhibit incomplete metamorphosis while caddisflies and diptera are examples of species that exhibit complete metamorphosis.

Most aquatic insects do not require water during all life stages. Hynes (1970) noted eggs of many aquatic insects could survive dry for many months and gave an example of several taxa that survived a D.D.T. treatment of a tropical stream, presumably as eggs. According to Hynes (1970), extended hatching periods are common in many aquatic stream insects. This as well as producing multiple generations per year allows them to escape flooding and drought conditions that could be deleterious to their populations.

Only 1 of the 3 perennial streams on the KNF has a resident fish population. Big Spring (Williams RD) commonly measures inches wide and does not sustain fish. The short stream portion of Big Springs (NKRd) is used for spawning by introduced rainbow trout

(*Oncorhynchus mykiss*), which spend the rest of their lives in pools below the springs. North Canyon Creek in the Saddle Mountain Wilderness (NKRD) supports Apache trout (*Oncorhynchus apache*), a species listed under the Endangered Species Act and introduced to the stream from Ord Creek, eastern Arizona, by the AGFD in 1963. North Canyon Creek typically flows for less than two miles within the KNF boundary. Because a listed fish species occurs in what many consider the only true flowing stream on the KNF, macroinvertebrates were selected to indicate stream health for North Canyon Creek. The AGFD periodically conducts surveys to investigate the viability of the population and environmental factors such as flow rate and evidence of erosion or sedimentation. Elevations range from about 7,000 to 8,000 feet for North Canyon Creek and it is primarily fed by winter snow pack. The surrounding forest includes mixed conifer forest with box elder, bigtooth maple (*Acer grandidentatum*), Gambel oak (*Quercus gambelii*), and New Mexico locust (*Robinia neomexicana*). The Civilian Conservation Corps (CCC) installed a series of log check dams in the 1930s to create plunge pools, resulting in habitat currently used by Apache trout.

Potential Management Impacts:

Macroinvertebrates are sensitive to impacts from many forest practices, and fluctuations in their relative abundances may be interpreted to determine whether water quality and aquatic habitats have been impaired relative to reference condition. Reference condition is defined as aquatic macroinvertebrate community composition in the absence of impacts from human activities such as timber harvest, grazing, and road building. Sensitivity to alteration of habitat for such features as water temperature, riparian vegetation, sedimentation, and water chemistry vary within the macroinvertebrate community, allowing identification of factors that may be compromising water quality and aquatic habitats. These aquatic insects work well as management indicators because they represent a diverse group, including long-lived and sedentary species, which react strongly and often predictably to human influences on aquatic systems (Cairns and Pratt 1993). These relationships have been thoroughly catalogued and the sampling protocol is relatively simple.

Because macroinvertebrates are tied to the aquatic portion of riparian areas, maintenance or enhancement of these systems will likely benefit macroinvertebrate populations. A hydrologically functional system occurs when degradational (erosional) processes are counter-balanced by aggradational (sedimentary) processes over time. The entire watershed influences quantities of sediments and nutrients moving in and out of an aquatic system. Ponding, flooding frequency and duration, and modification of inorganic and organic chemical distribution all affect the system's hydrologic stability.

The log check dams installed by the CCC are beginning to break down. Additionally, pool habitat was lost due to flooding in 2005. Although not directly affecting the macroinvertebrates, these essential elements for maintaining Apache trout are nearing the end of their functional life. Without replacement, the Apache trout population in North Canyon Creek is at risk.

Perhaps the biggest threat to maintaining vertebrate and invertebrate species in North Canyon Creek is fire. The watershed consists of forests shaped by over a century of fire suppression that are vulnerable to stand replacement, high-severity crown fire. Examples of such fire behavior that have recently threatened North Canyon Creek include the Aspen Fire, Outlet Fire, Poplar

Fire and Warm Fire. Should a similar wildfire reach North Canyon Creek, it would dramatically alter stream conditions and likely extirpate Apache trout. The lack of true riparian habitat means that the same general forest occurring across the watershed also forms the canopy over much of the creek. A high severity fire could disrupt the fundamental hydrology of the system and eliminate the resident fauna. Its isolation would retard or preclude recovery or recolonization by vertebrate and invertebrate species.

Population Data:

The trout in North Canyon Creek have been doing so well that twice in the 1990's fish were removed for reintroduction back into Ord Creek. In 2005, fish were again captured from North Canyon Creek for reintroduction into Hayground Creek in the White Mountains of eastern Arizona (Rinker et al. 2006). Although this population is non-native in North Canyon Creek, it represents an important conservation tool for future reintroduction efforts. Additionally, as long as the population is doing well, it can be interpreted that the creek is a fully functioning riparian system.

Two aquatic ecosystem inventories have been conducted at North Canyon Creek by AGFD (Mangum 1990 and 1998; Table 4, 5). Biomass data is used to assess benthic community productivity and health. The biotic condition index (BCI) indicates, as a percentage, how close an aquatic ecosystem is to an estimated potential. BCI is used for evaluating ecosystem integrity and health.

Table 4. Macroinvertebrate analysis of North Canyon Creek, Kaibab National Forest, Coconino County, AZ (Mangum 1990 and 1998).

Year	Station	Date	Diversity Index DAT ¹ (mean)	Biomass g/m ² (mean)	# of Taxa	Biotic Condition Index (BCI)
1998	1	8-98	5.1	0.5	22	81
	2	8-98	5.0	0.5	14	83
1990	1	6-90	6.4	0.4	18	85
	2	6-90	5.1	0.6	15	85

¹The diversity index (DAT) is a diversity measure combining dominance and number of taxa.

Table 5. Reference values for macroinvertebrate Diversity Index for the Kaibab National Forest, Coconino County, AZ.

Scale	DAT	Biomass	BCI
Excellent	18 - 26	4.0 - 12.0	Above 90
Good	11 - 17	1.6 - 4.0	80 - 90
Fair	6 - 10	0.6 - 1.5	72 - 79
Poor	0 - 5	0.0 - 0.5	Below 72

The survey data indicate relatively low biodiversity, which is likely due to the unusual isolation of this stream ecosystem (Mangum 1998). Low numbers of individuals sampled suggest an unstable ecosystem. Should stochastic events destabilize the ecosystem, e.g., high-severity wildfire, it could take many years before aquatic macroinvertebrate populations reestablished here. The U.S. Fish and Wildlife Service collected aquatic insects from North Canyon Creek in 2007. Species collected included a significant range extension for a mayfly (*Callibaetis falsus*), a new genus of stonefly recorded for Arizona (*Alloperla sp*), and a previously undescribed

species of stonefly (*Sweltsa sp*; Dave Smith, US Fish and Wildlife Service, personal communication). As of February, 2007, the samples were being examined at Brigham Young University.

Trend Estimate:

The data from the North Canyon Creek studies indicate stable conditions. The BCI suggest the habitat is in good condition and that spawning substrate for Apache trout is suitable. The ability to maintain these conditions over time is threatened in both the short- and long-term. In the short-term, failure of the CCC log dams will lead to a loss of key habitat for some aquatic invertebrate species (and Apache trout). In the long-term by risk of high-severity, stand replacement fire could eliminate Apache trout and the macroinvertebrate community upon which they depend.

Cinnamon Teal

The cinnamon teal (*Anas cyanoptera*) is one of the most common dabbling ducks in North America (Gammonley 1996b). The Latin word cyanoptera literally means “blue wing,” referring to the small spot of blue which is best seen when the wing is spread (Terres 1980). They can be seen in wetlands from southwestern Canada south to the tip of South America.

Life History:

Cinnamon teal were selected to represent species using late-seral wetlands (ponds, marshes, and ephemeral wetlands) within the Forest. Locally, this pan-American species is an uncommon to abundant breeder in wetland areas throughout the Great Basin and arid western U.S., including northern Arizona (Brown 1985). Dabbling ducks breed in freshwater, seasonal, and semi-permanent wetlands within these areas. Size and type of wetlands used varies and includes large marsh systems, natural basins, reservoirs, sluggish streams, ditches, and stock ponds. Of those, basins with stands of well-developed emergent vegetation are preferred. In Arizona, teal use shallow areas of wetlands and prefer vegetation that is either flooded or wetland associates (Corman and Wise–Gervias 2005). Additionally, Gammonley (1996a) found that cinnamon teal abundance is positively related to wetland size and habitat diversity, even if the species uses most wetland sizes.

Like most dabbling ducks, cinnamon teal are migrants. This species is typically a short to intermediate distance migrant. Unlike other dabblers, cinnamon teal migrate earlier in the fall with most birds gone from northern breeding areas by October (Bellrose 1980) and by November in southern breeding areas (Gammonley 1996a). Cinnamon teal winter in Mexico and Central America and use similar habitat such as tidal estuaries; freshwater, brackish, and salt marshes; agricultural fields; and mangrove forests (Saunders and Saunders 1981, Kramer and Migoya 1989).

Cinnamon teal aggressively defend their nest site and a small surrounding area. However, relative to other teal, the cinnamon teal is less aggressive in nature, having less intense territoriality with limited aggression displays (Gammonley 1996a). In Utah, cinnamon teal defended territories including the nest site and up to 100 m away, but most defended areas were approximately 30 m² (Spencer 1953). Territories also included the male’s favorite loafing sites for use while the female was on the nest. Territoriality is likely related to density and habitat quality (Gammonley 1996a).

Cinnamon teal place their nest in low, matted, dead stems of perennial vegetation about 12 to 15 inches high (Harrison 1979) after aerial scouting by the female (Gammonley 1996a). Cinnamon teal prefer to place their nest in the dead portion of vegetation to conceal it from all sides. Therefore, they select plants with similar structure rather than a particular species, e.g., baltic rush (*Juncus balticus*), saltgrass (*Distichlis spicatum*), spikerush (*Eleocharis macrostachya*), tufted hairgrass (*Deschampsia caespitosa*), western wheatgrass (*Agropyron smithii*), and foxtail barley (*Hordeum jubatum*) (Gammonley 1996a). When this vegetation is not available, nests have been observed at the base of greasewood (*Sarcobatus vermiculatus*), rabbitbrush (*Chrysomanthus spp.*), willows (*Salix spp.*), and small ponderosa pines. When terrestrial habitat is absent or degraded, nests have been found in dense bulrushes (*Scirpus sp.*), cattails (*Typha spp.*), and sedges (*Carex sp.*) adjacent to and over water. If available, female cinnamon teal place their nests under a matt of vegetation accessed by tunnels (Gammonley 1996a).

Cinnamon teals prefer to forage in shallow flooded areas along the edges of wetlands and are omnivorous, eating both vegetative and animal matter. As dabblers, they feed on the surface or in vegetation emerging from the water (Gammonley 1995). Connelly (1977) noted that cinnamon teals foraged twice as much in emergent vegetation relative to open water. Adults often feed together and throughout the day, but prefer the morning and late afternoon hours (Gammonley 1995). The main food items taken include seeds, aquatic vegetation, aquatic and semi-terrestrial insects, snails and zooplankton (Cox 1993, Migoya and Baldassarre 1993, Thorn and Zwank 1993, Hohman and Ankney 1994).

Predators are varied and include birds and mammals. Eggs are predated by common ravens (*Corvus corax*), American crows (*Corvus brachyrhynchos*), California gulls (*Larus californicus*), bald eagles (*Haliaeetus leucocephalus*), and black-billed magpies (*Pica pica*) (Spencer 1953, Myers 1982, Gammonley 1996a). Documented mammalian predators include coyote (*Canis latrans*), striped skunk (*Mephitis mephitis*), raccoon (*Procyon lotor*), mink (*Mustela vison*), weasels (*Mustela spp*), opossum (*Didelphis virginiana*), and Norway rat (*Rattus norvegicus*) (Spencer 1953, Miller and Collins 1954, Anderson 1957, Gammonley 1996a). Predators of adult ducks include coyote, red fox, mink, great horned owl (*Bubo virginianus*), peregrine falcon (*Falco peregrinus*), and northern harrier (*Circus cyaneus*) with nesting hens taken more often than drakes (Spencer 1953, Gammonley 1996a). Nests are also parasitized by mallard and ruddy ducks which have also been known to remove teal eggs (Ehrlich et al. 1988).

Potential Management Impacts:

According to the AGFD data, the KNF lands contain about 2.4% of total riparian acres in all 4th HUC watersheds occurring on the Forest. According to the AGFD riparian data, the KNF contains about 129 riparian acres, or 14% of total riparian acres in all intersecting 5th field watersheds. A Forest-wide Riparian Survey was conducted on approximately 566 acres of actual mapped wetlands, in 1990. During that survey, 129 springs and seeps and 492 stock tanks were identified. Only 16 stock tanks actually met classification requirements for wetlands and within these only a few contain actual vegetation suitable as wildlife habitat. In total, 88 actual wetland and riparian areas were identified. That survey was repeated in 2008, with the addition of 7 more riparian features bringing the total to 95. This included perennial and ephemeral springs, stock tanks, reservoirs, intermittent streams and seasonal wetlands. Of the riparian features surveyed,

over half (63) remained static since 1990. Twenty-two of the total riparian areas surveyed were considered functional but at risk, 18 were considered non-functional and 55 were considered to be in proper functioning condition.

Wetland areas in fair and poor condition do not provide optimum riparian habitat conditions. Of the wetland and riparian areas in good condition, most are ephemeral waters that are highly dependent on annual weather cycles. Where areas are properly fenced to prevent grazing, they are in functional condition. Observations and limited documentation indicate accessible, unfenced stock tanks capable of supporting emergent vegetation are not functioning well. Suitable vegetation may only occur every few years in normal conditions and less during drought conditions. Overall, the extent, diversity, and condition of wetland and riparian habitat on the KNF limits the contributions the Forest can make towards sustaining populations of cinnamon teal.

Cinnamon teal have been documented on the KNF within wetland areas supporting suitable habitat. However, the presence of individual breeding birds is not the same as having a resident population. The waters and adjacent uplands on the KNF that are suitable for breeding are ephemeral or are subject to annual fluctuations due to variation in annual precipitation. Consequently, this affects the number of individuals and breeding pairs on the forest each year. While the ephemeral habitat present on the KNF may contribute towards a regional population of cinnamon teal in wet years, it does not allow for tracking of populations on the KNF. Additionally, the ephemeral habitat may create sink habitat that, over the long-term, may negatively impact overall productivity and abundance.

Water scarcity throughout the arid West concentrates use by both grazers and humans in areas where water occurs. Human impact on wetland areas from fishing and other recreational activities causes increased disturbance of nesting teal (Gammonley 1996a) and could reduce productivity. Grazing can cause disturbance both directly and indirectly. Indirectly, grazing can increase erosion and silt runoff into wetlands, reducing riparian vegetation and thus cinnamon teal breeding habitat. Loss of vegetation, in turn, creates more indirect disturbance as it accelerates erosion, soil compaction, and sedimentation. Directly, riparian vegetation is degraded when trampled or reduced when browsed. This occurs at the water's edge and also in adjacent upland vegetation used for nesting and foraging by teal (Gammonley 1996a). In a recent study that included cattle and waterfowl interactions, they found little threat from trampling of waterfowl nests with 3 of 262 nests (1%) affected by cattle (Koper and Schmiegelow 2007). There was 3-7% increase in duck nesting success ($n = 136$) per 1-cm (0.4 in.) increase in vegetation height. However, the authors felt grazing intensity was too light to account for a cause and effect relationship between grazing, vegetation height, and nesting success (Koper and Schmiegelow 2007). On the KNF, both cattle and elk grazing occur on most ephemeral riparian and wetland areas. Steinke (2007) concluded grazing the ephemeral wetlands on the KNF reduces plant and animal diversity, negatively affecting riparian habitat and those species that depend on it for their survival.

In California pasturelands, Carroll et al. (2007) showed that rotational grazing with cattle present in the fall can result in vegetation density similar to or greater than that of non-grazed areas. Additionally, productivity by dabbling ducks, including cinnamon teal, did not differ between

grazed and non-grazed areas. While pastureland cannot be equated with riparian areas, it does suggest that careful timing of grazing may reduce impacts to riparian areas. National Forests have gone to great expense and effort to fence stock tanks and natural depressions where established or semi-permanent riparian habitat occurs. Additionally, fences can also reduce predation of nests, the foremost cause of reproductive failure in waterfowl (Ackerman et al. 2003). Pearse and Ratti (2004) found increased survival in mallard nests and ducklings when all predators, including coyotes, were prevented from accessing wetlands. On the KNF, Steinke (2007) notes that fenced wetland and riparian areas are in much better condition than those areas that are not. However, livestock fences are ineffective at excluding elk and impacts from this unmanaged grazing can be similar to those from cattle.

Drought can also cause reduced levels of upland and wetland vegetation, often eliminating cinnamon teal breeding habitat. On the KNF, drought conditions have existed for nearly a decade, with only a few years of normal precipitation. Drought induced reductions and drying of vegetation, surface organic matter, and ground cover, puts soils at risk of accelerated erosion and sediment delivery into wetlands. This lowers water levels and increases siltation, decreasing the likelihood of wetland recovery (Steinke 2007). Ultimately, this reduces plant diversity by limiting the species that can then occupy degraded habitat.

As vegetation dries out, there is increased risk of wildfire. Vegetation response to fire varies by vegetation type in addition to habitat type, i.e., grasses respond differently than shrubs which can be different than trees. Riparian areas are not fire-adapted ecosystems and high-severity fire can degrade or destroy riparian habitat. Stand-replacing fires in adjacent upland forests can accelerate erosion, create hydrophobic soils, and lead to sedimentation of wetlands. The absence of fire has created a banking of soil that would have otherwise been released incrementally had frequent, low severity burns occurred over the last century. This increases the soil load released after high-severity, high-intensity fires, exasperating sediment effects on wetland and riparian habitat. Conversely, patchy, low-severity fire can enhance habitat under moist conditions. Although ducks and ground nesting grassland songbirds selected nest sites with deeper litter, overall nesting success increased with decreased litter depths (Koper and Schmiegelow 2007). Nesting habitat may be enhanced with a mosaic or high interspersions of habitat characteristics.

Waterbirds have high nutritional and energetic demands (Alisauskas and Ankney 1992) and can be expected to select foraging habitats that have a high abundance of accessible foods (Laubham and Gammonley 2000). Because cinnamon teal females shift from a plant-dominated diet during the non-breeding season to a protein-rich aquatic invertebrate diet prior to and during egg-laying to meet protein demands, it is essential that their breeding habitat contain rich and substantial invertebrate prey resources (Gammonley 1995). Elevation and dominant vegetation among locations may influence invertebrate community characteristics (Reid 1985). Management that increases riparian plant diversity and the associated invertebrate community will likely benefit cinnamon teal as it will provide beneficial foraging resources.

Population Data:

Breeding Bird Survey data (Sauer et al. 2005) resulted in only 5 analyses for the cinnamon teal. All three time periods for Arizona show a declining but non-significant population trend (for

example, 1966-2005: trend = -11.5, $p = 0.35$, Fig. 1). Trends were also insignificant for the Southern Rockies regional analyses (for 1966-2005 and 1980-2005 trends = 0.6 and 1.2, $p = 0.92$

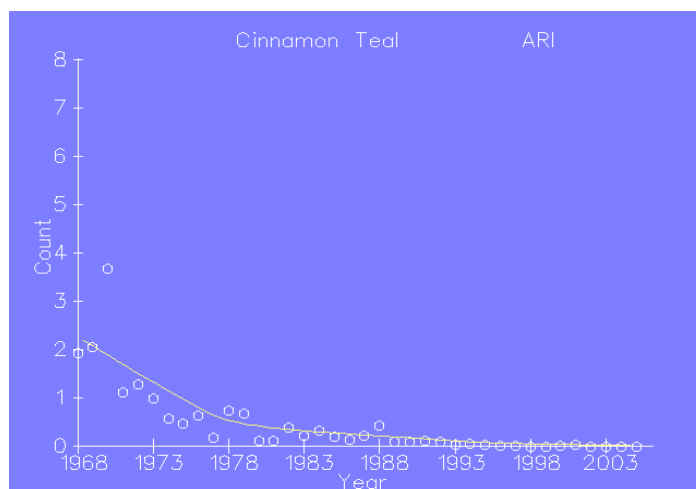


Figure 1. Cinnamon teal population trend data for Arizona from 1966-2005, BBS data (Sauer et al., 2005).

and 0.9). All analyses suffer from significant deficiencies, including low abundance (less than 1.0 birds/route), low sample size (less than 14 routes), imprecision (3%-year change would not be detected over the long term), and possible inconsistency in trend over time (sub-interval trends were significantly different [$P < 0.05$] from each other) (Sauer et al. 1999). However, no birds were recorded in Arizona from 1993 to present. While this doesn't allow for analysis, it does indicate that this species may no longer occur in areas where they were formerly present. Cinnamon teal are listed as G5, N5B N5N, and S5, by NatureServe. The species is considered to be demonstrably widespread, abundant, and secure globally, nationally, and statewide.

The North American breeding population of cinnamon teal is estimated at 260,000–300,000 (Bellrose 1980). Gammonley (1995a) states that few estimates are available for this species and those that are available are outdated or do not use modern abundance estimation techniques. At Ogden Bay Utah, Spencer (1953) recorded 6.9-7.2 pairs per 100 ha (247 ac). In Arizona's San Francisco Peaks and White Mountains, breeding-pair density estimates averaged 0.18 pairs per 100 ha, which translate to 1.32 and 1.27 pairs/wetland, respectively (Gammonley 1996b). Because reliable population estimates are lacking, long-term population trends remain poorly known. Locally, Northern Arizona Audubon has casually conducted repeat surveys on 8 lakes and tanks on the South Zone of the KNF (Kaibab Lake, Davenport Lake, Poquette Tank, Whitehorse Lake, JD Dam, Ruin Tank area, Scholz Lake, and Coleman Lake). In 2005, 4 male cinnamon teal were reported. Surveys during 2006 resulted in no sightings. In March of 2007, 4 cinnamon teal were recorded at JD Dam, but these individuals were considered migrants. These results illustrate the limited role the KNF plays in supporting local populations of cinnamon teal given the limited wetland habitat and its dependency on precipitation patterns.

Because the KNF supports individual cinnamon teal rather than *populations* of teal, this species is inappropriate as a management indicator. Although they represent important habitat, the climate-caused impacts at the edge of their breeding range are difficult to impossible to separate

from potential management impacts. Additionally, without a substantial population to monitor, trends cannot be ascertained. Even if similar numbers to those on the neighboring San Francisco Peaks were present on the KNF, there would only be 10.56 cinnamon teal across the forest (Gammonly 1996). Also, cinnamon teal have not been documented as having tight, inter-specific relationships with other species, further questioning their role as indicator species.

Trend Estimate:

The KNF is required to make estimates of population trends by law; therefore, if populations of cinnamon teal did occur on the KNF, they would likely reflect the same stable to potentially negative trend suggested by BBS data for Arizona. Overall, cinnamon teal use of the KNF is variable and there is limited breeding in the few suitable wetlands. However, it is likely that cinnamon teal use the KNF as stop-over habitat while migrating to breeding areas. Use by migratory birds will likely remain stable, although this trend is largely independent of vegetation management performed by the KNF. If fencing around ephemeral wetlands and surrounding meadows is increased, maintained, or built to exclude elk, migration stop-over use by cinnamon teal would likely increase. Most important however, is reconsidering the appropriateness of this species as a MIS.

Northern Goshawk

The northern goshawk is a pandemic species found across North America and Eurasia. While named for its nobility it is revered for its ferocity. Attila the Hun wore an image of a northern goshawk on his helmet to scare his enemies. This reflects the goshawks habit of commonly attacking people and other animals that approach their nests too closely.

Habitat characteristics:

The Kaibab Plateau holds one of the most concentrated populations of goshawks known in North America. The northern goshawk is classified as a Forest Service Sensitive species and is a MIS for the KNF, selected to represent species using late-seral, ponderosa pine (*Pinus ponderosa*) habitat. Goshawks occupy nearly every forest and woodland habitat type that occurs within the hawk's geographic range. In the west, goshawk populations occupy multiple forest types including Douglas fir (*Pseudotsuga menziesii*), various pines, and aspen (Reynolds et al. 1982, Younk and Bechard 1994, Siders and Kennedy 1996, Squires and Ruggiero 1996). In the southwestern United States, this species is primarily found in ponderosa pine forests (Erickson 1987, Reynolds et al. 1994).

Northern goshawks are considered partial migrants (Berthold 1993, Reynolds et al. 1994, Squires and Ruggiero 1995) and migrate during winter in response to food availability on breeding grounds (Squires and Reynolds 1997). Some goshawk populations exhibit short winter movements to lower elevations or to more open habitat types (Squires and Reynolds 1997). Also, irruptions of adults and juveniles from the breeding range during winter have been documented with adults often returning to breed after such movements (Campbell et al. 1990).

While on their breeding grounds, goshawks are territorial and defend the area around their nests, but pairs may overlap in other areas of their home range. Overall, home range can vary from 570–3,500 ha (1,408.5 – 8,648.7 ac), depending on sex and habitat characteristics (Squires and Reynolds 1997). Defended core areas, including the nest, are approximately 32% of home range

area (Kennedy et al. 1994). Male home ranges are usually larger than the female's home range (Hargis et al. 1994, Kennedy et al. 1994). In Arizona, male home ranges averaged $1,758 \text{ ha} \pm 500 \text{ SD}$ ($4344.1 \pm 1235.5 \text{ ac}$, Bright-Smith and Mannan 1994) and the average distance between neighboring nests was $3.0 \text{ km} \pm 0.83 \text{ SD}$ ($1.86 \pm 0.52 \text{ mi}$, Reynolds et al. 1994).

Descriptions of forests and woodlands used for breeding by goshawks show great variation in horizontal and vertical vegetation structure and where many of the areas do not produce closed forests with tall trees and continuous canopies that is purported to be required by the birds (Franklin and Dyrness 1973, Eyre 1980, Brown 1982, Barbour and Billings 1988). Thus, the old growth or late seral habitat type that this species is chosen for as a MIS is not necessarily the species preferred habitat type. Instead, it may be just one of many versions of forest types that fit into the broader structural context of its preferred breeding habitat. However, despite the wide diversity of habitats occupied by goshawks, within a habitat type, goshawk nest areas are consistently comprised of mature and older forests (Thomas et al. 1988, Habeck 1988, Bolgiano 1989, Hunter 1989, Franklin and Spies 1991, Kaufmann et al. 1992). These mature and older forests include, but are not limited to, old growth, and are typically concentrated within 30 acres surrounding the nest (Reynolds 2005). Typically, nest areas are composed of large, dense trees, closed canopies created by a variety of tree sizes, and open understories, but exact structure depends on forest type, elevation, and growth site potential (Bartelt 1974, Reynolds et al. 1982, Moore and Henny 1983, Hall 1984, Spieser and Bosakowski 1987, Lang 1994, Siders and Kennedy 1994, Daw 1996, Siders and Kennedy 1996, Squires and Ruggiero 1996, Desimone 1997, Daw et al. 1998, Keane 1999, Finn et al. 2002b). In Arizona, Crocker-Bedford and Chaney (1988) reported goshawks nested in ponderosa pine stands with $> 70\%$ canopy cover, but Lang (1994) found pairs occupying territories with 31-33% canopy cover. Although variable, habitat structure is more important than composition in the nest area (Reynolds 1983, Erickson 1987, Reynolds et al. 1992, Rissler 1995). Nests are built in coniferous or deciduous trees (Bent 1937, Reynolds et al. 1982), but western populations typically use conifers such as ponderosa pine, Douglas fir, white fir (*Abies concolor*), California red fir (*Abies magnifica*), western larch (*Larix occidentalis*), western hemlock (*Tsuga heterophylla*), and lodgepole pine (*Pinus contorta*, Reynolds et al. 1982, Moore and Henny 1983, Squires and Ruggiero 1996) although some deciduous trees are used such as aspen (Doyle and Smith 1994, Younk and Bechard 1994, Squires and Ruggiero 1996). In the southwest, goshawks use ponderosa pine extensively (Reynolds et al. 1992). Goshawks construct stick nests in the lower third of the largest tree available (Reynolds et al. 1982, Speiser and Bosakowski 1987, Hargis et al. 1994, Squires and Ruggiero 1996). Nest height significantly correlated with nest-tree height (Kennedy 1988), thus tree size and structure may be more important than tree species.

Overall, goshawks are closely tied to prey resources and less so to forest habitat type. If there is ample prey available in or adjacent to areas with preferred nesting structure, goshawks will nest regardless if the habitat type is forests, woodlands, or shrub lands (Swem and Adams 1992, Younk and Bechard 1992, 1994). Goshawks like to forage in habitat with relatively open understories so they can easily see and pursue their prey, or use open forest habitats because they can hunt from perch trees for rabbits or ground squirrels in openings between trees (Younk and Bechard 1992, 1994). The variety of foraging habitat lends to the variety of prey items taken. In general, goshawks primarily eat medium-sized birds (e.g., woodpeckers and jays) and small mammals (e.g., squirrels and rabbits, Reynolds et al. 1992).

Because adult northern goshawks are territorial they are, by nature, not a social bird. Not only are goshawks territorial against their own species, but also other raptors while on their breeding range and will readily kill neighboring raptors (Beebe 1974, Kostrzewa 1991). Adults actively defend their nests and have been known to attack red-tailed hawks (*Buteo jamaicensis*; Crannell and DeStefano 1992), short-eared owls (*Asio flammeus*; Lindberg 1977), and great-horned owls (*Bubo virginianus*, Squires and Reynolds 1997). They are solitary outside the breeding season, but fledglings remain together near the nest until dispersal (Reynolds and Wight 1978, Kenward et al. 1993, Squires and Reynolds 1997).

The ferocity of goshawks is likely why the species had few natural predators. However, great-horned owls have been documented killing adults and nestlings (Moore and Henny 1983, Rohner and Doyle 1992, Boal and Mannan 1994, Woodbridge and Detrich 1994). Loss of nestlings to predation may increase when other prey resources are low (Zachel 1985, Rohner and Doyle 1992). Nestlings have been killed by wolverines (Doyle 1995) and one-half of nestling mortalities in New Mexico were attributed to predation (Ward and Kennedy 1996). During the winter, eagles (Squires and Ruggiero 1995) and martens (Paragi and Wholecheese 1994) have been documented taking goshawks.

Potential Management Impacts:

The northern goshawk is considered a “Sensitive Species” in the Southwest by the U.S. Forest Service (Squires and Reynolds 1997). Sensitive Species designation requires biological evaluations to consider potential impacts of proposed management actions. Goshawks are also considered MIS because they are potentially sensitive to habitat change. However, as indicated above, the old-growth habitat they often represent is misleading because the species will use multiple habitat types as long as there is enough mature to old growth forest structure within the general forest.

In the Southwest, management over the past one hundred years has greatly altered forest structure and composition. Historical and current conditions differ considerably in that trees are much denser and in younger age classes. The resulting canopy closure reduces plant species abundance, understory composition is altered, and higher fuel loads currently exist. These habitat conditions can result in the loss of goshawk habitat through high-severity wildfire and epidemic infestation of insects and diseases (Reynolds et al. 1992). Because timber harvest has traditionally been the primary threat to northern goshawks and because the Forest uses the goshawk guidelines for timber management, this management impact will be the focus of this discussion.

Many forms of timber management have been identified as primary threats to nesting goshawk populations (Reynolds 1989, Crocker-Bedford 1990). Nests can be destroyed outright and drastic canopy closure reductions can effectively remove nesting habitat (Bright-Smith and Mannan 1994, Beier and Drennan 1997). These risks are addressed on the KNF by conducting pre-harvest goshawk surveys, avoiding known nest sites, and providing for multiple alternate nests. Research has documented that northern goshawks continue to occupy and breed successfully in the managed areas of the NKR. Virtually no part of the NKR contains forests in which tree harvest has not occurred (Burnett 1991, KNF 1993). Reynolds et al. (1994) and Reich et al. (2004) studied breeding goshawks on over 100 territories that produced over 600

young between 1991 and 2003 on the NKRD. This high density of goshawks and their reproduction strongly suggests that goshawks are not old growth obligates in the Southwest.

Past forest management has veered far from the historic range of variation. Reintroducing fire is key to creating sustainable forests in our fire dependent ecosystems and this can only be accomplished through active management. Reynolds et al. (1992) developed management recommendations for nesting goshawks in the southwestern United States. These recommendations describe desired forest conditions for nesting and foraging habitat while emphasizing conditions that support diverse prey populations. Recommendations prescribe habitat conditions at 3 spatial scales including nesting, postfledgling areas, and foraging areas. Overall, thinning under the goshawk guidelines results in a mosaic of vegetative structural stages interspersed across the landscape. To meet this end, the Kaibab Forest Plan prescribes the following leave percentages for each of 6 tree size-classes. First, because large trees are a critical nesting component, 40% of the trees across the landscape will be greater than 18 inches dbh (Vegetation Structural Stage [VSS] 5 and 6). These large trees will occur as small patches scattered throughout the landscape. The remainder of the landscape will include 10% openings occupied by grasses, forbs, and shrubs (VSS 1), 10% seedling-saplings (VSS 2), 20% young forest (VSS 3), and 20% mid-aged forest (VSS 4). Within each VSS class is a $\pm 3\%$ margin, i.e., post-harvest treatments can equal 17% VSS 5 and 23% VSS 6. This can be the difference between active management and focusing management efforts elsewhere. These structural stages are dynamic, growing from one stage to another and creating patches of mature trees that are available across the landscape through space and time. In addition, legacy trees are left on each acre of goshawk habitat that will not be harvested and instead are left to grow, die, and provide snags and logs.

Snag retention is another important habitat component for the northern goshawk because of the role they play in providing habitat for prey species. The Kaibab Forest Plan prescribes leaving snags in all three forested habitats including ponderosa pine, mixed conifer, and spruce-fir habitat types to support goshawk prey species (KNF Forest Plan 1996). The Forest Plan prescribes the goshawk guidelines to all forest and woodland habitat on the KNF, with the exception of Mexican spotted owl protected, restricted, and designated critical habitat, all of which have their own guidelines that take precedence (USFWS 1995).

Managed fires can create one to many acre patches of high-severity burns as tree canopies ignite. Fire can also reduce tree density, creating more open crowns. These openings can benefit many prey species (e.g., chipmunks, ground squirrels) while having mixed results for other species (tree squirrels). When managed fire is reintroduced into the ecosystem, snags and down logs will be reduced in the short term. They do provide immediate snag habitat and aid in replenishing downed woody debris. Snags are also created indirectly when trees weakened by fire eventually succumb to insects and disease. Weakened trees may last for years before becoming snags and many of these processes create longer lasting snags.

Given forest conditions and current fire behavior, goshawk habitat is not sustainable without active management. Past fires maintained forests with repeated, low-intensity ground fire. Snag and downed log resources under frequent fire return intervals would probably be considered limited by today's standards. Today fires commonly are assessed for opportunities to allow them

to continue to burn, but suppression is common to prevent stand replacing, high-severity crown fires. Prescribed fire, wildland fire use fires, and mechanical treatments are used to eventually attain sustainable forests. The goshawk food web approach, modeled using a 1,000-year time horizon, has strong parallels with the recommendations coming from forest restoration research through the Ecosystem Restoration Institute at Northern Arizona University (<https://library.eri.nau.edu:8443/>).

Population data:

BBS data (Sauer et al., 2005) for Arizona from 1966-2005 show a significant, positive population trend of 13 percent per year ($p = 0.03$; Fig. 2), but across the Southern Rockies, goshawks show a non-significant trend ($p = 0.690$). Both data analyses are from small data sets which exhibit several deficiencies, including low abundance (less than 1.0 bird/route), low sample size (less than 14 routes), imprecision (3%-year change would not be detected over the long term), and possible inconsistency in trend over time (sub-interval trends were significantly different [$P < 0.05$] from each other) (Sauer et al. 1999). Thus interpretation should be made with extreme caution and only used in light of surveys directly from the area of interest.

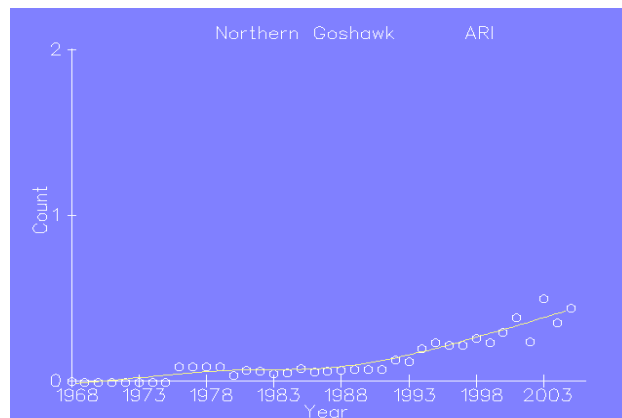


Figure 2. Goshawk population trend data for Arizona from 1966-2005, BBS data (Sauer et al., 2005).

Additionally, NatureServe globally ranks goshawks as G5, meaning their numbers greatly exceed 10,000 individuals and are demonstrably widespread, abundant, and secure. Nationally, they are classified as N4, or greater than 10,000 individuals with apparently secure populations. However, goshawks are considered vulnerable to extirpation or extinction in Arizona with a total estimated population between 3,000 and 10,000 birds.

Locally, the majority of the KNF goshawk population occurs within the North Kaibab Ranger District on the Kaibab Plateau, where surveys and studies have been conducted for close to 2 decades. The most intensive study has been conducted by the Rocky Mountain Research Station (RMRS), a sub-branch of the Forest Service. The Kaibab Plateau goshawk project started in 1991 and is expected to conclude at the end of the 2008 field season. In addition, all ponderosa pine and ponderosa pine/Gambel oak habitat on the Forest was surveyed by USFS personnel, following Forest Service Regional northern goshawks protocol. Although most of the NKRD appears to be at carrying capacity, goshawk reproduction on the Kaibab Plateau has been highly

variable over 15 years and overall showed a significant decline from 1991 to 2005, including the portions of the Plateau that have been managed by the National Park Service since the inception of the Grand Canyon National Park (Fig. 3, Reynolds 2005). Causes being investigated for the decline include change in forest composition and structure resulting from intensive forest management between the 1960s and early 1990s (large seed tree cuts) combined with catastrophic fire and wind throw, and natural environmental variation in prey abundance. Research to date indicates that as the amount of habitat changed by intensive management, fire, and wind-throw within goshawk territories increased, the frequency of reproduction decreased (Reynolds et al. 2006). Additionally, Salafsky et al. (2005) suggest that inter-annual fluctuations in precipitation and conifer seed production are correlated with, and may be responsible for, variation in prey abundance which in turn is strongly associated with goshawk reproduction. Together this suggests that goshawk demography is a complex interaction between vegetation composition and structure and natural variation in goshawk food resources, all of which may be confounded by ongoing drought conditions.

Swamping of habitat effects by large variations in food abundance, combined with the periodicity of wet versus dry weather in the Southwest, suggest that identifying the cause-effect responses of goshawks to forest management can only be accomplished through long-term research (Reynolds et al. 2005). In the 2005 summary report, Reynolds reported that precipitation and cone crops were high in 2005, suggesting 2006 would be productive for goshawks. Further, if the year produced a breeding rate greater than 50%, the goshawk population trend would change from a 15-year decline to stable over the last 16 years. Indeed, the 2006 breeding rate was 53%. However, 2007 is looking to be a poor year with breeding productivity in the 7% range, which will again result in overall negative trend. Breeding adult survival is 0.75 or declining. Reynolds et al. (2006) reports that goshawk reproduction over the 16-year study is not sufficient to replace adult mortality on the Kaibab Plateau. However, there seem to be enough juveniles to replace adults, suggesting the Plateau acts as source *and* a sink population. Given that the demographics appear influenced by precipitation patterns, it is difficult to judge if the population is stable or declining (Reynolds personal communication). It also appears that the goshawk pairs within the Grand Canyon National Park section of the Kaibab Plateau, which are also monitored as part of Reynolds ongoing research, have lower reproductive rates than those on the National Forest (Reynolds personal communication). There has been essentially no timber harvest or other forest structure management within this portion of the National Park.

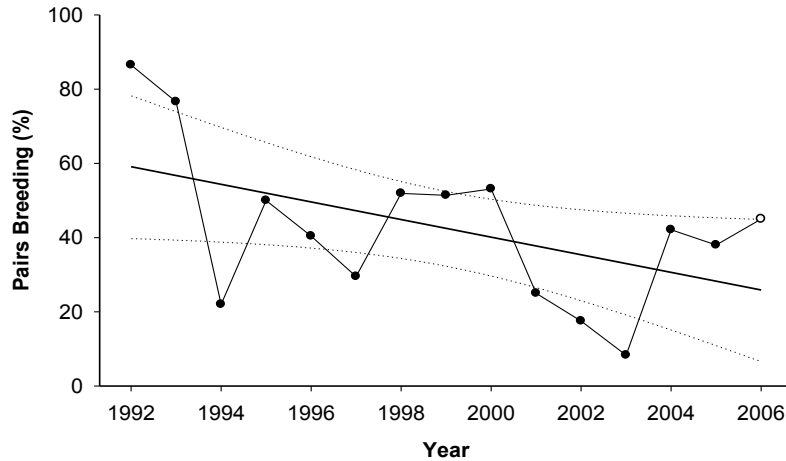


Figure 3. The annual percent of territorial goshawks breeding (laid eggs) on the Kaibab Plateau, between 1992 and 2005 (●) and the *minimum* percent of goshawks breeding in 2006 (○) needed to cause the current significant decline in reproduction ($P = 0.030$) to become non-significant ($P = 0.053$). From Reynolds (2005).

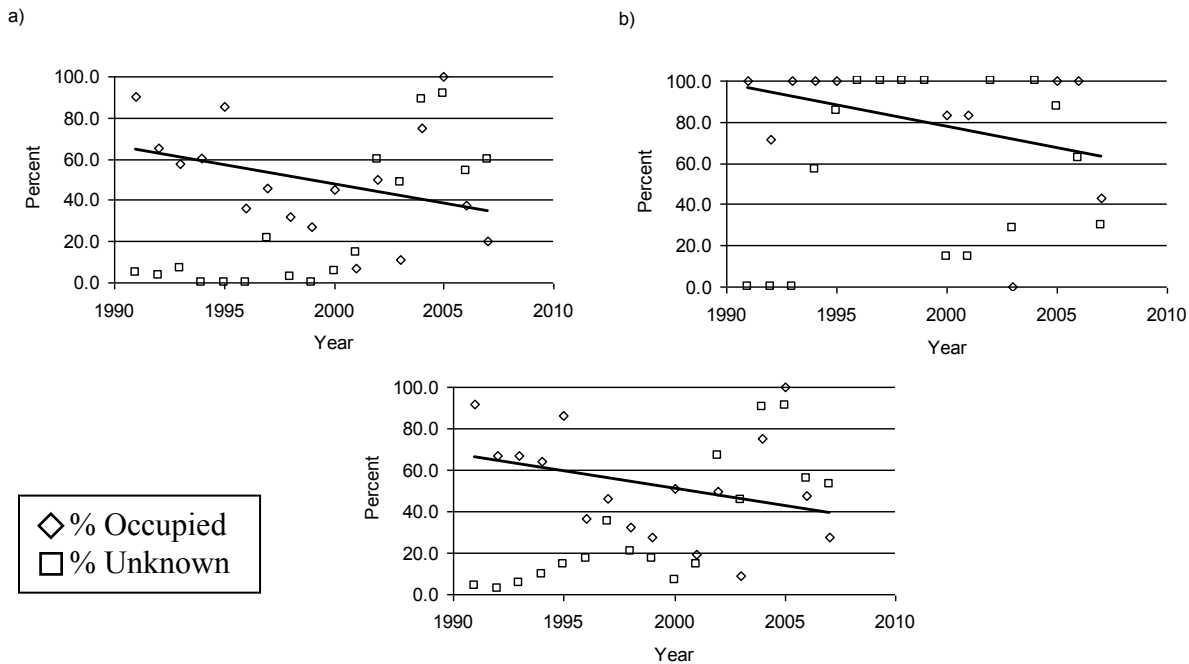


Figure 4. Percent of surveyed and occupied territories present on the Kaibab National Forest, Coconino County, AZ. a) William's Ranger District, b) Tucsyan Ranger District, and c) South Zone (William's and Tucsyan Ranger Districts combined).

Data for the rest of the Forest show a similar decline in occupied territories (Fig. 4). While a decline in territories does not translate directly into reproductive effort, it does indicate that the number of adults that could be breeding on the forest is decreasing and that this decrease would result in less offspring recruited into the population. Data should be interpreted cautiously as the number of nests with unknown occupancy does vary by year.

Trend Estimate:

Considering the information above, northern goshawks are assumed to be declining on the Kaibab National Forest. However, if future weather patterns produce good precipitation, the population could stabilize. Only precipitation can fuel forest productivity in terms of abundant seed crops which result in prey population increases that occur at greater frequencies. Continued reduction of forest stem density and basal area should ameliorate the stochastic nature of weather by reducing the threat of large-scale, high-severity crown fire, thereby helping stabilize the population. Continued monitoring of the population and its response to forest treatments need to continue to be measured over the long term.

Hairy Woodpecker

Hairy woodpeckers can be found in higher densities following wildfire than in unburned areas. Researchers believe it is because of the smorgasbord of insect prey that is attracted to the burned trees. Hairy woodpeckers find these insects by feeling the vibrations made by insects moving about in the wood. They can also hear the insects chewing on wood inside the trees.

Habitat Characteristics:

The ubiquitous hairy woodpecker (*Picoides villosus*) is one of the most abundant primary cavity nesters in northern Arizona. This species was selected to represent the snag component of the ponderosa pine, mixed conifer, and mixed conifer with aspen habitats within the KNF. They reside in forests from near the northern limit of boreal forest in Canada (Parker 1987) and central Alaska, south to Panama and northern Baja California, and east to the northern Bahamas Islands (American Ornithological Union 1998). The hairy woodpecker is widely distributed wherever there are mature forests with substantial snags, but they also occur rarely in small woodlots, wooded parks, cemeteries, shaded residential areas, and other urban areas with mature shade trees (Jackson et al. 2002). Hairy woodpeckers occur in both deciduous and coniferous forests, but may show preference relative to their availability and location, such as a preference for open pine forest in the southwest (Winkler 1979). Although its abundance is greater in Arizona's pine forests, hairy woodpeckers are also found in piñon-juniper woodland (*Pinus edulis* and *Juniperus* sp. respectively) of the north and some Upper Sonoran deciduous woodlands and riparian areas in the south (Monson and Phillips 1981). Hairy woodpeckers are strongly associated with burned areas (Covert-Bratland et al. 2006), an important historical component of northern Arizona's forests resulting from a frequent fire interval (Covington et al. 1997).

These habitats are typically used year-round as the hairy woodpecker is primarily non-migratory (Jackson et al. 2002). As a permanent resident, it both breeds and winters in the same area although populations at the northernmost extent of its range display irregular and unpredictable wandering in winter and in other areas short-distance movements take place after nesting (Ouellet 1977). In Arizona, color-banded resident birds have been seen in both breeding and winter seasons indicate they are likely year round residents (Bratland personal observation).

Resident birds maintain home-ranges both during the breeding and winter seasons, however, information on home range size in relation to habitat characteristics is limited. In Illinois' mature bottomland forest, Calef (1953) found territories ranged from 0.65 to 1.5 ha (1.6–3.7 acres) and in Michigan, Staebler (1949) documented winter territories to be approximately 324 ha (800 acres). In northern Arizona, Covert-Bratland et al. (2006) found that winter home range size increased with years post wildfire (7.85 ± 4.8 ha [19.4 ± 11.9 ac] 2 years post burn, 13.5 ± 16.4 ha [33.4 ± 29.4 ac] 3 years post burn, 65.3 ± 9.7 ha [161.4 ± 24.0 ac] 6 years post burn, and 146.4 ± 83.2 ha [367.8 ± 205.6 ac] 7 years post burn) and each contained portions of high- and moderate-severity burn. The authors suggest that decreasing prey resources accounts for the increase in home range size.

As primary cavity nesters, hairy woodpeckers are dependent on dead or dying portions of live trees and snags for nesting. Hairy woodpeckers excavate their nests in both live and dead conifers (Raphael and White 1984, Roberson 1993) and deciduous trees such as aspen with fungal heart rot (Conner and Adkisson 1976, Jackson 1976, Li and Martin 1991, McPeck 1994). Conifer species used include Jeffrey pine (*Pinus jeffreyi*), lodgepole pine, white fir, and red fir, and ponderosa pine (Raphael and White 1984, Covert 2003). In Idaho, hairy woodpeckers were found nesting in areas of burned ponderosa pine forest with mean snag densities of 258 ± 12 snags/ha (Saab et al. 2007). Preferred nest tree size varies but 35 cm (13.8 in) is typical in western conifer forests (Conner et al. 1975, Horton and Mannan 1988, Saab et al. 2007). Hairy woodpeckers prefer to drill their cavities on the underside of a curved limb in a somewhat open location (Jackson et al. 2002).

Hairy woodpecker foraging behavior as well as sexual differences in foraging sites and techniques have been studied extensively (Kilham 1961, 1965, 1970, 1973, 1983, Selander 1965, 1966, Lawrence 1967, Ligon 1968, Jackson 1970a, 1971b, 1979b, Short 1970, 1971, Kisiel 1972, Wallace 1974, Grubb 1975, 1977, 1978, Conner 1977a, 1993, Bull et al. 1986, Villard and Beninger 1993, Ouellet 1997, Weikel and Hayes 1999, Covert-Bratland et al. 2006). Hairy woodpeckers primarily eat insects (DeGraff et al. 1991) from the surface and subsurface of trees and consume a diversity of fruits and seeds (Jackson et al. 2002). In the western United States, hairy woodpeckers prefer to forage on conifers. Stallcup (1968) documented extensive use of conifers in Colorado. Similarly, Roberson (1993) found the same trend in California, but Kisiel (1972) and Conner (1980) found this use varies regionally relative to conifer availability. Within coniferous areas, hairy woodpeckers inhabiting riparian areas will forage on willows, aspens, and cottonwoods (*Populus spp.*, Jackson et al. 2002). In northern Arizona, hairy woodpeckers forage on ponderosa pine and are found in greater densities in burned areas (Covert-Bratland et al. 2006). Overall, research clearly reflects that hairy woodpeckers select trees based on availability by species, those of greater stature (e.g., Weikel and Hayes 1999), and species hosting high concentrations of prey (e.g., Kilham 1961, 1973) such as elms (*Ulmus*) during moth outbreaks (Neff 1928, Kilham 1973) and trees that contain bark beetles and wood borers (Hutchison 1951, Blackford 1955, Yeager 1955, Baldwin 1960, Bailey and Niedrach 1965, Otvos 1967, Koplin 1969, Crockett and Hansley 1978, Kroll and Fleet 1979, Murphy and Lenhousen 1998, Covert-Bratland et al. 2006).

Dead trees typically host more wood-boring arthropods, a staple in the hairy woodpecker diet (Otvos and Stark 1985). Several studies have shown that hairy woodpeckers select dead and

dying trees for foraging more so than live trees (Raphael and White 1984, Morrison et al. 1987). Hairy woodpeckers also select trees larger than the available average (Ouellet 1997, Covert-Bratland et al. 2006) and forage in areas with less tree density than average when forests are extremely dense (Covert-Bratland et al. 2006). During the winter and breeding season, hairy woodpeckers prefer burned trees because they harbor more arthropod prey (Murphy and Lenhausen 1998, Covert-Bratland et al. 2006). Males and females also segregate into different areas of selected trees (Kilham 1965, Conner 1977a, 1993, Morrison and With 1987, Ouellet 1997) but in western forests do not differ in how they obtain insects (Ouellet 1997, Covert 2003). When insects are plentiful, males and females exhibit less segregation suggesting that competition may be reduced by food availability (Hutchison 1951, Covert 2003). Across North America, the year-round diet of hairy woodpeckers consists of >75% insects, including ants, bees, wasps (Hymenoptera), and caterpillars (Lepidoptera), wood-boring larvae (Cerambycidae and Buprestidae), grasshoppers, crickets, and cockroaches (Orthoptera), and the remainder consisting of fruit and seeds (Beal 1911, McAtee 1911, Neff 1928, Bent 1939, Otvos and Stark 1985).

Hairy woodpeckers are generally solitary but occasionally occur in loose pairs or family groups (Jackson et al. 2002). During winter, hairy woodpeckers will join mixed species flocks for foraging, but remain on the periphery of groups. Hairy woodpeckers have agnostic interactions for nest sites with downy (*Picoides pubescens*, Kilham 1962) and red-bellied (*Melanerpes carolinus*, Stickel 1963) woodpeckers but have been found nesting in the same tree with red-naped sapsuckers (*S. nuchalis*, McClelland and McClelland 2000).

Hairy woodpeckers are an important prey resource to many raptors including the northern goshawk (Squires 2000), Cooper's hawk (*A. cooperi*, Hammerstrom and Hammerstrom 1951, Meng 1959), sharp-shinned hawk (*A. striatus*, Setterington 1997), great horned owl (*Bubo virginianus*, Errington et al. 1940), and barred owl (*Strix varia*, Hammerstrom and Hammerstrom 1951). Nestling are predated on by eastern screech-owl (*Otus asio*, Brown and Bellrose 1943), house sparrow (*Passer domesticus*, Bent 1939), European starling (*Sturnus vulgaris*, Jackson et al. 2002), and red-bellied woodpeckers (Grimes 1947).

Potential Management Impacts:

Hairy woodpeckers can inhabit a broad range of habitats. On the KNF, hairy woodpeckers likely inhabit ponderosa pine, dry mixed conifer, mixed conifer with aspen, and spruce fir habitat. Hairy woodpeckers prefer ponderosa pine and mixed conifer in Northern Arizona. There are approximately 681,158 acres of this habitat potentially available on the KNF. It would seem that the gross habitat requirements of the hairy woodpecker are readily available. However, degradation of habitat quality since European settlement has likely compromised its value as suitable habitat. There appears to be as many or more large trees (>18 in. dbh) and snags today as there were historically, but their arrangement on the landscape and the surrounding forests has been dramatically altered. "Green snags, or living trees with lightning strikes, dead tops, or cavities formed where branches used to attach, provide a wide array of nesting substrate across the forest. This aspect of habitat structure, although unquantified, has been greatly impacted by past management. For decades, these trees were selected for removal due to the presence of what we now consider habitat rather than abnormalities.

Fire suppression that has shifted forest structure from an open canopy, comprised of few large trees to a closed canopy comprised of many small trees has greatly decreasing the foraging value of these areas (Bock and Block 2005, Ghalambor and Dobbs 2006). Because of the intimate tie between hairy woodpeckers and burned areas, fire suppression and large scale salvage logging in burned areas that does not account for hairy woodpecker foraging behavior are the management practices that most affect this species in the Southwest today (Hejl 1994, Kotliar et al. 2002, Covert-Bratland 2003).

Attempts to restore ponderosa pine closer to its historical range of variation should positively affect the hairy woodpecker as long as such practices are careful to retain or encourage the creation of habitat conditions and components important to this species. Using the goshawk guidelines to direct management practices should positively affect the hairy woodpecker, as this prescription results in forest structure that more closely resembles historic forests than those present today, including large trees and an abundance of snags. Allowing wildfire to return to the system is also essential to maintain this species because burned areas not only provide ample food (Hutchison 1951, Blackford 1955, Baldwin 1960, Koplin 1969, Crockett and Hansley 1978, Kroll and Fleet 1979, Murphy and Lenhausen 1998, Covert-Bratland et al. 2006), but also provide snags used for foraging, winter roosting, and likely nesting (Covert-Bratland et al. 2007). Lastly, avoiding the green tree/burn area interface when salvage logging high-severity burn areas will retain important breeding and foraging habitat used by hairy woodpeckers.

Timber management and harvest practices can negatively and positively affect hairy woodpecker habitat. Direct negative impacts include removal of habitat when even-aged management is practiced. These management practices typically remove current snags and the live trees that represent future large snags. In general, even-aged stand management was replaced with uneven-aged management when the goshawk guidelines were adopted for National Forests in Arizona and New Mexico in 1996.

Returning fire to the ponderosa pine system will also produce direct and indirect positive affects for hairy woodpeckers. In northern Arizona, ponderosa pine forests historically experienced low-severity fires every 2–12 years and fires ranged from hundreds to thousands of acres in size (Covington et al. 1997, Fule' et al. 2003). Within these fires, small patches of snags were generated in high frequency across the landscape (Fule' et al. 2004). These small patches of high severity burns were beneficial to hairy woodpeckers because of their rich prey resources (Covert-Bratland et al. 2006). High-severity patches created from today's stand-replacing fires can reach several hundred hectares in size. For example, the Bridger Fire on the NKRD (1996) burned about 22,300 ha (53,500 ac), including over 1,300 ha (3,200 ac) of stand replacement fire in the ponderosa pine habitat. In 2006, the Warm Fire burned about 16,300 ha (39,100 ac) with 4,237 ha (10,170 ac) in ponderosa pine. Fulé and Covington (1997) estimated historical patch size at about 2 ha (5 ac) of stand replacement in ponderosa pine. Hairy woodpeckers do not use interior portions of larger burned areas, making these large patches less valuable than smaller more historical patches. This is due to the species behavior to readily use edges which likely balance the benefits of increased prey resources with increased predator exposure (Covert-Bratland et al. 2006).

Snags within high-severity burned areas are also used for winter roosts. They become available sooner than trees killed by other sources but are ephemeral in nature as they are susceptible to windthrow (Covert-Bratland et al. 2007). Moderate severity fires benefit the hairy woodpecker because moderately burned trees are susceptible to secondary mortality agents such as arthropod attack and drought. Snags created by mortality agents like these last longer, providing the structure and substrate used by woodpeckers for longer periods of time than severely burned trees that are lost by toppling in high winds (Salaman 1934, Flanagan 1996, McHugh and Kolb 2003). Prescribed burns, wildland fire use fires, and wildfires that mimic historical regimes and create a mixed severity burn across the landscape will benefit the hairy woodpecker more than low-severity burns alone. Low-intensity ground fires produce little scorch or injury and hence low susceptibility to secondary mortality agents (Furniss 1965, Ryan and Reinhardt 1988, Flanagan 1996, Santoro et al. 2001, McHugh and Kolb 2003). Thus, balancing the positive affects of wildfire with the difficulty of controlling high-severity burns is important to effectively manage current forested ecosystems (Covert-Bratland et al. 2007). This species requires more than simple forest structure, but also the forest function that results from having disturbance forces at work in the forest.

Decay within trees is what allows wood to become soft enough for excavation by primary cavity nesters, including the hairy woodpecker. Leaving snags without surrounding habitat exposes them to sun and wind, decreasing the moisture content needed for decay (Jackson and Jackson 2004). These exposed snags are also more vulnerable to windthrow than their counterparts within a forest (Morrison and Raphael 1993, Chambers and Mast 2005). If timber treatments retain snags and variable tree size and structure across the landscape, habitat quality is augmented by providing for nesting, roosting, and foraging (Ghalambor and Dobbs 2006). Maintaining large diameter trees ensures snag recruitment when they senesce either due to primary mortality agents, such as fire and drought, or secondary mortality agents, such as disease and insect attack. Encouraging the maintenance of large trees would likely increase the number of trees with dead tops as lightning strikes tend to hit larger trees. These trees are then used by multiple cavity nesters once the wood has rotted to a suitable level for excavation or natural cavities appear after the checking.

Following fire, salvage logging is one way many forests attempt to retain some economic value from lost timber. However, the manner in which salvage logging is conducted determines how species are affected. This may be more important for resident species that use treated habitat both in the winter and breeding season than for Neotropical migrants (Imbeau et al. 2001, Morissette et al. 2002). Saab et al. (2007) found that densities of hairy woodpeckers were 2.5 times lower in a partially logged burn in Idaho's ponderosa pine forest. Additionally, nest survival during the early post-fire period was significantly reduced in the partially logged burn relative to the unlogged burn. Similar responses were found in other wildfire locations (Raphael et al. 1987, Imbeau et al. 1999, Smucker et al. 2005, Hutto and Gallo 2006), and other logged and unlogged burns (Haggard and Gaines 2001, Johnson and Anderson 2002, Morissette et al. 2002). Covert-Bratland et al. (2006, 2007) showed that high-severity burned areas are beneficial foraging areas, support 18 times the abundance of hairy woodpeckers than unburned areas, and that burned trees provide readily available roost trees during difficult winter conditions. When salvage logging is done, it is important that burned trees are removed in large patches (greater

than 150 ha, 370 ac, Saab et al. 2007) of high-severity burn and that high densities of snags are left along the edges of high-severity burn patches for nesting and roosting.

Population Data:

BBS data produced a non-significant trend (2.7, $p = 0.440$, Fig. 5) for hairy woodpeckers in Arizona for the 1966-2005 survey period (Sauer et al. 2005). Out of 30 analyses, 26 resulted in a positive trend for the species but none are significant. Three guild analyses (cavity nester, woodland breeder, and permanent resident) showed negative trends in the Sierra Madre Occidental BCR where sample size was small. The data have significant deficiencies including low abundance, poor sample, and poor estimate precision. In total, these analyses likely indicate

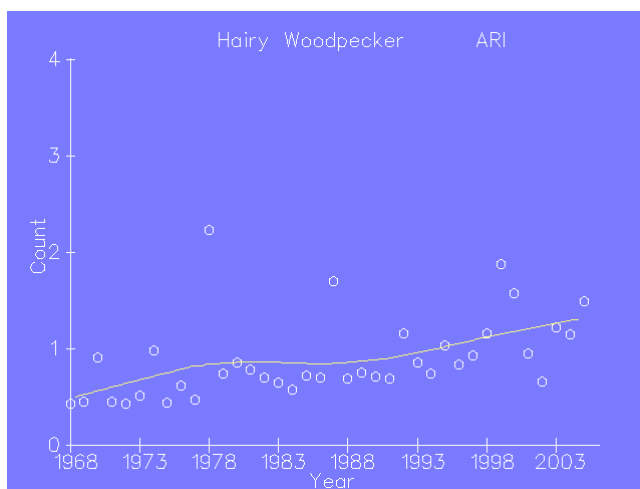


Figure 5. Hairy woodpecker population trend data for Arizona from 1966-2005, BBS data (Sauer et al., 2005).

hairy woodpeckers are at least stable in Arizona and the surrounding area. Because the trend is non-significant, further surveys need to be conducted to determine if the positive trend is significant, i.e., whether there is a stable or increasing population.

Hairy woodpeckers are listed by NatureServe as demonstrably widespread, abundant, and secure globally, nationally, and statewide.

Locally, several research projects conducted on both the Kaibab and Coconino National Forests determined hairy woodpecker abundance and density. Pope (2006) found winter hairy woodpecker density in unburned ponderosa pine forest to be 2.1 ± 0.3 S.E. per 100 ha (247 ac). Over the course of 6 years in 3 unburned ponderosa pine sites, Covert (2003) determined that hairy woodpecker density did not differ by site or year indicating a stable population. Between 1999 and 2001, Salafsky (2002) found similar densities (3.16 per 100 ha) in unburned ponderosa pine forest on the Kaibab Plateau within the KNF. This study also estimated density for mixed conifer habitat over 3 years and found comparable densities on average (3.58 per 100 ha), with an increasing trend across the years (2.6 in 1999, 3.2 in 2000, 4.9 in 2001). These relatively low numbers of individuals are markedly different from those found in burned ponderosa pine. Pope (2006) found a five-fold increase in density (10.8 ± 2.0 per 100 ha) in prescribed low-intensity

burned areas. Covert-Bratland et al. (2006) found a 14- and 18-fold increase in the 2 years following moderate to high-severity burns.

The KNF landbird surveys also found similar results for the hairy woodpecker. Densities in unburned ponderosa pine stands during 2005 surveys were 5.1 per 100 ha (or 13 per 100 ac) and 2.4 in 2006. Because methods for sampling and analysis used by the KNF surveys were similar to those from recent studies, comparisons should be valid. Considering all the surveys from the literature and the KNF, the data suggest that hairy woodpeckers have remained stable in ponderosa pine and may be increasing in the mixed conifer. It is likely that populations on the KNF exhibit similar population trends as the adjacent Coconino National Forest. Therefore, the stability determined by Covert-Bratland et al. (2006) would likely be found on the KNF. Continuation of the KNF landbird surveys is critical to ensure accurate trend assessments. It is hoped that future surveys with increased survey effort will result in more detections for estimating trend estimates. Forest wide trend assessments were estimated from the ponderosa pine population.

Trend Estimate:

Considering the information above, hairy woodpeckers are assumed to be stable on the Kaibab National Forest. Considering the KNF snag policy, leave tree guidelines for habitat manipulations, and the increasing severity of forest fires and number of acres burned in the Southwest, it is likely that hairy woodpecker populations will increase as all 3 habitat perturbations result in increases in hairy woodpecker density. Hairy woodpeckers will benefit from thinning practices that try to replicate historical forest structure and function. Allowing fire to burn within forested areas that leaves a mosaic of burn severities across the landscape ensures areas where populations will be greater than unburned areas. This helps maintain populations in unburned forests as densities in burned areas likely maintain those in unburned areas.

Lincoln's Sparrow

Lincoln's sparrow is one of the hardest sparrows to find owing to its skulking secretive behavior. Roger Tory Peterson said Lincoln's Sparrows are afraid of their own shadows, quickly taking flight at the sight of an intruder. John James Audubon even had trouble finding this bird. It was Thomas Lincoln, a young sharp shooter on a collecting trip to Labrador in 1833 that shot the first specimen, giving Audubon his first up close and personal look. Out of gratitude, Audubon named the sparrow after his helpful companion.

Habitat Characteristics:

Lincoln's sparrows were selected to represent species using the late-seral, high-elevation, riparian habitat within the Forest. Lincoln's sparrow breeds primarily across Canada and montane areas in the western U.S. and winters in the southwestern U.S. and into Mexico. In northern Arizona, the breeding range of this species is limited to the White and San Francisco Mountains (Monson and Phillips 1981). This species prefers thickets within montane forests, wet meadows, riparian habitats, and shrubby forest edges. At lower elevations, Lincoln's sparrow selects mesic willow shrubs, but can be found in mixed deciduous wood groves such as aspen and cottonwoods (*Populus spp.*), mixed shrub-willows, black spruce or tamarack bogs, as well as a variety of other riparian habitat types (Salt 1957, Erskine 1977, Ewert 1982, Douglas et al. 1992, Dobkin 1994). Lincoln's sparrows have been reported using forest clearcuts, but

usually clearcuts were situated near bogs (Erskine 1977). Contrastingly, Lincoln's sparrows are reported to avoid openings without shrub cover and densely forested riparian areas (Ammon 1995a).

Within these general habitat types and during the breeding season, territory boundaries are defined by males singing conspicuously from tree tops and shrubs (Ammon 1995a). Size of territory varies widely by location due to population density. Ammon (1995b) reported a home range diameter of 35 m (115 ft) within high-density populations of the Colorado Front Range. In lower-density populations in Ontario Speirs and Speirs (1968) reported an estimated diameter of >100 m (328 ft). These territories were defined for nesting only, as foraging often took place outside of estimated boundaries.

Lincoln's sparrow typically uses low willow for nesting as well as foraging, using the ground and base of trees to glean (Cody 1974). Lincoln's sparrow does use tall trees and exposed willow branches for singing and males will also use forest edges for singing and sentinel purposes (Ammon 1995a). Due to its secretive nature, which sex selects the nest site and builds the nest is unknown. Nests themselves are on the ground but the site is elevated more than random areas within a territory (Peck and James 1987) and has higher density ground cover and low-shrub (≤ 60 cm in height, 23.6 in) cover (Ammon 1995b). Often birds place their nests inside a willow shrub or mountain birch with dense sedge cover. Nest sites are typically wetter than those for sympatric species including white-crowned sparrow (*Zonotrichia leucophrys*) and dark-eyed junco (*Junco hyemalis*; Hadley 1970).

Foraging by the Lincoln's sparrow is usually restricted to gleaning from ground and from low vegetation (Cody 1974), using its feet to uncover litter-inhabiting invertebrates and seeds (Ammon 1995a). However, this species will also make limited use of tree foliage and branch tips (Ammon 1995a). During the breeding season, Lincoln's sparrow eats mostly arthropods, including insect larvae and adults of beetles, two-winged flies, leafhoppers (Homoptera), moths and butterflies (Lepidoptera), and May flies, as well as spiders (Araneae; Raley and Anderson 1990). When available, Lincoln's sparrow also eats small seeds. During moth outbreaks, Lincoln's sparrows will shift their diet to be nearly restricted to moths often flycatching to obtain prey. Lincoln's sparrows will use favored perched under protective cover to remove the wings from the moths creating piles of wings (Ammon 1995a). In the winter the species switches mainly to seeds, but maintains insects in its diet if they are available.

Giving up its secretive solitary nature, the Lincoln's sparrow is somewhat social during the winter, joining small loose flocks for foraging (Speirs and Speirs 1968, Amos 1991). It can be found fraternizing with other sparrows including white-crowned, song (*Melospiza melodia*), and swamp sparrow (*M. georgiana*) during migration, but does not form post-breeding aggregations like other species of sparrows (Ammon 1995b).

Across its range, Lincoln's sparrow falls prey to a variety of predators. As adults and fledglings, most are taken by sharp-shinned hawks, shrikes (*Lanius spp.*), domestic cats (*Felis domestica*), and short-tailed weasels (*Mustela erminea*) (Braund and Aldrich 1941, Ammon 1995a). Sharp-shinned hawks mostly take birds while perching or flying while shrikes and cats take those exhausted from migration that can not escape chase. Weasels capture fledglings and adults on

the ground (Ammon 1995a). Similarly, nest are preyed upon by many predators including short-tailed weasels, least chipmunks (*Tamias minimus*), shrews (*Sorex* spp.), and occasionally gray jays (*Perisoreus canadensis*) (Ammon 1995a). Predation rates seem to be positively correlated to predator abundances (Ammon 1995a) and negatively with concealment of nests and willow patch size (Ammon 1995b).

Potential Management Impacts:

Because this species breeds only in boreal regions, specializes on dense shrub cover, and is secretive in nature, much of its biology remains poorly documented (Ammon 1995a). One sorely lacking area is potential management impacts because there are very few empirical studies that have tested the effects of management practices. However, degradation of breeding habitat from any management practice, such as grazing and recreation, likely negatively impacts this species. In Arizona, the species is considered vulnerable to extirpation or extinction statewide for the breeding population. Thus understanding even what little we know could be important for managing this species. Although for the Kaibab National Forest, this is likely just an exercise because there has been only one breeding record (Troy Corman, AZ Game & Fish, personal communication) and one sighting (Elaine Morall, Northern Arizona Audubon Society, personal communication) of this species on the entire forest.

Because terrestrial species, particularly birds, are responsive to changes in the vertical diversity of vegetation structure (MacArthur 1964), alteration of vegetation by overgrazing decreases the quality of nesting and foraging habitat. The mechanism of decreased nesting and foraging habitat quality is by loss of vegetation volume (Knopf and Cannon 1982, Schulz and Leininger 1991, Ammon and Stacey 1997). Riparian habitat essential to the Lincoln's sparrow can be especially vulnerable. Schulz and Leininger (1990) found that canopy cover of willows was 88% greater at ungrazed sites, even though willow density was not significantly different. Grazing also changes hydrologic regimes when vegetation along a stream is trampled. Eventually stream depth is reduced to a level that does not support riparian vegetation, totally removing Lincoln's sparrow habitat.

Lincoln's sparrows are negatively affected by human disturbance at nest sites. In the Colorado Front Range, Ammon (1995b) found significantly greater rates of nest desertion at sites used by recreational visitors for picnicking, fishing, and hiking compared to those not used. Roads have also been documented to decrease species richness and especially decrease sensitive species with specific habitat requirements such as the Lincoln's sparrow (Miller et al. 1998, Ingelfinger 2001). As the number of riparian areas decrease in the west due to development and increased human use so to will Lincoln's sparrow habitat. Any management practice that protects or enhances boggy dense wetlands will likely benefit the Lincoln's sparrow.

Population Data:

BBS data (Sauer et al. 2005) show positive but non-significant trends for all 10 analyses (Appendix 2). The data for the regional analyses have deficiencies including low abundance, poor sample, and poor estimate precision. Analyses for groups were only available for the Southern Rockies/Colorado Plateau BCR and show similar non-significant trends. It seems reasonable that because all the trends are positive that more data would support the trend with a

significant result similar to that found for the Western BBS region from 1966 to 2005 (trend = 2.7, $p = 0.00$, Fig. 6).

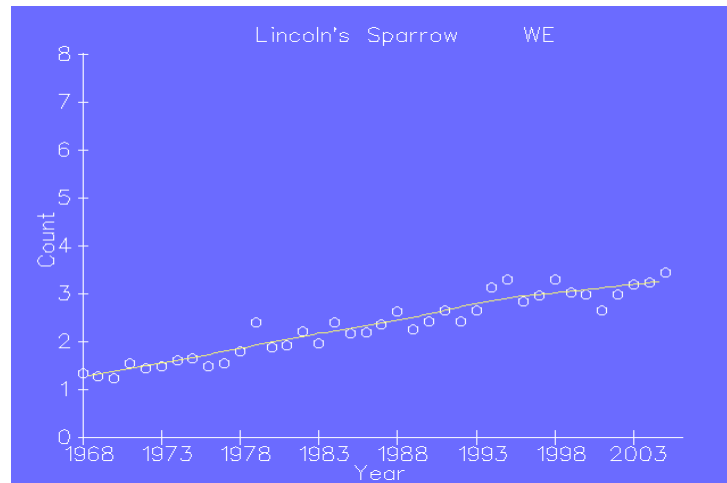


Figure 6. Lincoln's sparrow population trend data for the Western BBS region from 1966-2005, BBS data (Sauer et al., 2005).

Additional caution should be given to data because within Arizona, Lincoln's sparrow is considered vulnerable to extirpation or extinction statewide for the breeding population by NatureServe. On a larger scale, NatureServe's population assessment is similar to that shown by the BBS with Lincoln's sparrow listed as demonstrably widespread, abundant, and secure, globally, nationally, and statewide for the non-breeding population.

Breeding densities of Lincoln's sparrows varies greatly. Erskine (1977) reported densities less than 0.002 individuals/ha within the Maritime Provinces, while Ammon (1995b) reported 12-18 individuals/ha in the Colorado Front Range, a 7500 fold difference. The variation in species densities is unknown, but is likely due to its secretive nature and because it breeds in dense shrub cover, making it difficult to study at the population level. Locally, neither the KNF Landbird surveys nor the Arizona Breeding Bird Atlas has documented the bird's presence. There has been one documented nesting attempt at a high altitude lake (Troy Corman, AZ Game & Fish, personal communication) and one sighting at Sholz Lake in May of 2005, although the bird was suspected to be a migrant (Elaine Morall, Northern Arizona Audubon Society, personal communication).

Most important to assessing the population trend of this species is the validity of using it as a MIS. Because the boggy, dense willow riparian habitat that the Lincoln's sparrow typically nests in does not exist on the KNF and because the marginal habitat that does exist is limited to only a few tanks, it is likely that Lincoln's sparrows are rare or accidental on the Kaibab National Forest. Thus, the KNF does not support a significant population of Lincoln's sparrow. If the population is not significant, it can not be monitored, violating selection criterion for MIS. Because this species biology is relatively unknown, Lincoln's sparrows do not indicate trends of other species within riparian habitat or the health of riparian habitat, again violating MIS selection criterion. Because we know little about the affects of management activities on Lincoln's sparrows, we can not generalize what a change in its population indicates. All of these

conditions further violate selection criterion for MIS. Because ‘best science’ requires the use of some empirical evidence or at least some survey data, we can not expect this species to perform as an MIS for the KNF.

Population Trend:

Considering the information above, if a population of Lincoln’s sparrows did occur on the KNF, it would likely be stable. This species has been documented as colonizing new areas (Ammon 1995a). However, habitat is so limited on the KNF that a potential population expansion would be limited. Due to the limited habitat and an absence of a resident population, Lincoln’s sparrow is not a viable MIS for the KNF.

Lucy’s Warbler

Lucy’s warbler (*Vermivora luciae*) is only one of two warblers in the U.S. that nest in cavities. When cavities are abundant, this non-colonial species will breed in some of the densest concentrations with up to 11 pairs spaced as closely as 30 meters apart.

Life History:

Lucy’s warbler was selected to represent species using late-seral, low-elevation, riparian habitat and specifically the snag portion of this habitat type because this species is a secondary cavity nester. Lucy’s warbler usually nests below 1,000 m (3281 ft), ranging from 60 and occasionally up to 1,775 m (200-5825 ft, Johnson et al. 1997). In Arizona, this species is a common resident of low elevation mesquite (*Prosopis sp.*) bosques, cottonwood-willow forests, and densely vegetated xeric-riparian washes in southern and central Arizona (Swarth 1914, Phillips et al. 1964, Terres 1991, Johnson et al. 1997). They are also found in mid-elevation ash-walnut-sycamore-live oak associations (Phillips et al. 1964). Although classified as a generalist, the preferred habitat for Lucy’s Warbler is dense mesquite (Griscom and Sprunt 1957, Bent 1953, Rosenberg et al. 1991, Terres 1991, Curson et al. 1994, Johnson et al. 1997). Llyod et al. (1998) found that Lucy’s warbler is positively correlated with mesquite density. This significant relationship likely explains why Lucy’s Warblers tend to shun mature cottonwood-willow riparian associations (Dunn and Garrett 1997). Lucy’s Warbler has also recently begun breeding in tamarisk (*Tamarix ramossisima*) forests in the Grand Canyon region of Arizona (Johnson et al. 1997).

As a Neotropical migrant, Lucy’s warblers arrive on their breeding grounds earlier than most migrant species (Johnson et al. 1997) and in southwestern Arizona they arrive as early as the beginning of March (Phillips et al. 1964). Their arrival coincides with the leafing out of mesquites (Otahal 2006). Lucy’s warblers do not stay long with numbers greatly diminished by late June and are almost entirely gone by late July or early August (Otahal 2006). Lucy’s warbler winters in the tropical lowlands of the Pacific slope of Mexico (Gómez de Silva Garza 1996). Because Lucy’s warblers are limited to a single location for its wintering range making it vulnerable to extirpation, the bird’s conservation necessity increases. Physiognomy of Lucy’s warbler winter habitat is low scrub and weedy fields in coastal foothills and lower montane slopes of central western Mexico (Curson et al. 1994).

Lucy’s Warbler can nest in very dense colonies (Otahal 2006) and studies have shown spacing to be as close as 30 m (100 ft) in closed canopy bosques (Johnson et al. 1981) to 200 m (650 ft) in a desert wash near Tucson, Arizona (Johnson et al. 1997). On the Colorado River near Blythe

California, Grinnell (1914) found Lucy's warbler only within mesquite bosque and nests were uniformly spaced within 180 meters strips of habitat. The major factor contributing to the difference in nest density seems to be habitat with native mesquite habitat holding greater densities of birds than other habitat types. Brown (1989) found 228.8 pairs / 40 ha (100 ac) in mesquite dominated old-high water zone, while nesting densities in tamarisk was 200 pairs / 40 ha. Lucy's warbler densities were estimated by Stoleson et al. (2000) in mature riparian forest along the Gila River in New Mexico, which ranged from 1.7 to 3.3 pairs/ha (mean 2.3 ± 0.7 pairs/ha, 2.47 ac = 1 ha). Overall, regardless of habitat type, Lucy's warbler usually nests in association with riparian areas.

As a secondary cavity nester, Lucy's warblers build nests in natural cavities or those made by primary cavity nesters. They also utilize the space under loose tree bark, deserted verdin nests, and in roots along riverbanks (Ehrlich et al. 1988). If a cavity is deep, Lucy's warblers will fill it up with debris so that they can see out (Otahal 2006). Lucy's Warbler nests are small, well woven nests with a coarse exterior, lined with soft material such as fur, feathers and plant fibers (Johnson et al. 1997).

Lucy's warbler are a host for brown-headed cowbird (*Molothrus ater*) nest predation, but effects at the population level are unknown (Johnson et al. 1997). Rosenberg et al. (1991) believed that cavity nesting should reduced the incidence of cowbird parasitism on Lucy's warbler, however, Bent (1963), Harrison (1984), and Terres (1991) all recorded parasitism incidences. Predators can also hinder the success of Lucy's warbler nests. Nests have been destroyed by and eggs have been eaten by wood rats (*Neotoma spp.*), snakes (Howard 1899) and lizard's (Dawson 1923). The Gila woodpecker is also known to predate eggs (Bent 1953, Griscom and Sprunt 1957, Harrison 1984).

Lucy's warbler eats insects almost exclusively (Johnson et al. 1997). It forages mainly on mesquite, but also forages on other shrubs, desert vegetation, and tamarisk in the Grand Canyon, Arizona (Stevens 1985, Yard 1996). Lucy's warbler is a foliage gleaner, but is non-specific in where it forages on a tree. They have been observed feeding at the top of the canopy to the lowest branches with leaves (Johnson et al. 1997). Within the tree, Lucy's warbler tends to spend more time foraging on flowers (Bent 1953, Griscom and Sprunt 1957, Ehrlich et al. 1988, Terres 1991, DeGraff et al. 1991). Due to this generalist approach, Lucy's warbler diet consist of a variety of insects and spiders, true bugs (Hemiptera), leafhoppers, beetles, flies, moth larvae, wasps, biting lice (Mallophaga), and thrips (Thysanoptera) (Moody 1970, Rosenberg et al. 1991, Yard 1996). The diet is general and the species of insect consumed varies with season (Moody 1970).

Potential Management Impacts:

The Arizona Partners in Flight considers Lucy's warbler a priority species or one most in need of conservation (Latta et al. 1999). It was chosen because it is a representative of the cavity nesting guild in mesquite bosque, a declining habitat type. Its population is suspected to be significantly reduced from historic trends. Rea (1983) estimated that historically, several thousand pairs of Lucy's Warblers inhabited the Gila River Indian Reservation, but since the late 1970s and early 1980s, only scattered pairs have been found. Habitat loss has occurred through conversion to agriculture or residential use, wood cutting, and by modification of stream flows. It is believed

that there are extensive threats to wintering grounds in Mexico. Loss of breeding grounds in Arizona and across its range has been estimated to be 26-50% of its habitat (Latta et al. 1999). This degradation and loss of riparian mesquite habitat has extirpated local populations, but the loss may be offset by the species ability to use salt cedar on the lower Colorado River (Rosenberg et al. 1991). Therefore, it appears that current losses on the breeding grounds do not likely threaten the species as a whole, but threats on its wintering grounds may reduce the population without conservation (Johnson et al. 1997). How habitat is lost, converted for human use, and replaced by non-native exotics is complicated and how it will affect the species is unknown at this time. Even if we lack empirical evidence, Arizona is of primary importance to Lucy's warblers because it represents 51-100% of the species total breeding distribution (Latta et al. 1999).

As a secondary cavity nester, Lucy's Warbler is dependent on large trees for suitable nesting sites and usually nests in or in habitats adjacent and associated with riparian areas, such as mature mesquite bosque. Within its breeding habitat Lucy's warbler exhibits nonrandom use of tree species for nesting, which indicates that not all tree species are equally suitable (Johnson et al. 1997, Stoleson et al. 2000). In the Gila River valley of New Mexico, Lucy's warblers preferred large cottonwoods and willows (*Salix sp*; Stoleson et al. 2000). However, most studies found Lucy's warblers nesting in mesquite trees (Gilman 1909).

On the KNF, there are only 1,201 acres of cottonwood willow riparian forest located in Kanab Creek, Sycamore Creek, and North Canyon Creek. Of the three, only Sycamore Creek and Kanab Creek are potential habitats for Lucy's warbler. Steinke (2007) identifies Sycamore Creek, in the upper Verde River watershed, as containing 1,109 acres of riparian areas including streams and springs. The report states that the extent, diversity and condition of the riparian habitat only contribute a very small amount towards riparian ecological sustainability. Even so, Sycamore Creek is likely out of this species breeding range based on elevation range and lack of suitable habitat (Troy Corman, Arizona Game and Fish Department, personal communication). Expanding the analysis out to wetlands, the forest contains 31 riparian areas totaling 1,109 acres mapped during the 1990 Riparian Survey of the Kaibab National Forest. Of the 31 areas, 5 are riparian (stream type lotic systems). Forest-wide, 18 wetland and riparian areas are in poor condition (16%), 22 are in fair condition (71%), and 4 are in good condition (13%). Wetland and riparian areas in fair and poor condition do not provide optimum habitat conditions, including adequate vegetative diversity. These conditions indicate that the necessary snag or gallery tree component are likely missing from wetland areas as these conditions do not support the development of large mature trees.

Kanab Creek on the North Kaibab Ranger District does hold some non-gallery riparian vegetation. Lucy's warblers were detected by the Arizona Breeding Bird Atlas (ABBA) on lower Kanab Creek within the KNF. Vegetation is dominated by woody shrubs and comprised mostly of tamarisk (Steinke 2007). There are isolated pockets of willow and cottonwood within the drainage, but are limited in extent and do not qualify as actual cottonwood galleries. Water flow has been restricted by diversions upstream of the Forest boundary (Steinke 2007). Although the lower portion of Kanab Creek is perennial within the boundaries of Grand Canyon National Park, it is not perennial on the KNF (Steinke 2007). Lucy's warbler can thrive within

tamarisk (Johnson et al. 1997, Yard et al. 2004); therefore it is likely this species could persist on the KNF in Kanab Creek.

Population Data:

All 19 analyses conducted using BBS data (Sauer et al. 2005) resulted in non-significant trends (Appendix 2) for Lucy's warbler, including the regional trend for Arizona (1966-2005 = -0.3, $p = 0.71$, Fig. 7). Both the Sierra Madre Occidental and Arizona BCR with group analyses showed negative results (range = -0.091 to -2.497) but again none were significant, suggesting that the population is likely stable at each level.

NatureServe lists this species as G5, N5B, and S5, or demonstrably widespread, abundant, and secure globally, nationally, and within Arizona.

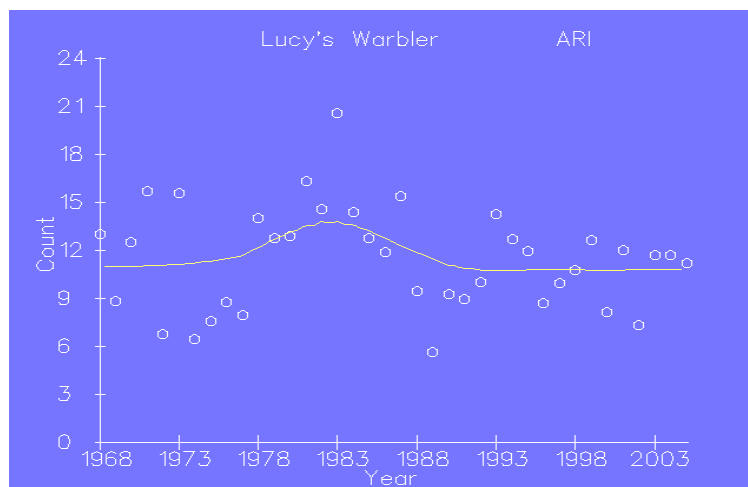


Figure 7. Lucy's warbler population trend data for Arizona from 1966-2005, BBS data (Sauer et al. 2005).

Locally, Lucy's warblers were detected by the ABBA on lower Kanab Creek within the KNF. The ABBA survey design is an area search within a 10 square mile area that attempts to document the presence of species breeding by establishing breeding behavior of birds through direct observation. Based on these observations and the observer's assessment of individuals, a rough estimate of abundance is given. The Kanab Creek block includes the lower section of Kanab Creek and surveys detected territorial behavior. Observers believe that the bird is a probable breeder and estimate the abundance of Lucy's Warbler to be between 11-100 breeding pairs. It is unclear where the National Forest/National Park falls within the surveyed effort and the habitat how much of this estimated population would occur on the KNF. It is also unknown how much of the survey focused on portions of Kanab Creek with perennial flow or how much included ephemeral water flow. This population is considered an extension of the upper Colorado River population (Troy Corman, Arizona Game & Fish, personal communication). While this estimate does not translate into a trend assessment, it does indicate that the species could be breeding on the KNF. Bird surveys conducted in Kanab Creek by the KNF in 2002 failed to detect any Lucy's warblers. The KNF surveys were not repeated so it is possible that the observers did not encounter or failed to identify the birds. It is likely that Lucy's warbler, if present at all, would be closer to the low end of the estimated range provided by the ABBA.

More important than the required trend statement is assessing whether the species is a valid MIS. Because riparian habitat is so limited on the KNF, it is likely that, at best, the KNF supports limited breeding at the edge of one Ranger District within a portion of a unique Wilderness Area. This is fundamentally different from having populations of Lucy's warblers occurring on the Forest. Even if a limited population (versus merely having individual birds or individual pairs of birds) occurred, it could not be easily monitored, violating one criterion for MIS. Lucy's warbler is also one of the least understood warblers of North America (Zimmerman 2004). Little is known about how habitat changes affect this bird and no empirical studies exist, violating yet other criteria for use as a MIS. This species does not serve as an indicator of any other species, being that it is a secondary cavity nester and has very different habitat requirements than other riparian birds, further violating MIS criteria. Because 'best science' requires the use of some empirical evidence or at least some survey data, we cannot expect this species to perform adequately as a MIS for the KNF.

Population Trend:

Considering the information above and given that the Forest Service is required to make a trend call, Lucy's warblers are likely stable within the limited habitat occurring on the KNF. More importantly, Lucy's warbler is not a viable MIS for the KNF.

Juniper Titmouse

The juniper titmouse (*Baeolophus ridgwayi*) was formerly considered to be the same species as its west coast counterpart and together they were called the plain titmouse (*Baeolophus inornatus*). In 1996, the American Ornithologists' Union split the plain titmouse into the oak titmouse (*Baeolophus inornatus*) and the juniper titmouse based on differences in songs, preferred habitat, and genetics (Sibley 2000). The name "titmouse" comes from the Old Icelandic *titr*, meaning "small," and the Anglo-Saxon word for bird, *mase*: "small bird" (Wells 2002).

Life History:

As an obligate secondary cavity nester (Latta et al. 1999) and obligate inhabitant of pinyon-juniper woodland (Phillips et al. 1964, Behle 1985, Andrews and Righter 1992, Small 1994), the juniper titmouse was selected to represent species using late-seral pinyon-juniper woodlands and the snag component within this habitat. Juniper titmice are most common where juniper is dominant and where large, mature trees are present to provide natural cavities for nesting (Cicero 2000). Few species are as closely tied to their habitat as juniper titmice. They are non-migratory and mainly reside throughout the year in pinyon-juniper woodlands. Juniper titmice use all of the many variations of pinyon-juniper habitats from thin, scattered stands of juniper at its lower limits to very dense stands of predominately pinyon pine heavily mixed with deciduous shrubbery at its upper limits (Latta et al. 1999). Within habitat variation, titmice seem to increase with increasing pinyon pine density (Pavlacky and Anderson 2001). Tree density used by breeding juniper titmice ranged from 155 to 380 trees per hectare (LaRue 1994, Masters 1980). Mature stands of pinyon-juniper are characterized by low densities of mature trees which allows for developed understory. Because mature trees have greater numbers of dead limbs there are more potential nest sites for Juniper titmice. This was exemplified by Pavlacky and Anderson (2001) who found that titmice are correlated with density of dead limbs in areas with significant ground cover. Juniper titmice occasionally wander into other habitats that are

adjacent to or near pinyon-juniper woodlands during the nonbreeding season including cottonwood, willow, buffalo-berry, and sagebrush shrublands (Phillips et al. 1964, Bradfield 1974, Brown et al. 1984, Gaines 1988, Andrews and Righter 1992, Small 1994, Sogge et al. 1998).

Juniper titmice may form permanent pair bonds and defend territories throughout the year (Ryser 1985), although less so in the winter (Cicero 2000). In western Nevada, Panik (1976) estimated mean territory size at 1.3 ha (3.2 ac). Within their territories, juniper titmice are more closely associated with the mature state of pinyon-juniper habitat because it typically contains more nesting cavities due to a higher density of snags and older live trees with dead/decaying parts (DeGraff et al. 1991). In a study using nest boxes in the White Mountains of California, juniper titmice used boxes in taller juniper trees with relatively sparse surrounding vegetation (Hall and Morrison 2003). In contrast, Shuford and Metropolis (1996) characterize juniper titmouse breeding habitat in the Glass Mountain area as moderately dense stands of pinyon woodland with fairly large trees, suggesting that the more critical element is larger trees than tree density. The variability in canopy cover is verified by LaRue (1994) who found it range from 11% to 26% a 2.4 fold increase.

Typically, juniper titmice nest in natural cavities such as knotholes or broken branches, but will also use woodpecker-excavated cavities or stump holes as well as readily using nest boxes (Panik 1976, Wilson 1992). Grinnell and Miller (1944) noted juniper titmice placing nests in crevices in twisted trunks of older, larger juniper trees. If holes are not suitable, birds can partially excavate the nest hole (Cicero 2000). Of 13 active nests found as part of the Arizona Breeding Bird Atlas, nine (79 %) were in junipers (Latta et al. 1999). Additionally, Corman and Wise-Gervais (2005) documented that 62% of nests were in pinyon-juniper woodlands and 25% were in three other vegetation communities all of which had strong juniper components (n = 526). Nests are typically low for a cavity nesting species, being only 1-3 m (3-10 ft) above the ground and in branches approximately 14-48 cm (5.5-1.5 in, Latta et al. 1999).

Titmice forage by gleaning insects from the bark of small branches and twigs within the canopy (Cicero 2000). They rely on the dense canopies for predator protection and will forage on the ground only where the understory and ground cover are thin (Ryser 1985). Juniper titmice forage on foliage, twigs, branches, trunks, and occasionally on the ground (Sibley 2000). Like other members of the family Paridae, the juniper titmouse has strong legs and feet, which allows it to hang upside down to forage. The juniper titmouse eats insects and spiders, sometimes seen catching insects in midair (Alsop 2001). It also takes berries, acorns, and seeds, sometimes hammering seeds against branches to open them (Cicero 2000). During the fall and winter months, the juniper titmouse switches more towards vegetable than animal matter. In the fall, Bradfield (1974) observed titmice feeding on juniper seeds and Balda (1987) considers the species to be a major pinyon pine seed predator that consumes large numbers of seeds. Christman (2001) found that the diet of juniper titmice is distinctive as compared to other tits, with large seeds (juniper and pinyon pine) and acorns making up most of the winter diet.

The switch in diet is likely a major contributing factor to how it interacts with other species. Typically, in the summer when food is abundant, juniper titmice occur as singles or pairs and do not typically form conspecific flocks (Phillips et al. 1964). However, in winter, titmice often

join mixed-species foraging flocks (Ryser 1985). Titmice and chickadees (*Poecile sp.*) forage on similar habitat components in winters with low food abundance (With and Morrison 1990). Titmice foraged more like chickadees, using ponderosa pine more than during periods of high food abundance. Chickadees may forage with titmice to gain knowledge of resource locations titmice develop as year round residents (With and Morrison 1990).

Predators of the juniper titmouse include birds and mammals (Bent 1946) and snakes (Wilson 1992). Typical predators include accipiters, small owls and Stellar's jay (*Cyanocitta stelleri*; Rowlett 1972, Panik 1976, Wilson 1992). Jays typically predate eggs, nestlings, and fledglings (Cicero 2000). Although it is a secondary cavity nester, the juniper titmouse is probably not subject to brood parasitism by brown-headed cowbirds (Latta et al. 1999).

Potential Management Impacts:

Conservation of the juniper titmouse is critical locally and throughout its range (Latta et al. 1999). Because the Colorado Plateau contains more than 40% of this species range, the conservation of this species a priority locally. Within the Colorado Plateau, changes in historic fire regimes and habitat conversion represent two major potential management impacts on the juniper titmouse. In addition, even-aged thinning and overstory removal could negatively impact juniper titmouse habitat, depending on objectives and implementation (Latta et al. 1999). Unrestricted snag removal through personal and commercial firewood harvests also represent a threat to the species.

Over the past 100 years, the suppression of fire has changed pinyon-juniper-woodlands from open diverse communities of trees, shrubs, perennial grasses, and forbs to dense woodlands (Dahms and Geils 1997). Suppression was initially indirect, using intensive livestock grazing to reduce understory fuels which carried fires. This strategy resulted in a reduction of fires and fire intensity allowing for a significant expansion of pinyon and juniper trees (Wright 1990) and an increase in tree density within historic pinyon-juniper woodlands. While it would seem that juniper titmouse habitat has expanded by increasing pinyon and juniper trees, dense forests limit the development of large mature trees and subsequent creation of snags, critical breeding habitat components for this species. Empirically, as tree density and canopy cover increases, juniper titmouse breeding density decreases (Balda and Masters 1980, LaRue 1994). Further, increased tree density and canopy cover increases the likelihood of high-severity stand-replacing fires (Brown 2000) rather than the low-intensity understory fires that historically were common in many pinyon-juniper woodlands. Therefore, as fire suppression and pinyon-juniper expansion continues, habitat quality is decreasing for this species or, in the case where high-severity fire is stand replacing, habitat is lost. Using fire as a tool within the pinyon-juniper woodlands will have to consider site-specific factors before concluding positive or negative effects to titmice. In addition to significantly affecting the natural fire regime, historic livestock grazing also affected ground cover and shrub regeneration in this habitat type (Latta et al. 1999). These effects eroded both habitat quality and the soils that support it. Additionally, large trees and snags were selectively removed for timber and fuel wood, significantly reducing the mature tree cohort within this habitat (Betancourt et al. 1993). Later, large tracks of mature trees were removed by chaining, bulldozing, and cabling methods to increase forage yield for livestock (Schmidt 1994). When this is practiced in historic grasslands invaded by pinyon and juniper, it likely has minimal effects on juniper titmice because of the lack of mature trees and large snags

in these younger stands. However, when mature pinyon-juniper habitat is removed to increase forage yield, this has a direct effect on juniper titmouse habitat. Although this no longer occurs on the KNF, it still occurs within the Colorado Plateau, negatively impacting juniper titmice in the general region (Sedgewick and Ryder 1987). Removal of large juniper trees has also occurred as a method to increase productivity of pine nuts on pinyon trees (Cicero 2000).

Because availability of natural or excavated cavities likely limit juniper titmice in pinyon-juniper woodland (Cicero 1996), retention of older juniper trees with large twisted trunks is especially important (Cicero 2000). Policies that ensure the retention of large trees and standing snags, such as limiting personal and commercial wood impacts to these resources, will help maintain necessary juniper titmouse breeding habitat. Similarly, limbing of old juniper can remove important habitat components on trees retained on site.

Thinning prescriptions and prescribed burns will likely not impact this species if snags and large trees are protected (Latta et al. 1999). However, thinning and burning can have negative effects if treated areas result in hot burns that damage the soil (Whisenant 1990). This could inhibit reestablishment of native plants and allow invasive species, such as cheatgrass (*Bromus tectorum*), to establish (Whisenant 1990). Cheatgrass, a non-native winter annual, germinates in the fall and grows slowly during winter. This creates a competitive edge where cheatgrass grows rapidly in the early spring, outcompeting native grasses. By early summer it has set seed and died, creating a continuous fuel bed of quick-drying, flashy fine fuel that can readily carry fire, even without wind. The presence of cheatgrass has made some pinyon-juniper areas susceptible to higher frequency wildfire or stand replacing fires in dense pinyon-juniper woodlands, creating a serious management concern (Whisenant 1990). By replacing native grasses, cheatgrass establishment can alter the invertebrate communities as well (Morrow and Stahlman 1984, Rosentreter 1994), resulting in habitat that no longer provides winter forage for titmice. This can be mitigated with native plant seeding following burns. Establishing herbaceous understories capable of supporting an arthropod prey base would likely benefit the juniper titmouse (Pavlacky and Anderson 2001). Where an adequate understory is present, timing prescribed burning with the phenology of the understory species could further benefit titmice (Monsen et al. 2004). Encouraging native grass and forb growth also alleviates soil erosion by stabilizing the shallow rocky soils common to pinyon-juniper woodlands.

The KNF Forest Plan provides guidelines that create openings, direct burns at appropriate intervals, and encourage understory seeding. This should result in a reduction tree density, which will help to enhance and re-establish understory grasses, forbs, and brows. In general, habitat treatments that encourage: a decrease in tree density and canopy cover; increasing native forage understory; and increasing snag and large tree densities should benefit juniper titmice. However, standing snags and large mature junipers are not currently protected. Current guidelines in the Forest Plan include removal of any dead standing juniper tree without green foliage. Because juniper titmice often depend on this habitat structure, changing this to trees < 5 in. root crown diameter/dbh or excluding juniper snags from collection would mitigate this impact.

Population Data:

BBS data produced 23 analyses for the juniper titmouse of which 52% significant and 96% had negative values (Appendix 2). Group analyses (successional or scrub breeders and permanent resident) for both BCRs indicate the juniper titmouse is significantly declining (trends ranged from -0.15 to -2.68, p-values ranged from 0.00 to 0.036). Group analyses for Arizona (successional or scrub breeders and cavity nester) showed non-significant negative trend values but permanent resident in Arizona was significant at the p-value of 0.10. Regional analysis resulted in a non-significant trend (-4.1, $p = 0.16$, Fig. 8) for the 1966-2005 survey period (Sauer et al. 2005). At the larger scale, all but one BCR and group analyses resulted in significantly declining trends ranging from -0.15 to -3.1. Together these results indicate populations are decreasing. This decline would reflect the trend within the greater pinyon-juniper ecoregional habitat analysis, which has significantly declined during the 1966-2005 time period (trend = -2.7, $p = 0.01$).

Conversely, NatureServe ranks the juniper titmouse as G5, N5, and S5, or demonstrably widespread, abundant, and secure globally, nationally, and statewide.

Within the literature, densities of juniper titmouse were variable. Note that for consistency between studies, densities of nesting pairs reported in the literature were converted to individuals by multiplying the number of pairs x 2. Densities of 2–10 birds per 10 ha (24.7 ac) were reported in western Nevada (Panik 1976). In Lassen County California, juniper titmice were as low as 0.5 per 10 ha (Pacific Research Bird Observatory data). Locally, Masters (1979) reported 14.4 to 26 per 10 ha in central Arizona and LaRue (1994) 3.8 to 5.75 per 10 ha in northeastern Arizona. The KNF landbird surveys found densities in pinyon-juniper stands were 2.22 in 2005 and 1.98 in 2006 per 10 ha.

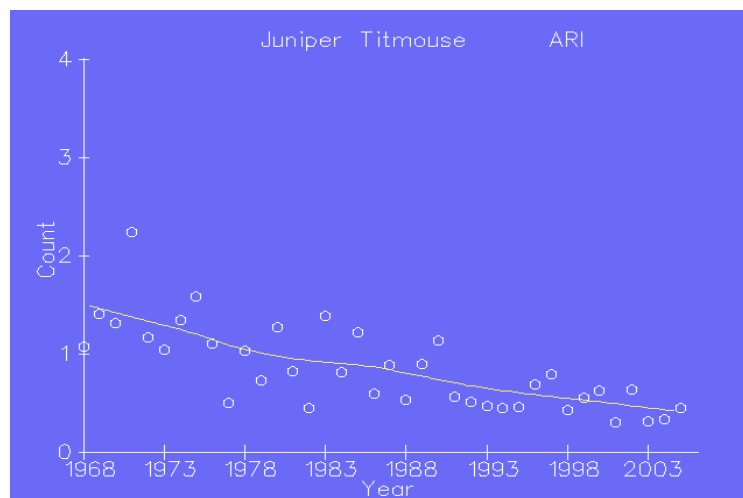


Figure 8. Juniper titmouse population trend data for Arizona from 1966–2005, BBS data (Sauer et al. 2005).

Trend Estimate:

Considering all the surveys from the literature and the KNF, the data suggest that juniper titmice are likely decreasing. This trend may be a reflection of long-term habitat trends in pinyon-juniper ecosystems. However, comparing local density estimates is difficult because of differences in survey and analysis methodologies. Current management practices on the KNF

encourage returning pinyon-juniper to its natural range of variation. Therefore, population trends on the KNF should continue to be surveyed while overall habitat restoration efforts continue.

Pygmy Nuthatch

The Pygmy nuthatch (*Sitta pygmaea*) is one of the most abundant species in ponderosa pine forests (Kingery and Ghalambor 2001) and one of the few cooperatively breeding songbirds in North America. Cooperative breeders use their progeny and other relatives to help feed incubating females, nestlings, and fledglings and to help defend the nest site. During the winter they communally roost with up to 30 individuals having been recorded exiting a single roost in Northern Arizona's ponderosa pine forests.

Life History:

Pygmy nuthatches were selected to represent species using late-seral, ponderosa pine habitat. They are regarded as one of the best indicator species for overall "health" of bird communities in ponderosa forests (Szaro and Balda 1982) because negative changes in its population status within managed ponderosa pine forests may reflect adverse changes in the community as a whole (Diem and Zeveloff 1980). Pygmy nuthatches are nearly limited to long-leaf pine systems, including ponderosa pine and Jeffery pine. In northern Arizona, they breed and feed in the vast expanses of ponderosa pine and also in shallow ravines that contain white fir, Douglas-fir, Arizona white pine, quaking aspen, and an understory of maple (Kingery and Ghalambor 2001). Pygmy nuthatches prefer old growth, mature forests (Szaro and Balda 1982), and unlogged forests support significantly larger populations than logged forests (Franzreb and Ohmart 1978, Brawn 1988, Sydeman et al. 1988). Conversely, this species can also be found in densely forested areas with smaller diameter trees as long as there is nesting and roosting sites available such as snags or trees with dead portions suitable for excavation (Hurteau 2007). Ponderosa pine foliage volume positively correlates with pygmy nuthatch abundance (O'Brien 1990), but abundance is inversely correlated with trunk volume, suggesting that the species prefers heterogeneous stands of well-spaced, old pines and vigorous trees of intermediate age (Balda et al. 1983).

Pygmy nuthatches are resident throughout their range, exhibiting little broad-scale movement in most populations in most years. The sedentary nature of pygmy nuthatches may be related to their highly social behavior. Families from post-breeding flocks and young males often remain on their natal territory to assist the parents in the following breeding season. Post-breeding wandering to lower and higher elevations, and to non-pine habitats, does occur irregularly from July to December, sometimes on a large scale (Kingery and Ghalambor 2001). Territory size ranges from 0.54 to 8.15 ha (1.3–20.1 ac, Norris 1958, Balda 1967, Storer 1977), varying with number of nuthatches in a family group, density of pine trees, and density and availability of nest sites. Territory size is significantly larger on heavily logged plots than on thinned plots (Brawn and Balda 1988a) and on plots with nest boxes in snag-poor habitats (Brawn and Balda 1988a, Bock and Fleck 1995) suggesting that both tree density and snag availability affects population size.

Pygmy nuthatches are primary cavity-nesters, excavating dead or well-rotted wood, but they also utilize existing cavities in northern Arizona. As both a primary and secondary cavity nester, pygmy nuthatches nest primarily in ponderosa pine, but occasionally use other conifers and

quaking aspen if cavities are present (Kingery and Ghalambor 2001). Nest cavities are often placed under or near existing broken-off branches (Hurteau 2007). In Arizona, 73 percent of nests were found in new excavations, 23 percent in secondary cavities, and 4 percent in natural cavities (n = 237 nests, Kingery and Ghalambor 2001). In central Arizona, 51.7 percent of pygmy nuthatch nests were in dead trees, and 48.3 percent were in dead sections of live trees (Kingery and Ghalambor 2001). During the winter, pygmy nuthatches use large tree-cavities for colonial roosts, which are typically associated with large snags. Because of this, roost cavity availability may be a limiting factor for local populations.

Pygmy nuthatches are primarily insectivorous (DeGraff et al. 1991). They forage in needle clusters, cones, twigs, branches, and trunks (Stallcup 1968, Bock 1969, McEllin 1978, 1979b, Ewell and Cruz 1998). During the breeding season, the pygmy nuthatch diet consists of 60 to 85 percent insects (Norris 1958, Anderson 1976), including beetles, wasps and ants, true bugs, and larvae of moths and butterflies (Beal 1907, Norris 1958). Winter diet is variable by location and long-leaf pine association. It can remain similar to summer diets (Norris 1958, Anderson 1976) or shift to primarily vegetable matter and pine seeds (Norris 1958).

Predators of adult and juvenile pygmy nuthatches include sharp-shinned and Cooper's hawks, northern pygmy-owl (*Glaucidium gnoma*), western scrub-jay (*Aphelocoma californica*), and Steller's jay (Kingery and Ghalambor 2001). While inside cavities, pygmy nuthatches are also vulnerable to predation by chipmunks, red squirrels (*Tamiasciurus hudsonicus*), and gopher snakes (*Pituophis melanoleucus*) (Kingery and Ghalambor 2001). Egg and nestling predators include hairy woodpecker, Steller's jay, house wren (*Troglodytes aedon*), gray-necked chipmunk (*Eutamias cinereicollis*), red squirrel, and gopher snake (Kingery and Ghalambor 2001).

At the community level, pygmy nuthatches likely experience strong interspecific competition for nest sites from other cavity-nesting species. Agonistic behavior has been seen near nest sites between pygmy nuthatches and hairy woodpeckers, violet-green swallows (*Tachycineta thalassina*), white-breasted nuthatches (*Sitta carolinensis*), mountain chickadee (*Parus gambeli*), house wrens, western (*Sialia mexicana*) and mountain bluebirds (*S. currucoides*) (Kingery and Ghalambor 2001), and Williamson's sapsuckers (*Sphyrapicus thyroideus*) (Dobbs et al. 1997). However, during winter, competitors such as the white-breasted nuthatch, hairy woodpecker, and mountain chickadee become allies as common flock associates (McEllin 1979b), helping each other to find food and keep watch for predators.

Potential Management Impacts:

Ponderosa pine habitat is common across the KNF and occurs on all three ranger districts. While the gross habitat requirement of the pygmy nuthatch is readily available across the KNF, the degradation of habitat quality since European settlement has likely compromised its value as suitable habitat. In the northwest, pygmy nuthatches are reported to prefer unmanaged mature ponderosa pine forests retaining historic fire return intervals, a habitat that has limited representation across the landscape (Hutto 1989, Wisdom et al. 2000). Selective logging of large trees, loss of large cavity substrate, and fire suppression represent significant threats to pygmy nuthatch foraging, breeding, and roosting habitat (Bock and Block 2005, Ghalambor and Dobbs 2006). Additionally, fire suppression and livestock grazing interact to reduce fire frequency, which creates an increased risk of stand-replacing wildfires that ultimately reduces habitat

availability and quality by reducing or eliminating sources of food and shelter (Ghalambor and Dobbs 2006). Ponderosa pine forests are likely to remain in this condition without active management.

Little information is available on populations of pygmy nuthatches prior to fire suppression policies, but evidence from Arizona and New Mexico suggests that the species was abundant (Scurlock and Finch 1997). Management strategies that move ponderosa pine forests closer to the historic range of variation should positively affect the pygmy nuthatch. Applying the goshawk guidelines to direct management practices should positively affect this species, as this prescription results in forest structure that more closely resembles historic forest conditions than those present today.

Timber management and harvest practices can positively affect pygmy nuthatch habitat quality by augmenting trees used for nesting, roosting, and foraging (Ghalambor and Dobbs 2006). For nesting and roosting, snags and dead portions of live trees are the most critical habitat component for pygmy nuthatches and augmentation can be accomplished directly and indirectly. This can be achieved by retaining snags and dying trees during timber operations. Also important is the indirect contributions to pygmy nuthatch habitat achieved through the maintenance of tree populations within large diameter classes that will ensure snag recruitment from primary mortality agents such as fire and drought, or secondary mortality agents, such as disease and insect attack. Large trees also increase the number of living trees with dead components because lightning tends to strike big trees. These trees are then used by multiple secondary cavity nesters once the wood has rotted to a suitable level for excavation or natural cavities form.

Returning fire to the ponderosa pine systems also benefits pygmy nuthatches directly and indirectly. Fire that mimics historical fire regimes in the form of prescribed burns or wildfires can create snags and foraging areas for pygmy nuthatches. It has been shown that removing litter or other flammables from the base of existing snags, living trees with dead wood, and dying trees can aid in their retention. Fire managers on the KNF often rake around snags. However, fire-killed trees create snags that are used sooner for nesting and roosting habitat because of faster decay rates than in snags created from other mortality agents (Covert-Bratland et al. 2007). However, severely burned trees are lost sooner by toppling in high winds (Salaman 1934, Flanagan 1996, McHugh and Kolb 2003). Trees that die from secondary mortality agents after moderate-burns help to maintain snags on the landscape by dying slower and lasting longer after a fire event. However, Hurteau (2007) found a 3-fold increase in pygmy nuthatch densities in core sampling areas within ponderosa pine sites that had been thinned and burned whereas pygmy nuthatch densities remained relatively unchanged on thin only and burn only sites. This suggests that it may not simply be forest structure, but also forest function that benefits this species.

Population Data:

Twenty-five of 28 (82%) BBS analyses conducted for pygmy nuthatches showed negative trends (Appendix 2). Pygmy nuthatches are significantly declining across the Southern Rockies (1966-2005: trend = -3.0, $p = 0.080$, 1980-2005: trend = -3.3, $p = 0.04$). Arizona had similar but non-significant trends, likely due to small samples sizes (1966-2005: trend = -1.5, $p = 0.53$, 1980-2005: trend = -2.2, $p = 0.5$; Fig. 9).

NatureServe lists this species as G5, N5, S5, or demonstrably widespread, abundant, and secure globally, nationally, and within Arizona.

Locally, several research projects conducted on both the Kaibab and Coconino determined pygmy nuthatch density. Hurteau (2007) found breeding density in untreated ponderosa pine to be 6.9 ± 1.3 SE, 6.3 ± 0.9 on thinned plots, 5.9 ± 0.9 on burned plots and 10.2 ± 2.2 on thinned and burned plots per 10 ha (24.7 ac). During a winter study, Pope (2006) found densities to be 4.5 ± 0.75 SE in burned areas and 3.97 ± 0.71 in untreated ponderosa pine per 10 ha. Disparities in density are minimal, but likely include differences in detectability of the species between seasons. These findings indicate lower densities in unmanaged forests, but

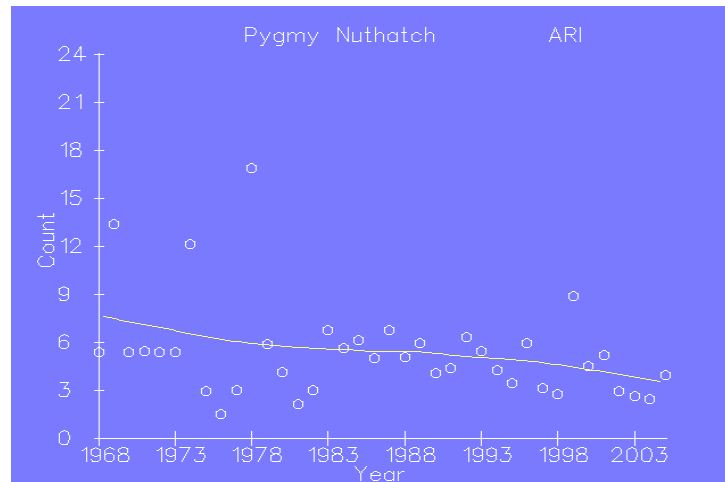


Figure 9. Pygmy nuthatch population trend data for Arizona from 1966-2005, BBS data (Sauer et al. 2005).

similar densities after thinning than previously determined in local historic studies. Brawn and Balda (1988) determined pygmy nuthatch density in areas unlogged since the 1920's, which would be comparable to those in the more recent studies, to be 9.4 and in moderately thinned plots and 1.65 in severely thinned plots per 10 ha (calculated by multiplying nesting pairs per 40 ha x 2 and dividing by 4). Scott (1979) found similar pygmy nuthatch densities: 8.6 in unlogged areas and 3.6 in thinned areas per 10 ha (calculated from nesting pairs as described above). This suggests that densities in unmanaged forests could be declining and that thinning has similar influences on pygmy nuthatch density regardless of surrounding population size. However, this must be interpreted cautiously since sampling and analysis methods were different between recent and historic studies.

The KNF landbird surveys found similar results to recent studies for the pygmy nuthatch. Densities in ponderosa pine were 6.18 in 2005 and 4.94 in 2006 per 10 ha. Because methods for sampling and analysis used by the KNF surveys were similar to those from recent studies, comparisons should be valid. Considering all the surveys from the literature and the KNF, the data suggest that pygmy nuthatches have declined over the long term. Research suggests that when thinning and burning are used in combination, densities may be similar to historic values.

Trend Estimate:

Considering the information above, pygmy nuthatches are assumed to be stable to declining on the KNF. In areas that were treated with thinning and prescribed burns or have been thinned and then burned naturally, pygmy nuthatches are likely stable to increasing. Pygmy nuthatches will benefit from forest management practices that result in stands more closely resembling historic forest structure. However, long term trends of this species will best be determined by continued monitoring on the forest.

Mexican Spotted Owl

Owls have comb-like features referred to as "flutings" or "fimbriae" on the leading edge of their flight feathers. This adaptation disrupts air flow into little groups called micro-turbulences during flight. Air rushing past wings of other bird species creates turbulence and the associated gushing noise. Micro-turbulences effectively muffle the sound of rushing air, allowing owls to fly silently. Perhaps due to their amazing stealth hunting, spotted owls can live to be 17 years old.

Life History:

Mexican spotted owls (*Strix occidentalis lucida*) are one of three spotted owl sub-species. They are the only sub-species to occur in the Southwest. They were listed by the US Fish and Wildlife Service as "threatened" in 1993. Following this listing, a recovery plan was written (USDI 1995) which was formally incorporated into the land management plans for all Southwestern National Forests (USDA 1996). Because this species is dependent on habitats containing large, old trees, the Mexican spotted owl was selected to represent species using late-seral, mixed conifer, and spruce-fir habitats. Mexican spotted owls can be found in forested mountain ranges and deeply incised canyons from southern Utah and Colorado south to the Guadalupe Mountains of west Texas and from 1,676 to 2,743 m (5,500 to 9,000 ft) elevation (Williams and Skaggs 1993, Young et al. 1994, USDI 1995). In Arizona, they are found north of the Mogollon Rim.

Although Mexican spotted owls are not obligated to migrate, many do so between their breeding and winter ranges. In six studies, 8 of 64 radio-marked owls migrated, including one that moved up in elevation during winter (Ganey et al. 1992, Willey 1993, Zwank et al. 1994, USDI 1995). When migration does occur, it is usually rapid and direct with birds passing through unsuitable habitat and moving 20 to 50 km with vertical displacement of greater than 1,000 m (3,280 ft; Gutiérrez et al. 1995). When owls move to lower elevations, they generally inhabit pinyon-juniper woodlands and riparian areas from November through March (Ganey 1998).

Mexican spotted owls respond aggressively to recorded or imitated vocalizations throughout the breeding season, suggesting that they are very territorial. However, disputes between neighbors are reported to be rare (Forsman et al. 1984) and Finton (1991) suggests that aggression may be alleviated by recognition of neighboring birds through their calls. Occasionally birds have been known to roost with members of opposite sex other than their mate (Forsman et al. 1984). Mexican spotted owls have home ranges less variable than other spotted owl subspecies (Ganey and Balda 1989b). Forested home range sizes are positively correlated with elevation, percent old-growth forest, and total amount of old forest (Ganey and Balda 1989b). In canyon habitats, home ranges appear to be influenced by topographic features (Willey 1993).

During the breeding season, Mexican spotted owl nest, roosts, and feed in a wide variety of habitat types and forest stand conditions throughout their range, but prefer those dominated by Douglas-fir, pine, true fir (*Abies*) and pine-oak forests (Ganey and Balda 1989a, 1994, Seamans 1994, USDI 1995, Ganey et al. 2003, Gutiérrez et al. 1995). In Arizona, the species breeds primarily in mixed conifer forest, although some breed in ponderosa pine with a well developed understory of Gambel's oak (Ganey 1998). Where forested areas are contiguous, Mexican spotted owls strongly select old-growth forests (Ganey and Balda 1989a) or forests with more complex structure than surrounding forests (Seamans 1994). Mexican spotted owls also use steep, narrow canyons with cliffs where suitable nest sites and perennial water sources are available (Rinkevich 1991, Willey 1993). Within these canyon habitats typical vegetation includes conifers or riparian forests, or clumps of trees, but also may be sparsely vegetated (Rinkevich 1991, Willey 1993).

In northern Arizona, Mexican spotted owls nest in areas with higher proportions of canopy closure ($\geq 55\%$) with mature trees and lower proportions of open ($< 10\%$) forest than the surrounding landscape (Grubb et al. 1997, May and Gutiérrez 2002). When nesting in pine-oak forests, nest sites are located in stands on steep slopes and in mature (> 45.7 cm dbh, 18 in) Gambel's oak or ponderosa pine (May et al. 2004). Mexican spotted owls do not make their own nests but instead use existing structures such as cliff ledges, cavities of debris, platforms on trees, and stick nests built by other birds. These nests are found in dense, multilayered, older portions of forest (Gutiérrez et al. 1995). When nesting in narrow steep-sided canyons, however, Mexican spotted owls will place their nests in areas with relatively little forest habitat (Rinkevich 1991). Females will scrapes out shallow depression in existing debris when laying eggs (Forsman et al. 1984).

Mexican spotted owls forage over greater areas and in a wider range of habitat types than those used for roosting (Ganey et al. 2003). Relative to what is available, Mexican spotted owls prefer areas with high canopy closure, greater live-tree basal area, greater snag density, and greater fallen logs (Ganey 1988, Ganey et al. 2003). This likely contributes to why these owls prefer to forage in unlogged forest more than expected and less than expected in selectively logged forest (Ganey and Balda 1989b). Over the course of a night, owls in forested habitats will search for prey in several stands (Ganey and Balda 1994), but owls in canyon habitat exhibit strong centers of activity (Willey 1993). In both forested and canyon areas, this species mainly eats rodents, but will also consume bats, birds, reptiles, and arthropods (Duncan and Sidner 1990, USDI 1995). Within a northern Arizona pine-oak forest, Block et al. (2005) determined that 94% of the owls diet consisted of animal matter including deer mouse (*Peromyscus maniculatus*), brush mouse (*P. boylii*), Mexican woodrat (*Neotoma mexicana*), and Mexican vole (*Microtus mexicanus*).

Being a large predatory bird, the Mexican spotted owl has few enemies outside other raptors. Predators of fledged young, dispersing juveniles, and rarely on adults include northern goshawks and great horned owls (Forsman et al. 1984, Gutiérrez et al. 1985, Miller 1989). Common Ravens have been observed attempting to predate eggs (Gutiérrez et al. 1995) and fisher (*Martes pennanti*) have been observed loafing in spotted owl nest trees and may prey on eggs and young in the nest (Gutiérrez et al. 1995).

Mexican spotted owls are solitary except when interacting with their mate but have been known to associate with other individuals on rare occasions (Gutiérrez et al. 1995). Like other predators they are often mobbed by many diurnal bird species including Steller's jay, American robin (*Turdus migratorius*), solitary vireo (*Vireo solitarius*), Anna's hummingbird (*Calypte anna*), Allen's hummingbird (*Selasphorus sasin*), acorn woodpeckers (*Melanerpes formicivorus*), and pileated woodpeckers (*Dryocopus pileatus*). Mexican spotted owls likely compete with great horned owls and barred owls, with the latter being known to displace spotted owls from their territories (Hamer 1988).

Potential Management Impacts:

Across its range, the spotted owl was subjected to loss and degradation of habitat from even-aged tree management. As a result of these practices, all 3 subspecies have experienced extensive loss of habitat in their ranges (Ganey and Balda 1989a, Thomas et al. 1990, USDI 1990, 1992, 1993, Bias and Gutiérrez 1992, Bolsinger and Wadell 1993, Dunbar and Blackburn 1994, Gutiérrez 1994a). Evidence exists that forests selectively logged in the past can be reoccupied by owls relatively soon (40–100 yr) if residual forest elements such as snags, coarse woody debris, and large trees with cavities are present (Forsman 1976, Forsman et al. 1977, Chávez-León 1989, Bart and Earnst 1992, Verner et al. 1992b, Folliard 1993, Seamans 1994). With rare exception, the KNF Forest Plan manages spotted owl habitat in accordance with Mexican spotted owl recovery plan.

Mexican spotted owls have had a long evolutionary history in forests structured by frequent, low-intensity fires. However, past forest management forest practices have changed forest structure from open, mature forests to extremely dense stands (148/ha increasing to 1265/ha, Fulé et al. 2003). According to habitat descriptions in the Recovery Plan (USDI 1995), these changes have likely improved and expanded Mexican spotted owl habitat above historic levels. Further complicating this deviation from the historic range of variation is the increased threat of habitat loss from wildfire, which is arguably the greatest threat to owls in the Southwest. Small diameter thickets have become increasingly abundant (Covington and Moore 1994), providing abundant fuel and connectivity for wildfires (Lowell 1996). Fuel connectivity has dramatically increased both vertically (i.e., ladder fuels that carry surface fires into the canopy) and horizontally (increasing canopy fuel loading). These changes result in many current wildfires burning hotter, covering larger portions of the landscape, and are often stand replacing such as the Pumpkin Fire on Kendrick Mountain in 2000, which burned most of the habitat in 2 of the 8 total PACs occurring on the KNF. In general, wildfire can decrease habitat availability by reducing or eliminating nesting, roosting, and foraging habitat (Sheppard and Farnsworth 1997).

In the case of low-intensity fire, owls tend to stay near the general area where fire occurred following the disturbance (Willey 1998). Additionally, Bond et al. (2002) documented high survival, site and mate fidelity, and reproductive success after "large" fires for 21 owls in 11 territories combined for California, Arizona, and New Mexico. Size of the fire does not indicate the severity of the fire and concern also remains over the potential cumulative loss of habitat to wildfire as a major threat to maintaining spotted owl populations (USDI 1995). Jenness et al. (2004) found that unburned areas tended to have more pairs and more pairs that reproduce young than burned areas. Unburned sites also had greater occupancy than burned sites, however, small sample sizes complicated analyses and resulted in insignificant affects. The authors suggested

fire has a significant effect on owls, citing probability of occupancy as 14% higher and probability of successful reproduction 7% higher in unburned sites. Further study needs to be conducted to determine more explicit results.

The Mexican spotted owl Recovery Plan (USDI 1995) recommends protection of and guidelines for actual and potential habitat, timber management, and forest restoration. Specifically around PACs, the plan recommends protecting 243 ha (100 ac, based on 75% adaptive kernel home range estimation) of habitat centered on the owl's nest site, roosting areas if nest site is not known, or subjective calls of habitat quality by US Fish and Wildlife Service personnel (W. Austin, personal communication). Over the broader landscape, the plan directs protection of habitat on steep slopes ($> 22^\circ$) and within research natural areas and wilderness areas. Timber harvest is recommended to be uneven-aged tree management in habitats outside of PACs and restoration efforts should focus on riparian zones with use restrictions to ensure the success of restoration.

Because timber management was the most prominent form of habitat alteration in most southwestern National Forests, the Mexican spotted owl Recovery Plan focused on providing a detailed description of the target and minimum threshold habitat conditions necessary to sustain Mexican spotted owl populations (Table 6). Habitat designated as suitable on the KNF is being managed to meet target/threshold conditions. Restricted habitat surrounding PACs is also managed for target/threshold conditions, but with less intensive treatments on the areas. In addition, Critical Habitat has been designated by the U.S. Fish and Wildlife Service, including pine-oak habitat on the Williams District and mixed conifer habitat on the NKRD. Interestingly, owls have never been detected in the pine-oak or mixed conifer on those respective Districts.

Table 6. Target/threshold conditions within restricted areas for the spotted owl on the Kaibab National Forest, Coconino County, AZ (USDI 1995). Both categories of mixed conifer conditions must be met.

Forest type	% area that must meet conditions	% trees 12-18" dbh	% trees 18-24" dbh	% trees >24" dbh	Tree basal area (ft ² /acre)	Large (>18" dbh) trees/acre
Mixed conifer Target	25	10	10	10	150	20
Mixed conifer Threshold	10	10	10	10	170	20
Pine-oak	10	15	15	15	150	20

The Forest Plan also provides directions to maintain and develop potential nesting and roosting habitat now and into the future, while providing a diversity of site conditions and tree group sizes across the landscape. The resulting landscape mosaic should ensure adequate nesting, roosting, and foraging habitat for Mexican spotted owls and habitat for a variety of prey species. Nesting/roosting stands include high tree basal area, large trees, multi-storied canopy, high canopy cover, and downed logs and snags. Occupied nesting and roosting PACs receive protection, as do steep slopes, unoccupied reserved lands, and restricted habitat. To date, this direction has not been integrated with maintaining sustainable forests relative to the historic range of variation for ponderosa pine and dry mixed conifer habitat.

Population Data:

Because BBS surveys are conducted during the day, nocturnal species are not surveyed. Thus, there is no BBS survey data for Mexican spotted owls. NatureServe has evaluated the species and considers it vulnerable to extirpation or extinction globally and nationally and vulnerable to extirpation or extinction Statewide. NatureServe states that while there are a “fairly large number of occurrences ... relatively few are of high quality, and the population trend is probably downward because of past and continuing loss and/or fragmentation of habitat, especially [resulting from] even-age timber management; threatened in some areas by the potential for catastrophic fire.” However, the same document cites “No undisputable evidence is available indicating that the population is declining or is significantly below historical levels” (USDI 1995).

Seamans et al. (1999) assessed trends for two Mexican spotted owl populations in the Upper Gila Mountains Recovery Unit, one in central Arizona on the Coconino Plateau and the other in west-central New Mexico. They estimated that both populations were declining at greater than or equal to 10% per year from 1991 through 1997. The total population size is not reliably known, but the Arizona-New Mexico population was estimated to be about 2,000 individuals (USDI 1995). In 2000, the population of the Upper Gila Recovery Unit was estimated to be approximately 1200-1650 individuals (White et al. 2001).

All Mexican spotted owl suitable habitat on the KNF has been surveyed according to USFS Regional protocol. The KNF initiated comprehensive surveys on the NKRD in 1988 and 1990 on the South Zone. Prior to these surveys, only sporadic surveys were conducted and there are no records of surveys prior to the 1970s. Despite extensive surveys since 1988, covering nearly all the mixed conifer habitat on the NKRD, no breeding or resident Mexican spotted owls have ever been confirmed on the District. Accordingly, the US Fish and Wildlife Service did not designate any PACs on the NKRD. Mexican spotted owls north of the Colorado River are in the Colorado Plateau Recovery Unit. All designated PACs in this Recovery Unit are in canyon habitat, not forest habitat like that designated and managed outside of the historic range of variation for spotted owls on the NKRD. Over 40 PACs occur around the edge of the Kaibab Plateau, inside Grand Canyon National Park. Assuming some percentage of the young fledged from Grand Canyon nests and foraging adults or those making seasonal movements have “discovered” the top of the Kaibab Plateau, the abundance of PACs around the Plateau brings to question the suitability of the Kaibab Plateau, including the unmanaged forests of Grand Canyon National Park, for Mexican spotted owl occupancy.

Mexican spotted owl PACs are absent on 2 of 3 Ranger Districts on the KNF. The Williams Ranger District has 6 functional PACs (not including those burned in the Pumpkin Fire). The Williams Ranger District is in the Upper Gila Mountains Recovery Unit. Because available funds for monitoring have been variable since 1990, the number of PACs surveyed on the KNF has varied and occupancy surveys were not typically conducted to protocol (Table 7). However, since 1994, all PACs were monitored annually using repeat visits with the exception of one year. One of these PACs crosses forest boundaries onto the Coconino National Forest and is thus administered by both Forests. Unlike goshawks, there is no information on reproductive success and nestling survival for Mexican spotted owls on the KNF. However, occupancy appears to be declining, but given the number of total PACs from which the trend line is drawn, this may not

be a meaningful change (Fig. 10). There is limited nesting habitat for Mexican spotted owls on this District, with the only confirmed nest sites occurring on steep canyon walls or in mixed conifer habitat on the 3 cinder cone mountains on the District. No Mexican spotted owls have ever been detected in the pine-oak designated across much of the Williams Ranger District. Surveys have been conducted to protocol on this District since 1990. The Tusayan Ranger District does not include spotted owl habitat and there are no records of spotted owls occurring on this District.

Trend Estimate:

In general, occupancy of the 6 designated Mexican spotted owl PACs is decreasing on the KNF. Based on this alone, Mexican spotted owl population trends appear to be decreasing on the KNF. Reproductive success for nesting birds on the KNF is unknown. Reproduction for Mexican spotted owls on other National Forests is known to be variable and is thought to relate to weather conditions (Seaman et al. 2002). It is unknown whether the apparent decline on the KNF is related to precipitation patterns or even whether a decline actually exists. Basing population trends on a sample of 6 has no scientific merit. More significant is the apparent lack of spotted owls, nesting or otherwise, across the 3 Ranger Districts. The KNF is at the extreme edge of occupied spotted owl habitat. MSO habitat appears to be limited to canyon and mixed conifer habitat occurring on the Williams District.

Table 7. Spotted owl monitoring data for the Kaibab National Forest, Coconino County, AZ. Not all seasons included full protocol.

Year	# PACs monitored	# PACs with MSO present	% PACs occupied
1990	5	5	100
1991	4	4	100
1992	5	4	80
1993	4	4	100
1994	6	6	100
1995	6	3	50
1996	6	5	83
1997	6	3	50
1998	6	5	83
1999	2	1	50
2000	6	2	33
2001	6	4	66
2002	6	1	17
2003	4	2	50
2004	3	2	66
2005	3	1	33
2006	3	2	66
2007	6	5	83

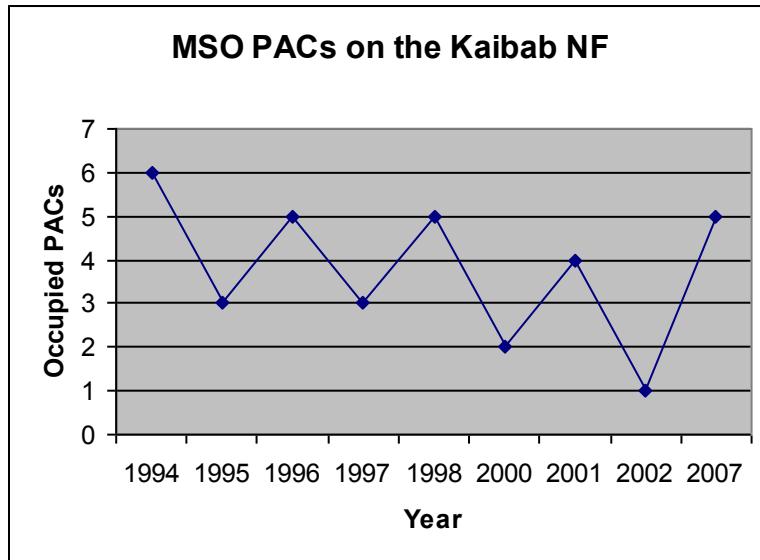


Figure 10. Number of occupied Mexican spotted owl protected activity centers (PACs) during years when all known PACs were monitored on the Kaibab National Forest, Coconino County, AZ.

Wild Turkey

Arizona is one of the few states that can boast wild turkeys (*Meleagris gallopavo*) of pure historic lineage (Phillips et al. 1964). Wild turkeys have inhabited Arizona since pre-Columbian times, when Native Americans kept them in domestication.

Life History:

Turkeys were chosen to represent late-seral ponderosa pine forests, but are also an economically and socially important species. National Forests contain the majority of turkey habitat in Arizona. They are found primarily in ponderosa pine forests with a mix of meadows, oak, and juniper. Roosting and nesting habitat consists of large, open-crowned trees often on steep slopes (Corman and Wise-Gervais 2005). Good brood rearing habitats include natural or created openings, riparian areas, abundant herbaceous vegetation adjacent to forest cover, and mid-day loafing and roosting areas. Turkeys are migratory in parts of their range, moving between lower elevations for wintering to higher elevations for breeding. Timing of movements can differ annually, depending upon snowfall (Hoffman et al. 1993). Typically they will stay in areas with a good mast crop until snow is too deep to allow for foraging (Wakeling 1991). Forage includes cone crops produced by mature ponderosa pine trees, hard mast from oak trees, seeds from grasses and forbs in early seral habitat, and invertebrates. Large woody debris is also used as cover (DeGraff et al. 1991).

Potential Management Impacts:

Current forest conditions currently provide necessary habitat for turkeys. Small scale thinning and prescribed burning creates open areas for foraging while preserving denser areas for nesting. However, current forest conditions are vulnerable to high-severity crown fire which eliminates turkey habitat. Tree harvest under the goshawk guidelines, which result in a mosaic of interspersed vegetative structural stages, will provide necessary habitat characteristics, such as roost stands, open areas for foraging, and downed woody debris for nesting. Reducing canopy

cover should increase invertebrate production, which may be a key element in maintaining wild turkey populations (Randal et al. 2007). Thinning under the goshawk guidelines also reduces the risk of stand replacing wildfire.

Population Data:

BBS data for turkeys from 1966-2005 in the Southern Rockies/Colorado Plateau region shows a significant ($p = 0.03$) positive trend of 12.8 (Fig. 11) across 21 Routes. In 1997, the AGFD began tracking population trends by using the number of turkeys seen per day by archery hunters during elk season (Table 8). AGFD believes this to be the most accurate trend information available for wild turkeys in Arizona (R. Miller, AGFD, personal communication 2002). Although the dataset is not large, indications for GMUs on the South Zone of the KNF are that populations have tended to remain stable or have increased (Fig. 12). Survey information is not available for the NKRD where elk hunts are rare and opportunistic. Continued surveys should allow a better interpretation of the changes in numbers. The KNF Landbird surveys are not designed to detect secretive birds and have rarely detected wild turkeys.

NatureServe lists turkeys G5, N5, S5, or demonstrably widespread, abundant, and secure globally, nationally, and within Arizona.

Trend Estimate:

Based on AGFD and BBS data, it appears that turkey populations on the KNF have a variable but overall increasing trend.

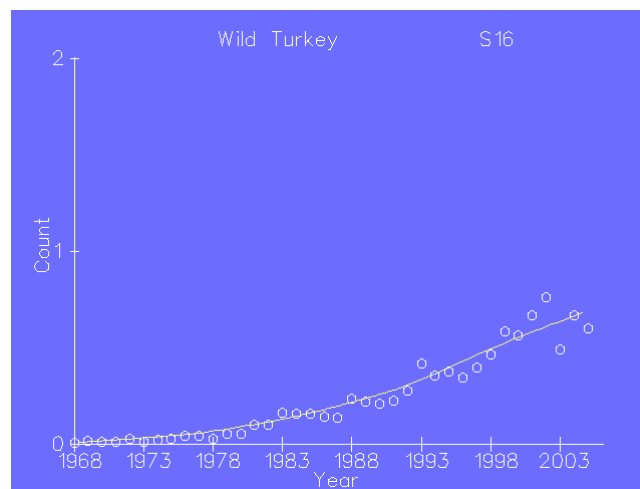


Figure 11. Wild turkey population trend data for the Southern Rockies/Colorado Plateau region, BBS Data (Sauer et al. 2005).

Table 8. Percentage of people hunting elk by bow reporting turkey observations, South Zone of the Kaibab National Forest, Coconino County, AZ (AGFD).

Year	Unit 6B	Unit 7	Unit 8	Unit 9
1997	39	34	57	22
1998	42	37	79	23
1999	64	31	77	24
2000	78	42	95	20
2001	62	40	54	50
2002	44	31	66	13
2003	55	35	80	48
2004	58	46	88	33
2005	73	41	96	49
2006	53	54	82	59

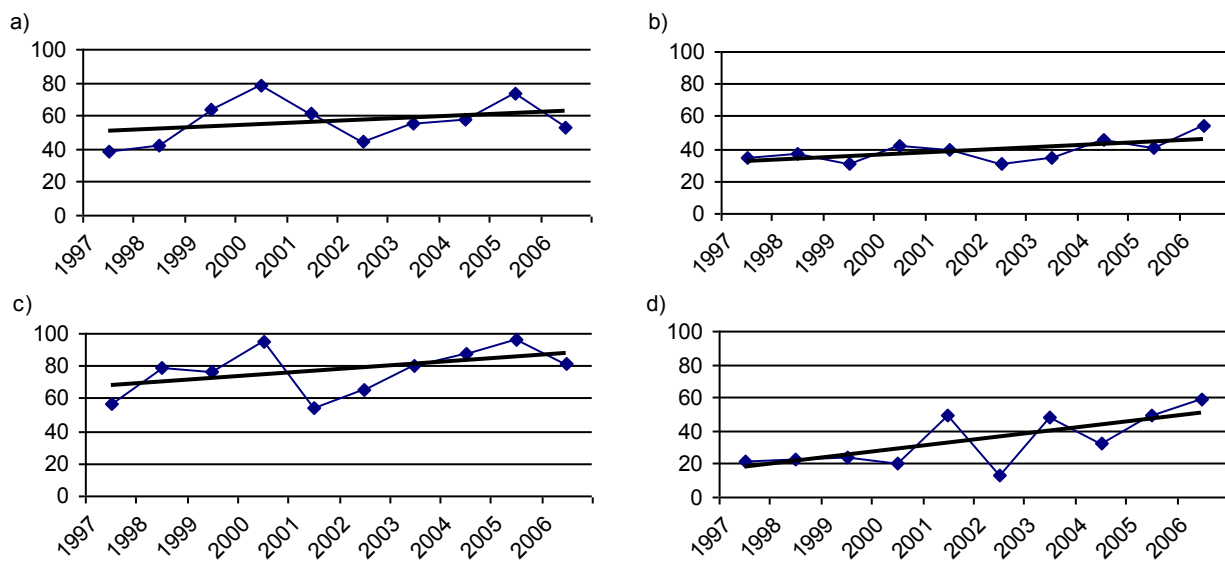


Figure 12. Percent of hunters per day seeing turkeys in hunt units a) unit 6B, b) unit 7, c) unit 8, and d) unit 9 on the Kaibab National Forest, Coconino County, AZ (AGFD).

Red-Naped Sapsucker

The red-naped sapsucker was formerly considered the same species as the yellow-bellied sapsucker (*Sphyrapicus varius*) and red-breasted sapsucker (*Sphyrapicus ruber*). Red-naped sapsuckers range across the Rocky Mountains and interior western states, including the Southwest. Red-breasted sapsuckers occur along the west coast while the yellow-bellied sapsucker ranges in the east of the Rocky Mountains to the Atlantic Ocean. Sapsuckers do not actually suck sap, but are specialized to sip it. The tip of their tongue has small hair-like projections that help collect the sap once they have drilled a well. Sap wells are also an important feeding resource for over 40 other species.

Life History:

Red-naped sapsuckers were selected to represent species using late-seral aspen habitat and the aspen snag component. Where this species is the only abundant woodpecker it is considered a “keystone” species as it provides the majority of cavities for secondary cavity-nesters (Ehrlich et al. 1988, Daily et al. 1993). In Arizona, this species is rare and primarily found in mixed conifer habitats containing aspen stands mixed with maple (*Acer sp.*) during the breeding season. Across its range, red-naped sapsuckers are also found in deciduous and mixed woodlands, including aspen groves in open ponderosa pine forests, aspen-fir parklands, aspen groves in open rangeland, birch groves, montane coniferous forests, and, occasionally, subalpine forest edges and residential gardens from 300 to 3,000 m (approximately 1,000-10,000 ft) elevation (Flanagan 1911, Hadow 1977, Short 1982, Campbell et al. 1990, Winkler et al. 1995). Because willow can be used for sap wells, they often associate with habitats containing or adjacent to willow (Daily et al. 1993, Walters 1996). In Arizona, breeding red-naped sapsuckers prefer deciduous and deciduous/coniferous forests along and north of the Mogollon Rim and in the White Mountains (Phillips et al. 1964, Monson and Phillips 1981). Within these habitats, they favor groups of large aspens near canyon heads at higher elevations (Terres 1996). Typically, they select diverse stands providing suitable diameter trees for nesting, insect diversity, and sap sources (Latta et al. 1999).

Red-naped sapsuckers are short distance migrants and winter from southern California, southern Utah, and New Mexico southward to central Mexico. Wintering habitat is similar to that used during the breeding season, including riparian (USDA Forest Service 1994), various forest and open woodland habitats such as pine-oak fir (Hutto 1992), parks, orchards, and gardens (AOU 1983). Red-naped sapsuckers are year-round residents in some areas of Arizona and New Mexico with suitable habitat and microclimate conditions (Walters et al. 2002).

During the breeding season, home range size varies from 1.67 ha (4.1 ac) in Colorado (Young 1975) to 13.2 ha (32.6 ac) in British Columbia with minimal distance between nest trees reported as 100 m (328 ft, Walters 1996). Aspen is the preferred nest tree species (Crockett and Hadow 1975, Harestad and Keisker 1989, Walters 1996) but red-naped sapsuckers have been found in many other tree species with rot or cavities including larch, pine, fir, birch, and spruce (Bull 1980, Tobalske 1992, McClelland and McClelland 2000). As a cavity nester, red-naped sapsuckers will nest in both live and dead trees, but prefer aspen which is often afflicted with heart rot fungus (Corman and Wise-Gervais 2005). Heart rot decays the center heartwood, facilitating cavity excavation but provides hard outer sapwood for protection from predators. Most often, red-naped sapsuckers use the same tree, but excavate new cavities each year (Li and Martin 1991, Daily 1993, Walters 1996). Because heart rot moves up a tree, these sequential nests also tend to move up the tree (Daily 1993). Snags are used for cavity sites mostly in conifer-dominated forests (McClelland and McClelland 2000), which are typically spruces, pines, or other conifers (Terres 1996). Whether live or dead, the minimum size of trees is 25.4 cm (10 in) on average (Thomas et al. 1979). Larger trees are preferred, possibly because they allow sapsuckers to excavate more cavities up the bole of the tree in successive years (Harestad and Keisker 1989). In central Arizona’s mixed conifer forest, Li and Martin (1991) found red-naped sapsuckers selected aspen exclusively for nesting with trees greater than 15 cm (6 in).

Sapsuckers feed primarily by creating sapwells or holes through the outer bark of a tree that then drip sap from the tree's xylem or phloem (DeGraff et al. 1991). Holes are drilled in parallel completely around a stem or trunk collecting large quantities of sap, which attracts many species of insects (Walters et al. 2002). Sapsuckers then eat the sap and insects using a specialized feathered tongue. Wells are characteristically drilled in conifers such as Rocky Mountain juniper (*Juniperus scopulorum*), Douglas-fir, lodgepole pine, and white spruce (Loose and Anderson 1995). They will also use deciduous trees such as aspen, willow, birch, and black cottonwood once they leaf out (Walters 1996). Sapsuckers supplement their diet by gleaning and fly-catching insects. Insect prey include moths of the forest tent caterpillar, spruce budworm, and other bark and tree insects (DeGraaf et al. 1991). They will also feed on aspen and other buds, fruit, berries, and nuts, at times caching nuts and fruit (Ehrlich et al. 1988).

Because red-naped sapsuckers create such a remarkable food resources, it is not a surprise that they are not very social. The species tends to guard its sapwells from other birds and small mammals (Ehrlich et al. 1988). During the breeding season, pairs tend to avoid each other (Walters 1996), but during migration, small loose flocks may occur (Walters et al. 2002). Red-naped sapsuckers are not the dominant sapsucker, often enduring aggressive confrontations with Williamson's Sapsucker (Young 1975).

Predators of red-naped sapsucker adults include Cooper's hawk (Walters and Miller 2001), sharp-shinned hawk, and northern goshawk (Reynolds and Meslow 1984, Squires 2000). While in the nest, red-naped sapsucker adults, nestlings and eggs are also preyed upon by black bear (*Ursus americanus*; Franzreb and Higgins 1975, Walters and Miller 2001), deer mice (*Peromyscus maniculatus*; Walters and Miller 2001), weasel (*Mustela sp.*; Daily 1993, Walters and Miller 2001.), house wren (Walters and Miller 2001), and gopher snakes (Walters et al. 2002).

Potential Management Impacts:

Unlike the Coconino National Forest where aspen occurs in extensive stands, it is not a distinct habitat type on the KNF. Aspen is a co-dominant species on the NKRD where it grows in stands of ponderosa pine and mixed conifers. Depending on the amount of ponderosa pine in the stand, these are characterized as dry mixed conifer (more pine present) or moist mixed conifer vegetation types. Outside of 2 limited stands on the Tusayan, aspen on the South Zone is found on the Williams Ranger District where it typically occurs as small (typically < 5 ac), scattered stands within the ponderosa pine forest type. Although limited in occurrence, it provides some of the only diversity in the ponderosa pine forest overstory.

Threats to red-naped sapsuckers are largely unknown (Walters et al. 2002) and empirical studies on management treatment effects show varying results by location, making generalization difficult. However, the dependence on aspen for nesting sites in many areas, including northern Arizona, is a concern as this habitat is severely threatened across the west (Dobkin et al. 1995, Latta et al. 1999, Walters et al. 2002, Lynch et al. 2006). Aspen is especially important to red-naped sapsuckers because it is the only upland deciduous tree that grows extensively in the Rocky Mountains (Finch and Ruggiero 1993). The rate of aspen regeneration loss on the KNF over the last five years is likely similar to that for neighboring Coconino National Forest and is estimated at 97% for sites below 7,500 feet elevation, 50% at 7,500-8,500 feet and 25% above

8,500 feet (Fairweather, USDA Forest Service, personal communication). Regeneration in many areas has been reduced or completely eliminated (Muldavin et al. 1999). Elk grazing and browsing appears to be the limiting factor. Much of the older aspen is now dying due to weather and insect interactions (Lynch et al. 2006) or being converted to mixed conifer from lack of natural disturbance agents, mainly fire (Bailey and Whitham 2002). As an early seral species aspen is adapted to overstory loss and replacement, but cannot maintain itself on the landscape if regenerating sprouts are continually consumed by grazing ungulates.

Fire is the crucial disturbance agent for aspen. High and moderate severity fires typically kill aspen and low intensity fire removes decadent individuals from the population (Dahms and Geils 1997). While fire kills, it also regenerates opening up the canopy and removing shade tolerant conifers from the understory. This allows regeneration of aspen by ramets that are stimulated to grow by fire and need the reduced canopy cover and cleared soils to propagate (Patton and Jones 1977, Walters et al. 1982). For the red-naped sapsucker, this process either directly creates snags by outright killing live trees or indirectly by severely weakening trees that then become susceptible to secondary mortality agents. Without fire, shade tolerant conifers easily overtop aspen, closing the canopy and eventually killing even mature trees and eliminating regeneration due to canopy closure. Latta et al. (1999) suggested managing for groups of aspen of different age classes within the greater forest matrix and suggested the use of fire as the most economical way to clear areas of mixed conifer for aspen regeneration. To be effective the authors suggest moderate to high intensity prescribed fires. This could be affective for red-naped sapsuckers as they will use burned areas if snags are created or are protected, some live hardwood trees remain, and adjacent forest is available for foraging (Bock and Lynch 1970, Tobalske 1992). Therefore, using moderate to low intensity fires that did not completely remove aspen stands but adequately removed shade tolerant conifers could also be affective. As an alternative to fire, Patton and Jones (1977) suggested small patch clearcuts or specified tree species removal (conifers) to stimulate ramet sprouting in areas that have or previously had aspen. Red-naped sapsuckers do use forest edges and logged forests, but extensive clearcuts or the removal of snags and preferred tree species would be detrimental. This indicates that the size of the clearcut and the resulting patch area of aspen could affect red-naped sapsucker presence. Rosenberg and Raphael (1986) showed a significant correlation with stand area and the presence of adjacent hardwoods in Douglas-fir forests and Dieni and Anderson (1999) showed a positive relationship between red-naped sapsucker density and patch size for prescribed burned aspen groves in Montana. However, in an Idaho cottonwood gallery forests, no significant sensitivity to patch size was observed (Saab 1998).

While aspen ramets successfully regenerate after fire, they are then highly susceptible to grazing. In northern Arizona, non-native Rocky Mountain Elk have practically eliminated regeneration of aspen suckers (Bailey and Whitham 2002). Bailey and Whitham (2002) found a positive relationship between aspen and fire severity with more ramets sprouting in high-severity burned areas than areas that experienced lower severity and that elk foraged more heavily in these areas where forage was more available, killing 85% of all ramets. As a result, net biomass was greatest in intermediate burn severity areas suggesting that this burn severity might be more successful for management in Northern Arizona. In addition, deer and livestock can also destroy entire areas of regeneration (Dahms and Geils 1997). National Forests of northern Arizona, including the Kaibab, have resorted to installing very expensive fences capable of excluding not

only cattle, but also deer and elk. Dhams and Geils (1997) suggest that large fires in the mixed conifer could swamp these animal predators if regeneration occurred over a large enough area, but that has not happened to date, despite extensive burn areas.

Harvesting of aspen can also present a management impact for red-naped sapsuckers. Recently, aspen habitat has been increasingly harvested across the Intermountain West for furniture and chopsticks, which directly reduces available habitat (Walters et al. 2002). In addition, because aspen provides a relatively clean burning fuel, snags and nest trees have been cut down for firewood (McClelland 1977, Mills et al. 2000). The KNF Forest Plan allows cutting of trees 31 cm (12 in) or less and red-naped sapsuckers require mature large trees (>25 cm diameter at breast height, 10 in, Thomas et al. 1979) for nesting. The overlap in use can be detrimental to sapsuckers if large (>25 cm) aspen are targeted for harvest in areas where they are limited. Although at this point in time there does not appear to be a significant conflict in resource use, there has been increased interest in a potential aspen market on the KNF. Simultaneously, the older aspen have been declining due to interrelated weather, insect, and disease events.

Population Data:

In Arizona, BBS regional analyses show red-naped sapsucker as significantly declining for two of three time periods (1996-2005 = -14.7, $p = 0.07$, 1980-2005 = -14.8, $p = 0.07$) but the analysis are considered to have significant deficiencies, including low abundance, poor sample size, and poor estimate precision (Fig. 13, Appendix 2). On a larger scale, sapsuckers in the Southern Rockies from 1966 through 2005 show a significant positive trend regionally (1996-2005 = 7.4, $p = 0.0$) with strong data. Grouping the species as cavity nesters, woodland breeders, and sort distance migrants within the Southern Rockies BCR also results in significantly positive trends ranging from 0.78 to 7.0 (Appendix 2).

NatureServe lists this species as G5, N5B, and S5, or demonstrably widespread, abundant, and secure globally, nationally, and within Arizona.

Within Arizona, density of red-naped sapsuckers in Arizona has been reported as 10-20 birds per 40 ha (100 ac) (Yanishevsky and Petring-Rupp 1998). We could find no other resources for Northern Arizona and the KNF landbird surveys only detected one red-naped sapsucker within the mixed conifer forest in 2005 and six in 2006 (1 in mixed conifer and 5 in ponderosa pine forest). While sampling methods used in the KNF Landbird surveys could be appropriate for estimating densities of red-naped sapsucker, aspen is not a targeted habitat type. Therefore, we did not have adequate sample size for density estimation. Current funding does not allow for sampling of every habitat type and other forms of monitoring such as nest searching or spot mapping is likely more appropriate for this species.

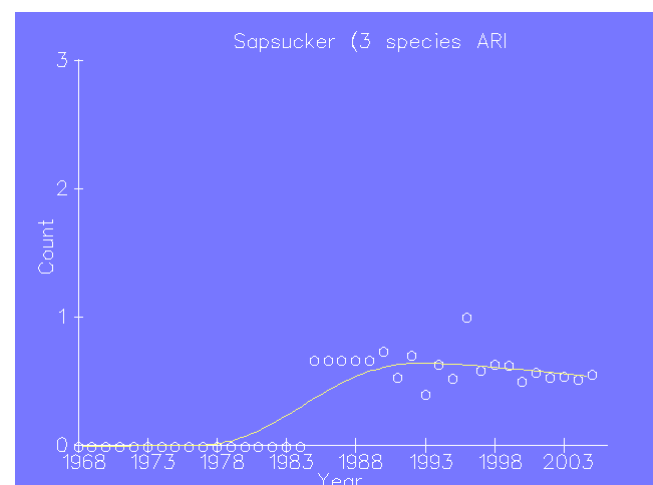


Figure 13. Sapsucker population trend data for Arizona from 1966-2005, BBS data (Sauer et al. 2005).

Regulations within the Forest Plan are very coarse and assessing if they would actually benefit this species is difficult because if applied in one way could be beneficial or if in another way detrimental. For example, the Mexican Spotted Owl guidelines for restricted areas that encourages prescribed and wildland fire with thinning from below could help regenerate aspen patches and aspen snags if applied in mixed conifer where aspens are present. However, the KNF has been advised to work outside of Mexican spotted owl habitat by the local office of the Fish and Wildlife Service. Another potentially positive guideline includes snag retention. Aspen snags 12 inches or greater should be retained in patches to maximize benefits to cavity nesting birds, but this is not currently stated in the Forest Plan. Revising cutting regulations could benefit red-naped sapsuckers by retaining aspen snags on the landscape through time. Lastly, aspen should be a priority habitat to maintain and fence from elk predation because of the alarming loss of regeneration (Lynch et al. 2006, Muldavin et al. 1999, Bailey and Whitham 2002). While treatment of aspen habitat is mentioned in the current Forest Plan, the specifics should be reviewed to assess impacts to this MIS.

Trend Estimate:

Considering the information above, red-naped sapsuckers are assumed to be stable to increasing on the KNF. However, the loss of aspen regeneration may change this to stable (on the North Kaibab where elk do not typically occur) to decreasing (on the South Zone) in the near future. Allowing fires to burn in areas with aspen and limiting elk browsing impacts should secure red-naped sapsuckers across the forest. Attempts to restore aspen within the historic range of variation should benefit this species. However, long term trends of this species will best be determined by continued monitoring on the forest for substantial time periods.

Yellow-Breasted Chat

Yellow-breasted chats are the giants of the warbler world, weighing in at 10 grams (0.35 ounces) and reaching a length of 7.5 inches and a wingspan of 9.75 inches. This is significantly larger than the smallest member of its family, the Lucy's warbler that is only 7 grams (0.25 ounces), 4.2 inches in length, and has a wingspan of 7 inches. Although smallest, Lucy's is more typical of warbler size. Yellow-breasted chats do not only deviate in size but deviate from monogamy, the more common mating system for warblers. DNA fingerprinting has revealed that nearly a quarter of nestlings are not sired by the male attending the nest.

Life History:

Yellow-breasted chats (*Icteria virens*) were selected to represent species using late-seral, low-elevation, riparian habitat. However, this is not the habitat that the species prefers to breed in. Yellow-breasted chats prefer early seral, shrubby thickets that are comprised of low, dense vegetation with sparse canopy cover (Eckerle & Thompson 2001). This habitat type can be found along forest edges, the margins of riparian or wetland habitat, regenerating burned areas, partially clearcut forests, and fencerows and thickets on abandoned farmland. Because shrubby vegetation occurs in many habitats, the yellow-breasted chat can be found over much of the United States. Breeding habitat is extensive across the eastern United States and so too is the chat's range. However, in the western United States the habitat and bird are more patchily distributed. In the arid west, chats are mainly confined to riparian and shrubby habitats, but can use many forms of such habitat as it is considered more of a generalist than its other riparian bird counterparts (Brown and Trosset 1989). Nesting occurs from near sea level to about 1370 m

(4495 ft) in the Lower Sonoran Life Zone, through the Upper Sonoran Life Zone, and rarely in the White Mountains to 2050 m (6725 ft, Small 1994). Within the Lower Sonoran Life Zone, the chat prefers dense mesquite and willow associations along rivers and ponds (Phillips et al. 1964). Along major rivers, including the Pecos and Colorado, yellow-breasted chats are increasingly using tamarisk relative to native cottonwood-willow, a trend reflected by greater abundance of chats in tamarisk (Hunter et al. 1988, Rosenberg et al. 1991). In Arizona, yellow-breasted chats occur primarily below the Mogollon Rim and in the southeastern corner of the state in cottonwood-willow associations with a dense understory of mesquite and tamarisk along major rivers and ponds (Eckerle & Thompson 2001, Corman and Wise-Gervais 2005).

The yellow-breasted chat sticks to its habitat for most of its life cycle, including while on migration and on its wintering ground. For example, shrubby habitat along the San Pedro River serves as a migration corridor (Skagen et al. 1998). The chat overwinters in Mexico and Central America (Eckerle & Thompson 2001) using habitat similar to that in its breeding range including shrub-steppe, with dense, low cover of woody vegetation (Rappole et al. 1995), savanna or pasture with scattered clumps of trees (Rappole and Warner 1980, Rappole et al. 1998), old pasture, where canopy height was 1–3 m (approximately 3–10 ft, Saab and Petit 1992), and pine-savanna that had dense patches of shrubs and that experienced recurrent fires (Petit et al. 1992). Overall, this suggests that this species selects habitat based on structure (James 1971, Whitmore 1977).

Within its habitat, density of birds is generally positively related to the density of shrubs (Crawford et al. 1981, Connor et al. 1983). This suggests that territory size would likely be a function of bird density and habitat quality. In a low-density population, territory size was 1.2 ha (2.9 ac) on average and aggressive interactions between males were low (Thompson and Nolan 1973). In another area, Thompson and Nolan (1973) reported mean territory size as 1.24 ha (3 ac) but that territories shrank as more males arrived on the breeding grounds. With increasing male density, territorial disputes increase. In contrast, in a high-density population where territory size was 0.75 ha (1.9 ac) on average, aggressive interactions were common (Dennis 1958).

Yellow-breasted chats place their nests in dense bushes, brier tangles, vines, and low trees less than 2 m (approximately 6 ft) above ground (Eckerle and Thompson 2001). Chats tend to place their nest in larger patches of suitable vegetation surrounded by larger numbers of woody stems than the average found within the overall habitat (Burhans and Thompson 1999). This apparently protects them from predators as nests in larger patches were less likely to be depredated. In the arid west, yellow-breasted chats build cup nests in dense, brushy, low lying trees and shrubs, including Arizona alder (*Alnus oblongifolia*), Arizona ash (*Fraxinus velutina*), Russian olive (*Elaeagnus angustifolia*), Siberian elm (*Ulmus pumila*), box-elder (*Acer negundo*), Goodding's willow (*Salix gooddingii*), coyote willow (*S. exigua*), blue-stem willow (*S. irrorata*), seep willow (*Baccharis salicifolia*), canyon grape (*Vitis arizonica*), Virginia creeper (*Parthenocissus quinquefolia*), net-leafed hackberry (*Celtis reticulata*), 3-leaf sumac (*Rhus trilobata*), and New Mexico Forestiera (*Forestiera neomexicana*) (Ricketts and Kus 2000). In early successional shrubby habitats where chats were more abundant, the preferred nesting substrates appear to be seepwillow, coyote willow, and canyon grape (Ricketts and Kus 2000).

The primary foods of yellow-breasted chats are insects and fruits of which they eat about equally (Ehrlich et al. 1988). Soft-bodied grasshoppers, larval moths, and butterflies gleaned from foliage are typically fed to nestlings (Petrides 1938). Following the breeding cycle, chats feed mostly on small fruits, including honeysuckle (*Lonicera spp.*), wild strawberry (*Fragaria virginiana*), blackberry (*Rubus spp.*), mulberry (*Morus spp.*), chokecherry (*Prunus virginiana*), sumac (*Rhus spp.*), and nightshade (*Solanum spp.*; Dunn and Garrett 1997). Outside the breeding season, regional and seasonal differences in food is relatively unstudied. Adult diet includes beetles and weevils (Coleoptera), true bugs, ants, bees and wasps, mayflies, and various caterpillars (Howell 1907, Howell 1932, Sprunt 1954, Oberholser 1974).

The major threat to yellow-breasted chat productivity is parasitism from brown-headed cowbirds and rarely bronzed cowbirds (*Molothrus aeneus*; Ricketts and Kus 2000). The frequency of nest parasitism ranges from 0-91% (Eckerle & Thompson 2001). Along the Gila River in southwestern New Mexico, 32% of chat nests were parasitized (Ricketts and Kus 2000). In contrast, none of the 57 chat nests located in east-central Kentucky was parasitized by cowbirds (Ricketts 1999). Parasitism rates vary by habitat structure, with greater rates when more large stems are adjacent to nests (Burhans & Thompson 1999).

Typical predators of yellow-breasted chats include snakes, blue jays (*Cyanocitta cristata*), and chipmunks (*Tamias sp.*; Thompson and Nolan 1973). Potential nest predators in California included Western scrub-jays (*Aphelocoma californica*), dusky-footed woodrats (*Neotoma fuscipes*), raccoons (*Procyon lotor*), and several species of snakes (Ricketts and Kus 2000). Typically, across its range, the frequency of nest depredation is high. Frequency of nest depredation appears to be highest in the egg-laying and incubation periods, and declines after hatching (Thompson and Nolan 1973). Both predation and parasitism vary with patch size of nesting habitat.

Potential Management Impacts:

Chats use transitional forests that are created in many ways, and multiple management prescriptions can create chat habitat (Eckerle and Thompson 2001). However, because these habitats are short-lived, either management or successional processes that create habitat for chats must be actively continued or encouraged. In contrast, other management activities threaten chat habitat such as development, damming and reduction of riparian areas.

Yellow-breasted chats tend to vacate breeding areas readily but rapid resettlement of experimentally vacated territories has been observed (Thompson and Nolan 1973, Thompson 1977). Thus, chats have the ability to easily colonize new habitat, a necessary behavior for exploiting short-lived habitats. Wherever marginal cropland is abandoned, yellow-breasted chats benefit until canopy closure (Eckerle & Thompson 2001). Similarly, timber harvest strategies that promote the growth of a dense shrub layer in regenerating forest patches are beneficial to chats. Tree removal from power-lines creates a corridor of brushy habitat suitable for chats, a habitat that can be maintained indefinitely (Eckerle & Thompson 2001). In the Missouri Ozarks, Gram et al. (2003) found greater chat abundance in regenerating even-aged clearcuts and uneven-aged selective logging plots than on control plots. Annand & Thompson (1997) also found greatest chat numbers in regenerating southeastern Missouri clear-cuts relative to shelterwood, group selection, and single-tree selection. The key element was patch size of

created habitats, with upper and lower area limits negatively affecting the species (Robinson and Robinson 1999). Additionally, burning or clearing of shrubs after clear-cutting will likely delay colonization by chats (Eckerle & Thompson 2001).

Non-native plant invasion in the west has also appeared to benefit the yellow-breasted chat. Hunter et al. (1988) found that chats will use non-native tamarisk and suggest use may be preferential to native habitat. However, use likely differs due to availability as suggested by Brown and Trosset (1989), who reported chats nest in tamarisk and native shrubs in proportion to the occurrence of the different types of vegetation. Use of Himalaya blackberry (*Rubus discolor*) as breeding habitat has also been noted in California due to its dense brushy structure (Ricketts & Kus 2000). Management efforts to remove these plants from riparian areas should include assessments and commitments on reestablishing native shrubs and coordinate the removal work with the restoration work.

Management activities can affect whether habitat is used by breeding yellow-breasted chats and/or whether it affects nest parasitism by brown headed cow-birds. Riparian alteration, including flood control and river channelization eliminates early successional riparian habitat used for nesting (Ricketts & Kus 2000). This may be especially important in the arid west where water is a limited resource and riparian areas are under heavy pressure by humans, development, and grazing. Grazing often leads to the reduction or disappearance of dense, shrubby areas, in both upland and riparian areas (Eckerle and Thompson 2001). However, the response of birds to grazing is not consistent. Saab et al. (1995) reported mixed affects in Colorado, but moderate-intensity grazing in cottonwood floodplain during late autumn had no significant effect on abundance. However, Sedgwick and Knopf (1987) suggested heavy grazing leads to declining abundance due to habitat reduction. Development also reduces habitat directly, but can also reduce the quality of the habitat because of increases parasitism. Limited breeding bird survey (BBIRD) data suggests a positive correlation between developed land cover and cowbird parasitism (Eckerle and Thompson 2001). Lastly, thinning prescriptions had a slight negative impact on yellow breasted chats by increasing parasitism and nest predation in a pine plantation in Arkansas (Barber et al. 2001).

Very little riparian habitat appropriate for this species exists on the KNF. What does occur on the forest primarily consists of dense, non-native tamarisk and other limited shrubs along Kanab and Sycamore Creeks. This habitat has been documented as being in fair and poor condition and does not provide vegetative structural diversity. This is mainly due to historic livestock grazing which likely occurred during the late 1800s and through much of the 1900s. While grazing no longer occurs in Kanab Creek and only occurs within the Prescott National Forest portion of Sycamore Canyon, the lack of restored habitat along the riparian areas indicates that it will likely not change without active restorative management.

Population Data:

Regional BBS data (Sauer et al. 2005) for Arizona from 1966 through 2005 shows a non-significant positive population trend of 0.9 percent per year ($p = 0.55$, Fig. 14), but for the time period of 1980-2005 the population trend is significantly positive (trend = 2.4, $p = 0.03$). However, this data exhibits several deficiencies, including low abundance (less than 1.0 birds/route), low sample size (less than 14 routes), imprecision (3%-year change would not be

detected over the long term), and possible inconsistency in trend over time (sub-interval trends were significantly different [$P < 0.05$] from each other) (Sauer et al. 1999). At the larger BCR scale, yellow-breasted chats show non-significant negative trends for both the Sierra Madre Occidental and Southern Rockies regions with moderate precision and moderate abundance. Overall, BBS data for this species varies widely depending on how it is analyzed (79% positive, 21% negative trends, $n = 28$), with negative and positive trends for the same grouping but different time periods, and does not present any consistent trends (Appendix 2). Thus, BBS data should therefore be interpreted with extreme caution.

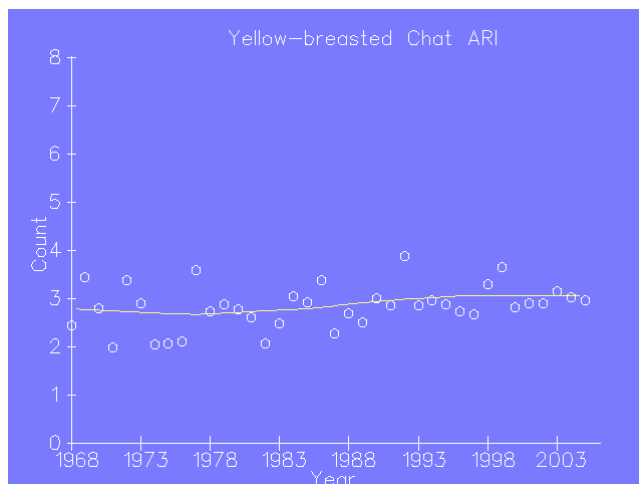


Figure 14. Yellow-breasted chat population trend data for Arizona from 1966-2005, BBS data (Sauer et al. 2005).

Yellow-breasted chats are demonstrably widespread, abundant, and secure globally and nationally and secure statewide according to NatureServe.

Across its range, densities of yellow-breasted chats vary considerably among different habitats for unknown reasons. In California riparian habitat, densities ranged from 6.5 to 26 males per 100 ha (247 ac) over a 7-yr period (Eckerle and Thompson 2001). In Colorado, Sedgwick and Knopf (1987) reported densities ranging from 2.4 to 7.5 males per 100 ha in ungrazed pastures within cottonwood floodplain. In eastern Arizona, yellow-breasted chat densities were 12.5–25 males per 100 ha in introduced salt-cedar habitat and 150–225 in cottonwood-willow (Rosenberg et al. 1991). Additionally, Hunter et al. (1988) reported declines along the lower Colorado River when native habitat is destroyed.

According to the Eckerle and Thompson (2001) range map, yellow-breasted chats do not occur on the Kaibab National Forest. However, range maps are a coarse filter approach to determining presence. Possible chat habitat on the KNF includes the dense, non-native tamarisk and shrubs within the riparian areas along Kanab and Sycamore Creeks. No yellow-breasted chats were detected during surveys by the Arizona Breeding Bird Atlas (ABBA) on lower Kanab Creek, the KNF landbird surveys in upland habitat, and annual surveys of key wetlands on the Williams Ranger District by members of the Northern Arizona Audubon Society. While this does not translate into absolute absence, it is likely that this species is not breeding on the KNF because the habitat is generally lacking and possibly out of the species breeding range.

More important to assessing population trend of this species is the validity of using yellow-breasted chats as a MIS. Because riparian and dense shrubby habitats are so limited on the KNF, it is likely that the Forest does not support a population of yellow-breasted chats. Further, it is more likely that the species does not occur on the KNF at all. If individual birds do occur on the Forest, they would not be easily monitored, violating one criterion for MIS. Yellow-breasted chats are not indicators of other species within riparian habitat or riparian health, as its population is likely more driven by presence of habitat, predation, and cowbird parasitism than current management activities. Only direct measures of this species within the local area would be effective in determining any management effects. Both of these conditions further violate selection criterion for MIS. Because ‘best science’ requires the use of some empirical evidence or at least some survey data, we can not expect this species to perform as an MIS for the KNF.

Trend Estimate:

Considering the information above, the KNF does not support a population of yellow-breasted chats. However, at the state level, it appears that yellow-breasted chats are stable, but that this stability includes populations with both increasing and decreasing trends that may be balancing each other out. Overall, yellow-breasted chats are not a functional MIS because they do not occur on the KNF in numbers that provide any biological or ecological insight, if in fact they occur at all on the forest. Only repeat surveys specifically targeting this species and its potential habitat would be able to document their precise status.

Elk

Elk (*Cervus elaphus*) are sometimes called wapiti, a Shawnee word meaning “white rump,” and are the largest and most phylogenetically advanced species of *Cervus* (Nowak 1999).

Life History:

They were selected to represent big-game use of early-seral ponderosa pine and mixed conifer habitats, but are also an economically and socially important species. In addition to occupying pine forests, they graze grassland and woodland habitats occurring within the forest. Although elk prefer grasses over forbs, they are associated with deciduous thickets and early-seral stages that contain an interspersed of grasses and forbs. They occupy mountain meadows and forests in summer and move to lower-elevation pinyon-juniper woodland, conifer forest, and grasslands in winter where they will browse woody shrubs (Hoffmeister 1986). There is no historic evidence of elk occurring on or near the lands of the KNF (Davis 2001).

Potential Management Impacts:

Current forest conditions provide year-round habitat for elk. Under current management direction, elk will likely continue expanding their numbers. Tree harvest under the goshawk guidelines, which results in a mosaic of interspersed vegetative structural stages and openings, will increase the quality of elk habitat. Wildland Urban Interface projects that strive to open forest canopies to reduce fire threat to private land adjacent to National lands will likely increase elk forage. Taken together, forest management will likely increase potential carrying capacity for elk, thereby increasing pressure on palatable shrub species and aspen ramets. Projects that add or expand existing water sources will contribute to the ability for elk to increase in numbers. Current forest management is expected to exacerbate elk over-utilization of aspen and shrub

species across the South Zone. Additional water developments could increase the risk elk establishment on the Kaibab Plateau.

Population Data:

Elk are common on the South Zone Districts, but only occur intermittently on the NKRD. Management objectives by the AGFD call for no elk on the Kaibab Plateau in order to minimize negative effects on the deer herd there. Elk on the South Zone of the KNF are managed as three functional herds. The herd in GMU 8 has enough interchange with individuals in GMU 6B and Camp Navajo that, in terms of population trends, the three GMUs are tracked as one herd. However, demographic exchange between GMUs 7 and 9 is limited enough that management remains separate for these game units. Elk population estimates are based on a model developed by AGFD that relies, in part, on annual harvest data.

Data compiled by AGFD for the GMUs on and around the KNF show an increase in elk numbers from the late 1980s into the mid- to late-1990s. By then, elk numbers were considered too high and new management objectives were developed in cooperation with the Kaibab and Coconino National Forests to reduce population numbers to about the 1988 level. That baseline was selected because few complaints were received then relative to successive years. This effort did not include Unit 8, which is where most of the aspen on the South Zone occurs. While the reduction effort was successful, numbers continue to increase (Table 9 and Fig. 15).

Elk are considered to be demonstrably widespread, abundant, and secure at the global, national, and statewide levels.

Table 9. Elk population simulation results, by game management unit on the South Zone, Kaibab National Forest, Coconino County, AZ. Harvest numbers come from Arizona Game and Fish Department and include different methodologies (Timothy Holt, AGFD, personal communications 2007).

Year	6B+CN+8	Unit 7	Unit 9	Year	6B+CN+8	Unit 7	Unit 9
1986	NA	2269	NA	1997	3026	3386	2316
1987	NA	2466	NA	1998	3069	3575	2488
1988	2383	2649	NA	1999	3047	3804	2642
1989	2598	2745	NA	2000	2798	3681	2420
1990	2747	2840	NA	2001	2270	2938	1902
1991	3124	3123	1523	2002	3052	2942	1767
1992	3084	3375	1750	2003	3008	3309	2774
1993	3367	3890	1930	2004	3062	3434	3094
1994	3445	3967	2169	2005	2902	3668	2390
1995	3473	3776	2308	2006	2941	3656	2141
1996	3220	3545	2475				

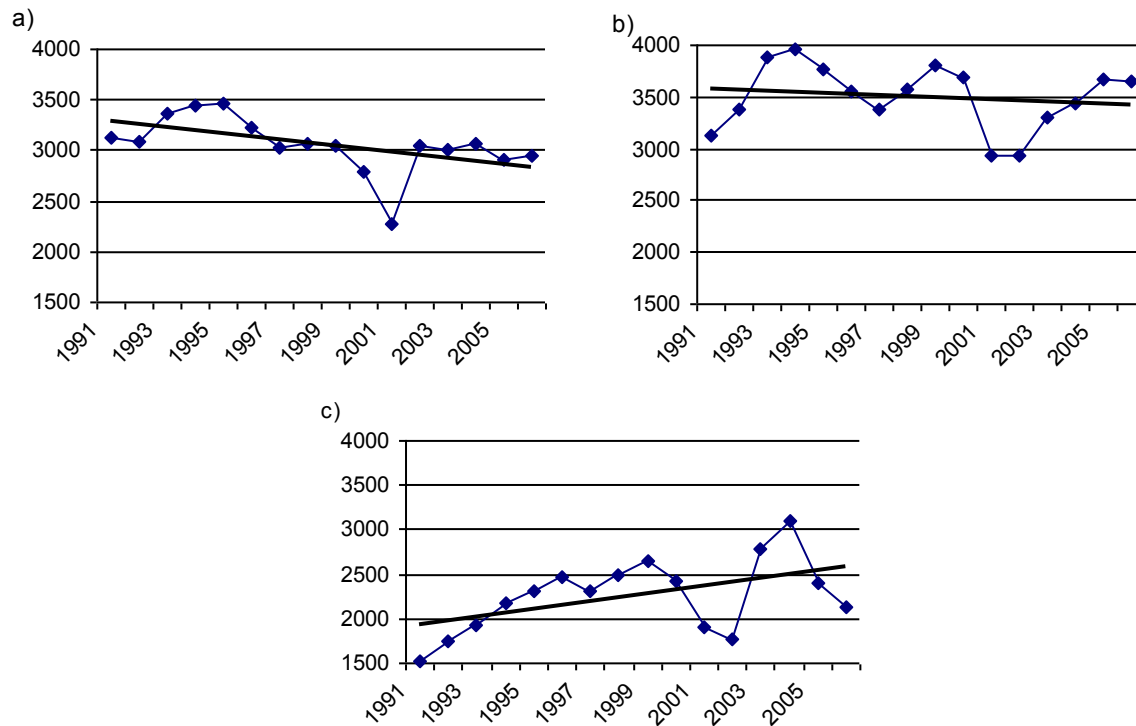


Figure 15. Estimated elk populations by hunt units a) 6B, CN, and 8, b) 7, and c) 9, on the Kaibab National Forest, Coconino County, AZ (AGFD).

Trend Estimate:

The population trend for elk has been stable to positive on the KNF. Overall numbers are such that there is continued debate on appropriate population goals and acceptable levels of resource impacts. Ultimately however, this is an AGFD decision.

Mule Deer

Often referred to as “large charismatic megafauna” the mule deer (*Odocoileus hemionus*) is one of the most sought after species for both hunting and viewing opportunities. They are called mule deer because of their large mule like ears that help radiate heat in the summer months.

Life History:

Mule deer were selected to represent species using early-seral stages of aspen and pinyon-juniper habitats. Mule deer are also an economically and socially important species. They are a generalist species that use ponderosa pine, mixed conifer, woodland, and chaparral habitats. Forage items mostly consist of a variety of woody browse, but they feed more on grasses and forbs during the spring and summer months (Hoffmeister 1986, Nowak 1999). Important plants in a mule deer's diet include mountain-mahogany (*Cercocarpus ledifolius*), buckbrush (*Ceanothus cuneatus*), cliffrose (*Cowania mexicana*), sagebrush (*Artemisia spp.*), buckthorn (*Rhamnus spp.*), juniper, and oak. Home range size varies, depending upon availability of forage and cover. The typical home range size is about two square miles (Hoffmeister 1986). Migrations of any great distances have apparently not been documented.

Mule deer occur across the KNF, but are especially important on the NKRD, much of which is within the boundaries of the Grand Canyon Game Preserve. The North Kaibab deer herd is famous for providing quality hunts and has a long history of management aimed at promoting large numbers of deer.

Potential Management Impacts:

Current forest conditions do not provide optimal cover and foraging conditions. Thinning under the goshawk guidelines, which result in a mosaic of interspersed vegetative structural stages will provide necessary habitat characteristics, such as bedding sites and open areas for foraging and increased forage species. Water maintenance and developments should expand available habitat, particularly on the NKRD (currently new water developments are planned in mule deer winter range to expand use of winter habitat) and Tusayan Ranger District (the Tusayan waters project that is creating up to 24 new stock tanks and developing over 12 miles of pipeline to store and distribute reclaimed waters). Redistribution of deer can benefit both forage availability and deer numbers, depending on deer population objectives. Deer may be negatively affected by elk competition on shared forage species if widespread hedging (e.g., shrubs on the Tusayan Ranger District) or actual elimination of forage occurs (e.g., aspen regeneration on the Williams District). Fire suppression over the last century has led increased tree densities and canopy closure, reducing forest openings, meadows, and grasslands. These changes have reduced both groundcover and the shrub layer, likely decreasing the carrying capacity of lands on the KNF. Forage abundance is also positively affected by application of the goshawk guidelines, grassland restoration efforts, and reducing tree overstory and stem densities in Wildland Urban Interface areas on the KNF. Within the context of the Forest Plan, these changes probably do not yet account for changes in deer population trends across the Forest. However, the continuing drought since about 2001 may be impacting deer on the KNF and across much of Arizona as well.

Population Data:

GMUs for the South Zone display a stable to decreasing trend in mule deer numbers (Fig. 17). The survey data reflects decreasing sightings per hour of survey effort. Although different survey techniques make comparisons between GMUs difficult, trends can be assessed within individual GMUs (Table 10). GMUs 6B, 7, and 8 show decreasing to stable trends. This is consistent with mule deer numbers around Arizona. GMU 9 displays a variable but decreasing trend.

Data from the NKRD indicate an increasing trend since the early 1990s. This is also a GMU with relatively high precipitation rates, relative to the rest of Arizona. Figure 16 displays data from the same model as those used in Table 10, but uses numbers from later in the season. These values include animals harvested and fawn recruitment. Although deer numbers have decreased in recent years on the North Kaibab, this has been due in part to a deliberate effort by AGFD. Research done by AZGFD on the North Kaibab has documented high pregnancy and fawning rates. Collaborative efforts are underway between the AGFD and the KNF to increase functional winter habitat for deer that will likely allow deer numbers to increase in this GMU. In addition, more habitat is expected to come on line as the Warm fire impact area recovers from the 2006 burn that took place on the east side of the Kaibab Plateau. Overall, deer numbers are expected to increase in this GMU.

Mule deer are considered to be demonstrably widespread, abundant, and secure globally, nationally, and statewide.

Trend Estimate:

Mule deer population trends on the KNF vary by Ranger District/GMU. The SZ of the forest appears to be following the Statewide trend of decreasing numbers. Deer on the Kaibab Plateau are variable to increasing and, if not for management, would likely be higher. Overall, mule deer trends on the KNF appear to be stable to increasing.

Table 10. Population trends of mule deer by Game Management Unit, located on the Kaibab National Forest, Coconino County, AZ (AGFD).

Year	6B^a	Unit 7^a	Unit 8^b	Unit 9^a	Unit 12A^c
1983	NA	NA	NA	NA	9,172
1984	NA	NA	NA	NA	11,869
1985	NA	105.8	NA	NA	14,654
1986	NA	NA	8.3	65.8	16,831
1987	63	NA	20	68	15,189
1988	72	51	11	69	14,129
1989	63	NA	12.9	NA	12,796
1990	58.7	102.9	NA	171	11,983
1991	27.4	NA	NA	NA	9,036
1992	62.3	50	16	63.8	9,379
1993	39.4	3	8.4	34.9	9,797
1994	58.8	28.2	11.6	84.8	9,853
1995	36.7	41	7.7	55.8	9,778
1996	NA	36.1	4.2	97.5	10,750
1997	17	16.3	4.4	65.3	10,894
1998	8.3	16.5	5.3	80.7	11,262
1999	4.2	14.3	7.8	47.6	13,151
2000	18.9	50.5	9.3	75.2	12,441
2001	6.2	28.8	6	120	11,524
2002	0	NA	50.	62.6	4813
2003	10.6	58.1	4	33.2	8913
2004	15.3	14.7	3.5	44.2	6694
2005	19.2	17.5	5	30	8144

^a Counts based on mule deer seen per hour from rotary wing surveys.

^b Counts based on mule deer seen per hour from ground surveys.

^c Counts based on population estimates from simulation models.

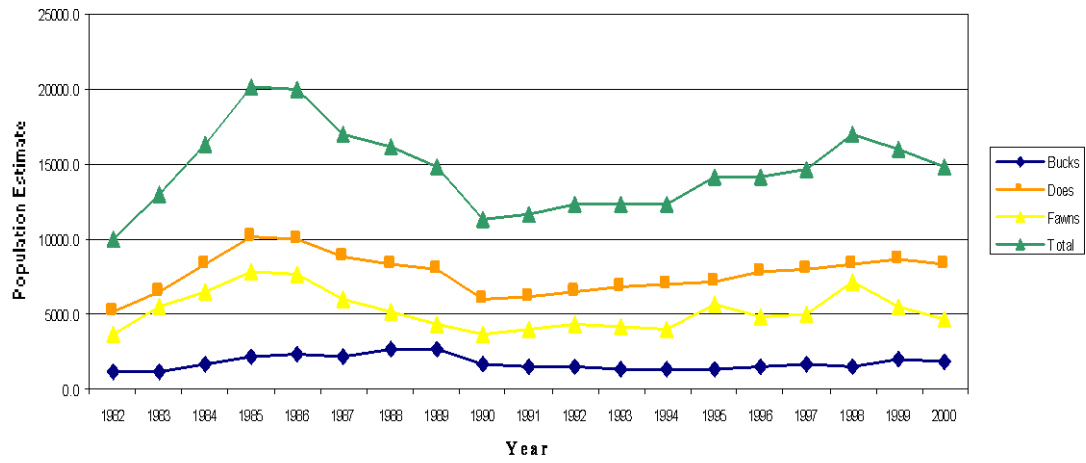
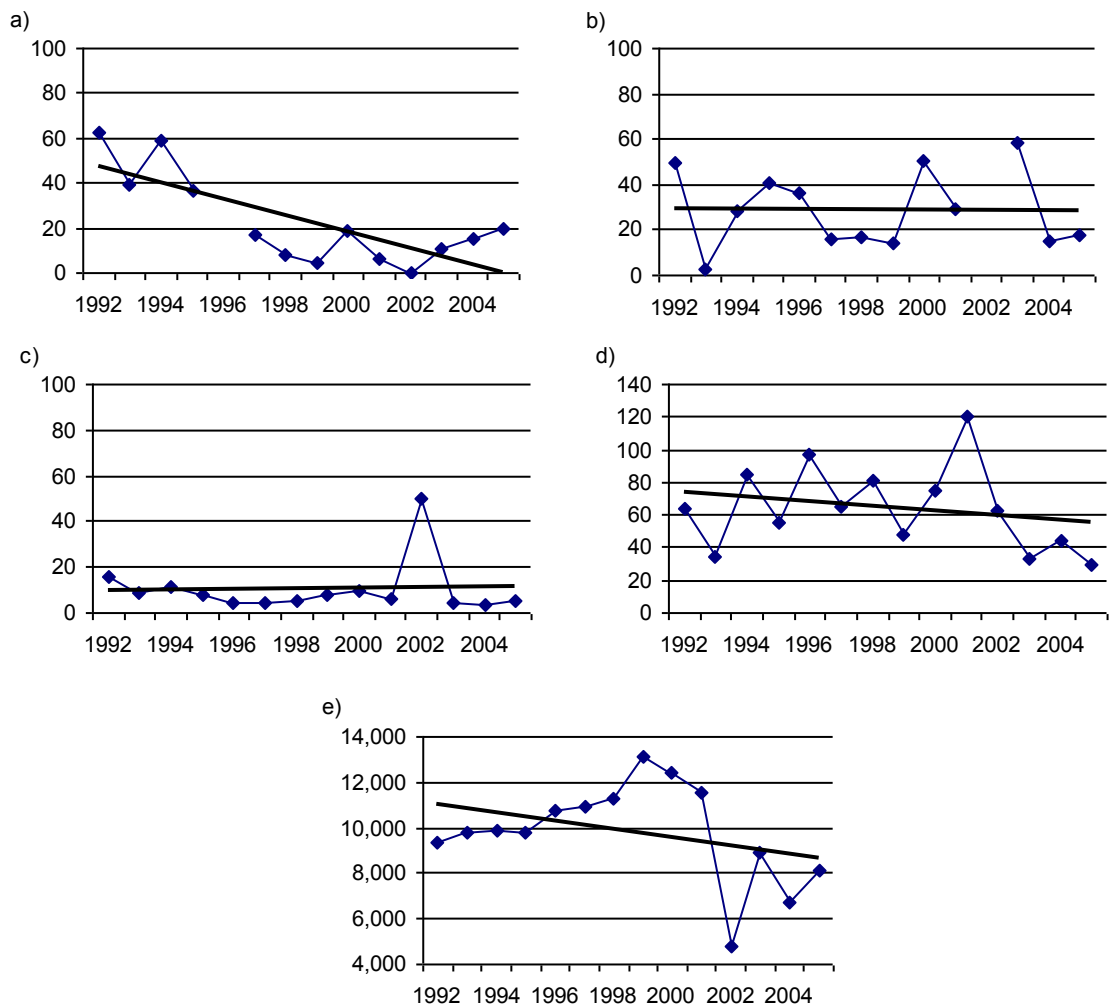


Figure 16. Mule Deer Population Estimates for the North Kaibab Ranger District, Kaibab National Forest, AZ (Buck 2002).



Figures 17. Estimated abundance of mule deer by hunt units a) 6B, b) 7, c) 8, d) 9, and e) 12A on the Kaibab National Forest, Coconino County, AZ (AGFD).

Pronghorn

Pronghorn (*Antilocapra americana*) are the only living genus and species of *Antilocapra*. They historically occurred in open country from Canada to Mexico, but now are limited in distribution due to a variety of habitat changes. The fastest of all North American land mammals, they have been clocked at over fifty miles per hour and at 4 days a fawn can outrun a human (Novak 1999).

Life History:

Pronghorn were selected to represent species using grassland habitats; however they are also an economically and socially important species. Pronghorn are associated with grasslands and savannahs with scattered shrubs and rolling hills. They prefer forbs and grasses as forage but will eat woody browse when forbs and grasses are not available (Hoffmeister 1986). Fawns are typically born in May and rely on good grass cover to escape coyote predation (Hoffmeister 1986). However, there is not a clear correlation between hiding cover and fawn survival (Richard Ockenfels, AZGFD, personal communication). Rangeland with a low vegetative structure, averaging 15-24 in. is considered prime pronghorn habitat (Kindschy et al. 1982). Pronghorn movements vary seasonally. Animals using habitat on the South Zone spend time on different GMUs, including seasonal use south of the KNF. Pronghorn in GMUs 6B and 8 constitute different bands within a single herd. Animals occupying GMUs 7 and 9 also interact as a herd. However, Interstate-40 (I-40) is a barrier between the two South Zone herds. Unit 12A occurs on the NKRD.

Potential Management Impacts:

Historic forest development over the last century has likely negatively affected pronghorn populations. Fire suppression has allowed encroachment of grassland systems by ponderosa and pinyon pines and junipers. Effects from historic grazing, used to assist fire control, resulted in soil loss and decreases in habitat quality. Development of private lands, fence lines, railroads, roads, and highways has fragmented pronghorn habitat. Forest management incorporating the goshawk guidelines, grassland restoration, and fence removal or modification efforts continue to improve habitat conditions for pronghorn. Key to the restoration of meadows is reintroducing fire to the ecosystem. However, the Forest Service management emphasis is on the wildland-urban interface to safeguard people, property, and communities. From the perspective of pronghorn habitat, it is important to find ways to treat more grasslands and savannas with fire despite the limits imposed by funding and staffing.

Population data:

Survey numbers from the AGFD indicate different trends for different herds (Table 11 and Fig. 18). There appears to be an increasing trend for pronghorn in GMU 8, located primarily on the Williams Ranger District. Pronghorn in GMU 6B have been in decline since the mid-1990s. Pronghorn numbers on GMU 7, north of I-40, indicate a slightly decreasing trend since the early to mid-1990s. However, there has been such large annual variation in these GMUs that what appears to be a trend may in fact be a spurious correlation. GMU 9, also north of I-40, has demonstrated a relatively consistent increasing trend since 1985. The GMUs north of I-40 do not have discrete herds. Likewise, pronghorn south of I-40 interact between GMUs. Therefore, pronghorn north and south of I-40 each display areas with an increasing trend and an area with a decreasing trend. Both trends occur within the same herd, further complicating trend descriptions. Pronghorn numbers on the NKRD (GMU 12A) appear to be sustaining an

increasing trend, with animal counts consistently larger in recent years than they were in the late 1980s. However, pronghorn from Utah were released near the NKRD and may have contributed to increases in numbers of animals.

The fawn/doe ratio is a critical aspect of pronghorn ecology (Table 12 and Fig. 19). When births equal mortalities in a given year, there is no change in a population. The AGFD estimates the equilibrium point to be about 25 fawns per 100 does (Goodwin 2002). The fawn/doe ratio varies from 18.4 to 31.1 fawns/100 does on the KNF. The annual variation and relatively low fawn/doe value ratios indicates concern for all three herds.

GMU 6B has the lowest recruitment south of I-40. This GMU lies east of KNF lands, but these animals interact with pronghorn in GMU 8 where the fawn/doe ratio is about 31 fawns per 100 does. South Zone pronghorn north of I-40 are at or below 25 fawns per 100 does.

Table 11. Pronghorns seen per hour on fixed-wing surveys by game management unit, Kaibab National Forest, Coconino County, AZ (AGFD).

<u>Year</u>	<u>South Zone</u>			<u>North Kaibab RD</u>	
	<u>Unit 6B</u>	<u>Unit 7</u>	<u>Unit 8</u>	<u>Unit 9</u>	<u>Unit 12A</u>
1985	NA	16.7	29.7	16.7	NA
1986	6.0	25.9	26.4	18.1	7.5
1987	3.0	28.1	46.0	NA	7.0
1988	30.0	33.3	49.0	18.3	7.0
1989	NA	24.7	NA	11.6	7.0
1990	13.0	33.9	65.7	18.6	10.9
1991	NA	NA	NA	NA	NA
1992	NA	30.6	33.6	18.9	17.2
1993	26.0	31.6	52.4	23.9	4.4
1994	NA	25.7	37.5	17.5	11.4
1995	19.4	27.9	41.5	26.5	13.4
1996	NA	14.3	50.7	32.1	NA
1997	14.3	36.2	23.8	15.5	12.9
1998	7.5	23.2	20.5	24.6	21.0
1999	17.5	29.5	35.9	21.5	10.6
2000	5.3	13.6	35.4	20.3	10.0
2001	9.1	30.1	63.9	23.5	18.3
2002	2.4	8.7	42	14.4	10
2003	8.7	23.7	39.3	20.2	10.2
2004	16.1	17	32.5	16.9	11.4
2005	6.3	17.6	43.3	17.8	12.7
2006	7.8	14.1	24.7	11.4	10

Most of the herd from GMU 8 summers on the Kaibab and winters south of the Forest via a westward movement to highway 89 and then south to the Verde River. It has been hypothesized that these animals may have traditionally moved up towards the Tusayan Ranger District and near the Peaks as a winter migration pattern, but construction of I-40 has forced them to adapt to a new migration pattern (R. Miller, AGFD, personal communication). Whatever the reason, the current movement patterns forces them to cross through forested land rather than more traditional open grasslands. Human development, including roads and fence lines, has had a significant impact on pronghorn throughout their range (Kitchen and O’Gara 1982). Pronghorn reproduction also seems to decrease as a result of drought conditions (R. Miller, AGFD, personal communication).

While still low, the NKRD herd is above the break-even point with an average of about 30 fawns per 100 does. Although the trend appears to be increasing and the ratio is greater than 25 fawns per 100 does, the long-term health of this herd remains a concern.

Pronghorn are considered demonstrably widespread, abundant, and secure globally, nationally, and Statewide.

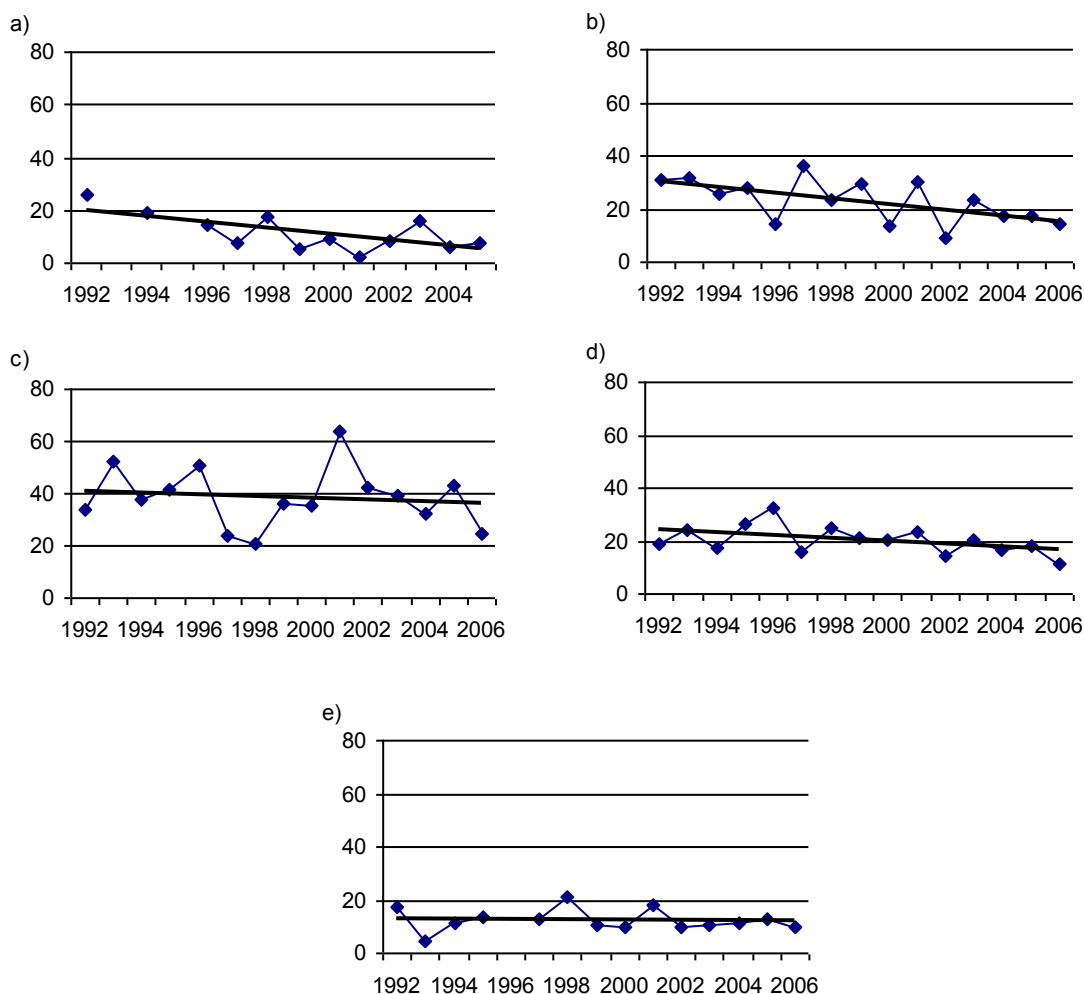


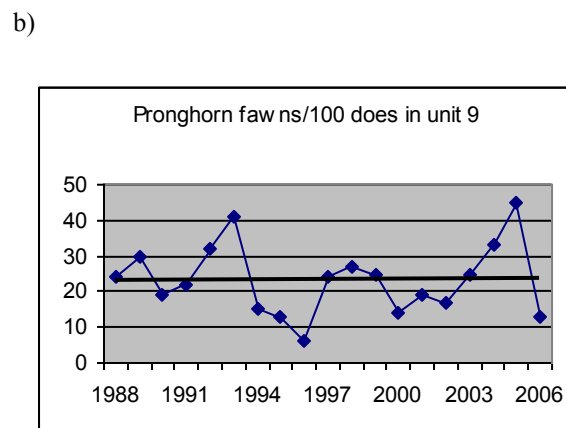
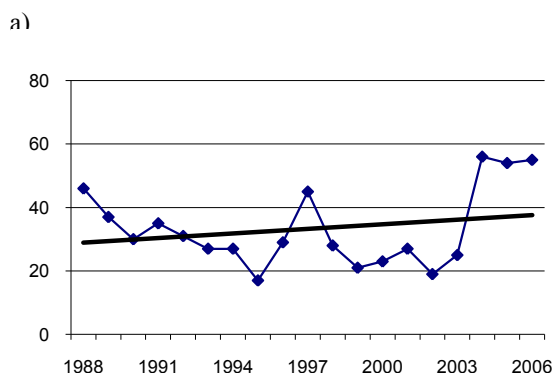
Figure 18. Pronghorn seen by air per hour by hunt unit a) 6B, b) 7, c) 8, d) 9, e) 12A, on the Kaibab National Forest, Coconino County, AZ (AGFD).

Table 12. Pronghorn fawns per 100 does by Hunt Unit on the Kaibab National Forest, Coconino County, AZ (AGFD, 1993, 1998, 2000, 2006).

Year	<u>Coconino NF</u>		<u>South Kaibab</u>		<u>North Kaibab</u>
	Unit 6B	Unit 7	Unit 8	Unit 9	Unit 12
1988	24	37	46	24	42
1989	18	23	37	30	10
1990	0	36	30	19	29
1991	N/A	26	35	22	36
1992	N/A	26	31	32	32
1993	38	28	27	41	17
1994	0	21	27	15	34
1995	13	23	17	13	44
1996	0	7	29	6	14
1997	35	35	45	24	51
1998	28	32	28	27	25
1999	28	24	21	25	24
2000	19	23	23	14	N/A
2001	62	24	27	19	28
2002	50	11	19	17	3
2003	115	48	25	25	10
2004	43	44	56	33	11
2005	27	56	54	45	59
2006	20	24	55	13	11
Average	30.5	28.8	33.3	23.4	26.6

Trend Estimate:

Actual population numbers are difficult to obtain and the scale of variability in numbers may not always reflect real change. The low numbers, inherent variability, and questionable data make identifying trends difficult and the accuracy of the calls uncertain. Fawn recruitment varies from about 23 to 33 fawns per 100 does on the KNF. The annual variation in survey results and the low fawn/doe ratios indicates a need for cautious management in regards to pronghorn habitat. Overall, the forest-wide trend is probably decreasing although numbers on the Williams Ranger District may be increasing.



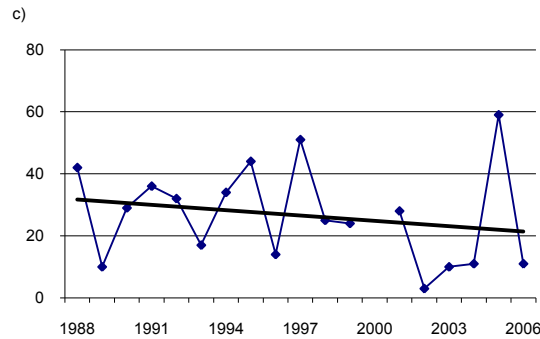


Figure 19. Pronghorn fawns per 100 does on hunt units a) 8, b) 9, and c) 12 on the Kaibab National Forest, Coconino County, AZ (AGFD, 1993, 1998, 2000, and 2006).

Red Squirrel

Red squirrels are easily distinguished from other tree squirrels by the smaller body size, reddish color, a variety of vocalizations and its territorial behavior. When you are scolded by a squirrel while hiking on the KNF, you are listening to a red squirrel defending its space. The species relies heavily upon the middens where they store food. Middens serve as a central larder that is defended against competitors and provides a moist, cool environment that preserves and prevents cones from opening (Steele 1998).

Life History:

Red squirrels were selected to represent species using late-seral mixed conifer habitat. They prefer boreal coniferous forests that provide abundant conifer seeds, fungi, and interlocking canopies for efficient travel, foraging, and escape from predators (Steele 1998). Red squirrels depend on the upper layer of the forest floor associated with tree litter, roots, and mycorrhizal fungi. They focus much of their activity on collecting conifer cones and storing them in middens, which tend to be located in the center of circular territories (Wood et al. 2007). Dense, mature forests are selected for midden and territory locations (Wood et al. 2007). Red squirrels use cavities in trees for nesting and so need large snags or live trees with dead/decaying wood (Hoffmeister, 1986). Douglas-fir trees with dwarf mistletoe witches brooms are also important for red squirrel nesting (Hedwall et al. 2006).

Potential Management Impacts:

Most red squirrel habitat on the KNF is on the NKRD. Mixed conifer forest occurs as high elevation belts on the South Zone and can be found on the highest volcanic cones, including Bill Williams Mountain, Kendrick Peak, and Sitgreaves Mountain. Current forest conditions, which contain extensive areas of closed canopy overstory and abundant snags, create favorable conditions for red squirrels. However, these conditions are not sustainable through time because they are an artifact of fire suppression. The abundance of accumulated fuels, combined with multi-storied fuels that carry ground fire into the canopy, and the high rate of seasonal lightning leaves the forests vulnerable to stand replacement fires. The large tree component, key for producing cones and supporting nest sites, is less able to compete for limited moisture, leaving an additional vulnerability to insects and disease.

Thinning under Forest Plan guidelines will result in a mosaic of interspersed vegetative structural stages, interrupt canopy closure, and allow more sunlight to reach the forest floor. If managed fire is reintroduced into the ecosystem, snags and down logs will be reduced. All these actions will likely cause declines in squirrel populations. Reducing canopy connectedness and opening the understory to direct sunlight will have negative effects on safe travel routes and mycorrhizal production, respectively. These actions will also provide for sustainable forests that include large, cone-bearing trees either as individual legacy trees or in groups and clumps of mature and old-growth trees interspersed with patches suitable for fungi production. Canopy connectivity will be retained, but in small groups rather than across whole landscapes. Currently, red squirrel numbers may be above historic populations due to changes in forest structure over the last century.

Managed fires can create openings and reduce tree density, opening the canopy closure. Both actions decrease some aspects of squirrel habitat effectiveness while simultaneously increasing other elements of their habitat such as improved health of mature trees and overall forest sustainability. When managed fire is reintroduced into the ecosystem, snags and down logs will be reduced in the short term. They do provide immediate snag habitat and aid in replenishing downed woody debris. Snags are also created indirectly when trees weakened by fire eventually succumb to insects and disease. Weakened trees may last for years before becoming snags and many of these processes create longer lasting snags.

Population Data:

Red squirrels are considered demonstrably widespread, abundant, and secure globally, nationally, and statewide.

Count data for red squirrels on the Kaibab Plateau indicate red squirrel numbers have been variable over time, including a sharp increase in numbers in the late 1990s followed by an abrupt decline around 2000 (Fig. 20). Red squirrels rely on mixed conifer habitat. The count data in Figure 20 primarily includes survey results from ponderosa pine habitat and so the scale of change may not be representative of the overall population. However, Salafsky (2002) developed population density estimates specifically for mixed conifer habitat that indicate a decline in numbers of red squirrels on the Kaibab Plateau (Table 13). Red squirrels respond to forage availability and forage is affected by weather. The decline in squirrel numbers may be related to the drought conditions of recent years.

The KNF landbird surveys were modified to incorporate squirrels detections (both audible and visual) and surveying for squirrel sign along the bird transect lines. Squirrel sign includes branchlet clippings, groups of cone cores, piles of cone scales, middens, and peeled twigs. Three years of surveys have been completed, although the survey effort and habitat focus has evolved from a pilot effort in 2005 to a final study design in 2007. The data has not yet been analyzed for trends, given the limited effort to date. Annual surveys following the 2007 field effort are expected to continue into future years.

Trend Estimate:

Since spiking in the late 1990s, red squirrel numbers have declined on the KNF. The apparent effects of precipitation patterns have yet to be verified as cause-and-effect versus being a

spurious correlation. Limited data does not allow assessment of whether the sudden population increase in the 1990s was unusual or how those population numbers compared to previous long-term trends. Given the predicted long-term drought, we expect a decreasing trend to continue for red squirrels on the KNF. Squirrel numbers may also decline with continued implementation of the Forest Plan. However, because the Forest Plan does not call for full restoration and the goal is to manage for more trees than likely occurred historically, squirrel numbers are expected to remain above historic, hence sustainable, levels.

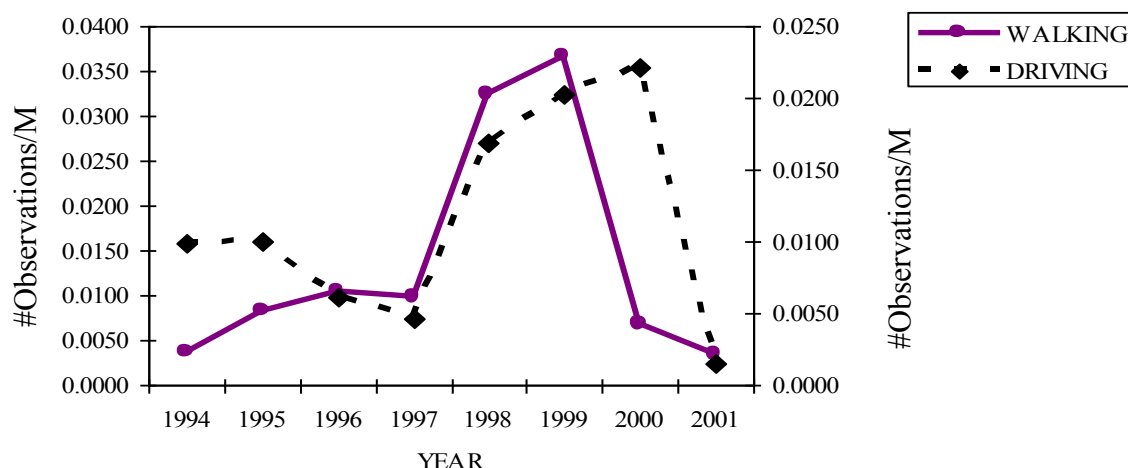


Figure 20. Comparison of walking and driving counts for estimating red squirrel abundance for the Kaibab Plateau, North Kaibab Ranger District, Kaibab National Forest, Coconino National Forest, AZ (Salafsky 2002).

Table 13. Red squirrel population density data for the Kaibab Plateau, North Kaibab Ranger District, Kaibab National Forest, Coconino County, AZ (Salafsky 2002). Density equals the number of red squirrels per hectare.

Year	<u>Mixed Conifer</u>	
	Number	Density
1999	405	0.9957
2000	384	0.9441
2001	27	0.0664
Average Density		0.6687

Tassel-Eared Squirrel

Tassel-eared squirrels include Abert's squirrel (*Sciurus aberti aberti*) south of the Grand Canyon, and the Kaibab squirrel subspecies (*Sciurus aberti kaibabensis*) north of the Grand Canyon. Kaibab squirrels were originally thought to be a unique species endemic to the Kaibab Plateau. They have since been described as a subspecies. Abert's squirrels were first collected near the San Francisco Peaks by Dr. Woodhouse, zoologist on the 1853 Sitgreaves expedition that crossed northern Arizona. They were named in honor of Col. John J. Abert, Chief of the Corps of Topographical Engineers, "to whose exertions science is so much indebted" (Sitgreaves 1853 in Davis 2001).

Life History:

Tassel-eared squirrels were first selected to represent species using mid-seral ponderosa pine habitat. When mid-seral stages were dropped from the first Forest Plan analyses, tassel-eared squirrels were assigned as indicators to early-seral ponderosa pine habitat. Tassel-eared squirrels live, nest, and forage in ponderosa pine forests. Preferred habitat structure is composed of intermediate-aged ponderosa pine forest (20-46 cm dbh, 9-18 in.), intermixed with larger trees, where groups of trees have crowns that are interlocking or are in close proximity (USDA 1994). In contrast, Dodd et al. (1998) demonstrated that thickets of medium-sized trees, with fewer large trees per acre, also create favorable habitat for tassel-eared squirrels.

Nests are typically built in the branches of large ponderosa pine between 5 and 27 m (16-90 ft) high. Other nest sites include cavities in Gambel's oak and in witches' brooms caused by dwarf mistletoe (Nash and Seaman 1977). Nests are made of small pine branches and needles and lined with shredded grass, bark, and pilfered paper or cloth when near human development (Hoffmeister 1986). Nests are often placed on the south-side of trees, presumably to take advantage of the sun's warmth (Hoffmeister 1986).

Tassel-eared squirrels depend on the interspersed habitat types within the Forest to provide arboreal travel routes and food both on the ground and in the trees. Tassel-eared squirrels are strictly diurnal. They frequently forage on the forest floor eating roots, mycorrhizal fungi, carrion, bones, and antlers. They also depend heavily on mature ponderosa pine as a food source the entire year, feeding on the inner bark of twigs, seeds, terminal buds, and staminate flowers (Nash and Seaman 1977). The Kaibab squirrel has been known to cache mushrooms (Hoffmeister 1986).

Tassel-eared squirrels are solitary much of the year (Farentinos 1974). Vocalizations and behavioral stances infer territory boundaries (Hoffmeister 1986). Territories have been estimated to be about 2 ha (5 ac) in winter and 3 ha (7.5 ac) in summer. The Kaibab sub-species appears to have smaller home ranges (4.4 in the summer), but this could be an artifact of small sample size (Hall 1981).

Potential Management Impacts:

Current forest conditions, which contain closed canopies and abundant snags, create favorable conditions for Abert's squirrels. However, these conditions are not sustainable through time because they are an artifact of fire suppression. The abundance of accumulated fuels, combined with multi-storied fuels that carry ground fire into the canopy, and the high rate of lightning strikes during monsoon season leaves the ponderosa pine forests vulnerable to stand replacement events. The large tree component, key for providing cones and sources of cambium feeding, are less able to compete for limited moisture in stands with high tree densities, leaving them vulnerable to insect and disease.

Thinning under Forest Plan guidelines will result in a mosaic of interspersed vegetative structural stages, interrupt canopy closure, and allow more sunlight to reach the forest floor. These actions will likely cause declines in squirrel populations. Reducing canopy connectedness and opening the understory to direct sunlight will have negative effects on safe travel routes and mycorrhizal production. However, these efforts will also provide for sustainable forests that include large, cone-bearing trees either as individual legacy trees or in groups and clumps of mature and old-

growth trees interspersed with patches suitable for fungi production. Canopy connectivity will be retained, but in small groups rather than across whole landscapes.

Managed fires can create openings and reduce tree density, opening canopy closure. Both actions decrease some aspects of squirrel habitat effectiveness while simultaneously increasing other elements of their habitat such as improved health of mature trees and overall forest sustainability. When managed fire is reintroduced into the ecosystem, snags and down logs will be reduced in the short term. They do provide immediate snag habitat and aid in replenishing downed woody debris. Snags are also created indirectly when trees weakened by fire eventually succumb to insects and disease. Weakened trees may last for years before becoming snags and many of these processes create longer lasting snags.

Population Data:

Abert's squirrels are considered demonstrably widespread, abundant, and secure globally, nationally, and statewide. However, the Kaibab subspecies is considered to be vulnerable to extirpation or extinction globally, nationally, and statewide by NatureServe. This ranking appears to be based on the geographical isolation of the sub-species.

Surveys for Kaibab squirrels indicate a fairly stable trend with annual fluctuations on the Kaibab Plateau (Fig. 21 and Table 14). Data compiled by AGFD uses hunter harvest surveys rather than count data for tree squirrels. Results indicate the popularity of hunting squirrels rather than an index of density. Thus interpretation of this data is with serious caveats. The results indicate a decreasing trend for all squirrels from 1988-1999 (Fig. 22). This data includes red squirrels, but the vast majority of the tree squirrels harvested, i.e., most of the data, is from tassel-eared squirrels (Dodd 2002).

Continued implementation of the Forest Plan should create increasingly sustainable forests while retaining the key elements of tassel-eared squirrel habitat. Canopy connectivity will be reduced, but will exist across groups and clumps of mature trees estimated to cover 40% of the landscape post-treatment. Six 30-ac nest stands will be retained per goshawk territory and each nest stand will retain canopy connectivity for mature to old-growth trees. Additional legacy trees will provide food (staminate and seed cones and branchlets for cambium feeding) and stands of dense mid-sized trees (covering 20% of the landscape) will provide mycorrhizal fungi. These habitats, along with openings and early seral stages, will be interspersed, allowing for breaks in the canopy and ground fuels so fire can be reintroduced into the system. Sustainable forests will have a lower carrying capacity versus simply maximizing squirrel habitat, but the management goal is to have sustainable squirrel populations and habitat in this fire dependent ecosystem.

Under the Forest Plan, tree densities will be managed above historic levels and forest patches will retain key elements of squirrel habitat. Therefore, forests are expected to support more squirrels than what had occurred prior to Euro-settlement. In addition, squirrel populations should be maintained through time. Dodd et al. (1998) evaluated and defined tree densities and size-classes relative to tassel-eared squirrel habitat. Their results were compared with surveys conducted on the Kaibab Plateau in 1910 (i.e., historic conditions), and with current forest conditions using Forest Inventory and Analysis data. The number of trees per acre reported by

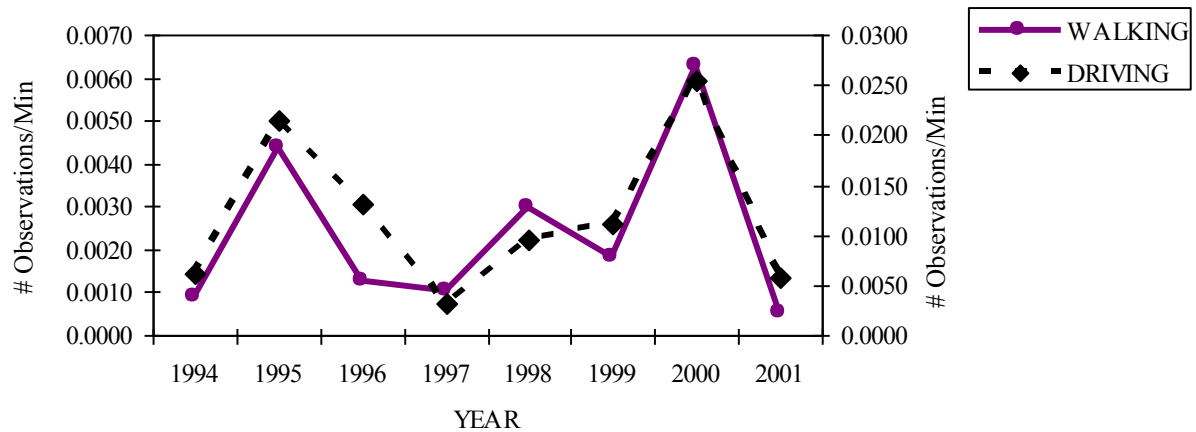


Figure 21. Comparison of walking and driving counts for estimating Kaibab squirrel abundance for the Kaibab Plateau, North Kaibab Ranger District, Kaibab National Forest, Coconino National Forest, AZ (Salafsky 2002).

Table 14. Density estimates for the number of Kaibab squirrels per hectare (Salafsky 2002) on the Kaibab Plateau, Kaibab National Forest, Coconino County, AZ.

Year	<u>Ponderosa Pine</u>	
	Number	Density
1999	19	0.0485
2000	44	0.1123
2001	15	0.0383
Average Density per ha		0.0664

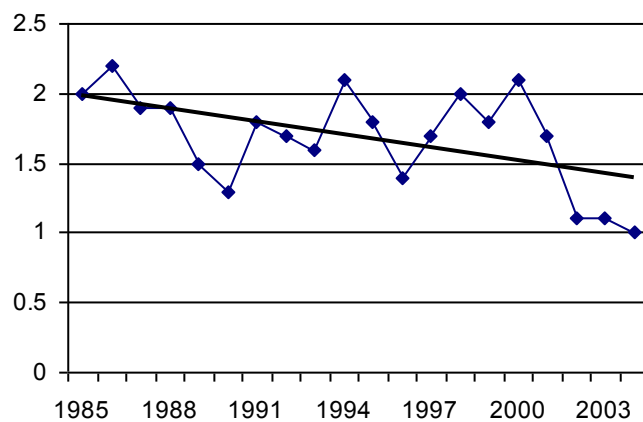


Figure 22. Tree squirrels harvested by hunters in Arizona (AGFD 1993, 1998, 2000, and 2006).

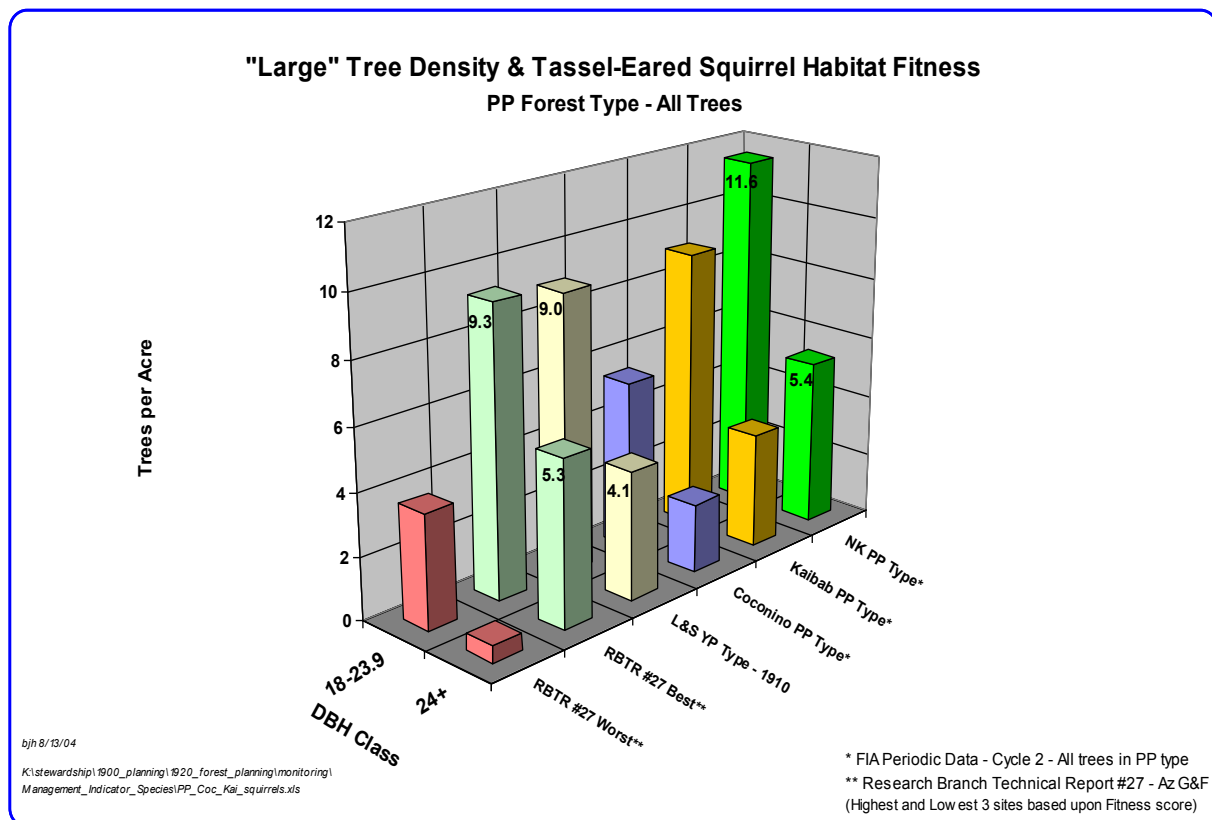


Figure 23. Numbers and density of large ponderosa pine trees as described for tassel-eared squirrels by Dodd et al. 1998 (RBTR #27 Best and Worst), recorded by Lange and Stewart during their 1910 survey of the Kaibab Plateau (L&S YP ["Yellow Pine"] Type – 1910), the Coconino National Forest (where Dodd et al. conducted most of their research), Kaibab National Forest, and North Kaibab Ranger District using Forest Inventory and Analysis data from the U.S. Forest Service.

Dodd et al. (1998) for their best habitats, defined by dbh size-classes, were similar to but greater than those stands from 1910 that had only minimal alteration following Euro-settlement. Current forest structure is similar to the 1910 surveys. Interestingly, current conditions on the NKRD exceed both the historic numbers as well as the description by Dodd et al. (1998) for "best habitat" (Fig. 23).

Trend Estimate:

Although inherently variable, tassel-eared squirrels are currently stable on the KNF. With increased forest treatments that follow Forest Plan guidelines, populations may decline through time. However, it is likely that current populations are artificially inflated due to unnatural forest structure resulting from anthropogenic influences. Any future declines in squirrel numbers are expected to stabilize above historic (pre-Euro settlement) levels.

Arizona Bugbane

Arizona bugbane (*Cimicifuga [Actea] arizonica*) belongs to the Buttercup family (Ranunculaceae) and is a relict from the Miocene. It is an endemic, with 4 disparate populations, including one on Bill Williams Mountain, Williams Ranger District. Nearly all the pollinating is accomplished by bumblebees (order Hymenoptera). They sometimes "buzz" the flower by

grabbing unopened stamens and briefly buzzing their wings. The resultant vibration can cause mature anthers to burst open, providing the bees with pollen (Phillips 2007).

Life History:

Bugbane is found in montane, riparian habitats characterized by Douglas-fir, maples, bracken fern, arboreal lichens, and moist, loamy soil. Canyon walls and cliffs typically provide shade that helps maintain humidity in summer and snow in winter. Typical habitat also has a diverse herbaceous understory and deep duff. It is a perennial, rhizomatous plant with palmately compound leaves. It grows up to 1.8 m (6 ft) and includes a flowering raceme containing small white flowers, 50 – 70 stamens, and long, showy filaments (Phillips 2007).

Population Data:

This plant species was not listed as an MIS in the Forest Plan Environmental Impact Statement. However, it is listed in the Forest Plan for EMA 6, which is a botanical area created for the protection of this species. The USDI Fish and Wildlife Service listed it as a candidate species for threatened status in 1980, and it is still a candidate species. It is also a Forest Service Sensitive species and is classified as Highly Safeguarded by the Arizona Native Plant Law. It occurs on the Coconino and Tonto National Forests.

Arizona Bugbane is listed as G2, N2, and S2 by NatureServe (2001). This means that Arizona bugbane is considered to be imperiled globally, nationally, and Statewide. The ranking is based on the geographical isolation and small population of this species rather than any direct human-caused threats. The Arizona Rare Plant Field Guide (2001) ranks Arizona bugbane as Rare, defined as 6 to 20 occurrences in the state or few individuals or acres within the state.

Arizona bugbane is managed under U.S. Fish and Wildlife Service (USFWS) direction through the Arizona Bugbane Conservation Agreement (USDI FWS et al. 1999). This agreement represents a commitment by the Forest Service and the Fish and Wildlife Service to manage this species to ensure that it does not become threatened or endangered, as stated in the Arizona Bugbane Conservation Assessment and Strategy for the Coconino and Kaibab National Forests (USDA FS 1995).

Trend Estimate:

There is only one population known to occur on the KNF is on. The population extends for approximately one-half mile and generally occupies a strip of habitat measuring 20 to 50 feet wide on the slopes of Bill Williams Mountain (Warren 1991). In 1988, there were approximately 1,150 plants in this population (Galeano-Popp 1988). In 1998, there were at least 1,200 total plants (USDA FS 1998), suggesting a stable population. At this time, there is less concern for the immediate bugbane population than there is for the health of the forest surrounding and supporting the population.

2. Habitat Trends on the Kaibab National Forest

Habitat is fundamental to the existence of all species and is a central unifying theme within the field of ecology. Ecology, derived from the Greek word “oikos”, means the household. Thus habitat can be considered the house for a given population, representing the physical and biological components that determine species presence or absence (Southwood 1977). Habitat can be used to describe requirements at various spatial scales ranging from broad landscape-level characteristics to more site specific locations. Making this distinction is critical to accurately assess habitat use and needs. While vegetation alone does not constitute habitat, it is one of the primary factors that determines habitat use.

INTRODUCTION

In this section we focus on the major habitat types associated with the Management Indicator Species (MIS) described in the first portion of this document. An earlier version of the habitat report (version 1.2) was completed in October of 2003. Building on the data from this previous document, we identify habitat changes that may have occurred since that time. At the time this report was being revised the Kaibab National Forest (KNF) was developing a new Forest Plan and many of the ideas, concepts, and tools used during the forest plan revision process were used to inform this document. A guiding principle of the forest plan revision process is ecological resilience and sustainability. Accordingly, the Forest Service has adopted a new conservation strategy aimed at promoting greater biodiversity through ecological restoration. Central to this concept is the idea that conservation of a variety of vegetation types will benefit plant and animal species diversity (interim directive FSM 2020).

If the goal of ecological restoration is to return an ecosystem to its *historic* trajectory (SER 2004), then management efforts that take a holistic approach involving vegetation, wildlife, disturbance regimes, and ecosystem processes and function should be more successful than those that focus on single species removal and management (Harms and Hiebert 2006). To that end, a primary emphasis of the revision process has been to evaluate existing habitat trends and to future habitat trends across the forest based on each habitat’s potential natural vegetation type (PNVT, described under habitat trend sources below). Use of historical reference conditions, combined with TES survey data (soils topography, climate) as a base from which to formulate management and restoration practices, should be an effective means for fostering better wildlife habitat, biodiversity, and fire management goals (Brown 2000, Egan 2008). This is crucial for scientifically based ecosystem management. This information is integrated with species diversity information across the forest and used to evaluate future desired conditions, while taking into consideration current management policies and predicted long-term changes over time. Many of the habitat evaluations and predictions for future habitat trends used in the revision process were used to supplement habitat trend estimates for MIS.

Background

The 1998 Code of Federal Regulations states that “on the basis of available scientific information, the interdisciplinary team shall investigate the effects of changes in vegetation type,

timber age classes, community composition, rotation age, and year-long suitability of habitat related to habitat of indicator species (36 CFR 219.19).” To date, “available scientific information” that lends itself to the *consistent* analysis of habitat trend data has been difficult to assess for several reasons. The KNF lacks funds to address habitat monitoring specific to all MIS. Instead, the KNF relies on habitat data from a variety of sources (both internal and external), many of which use differing methodologies. Existing Forest habitat surveys do not focus on collecting species specific habitat data but are rather opportunistic in nature, e.g., stand exams. Understanding habitat use is paramount to understanding species population trends and demographics. However, it is difficult to capture this type of information without species-directed habitat monitoring. A tenet of the new forest plan is that a diversity of vegetation types within a range of “desired conditions” should benefit species diversity. However, measuring habitat and vegetation cover types alone are weak proxies for species abundance and are of limited utility when managing a variety of species across the landscape (Cushman et al. 2008). Because the relevant importance of habitat components varies by both spatial and temporal scale, it becomes necessary to address fine-scale habitat features, in addition to broader features, to accurately assess habitat use and needs (Hobbs et al. 2003). Species-directed monitoring, such as the KNF land bird surveys, are an effective means for detecting population level changes for some species. However, species directed monitoring alone does not explain causal pathways and the mechanisms associated with change. Forest and wildlife management programs that combine species-directed monitoring with vegetation surveys and analyses can begin to explain habitat related change if the right questions are asked and surveys are properly designed and implemented. If ecosustainability is a guiding force behind the new forest plan, then implementation of habitat surveys that follow standard habitat-specific methods focused on understanding plant-animal relationships (see methods of James 1971, Dueser and Shugart 1978,) would greatly enhance future forest management for the benefit of wildlife and plant species. General forest classifications (i.e. stand exams) alone do not address the nuances of fine scale habitat heterogeneity that are unique to a species niche (James et al. 1984, Niemi et al. 1997). Nonetheless, conclusions about habitat and current forest management practices can be drawn from existing forest stand level data and, when supported by local and regional data sources, it is possible to estimate trends. This information can be used to supplement data collected at much broader scales such as is collected by national and regional inventory and monitoring programs. Although more coarse in resolution, when data collected at regional levels are collected systematically and standardized across both spatial and temporal scales it allows for more consistent comparison of data sets and evaluation of long-term trends over time. Specific data assessment sources used in the following analyses are discussed in more detail below.

Changes in Management Direction

Timber

Since its inception in 1987, there have been two primary amendments that have significantly changed timber management practices on the KNF such that they affect wildlife habitat. Jointly, these amendments have helped to facilitate changes that promote better habitat conditions for a variety of species while also setting a trajectory for improved forest health over time. This new direction is fundamental to achieving the long-term goals of an ecologically sustainable and biologically diverse forest, particularly in light of recent changes regarding fire severity, climate and drought in the Southwest.

Amendment 2

A 1990 amendment to the 1987 Forest Plan called for 15% of the suitable timber base to be reserved as blocks of old-growth. Old-growth promotes biodiversity and provides specialized habitat for certain vertebrate and plant species. It increases within-stand genetic diversity and provides downed woody debris and snags, important habitat components for some wildlife (Kaufman 1992, Abella 2008). Old-growth forests have been particularly affected by anthropogenic induced changes. Wildfire, competition with younger trees, drought and bark beetle infestations currently threaten old-growth throughout the Southwest (Kolb et al. 2007, Abella 2008).

Old-growth is a relative term with definitions varying among forest type and by spatial and temporal scale. For example large trees are frequently perceived as “old,” but size may vary depending on species and site specific growing conditions, i.e., young trees may be large and old trees may be small. Old growth ponderosa pine forests are commonly described as a mosaic of uneven-aged stands and smaller even-aged groups (Kaufman et al. 2007). On the KNF, old-growth is defined as sites dominated by trees greater than 120 years old and greater than 18 inches DBH.

Amendment 3

The 1987 Forest Plan describes even-aged harvest strategies for timber management on suitable timberlands. Suitable timberland is defined as the balance of lands supporting coniferous forest after subtracting out those lands withdrawn from timber management (e.g., wilderness areas, recreation sites, and Research Natural Areas) and lands either not capable or not suitable for timber management. The suitable timberlands on the KNF total about 493,000 acres. Under the 1987 Forest Plan, typical cutting units were 30-50 acres in size. Areas on the Forest were set aside as old-growth reserves and there were no constraints for operating within northern goshawk foraging areas. Northern goshawks were just emerging as a Regional issue around the time the Kaibab Forest Plan was completed. The Forest Plan was officially updated in 1996, in part, to incorporate changing societal values. The actual changes in forest management, including group selection on suitable timberlands, were beginning to be implemented on the KNF by 1992. Using the newly completed goshawk management recommendations (Reynolds et. al. 1992), group selection for site regeneration and stocking control was applied to areas of 0.5-4 acres in size. Rather than creating even-aged stands across the landscape, the desired future condition was a mix of varying age-classes (Figure 24).

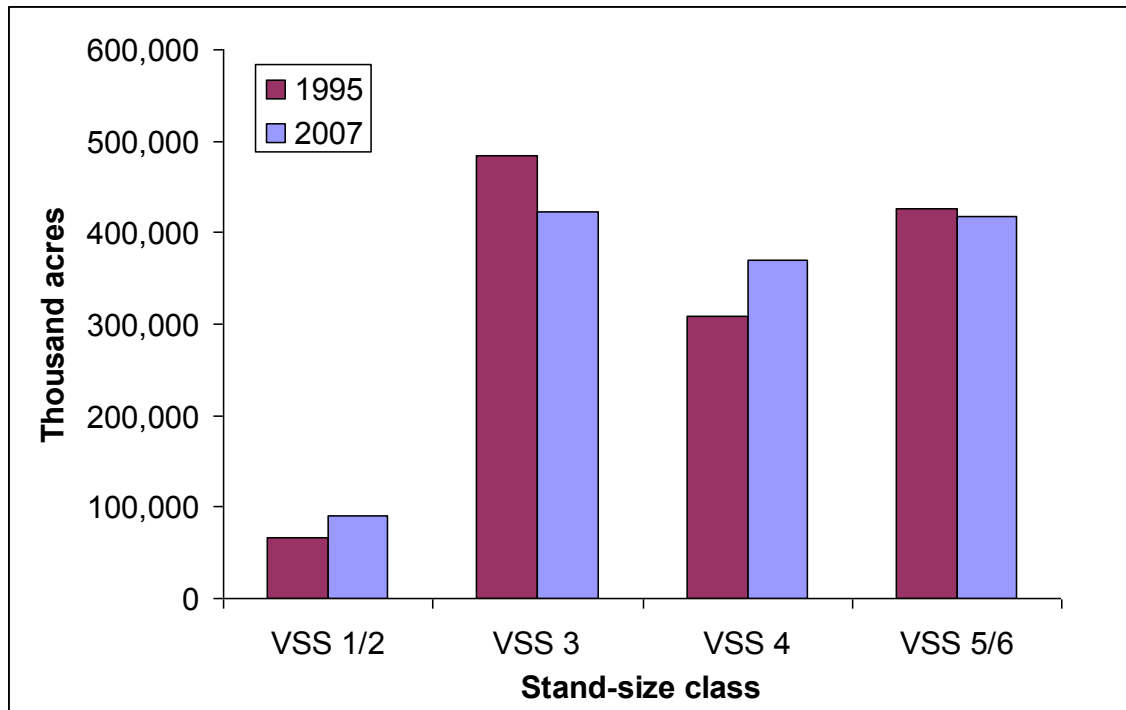


Figure 24. Acres of forest by VSS class across the KNF (FIA, unpublished data).

In 1996, acreage occupied by old-growth was increased from 15-20% of the suitable timber base. Old-growth structure now occurs within most forested areas over 10 acres in size. An additional 20% of the Forest is to be dominated by trees greater than 200 years old and greater than 24 inches DBH. This old-growth structure will occur throughout the forest, but areas within goshawk post-fledging family areas (PFAs) will be managed with higher canopy densities than areas outside of designated PFAs. In total, the 1996 Forest Plan amendment calls for 40% of the Forest to be managed for old-growth conditions. However, the old-growth is not delineated in defined polygons or “set asides.” The heterogeneous approach applies at the patch or group scale. The old-growth conditions will be tracked as a percentage of the area; the precise location of these conditions will move over time as trees grow, die, or are harvested. In regenerated patches greater than one acre, the amended guidelines call for retaining 3-6 large, residual trees. In actual practice, 3-6 residual trees have also been left in some regeneration harvest patches less than an acre in size. With the addition of 3-6 large, old trees per acre in perhaps half of the remaining area, the entire forest will have components of old growth structure present within any 5-10 acre area.

The management plan still results in commercial harvest of trees greater than 18 inches DBH, particularly where existing numbers of large trees exceed desired conditions. Desired conditions are based upon tree size and canopy density described in the goshawk management recommendations (Reynolds et al. 1992). Harvest will occur at the 1-4 acre scale with the objective of providing timber while enhancing forest health and growth. For example, on the Kaibab Plateau there is more ponderosa pines in the 18-24 inch DBH size-class than there were historically (Figure 25). The number of trees greater than 18 inches per acre is expected to increase over time. The intent of management is to produce more trees over 24 inches and more

trees over 18 inches than existed before logging began on the Forest. Uneven-aged management at this scale will eventually lead to less structural variation between stands while increasing structural variation within stands, including an increase in vertical diversity of forest structure. About 40% of any one area will be dominated by large (greater than or equal to 18 inches DBH), old (greater than or equal to 120 years) trees. Eventually, referring to individual “stands” should no longer be applicable in the KNF.

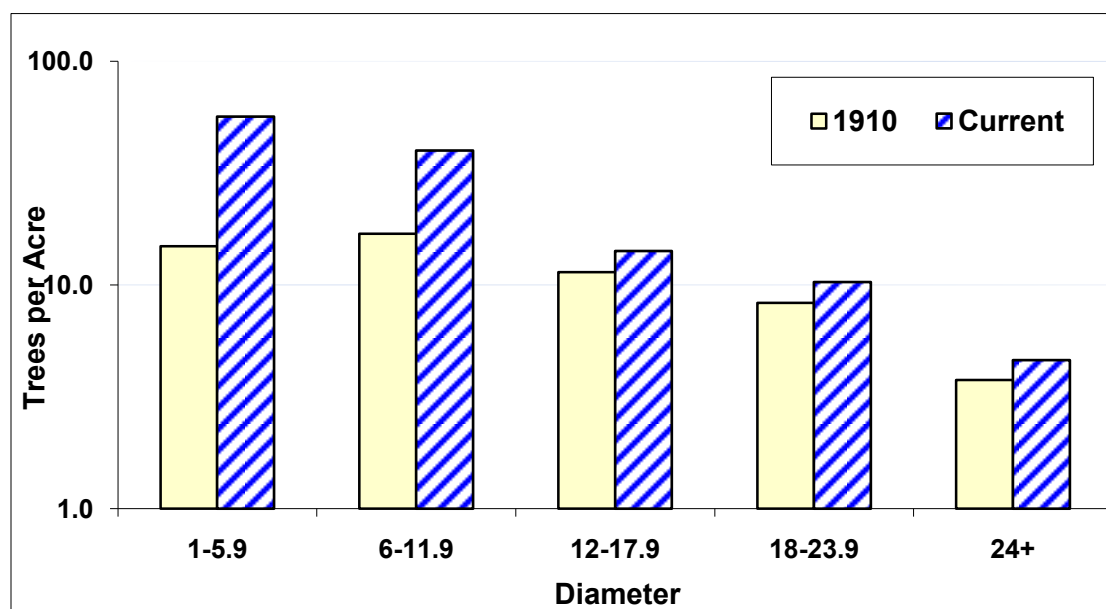


Figure 25. Ponderosa Pine Trees per Acre by Diameter in Historic & Contemporary Ponderosa Pine Forest North Kaibab (Adapted from FIA data and Lang and Stewart 1910; North Kaibab).

The current amended Forest Plan is expected to result in a constant recruitment of seedlings over time. Planned entries within an assessment area for timber harvest are expected to occur at 20-year or greater intervals. Exceptions would be prescribed burning, small-scale operations such as planting, or addressing outbreaks of insects or disease.

The change in management direction per the 1996 amended Forest Plan was intended to shift into a sustained-yield approach for long-term renewable resources applied at the ecosystem scale. Because management activities are intricately tied to wildlife habitat conditions, alterations of forest structure and composition can have significant impacts on wildlife. For example, habitat conditions that support a wide variety of predators and prey can be improved by increasing old growth forest components (large trees, snags and downed logs), managing for groups of trees that contain smaller patches of mixed ages, restoration of a diverse, well developed understory, and reduction of fuel loads and tree densities that may contribute to uncharacteristic wildfires and insect epidemics. Many of these conditions can be achieved through uneven-aged timber management practices that retain larger trees, logs and snags, prescribed burning, understory thinning, and practices that promote natural soil regimes (Reynolds et al. 1996). It follows then that the current focus in terms of wildlife habitat is to provide for a diversity of vegetation structural stages by including habitat goals in the design of all vegetation treatments. Reducing tree densities should increase the area occupied by

understory plants, i.e., grasses, forbs and shrubs. The Forest Plan is now based more on forest ecology rather than timber management. Harvest objectives have become more ecologically sustainable by recreating patterns of natural forest patch dynamics. Reynolds et al. (1996) state “native plants and animals of Southwestern forests were adapted to environmental conditions that existed before the beginning of intensive forest management. Management prescriptions that restore the composition, structure and spatial patterns of these forests should improve the habitats of native species.”

The average treatment now cuts about 25% of the total board feet per acre than was cut under the original Forest Plan. Total acres treated per year have decreased with the implementation of these guidelines. Although the trend in forest development resulting from the last century of management has been reversed, current management does not seek to return to pre-European settlement conditions. However, managing for this level of heterogeneity is expected to move the KNF much closer to pre-settlement conditions than today’s forests. A query of the Forest Activities database (FACTS) highlights some of these changes (Table 15).

Table 15. Primary timber harvest treatments for the KNF, 1987-2008.

Treatment	Acres Cut 1987-1991 ¹	Acres Cut 1991-2008 ²
Commercial Thinning	28,392	27,110
Group selection cut	0	2,603
Overstory removal cut	2,454	986
Sanitation/Salvage ³	1,800	5,924
Shelterwood	6,137	1,341
Clearcut	145	87
Total Treatment Acres	38,928	38,051
Average Acres Treated/Year	7,786	2,114

¹Timber harvest under the 1987 Forest Plan

²Timber harvest following the shift to uneven-aged management

³This category relates primarily to uncharacteristic conditions related to wildfire and mistletoe

Restoring the grouped nature of trees and the interspersed nature of small patches of different age classes should also reduce current threats to forest health. By reducing fuel ladders and stand stocking densities, the resulting forest should be more resilient to insect epidemics and catastrophic fire (Reynolds et al. 1996). However, the ability to treat enough acres across the landscape to reduce or avoid ecosystem-scale impacts from fire, insects, and disease has yet to be demonstrated. Pre- and post project surveys from timber sales implemented under the 1992 Amendment have shown that, within the acres treated, more volume has grown during project planning and implementation than was actually harvested. The Lookout Canyon project, for which numbers were readily available, had 6.70 trees per acre greater than 24 inches DBH three years before harvest and 6.78 trees per acre greater than 24 inches DBH three years after the sale. This amounted to nearly 200 more trees greater than 24 inches in the project area.

The shift in management direction has impacted wildlife habitat by the scale of the projects. The average number of acres treated per regeneration area on the KNF peaked in the late 1980s and dropped significantly in the early 1990s. Project planning under the 1987 Forest Plan began in 1988. Even though the Plan was not amended until 1996, uneven-aged prescriptions began in 1992. Most treatments after 1992 were commercial thinning or group selection cuts rather than shelterwood cuts (Figure 26), although some shelterwood cuts that were planned before 1992 were not harvested until later in the 1990s and even 2001. Commercial thinning projects help to open up the canopy and reduce ladder fuels that may lead to crown fires. Clearcutting is only done as a means to mitigate the effects of damaging agents such as insects, disease, fire, and windthrow on the health of uneven-aged stands. The glossary in Appendix 5 lists definitions for some of the more common silvicultural treatments mentioned in this document.

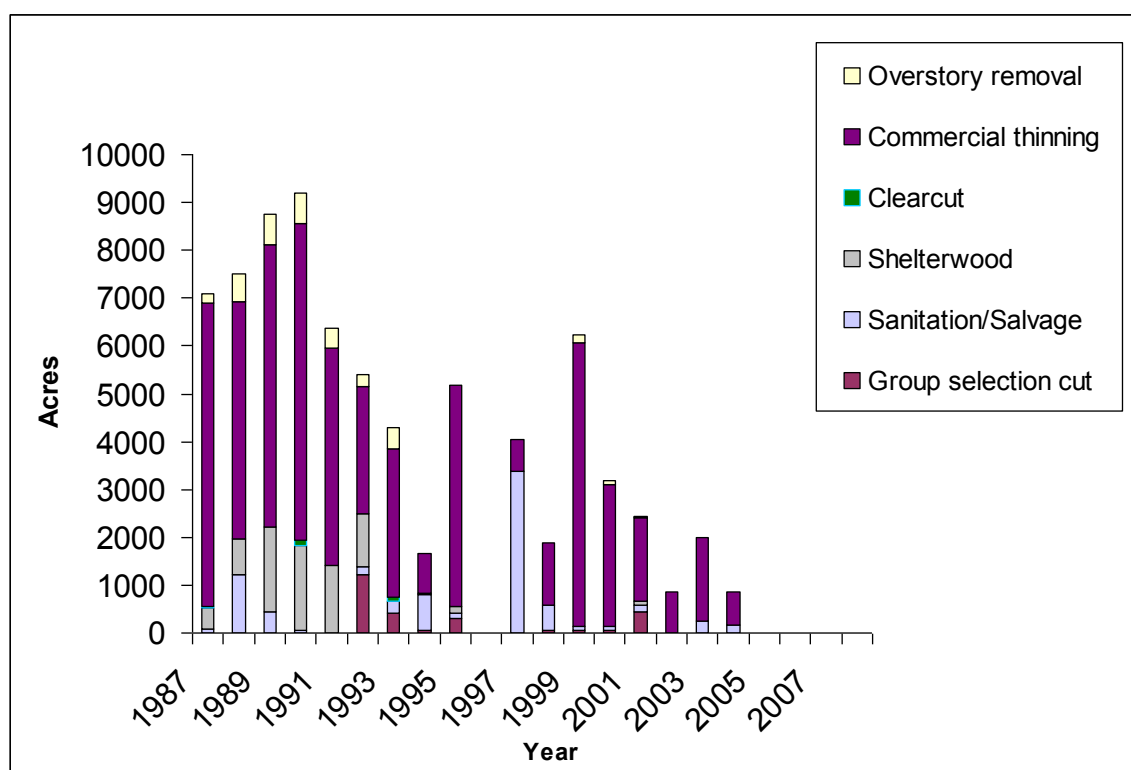


Figure 26. Timber treatments since 1987 (KNF, unpublished data from FACTS).

Fire

In addition to timber management policies, there have been fundamental changes to fire management programs throughout the West. Implementation of the National Fire Plan in 2001 and the Healthy Forests Act of 2003 facilitated the use of prescribed fire and mechanical thinning to reduce fuel loads and decrease hazardous fire conditions. In 1995, the Forest Service adopted a Federal Wildland Fire Policy that stated that “Fire, as a critical natural process, will be integrated into land and resource management plans and activities on a landscape scale, and across agency boundaries.” Implementation of this policy was recently revised at the start of the 2009 fire season. Although initial action on human-caused wildfire continues to be suppression, revised policy will allow for a more flexible approach for managing fire to meet human safety as well management goals to benefit natural resources. This revised policy includes more flexibility

in decision making and moves toward two types of wildland fire: planned ignitions (prescribed fire) and unplanned ignitions (wildfire). These changes will allow fire managers to better manage fire for resource benefit by allowing them to respond more effectively to changing and unpredictable conditions. This new Wildland Fire Decision Support System should improve decision documentation, risk assessment, and operational implementation (National Wildfire Coordinating Group 2009).

Stephens (2005) reported a significant increase in relative areas burned from 1940-2000 in the Southwest. Region 3 experienced the second highest total number of fires in the nation, surpassed only by Region 8 in the Southeast. The southwest in general has a growing trend toward uncharacteristic stand replacement fires. Heavy fuel loads have led to hotter, faster fires. Forest fires will likely continue to increase in intensity and frequency as a result of climate induced changes. Wildlife species dependent on late-seral habitat may be especially hard hit if forest structure remains uncharacteristically dense while climate change exacerbates longer and more severe fire seasons. Late-successional stands comprised of species such as ponderosa pine and Douglas-fir will be more susceptible to mortality, decreasing available habitat for wildlife species which depend on that habitat type. Conversely, species dependent on early-successional habitat should benefit (McKenzie et al. 2004). On the KNF for example, these changes could affect the Northern Goshawk and favored prey such as the red squirrel. Mexican spotted owls could also be affected, particularly on the SZ where dry-mixed coniferous forest and spotted owl PACs (Protected Activity Centers) are present. Although critical habitat for spotted owls is found on the NKRD, the forests there are wetter than on the SZ, seemingly less suitable habitat for spotted owls, (no PACs are currently found there) and thus less likely to be affected by these changes.

According to KNF fire records, the average number of both human and lightning caused wildfires has remained stable over the past 20 years with about 200 fire starts per year (Figure 27). However, there has been a dramatic increase in the total number of acres burned by uncharacteristic wildfire across the Forest, particularly since 1995 (Figure 28). This extreme fire behavior and resulting severity is uncharacteristic and well outside the historic range of variability. For example, the Bridger Knoll Fire in 1996, the Pumpkin Fire and Outlet Fire in 2000, and the Warm Fire in 2006 all burned thousands of acres in a short period. They sustained crown fires that resulted in overstory mortality across large areas, where historically there was frequent low intensity, or mixed-severity fires (Kaibab National Forest 2008a).

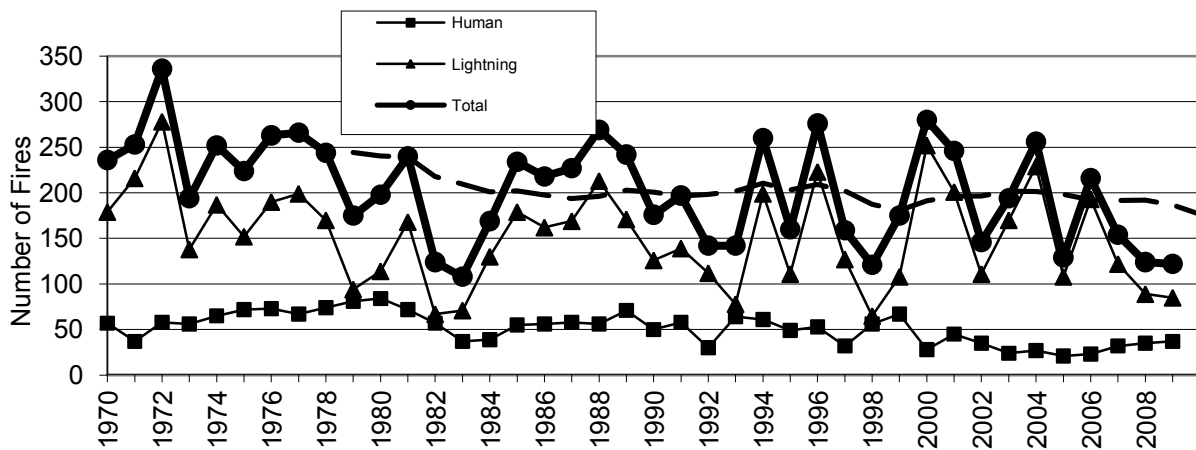


Figure 27. Fire occurrence by cause, total, and 10-year average on the KNF, 1970 - 2009.

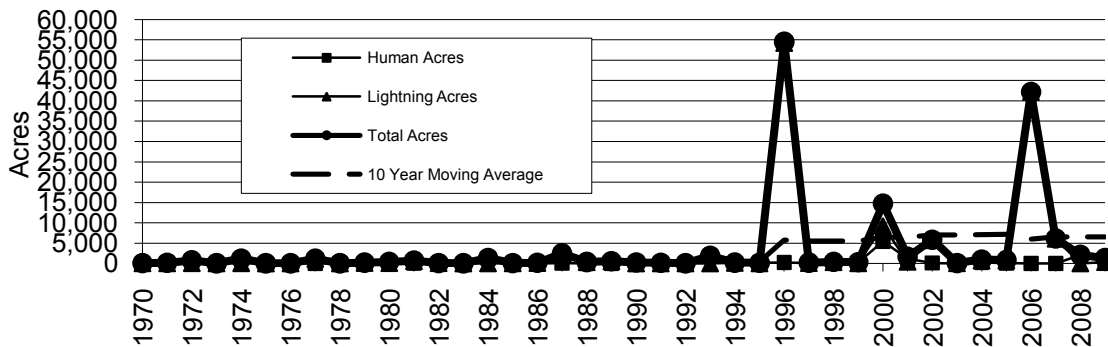


Figure 28. Acres burned with stand replacing wildfire on the KNF (by cause, total, and 10-year average), 1970 - 2009. Does not include acres from wildfires managed for resource benefit.

Climate Change

Many of the most urgent habitat management issues presented in this document have been driven in part by climate change. These include wildfire severity, large-scale bark beetle epidemics, and changing water conditions. Breshears et al. (2005) note that ongoing drought resulting from increasing temperatures and decreasing precipitation has the potential to widely alter ecosystems and cause overstory tree mortality. In a recent review, Periman (2008) found that most current models of climate change are broad and still evolving, making it difficult to assess associated impacts at the forest scale. This distinction is important if locally implemented management strategies are to be effective. In general though, most climate modelers agree that the Southwest

is trending toward prolonged drought. Future potential ecological effects in the Southwest may include an increase in more intense disturbance events such as wildfires, monsoons, and wind. Drought conditions may affect wildlife habitat directly by stressing vegetation and indirectly by exacerbating the effects disease, insects, fire, and low recruitment have on different habitat types. Changing ecological conditions could provide opportunities for invasion by non-native species with potential subsequent negative impacts on various taxa. General trends toward decreased precipitation could limit overall forest productivity. Subsequent changes in vegetation patterns could affect overall distribution and range of flora as well as fauna. Cumulatively, these factors would likely impact biodiversity, however to what extent is currently uncertain (Periman 2008 and references therein). The US Forest Service is currently developing a strategic framework that responds to climate change (USDA 2008a) and it is expected that future iterations of the KNF Land Management Plan will incorporate guidelines to mitigate negative impacts.

Habitat Trend Estimates

Overview

The first section of each habitat account starts with a historical overview, general information on each habitat component, a description of its importance for particular MIS, and how the habitat has changed, in a general sense, from historic times. A discussion on current management practices that may alter conditions/resources follows. The third section reports on both quantitative and qualitative trend data. If available, we used multiple sources that included National Forest survey data as well as data gathered from local researchers and/or Forest-level activities. We consulted the published literature as well as technical reports and gray literature such as theses and project level summaries. The final section of each account is a trend assessment call that was estimated for each habitat type across the entire forest. In most instances, habitat types were assessed to be trending away, toward, or static with regard to historic reference conditions. We also assessed how these trends might impact MIS species.

Habitat Trend Assessment Sources

We made every effort to use data that would be conducive to projecting long-term data trends and that would be replicable over time. In most instances this included information collected as part of regional or national sampling programs. For some MIS habitat types however, this information was not always available or had large sampling errors. In those circumstances, we supplemented with local or best available information. Below are descriptions of the major sources used for assessing habitat and habitat trend calls.

Forest Inventory Analysis

The Forest Service's Forest Inventory and Analysis (FIA) program is a national vegetation sampling program. First established in the 1930s, FIA has evolved from a timber-centric sampling program to one that incorporates multi-resource management issues such as wildlife, rangelands, and forest (LaBau et al. 2007). Primary areas of focus include status and trends of tree growth, mortality, thinning, and forest cover and health. Initial surveys were conducted on a semi-annual basis with annual surveys implemented in 2001. FIA surveys consist of three phases: 1) Aerial images used to stratify sampling regions; 2) Systematic sampling conducted on the ground across every 6000 acres of forest; and 3) Surveys are repeated annually on 10% of the total plots that occur on forested lands in each western state. This includes public and private forest land (Brand 2005). For the KNF, roughly 20-30 plots are sampled each year, with

individual plots sampled once every ten years. Although data collected is at a broad scale it is collected randomly and systematically. This allows for more consistent evaluation of data over time. FIA data presented in this report are from surveys completed in 1995 and 2007. For additional information visit the FIA website: <http://www.fia.fs.fed.us>.

Potential Natural Vegetation Type

“PNVTs represent the vegetation type and characteristics that would occur when natural disturbance regimes and biological processes prevail” (TNC 2006). For the Southwest Region, PNVTs were developed collaboratively by The Nature Conservancy and the US Forest Service’s Mid-Scale Vegetation Mapping Project. The original descriptions can be found in the Nature Conservancy PNVT assessments at www.azconservation.org (TNC 2006). During the forest plan revision process, an evaluation of the vegetative conditions of the KNF and the surrounding area was completed using these PNVTs (Appendix 6). This evaluation allowed for comparisons between historic or reference conditions, current conditions, and projected future conditions (Kaibab National Forest, 2008a). The new forest plan will incorporate these evaluations. PNVTs combine potential vegetation (determined by historical soil and climate data) and historic fire regime to form ecosystem classes useful for landscape assessment. A PNVT has successional states that range from an herbaceous state to a fire maintained climax state. States are a mosaic of populations that represent different positions along a complex environmental gradient. Uncharacteristic states can occur as the result of an introduction and spread of invasive species or a change in the frequency of natural disturbance processes. The Vegetation Dynamics Development Tool (VDDT) was used to project future conditions, given the current states and trajectory. High, medium, and low departures from reference conditions were noted. Since the PNVTs are, in most instances, similar to MIS habitat types, they were used to help assess trends for some of the MIS habitat sections. A crosswalk between PNVTs and habitat types is described in Table 16. The main fundamental difference is that cover types do not include historical reference conditions (e.g. soil type, fire), therefore current acreages may differ slightly from PNVT results, i.e., cover type is what is currently there while PNVT is what is there or could be there depending on soil type, historic fire regime, and natural disturbance.

Table 16. Crosswalk for PNVTs used in the forest plan revision process and corresponding habitat types listed in the MIS document.

PNVTs (20 acres or more)	Acres	Cover Type (10 acres or more)	Acres
PJ Woodland	647,604	PJ Woodland	699,649
Ponderosa Pine	553,310	Ponderosa Pine	515,149
Dry Mixed Conifer and Mixed Conifer w/Aspen	127,848	Mixed Conifer	39,122
Spruce-Fir Forest	29,146	Spruce-Fir	28,635
Great Basin, Montane- subalpine and Semi Desert	117,880	Grasslands	216,157
Cottonwood-Willow Riparian Forest and Wetland Cienega	2,676 ¹	Riparian Associations Water	1216 ²

¹Likely includes subalpine meadows

²Includes natural riparian/wetlands and reservoirs/impoundments

Forest Level Data

Several corporate databases were used to obtain Kaibab data. These included the FACTS database that tracks vegetation treatments over time. The infrastructure and range databases were queried for information on livestock management and fencing activities. These corporate databases are meant to be a central repository for forest-wide data which should allow for more consistent data tracking over time. However, because the migration of archived data to these databases was relatively recent, it is possible that not all existing data was migrated over and this has been noted in the text where relevant. As a result, figures may not always reflect the true total numerical value, but rather the best value available at the time. These discrepancies are not expected to affect the overall trend calls. Finally, project records, NEPA documents, completion reports, and the results of contracted surveys were also used to assess forest level data.

HABITAT ACCOUNTS

Ponderosa Pine Cover Type and PNVT



Figure 29. Ponderosa pine PNVT.

The largest ponderosa pine forests in the world can be found in the southwestern United States, including northern Arizona (Kaufman 1992). The oldest ponderosa pine remains are 600,000 year old fossils found in west central Nevada (Moir et al. 1997). One of the most remarkable traits of ponderosa pine is its insulating bark that allows it to survive low intensity wildfires (Reynolds et al. 1996, Graham and Jain 2005, Graham 2005).

Historical Info and Background

Covering about 3 million acres statewide, ponderosa pine (*Pinus ponderosa*) is the third most abundant forest type in Arizona (O'Brien 2002). Ponderosa pine occurs between 6500 and 8500 feet in elevation between mixed-conifer forests and pinyon-juniper and oak woodlands (Reynolds et al. 1996, Reynolds and Boyce 1996). It is used by a variety of wildlife including numerous birds, small mammals, elk, mule deer, and is particularly important for tassel-eared squirrels (Graham and Sisk 2002). On the Colorado Plateau, ponderosa pine forests are dominated by Rocky Mountain Ponderosa Pine (*Pinus ponderosa* var. *scopulorum*) and are frequently associated with Gambel oak (*Quercus gambelii*) at lower elevations. It may also co-occur with Southwestern white pine (*Pinus strobiformis*), Douglas-fir (*Pseudotsuga menziesii* var. *glauca*), Rocky Mountain white fir (*Abies concolor* var. *concolor*), and quaking aspen (*Populus tremuloides*).

On the KNF, ponderosa pine forest covers approximately 515,148 acres (about 34% of the total Forest acreage) and occurs on all three Ranger Districts. Similarly, the corresponding PNVT covers 541,000 acres, approximately 35% of the total land area.

Aspen occurs in small, scattered patches on the Williams Ranger District (RD) and is common in the overstory on the North Kaibab (NK).

There is wide consensus that pre-settlement ponderosa pine forests in northern Arizona and New Mexico were parklike in nature, characterized by grassy openings interspersed with groups of large diameter trees (Cooper 1960, Covington and Moore 1992, Covington 1992, Covington and Moore 1994, Reynolds et al. 1996, Kaufmann et al. 2007). Explorers in the 1800s marveled at the open nature of early ponderosa pine forests (Cooper 1960). These conditions were maintained largely due to low intensity ground fires that occurred every 2-12 years (Cooper 1960, Covington and Moore 1994). These open parklike stands promoted greater faunal diversity and fire resilience than the dense stands of today which are more prone to severe fire events and insect epidemics (Covington 2000). Indeed, today's forests have changed drastically from pre-settlement conditions, largely due to fire suppression, and historic livestock grazing and timber harvest activities (Cooper 1960, Covington and Moore 1992, Reynolds et al. 1992, Reynolds et al. 1996, Abella and Covington 2006). These changes have had ecosystem wide effects, including negative ramifications for sustaining wildlife habitat and populations (Reynolds et al. 1996). It is likely that ponderosa pine forests have been burned more severely than any other type of forest in the last 30 years (Arno 2008).

Tassel-eared squirrels and elk were selected to represent early-seral ponderosa pine forest. Northern goshawks, pygmy nuthatches, and turkeys were selected to represent late-seral ponderosa pine forest.

Potential Management Implications

Wildfire

Low intensity wildfires resulting from natural lightening strikes and/or American Indians were historically common among the grasslands and forests of the Colorado Plateau (Arno et al. 2008, Graham and Sisk 2002). Frequent low-intensity fires have many benefits including insect control, impeding overly dense regeneration of young trees, and decreased competition. Forests characterized by fires of this type are less dense and have smaller fuel loads beneath the canopy (Kaufman et al. 2007).

Under historic conditions, stand-replacement fires were not common in most ponderosa pine forests, although crown fires did occur sporadically (Cooper 1960). Presettlement fires were characterized by slow moving low flames in the understory. In contrast, today's fast moving fires can have flames heights of 80-150 feet that can facilitate crown fire in the overstory (Covington and Moore 1994). The USFS started suppressing wildfire in the 1900s to decrease its effect on urban areas (Arno et al. 2008). Cooper (1961) noted that fire exclusion from southwestern forests was the most significant change brought about by early settlers. As a result, stand densities increased and the threat of high intensity stand replacement wildfires became increasingly prevalent. These uncharacteristic stand-replacement fires have resulted in drastic changes in forest composition, structure, and function. On the KNF, large (> 500 acres) stand replacing fires became evident in the 1960s (Kaibab National Forest 2008a). The most severe fires (Willis, Bridger-Knoll, Pumpkin, Warm, Point) created broad areas of early seral stage communities. In some extreme cases, such as the Point, Pumpkin and Warm fires, uncharacteristic fire conditions seared the existing vegetation on steep slopes. Subsequent rains led to mass-wasting events that washed the soil off the upper slopes, leaving bare rock where forests once stood. It will take many centuries for soil to rebuild on these steep, rocky slopes and perhaps centuries more before forests eventually occupy the sites. About 46% of the total acres converted to early seral stage communities since 1987 (78,435 acres) were a result of crown fire in the ponderosa pine cover

type. Furthermore, the effects from such severe fires are not static, but continue through time. For example, research on the Bridger-Knoll wildfire showed that mortality actually increased 2-3 years post fire. Forty-eight percent of mortality occurred in the severe burn areas even though only 26.5% of that area was sampled (McHugh et al. 2003). Artificial seeding in these areas has had moderate success below 8,700 ft elevation. Recent field visits to the Summit, Willis, and Bridger-Knoll burns have shown regeneration to at least historic conditions (Kaibab National Forest 2008a). Within the Bridger-Knoll burn area, on the north end at the edge of the pinyon-juniper ecotone, planting efforts have produced more seedlings than the number of dead pre-settlement trees evident, but natural reforestation has been essentially nonexistent (B. Higgins, *pers. comm.*).

Stand-replacement fires have the ability to alter natural stand development trajectories by perpetuating dense forest, further increasing the potential for crown fires, or creating altered community states such as grass and shrublands in the face of severe fire impacts (Savage and Mast 2005). One 2-year study following a stand-replacing event in Arizona found that total understory plant community cover increased significantly in areas of high severity fire. Seeded and early successional species were indicative of severely burned habitat while cool season bunch grasses predominated in areas of low severity fire (Kuenzi et al. 2007). Strom and Fulé (2007) studied forest recovery following the Rodeo-Chediski fire, the largest wildfire in Southwest history. They found that forest areas that had been thinned prior to the fire event experienced significantly better post-fire recovery in terms of forest structure than areas that had not experienced fuel reduction treatments. However, regeneration was dominated by Gambel oak, alligator juniper, and New Mexico locust rather than ponderosa pine, suggesting a switch to an alternative stable state. Follow-up modeling exercises found that 100 years post-fire ponderosa pine still comprised most of the basal area in treated areas, but only 35% of the basal area in untreated areas. These altered trajectories in vegetation structure and composition can have ecosystem-wide affects including altered habitat conditions for wildlife species such as tassel-eared squirrels. Other effects of high-intensity fire include the accumulation of snags and downed woody debris that can cause severe reburning, further impeding regeneration and recovery (Passovoy and Fulé 2006, Monsanto and Agee 2008). Older larger trees may be especially vulnerable to severe fires. McHugh and Kolb (2003), assessed tree mortality of ponderosa pine following wildfire in 3 northern Arizona forests. Contrary to most studies, they found that tree mortality was highest among larger trees 10.1- 12.4 inches DBH (25.6-31.5 cm). They suggest that larger trees are physiologically disadvantaged to survive heavy crown damage. In their study, crown scorch was the most important predictive variable ($p \leq 0.001$). This is not surprising given the effects crown damage may have on whole-tree photosynthesis. Other researchers have also noted that crown scorch, in addition to DBH size class and *Ips* prevalence are strong predictors of ponderosa pine survivorship following intense wildfire (Sieg et al. 2006).

Fuels Reduction

The USFS has responded by increasing restoration efforts in ponderosa pine forests, although it may take decades to reverse the ecological imbalance that has resulted from over a century of human interference. The Healthy Forests Restoration Act was passed in 2003 to help mitigate negative impacts resulting from overly dense forests. Somewhat contentious, this act mandates forest restoration largely in the form of fuel-reduction treatments. Restorative treatments that include thinning and prescribed burning have become a focus of many forest management efforts

(Martin et al. 2005, Kolb et al. 2007). If carelessly executed however, such treatments could place additional stress upon the landscape by facilitating erosion, infrastructure development, non-native plant invasion, and habitat loss, particularly for imperiled species (Abella et al. 2006).

Controlled burning improves ecological integrity of the forest by decreasing stand density, improving the diversity of forest composition and structure, and aiding in beneficial nutrient cycling (Arno 2008). These effects can be positive for many wildlife species. Dickson et al. (2009) found that breeding birds in Southwestern ponderosa pine forests respond favorably to prescribe fire of low to moderate intensity. They compared pre- and post-burn densities of a suite of bird species in four local forests (including the KNF) to identify “fire response” species that could be predictably associated with different levels of fire severity. Pygmy nuthatches and hairy woodpeckers were among the species that exhibited strong response to such treatments. On the Kaibab and Coconino National Forests, Hurteau et al. (2008) found that prescribed burns as a follow-up to mechanical thinning can have positive effects on pygmy nuthatches. In thin-and-burn areas, densities of pygmy nuthatches increased by > 500%. Conversely they found that densities of mountain chickadees decreased, especially in thin-only areas. In general, those species that responded most favorably to thinning treatments used areas that had an average basal area from 52-65ft²/acre (12-15m²/ha) and stem density between 57-77 trees/acre (140-189 trees/ha). Merriam’s turkeys can also respond favorably to such efforts. Martin et al. (2005) compared historic turkey roost sites from 1985 to roost sites in forest that was thinned and burned from 1995-2002. They found that improvement of nesting and roosting sites for turkeys can be achieved by decreasing stem density, basal area and canopy cover by thinning and burning of small trees. While the authors identify caveats associated with the study, they concluded areas that had been thinned supported more active turkey roost sites than untreated areas (Martin et al. 2005).

Since 2003 some naturally-occurring fires have been allowed to burn on the KNF; most of these grow up to 100 to 500 acres, with the largest growing to nearly 21,000 acres (Kaibab National Forest 2007b). Prescribed fire is also used as a tool to reduce fuel loading and the potential fire severity of a resulting wildland fire. Currently, the majority of funding for prescribed burning is used to reduce fuel loading in the wildland urban interface. The KNF has burned over 10,000 acres per year for 2005, 2006, and 2007. The target for fuels reduction is expected to grow by 10% every year for the next 10 years. The following charts display the annual amount of fuels reduction accomplishments (Figures 30 and 31). The majority of this work has occurred within the ponderosa pine PNV (Kaibab National Forest 2008a).

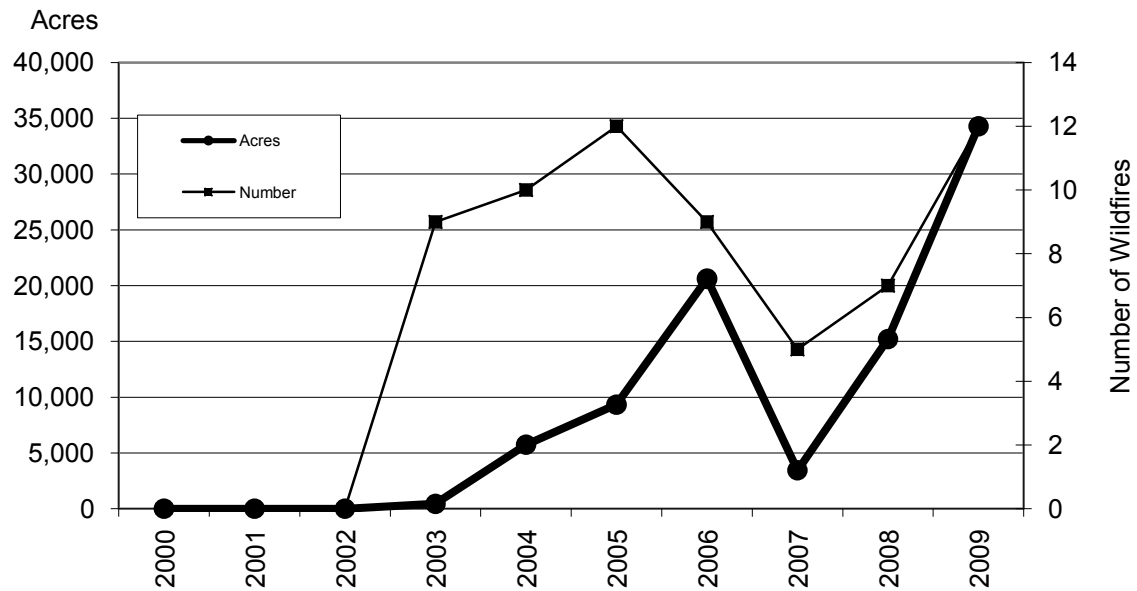


Figure 30. Annual acres treated and number of fires managed across the KNF 2000-2009.

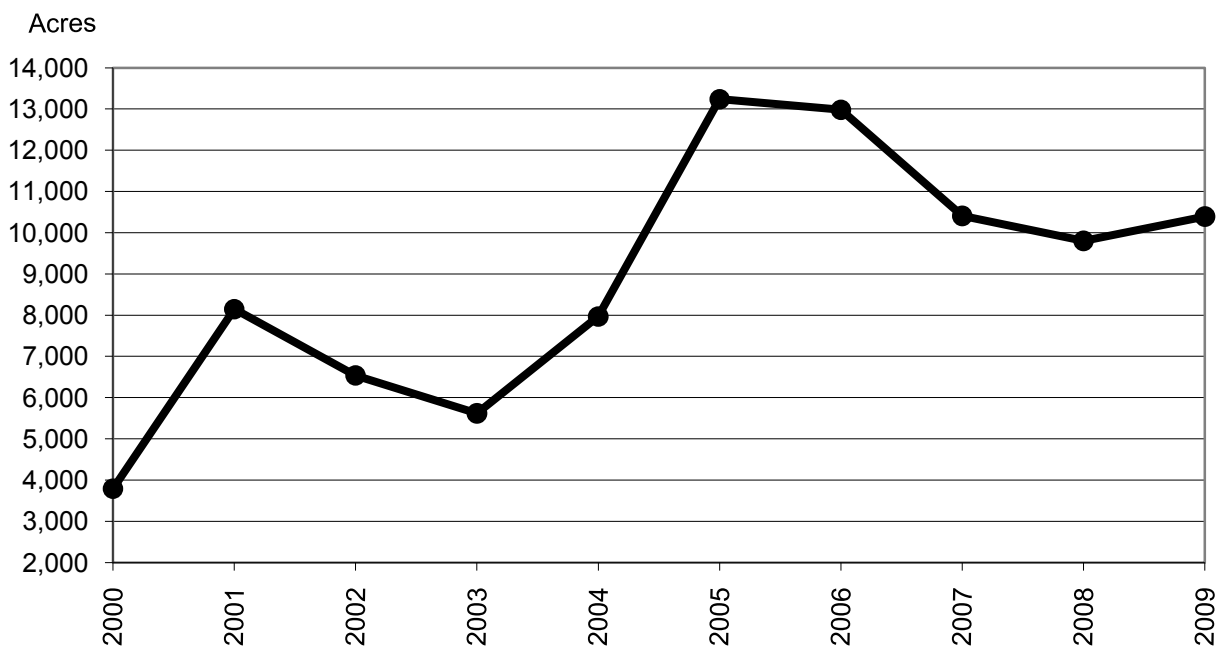


Figure 31. Annual number of prescribed fires and acres treated for the KNF 2000-2009.

Insects/Disease

The KNF plan revision team identified insect/disease outbreaks as a moderate risk to ponderosa pine forest (Kaibab National Forest 2008a). This risk is largely a function of stand density. Across the southwest increased stand densities resulting from years of fire exclusion have created prime conditions for insect epidemics and disease outbreaks, particularly among older trees (Arno 2008). Fungi and parasitic plants may weaken the structural integrity of already stressed trees, making them even more susceptible to bark beetle attacks (Filip 2005). In general, ponderosa pine mortality in the southwest has increased as a result of drought and more frequent bark beetle attacks (Kolb et al. 2007).

Historically, the western pine beetle (*Dendroctonus sp.*) was the most aggressive damaging agent to ponderosa pine (Lynch et al. 2008). On the KNF, mountain pine beetle outbreaks occurred in the 1910s and 1970s, however mortality was reported as less than 3% (Lang and Stewart 1910, Lynch et al. 2008). In general, tree mortality from mountain pine beetle outbreaks has decreased on the Kaibab plateau since 1997 (USDA 2008b). Since 2003 however, damage by *Dendroctonus* has been surpassed by the *Ips* genus, an aggressive beetle that favors denser forests and smaller tree diameters. It is expected the high levels of ponderosa pine mortality will continue to occur throughout the region as a result of high population sizes and dispersal distances associated with *Ips* and other aggressive bark beetles (Allender et al. 2008). An unprecedented *Ips* outbreak damaged 60,000 acres on the KNF from 2002-2004 with 100% mortality occurring in some stands. Some defoliators do occur on the KNF; however they are not considered major pests (Lynch et al. 2008).

High intensity wildfire with heavy crown damage can weaken or kill trees, and decrease resin production that further perpetuates massive bark beetle attacks, however research suggests that low intensity fire used during prescribed burns should not have such adverse impacts. McHugh et al. (2003) compared bark beetle outbreaks in areas of prescribed burns with wildfire. They found that in Arizona, *Dendroctonus* and *Ips* outbreaks in ponderosa pine were lowest during a light prescription burn (found in 11% of trees burned), but increased for high intensity wildfires in the summer (found in 19% of trees burned) and spring (found in 41% of trees burned). They found that bark beetle attacks were most severe in forest which experienced high levels of crown damage rather than trees lightly damaged during prescribed burning. Drought which occurred during their study was also likely to have perpetuated beetle attacks. In another study, Breece et al. (2008) studied tree mortality resulting from bark beetle attacks and prescribed fire. They found that bark beetle induced tree mortality was minimal in Arizona and New Mexico. Cool season prescribed fires increased tree mortality from bark beetle attacks in trees > 5 inches DBH (13 cm) from 0.2% in unburned to 2.8% in burned areas (n = 2091). Heavy crown damage was the most significant variable increasing susceptibility to bark beetle attacks. Prescribed fire in conjunction with mechanical thinning may be optimal for forest restoration. The results of recent research suggest that restoration treatments that use thinning hold great promise for bark beetle control (Kolb et al. 1998, Wallin et al. 2003). Wallin et al (2008) found that thinning in ponderosa pine stands can increase resin production in as little as 2 years. In addition they found that the number of emerging beetles decreased as resin volume increased.

Dwarf mistletoe (*Arceuthobium vaginatum*), a common parasite of ponderosa pine, is considered the most widespread pathogen throughout the southwest (Filip 2005). It infests approximately

2.2 million acres (0.9 million hectares) of forest (Hoffman et al. 2007) causing an even greater loss to the timber industry than insects (USDA 2008b). Paradoxically, dwarf mistletoe can have positive benefits for wildlife by providing nesting and foraging habitat through broom production, increased tree mortality and snag densities (Fairweather 2004, Garnett et al. 2006, Hoffman et al. 2007). This poses an interesting management dilemma because while the increases in mistletoe and snags are associated with increased fuel loads, they may also provide important nesting habitat for woodpeckers and sapsuckers, or foraging habitat for birds and small mammals. Garnett et al (2006) found that ponderosa pine trees ≥ 59 ft (18m) in height with brooms containing > 7 branches, were most utilized as foraging and caching sites by Abert's squirrels. These criteria can guide selection of trees in forest thinning treatments; however such criteria should also be weighed against landscape level restoration objectives.

In 2007, dwarf mistletoe affected approximately one-third of ponderosa pine forest in the states of Arizona and New Mexico (USDA 2008b). The resulting increase in tree mortality and flammability subsequently altered fire behavior and increased fuel loadings. Locally, in Flagstaff AZ, Hoffman et al. (2007) found that severely infested stands had higher snag densities and lower tree densities. Although snags created by mistletoe typically remain standing longer than fire or beetle killed trees (Joy Mast *pers. comm.*, C. Chambers *pers. comm.*). On the NKRD, Fairweather (2004) found a 32% dwarf mistletoe infection rate, where 25% of the stands examined had heavy to severe levels of infection. Infection rates were highest among ridge tops that offer favorable light and moisture conditions for mistletoe establishment. Hessburg and Beaty (1985 *in* Lynch et al. 2008) estimated an annual growth loss of 20-30 million board feet annually for the KNF in the 1980s. Hawksworth and Geils (1990 *in* Lynch et al. 2008) found that more than 90% of uninfected or lightly infected trees survived over a 32-year period in the Grand Canyon National Park (NP). However, only 5% of heavily infected trees over 9 inches DBH (22.9 cm) survived and no trees less than 9 inches DBH survived (Hawksworth and Geils 1990 *in* Lynch et al. 2008). Bark beetle infestation may place additional stress on heavily infected trees as was the case during a beetle outbreak in ponderosa pine forest on the KNF from 2002-2003 (Lynch et al. 2008).

The prevalence of dwarf mistletoe among stands should be considered when evaluating silvicultural prescriptions, since infestation rates will determine if even-aged or uneven-aged management is more appropriate. Infection rates tends to be highest among uneven-aged stands (Fairweather 2007). Burning may be used to reduce the spread of infection among heavily affected stands while thinning and group selection cuts can be used to control light to moderately affected stands, particularly those that are uneven-aged (Fairweather 2007, Hessburg et al. 2007, Conklin and Geils 2008). The KNF began silvicultural prescriptions to mitigate the seriousness of southwestern dwarf mistletoe in the 1980s.

Noxious and Invasive Weeds

Cheatgrass (*Bromus tectorum*) is found throughout western North America and is especially threatening at lower elevations. It commonly invades areas that have been severely burned, outcompeting native grasses (James 2007). On the KNF, cheatgrass and bull thistle (*Cirsium vulgare*) occupy up to 1% of the ponderosa pine PNV, primarily along roads (Kaibab National Forest 2008a). Fowler et al. (2008) documented significantly greater exotic plant species richness along roadsides than in adjacent interior habitat. Wheatgrass (*Thinopyrum intermedium*),

common knotweed (*Polygonum aviculare*), black medick (*Medicago lupulina*), and smooth brome (*Bromus inermis*) were among the most abundant species. For that same study, they also found that most exotic species showed little increase resulting from low-intensity *prescribed* fire in ponderosa pine forest. However, other studies have shown that thinning and burning can increase diversity and abundance of invasive plants (James 2007), particularly if the resulting slash is piled and burned (Neal 2007, Owen et al. 2009). Some areas on the KNF that have experienced large wildland fires can have extensive populations of invasive weeds (Kaibab National Forest 2007b). The resulting changes in understory plant composition can cause trophic cascades that may be detrimental to wildlife. For example, replacement of heterogeneous vegetation by a monotypic cover such as cheatgrass may lead to decreases in the invertebrate prey base necessary for some bird species (Whiteman and Germaine 2006). Likewise, cheatgrass can shorten fire return intervals thereby altering shrub communities and decreasing small mammal populations and the birds that prey upon them (Delasalla et al. 2004).

While thinning and burning have become common restoration tools in ponderosa pine forests, incorporating management procedures to mitigate the spread and invasion of cheatgrass and other noxious weeds is necessary to offset negative effects that may result from such activities. A 2003 environmental impact statement for the treatment of noxious or invasive weeds identified 187,500 acres (out of 4.9 million total, 3% of the land area) across the Coconino, Kaibab, and Prescott National Forests as being impacted by noxious weeds. Weeds identified for ponderosa pine habitat included leafy spurge (*Euphorbia esula*), spotted knapweed (*Centaurea maculosa*), diffuse knapweed (*Centaurea diffusa*), Dalmatian toadflax (*Linaria dalmatica*), Scotch thistle (*Onopordum acanthium*), and bull thistle (USDA 2005). On the KNF, the Range Management program mitigates the spread of cheatgrass and other undesirable species by treating those areas that are most susceptible to invasion, particularly those areas that have experienced recent disturbance events such as fire. In general, the management emphasis is in the pinyon-juniper and grassland habitat types (see those sections, this report), with less than 200 acres of cheatgrass treated on the North Kaibab RD in ponderosa pine forest. However, new funding may become available as additional areas susceptible to cheatgrass invasion are identified in ponderosa pine, increasing this amount by as much as 500 acres by the fall of 2009 (M. Hannemann *pers. comm.*). Severe fires can facilitate the spread of other highly invasive plants like Dalmatian toadflax. Dodge et al. (2008) found that post-wildfire conditions in severely burned areas had increased density, cover, reproduction, and spread of Dalmatian toadflax while native species decreased. These effects were most significant 2 years post-fire. Additional measures proposed as mitigation for noxious weeds include burn size and seasonality, adjusting livestock management, minimizing soil disturbance, cleaning vehicles and personnel after working in cheatgrass infested areas, and inventories of infested areas may minimize additional invasion and effects James (2007).

Grazing

Historically, grazing that began in the late 1800s adversely affected plant cover through consumption and trampling and indirectly affected plants by impacting soil and water quality (Graham and Sisk 2002). In ponderosa pine forests, cattle and sheep browsing diminished recruitment of seedlings and reduced levels of fire resistant grasses on the forest floor (Cooper 1961). Unmanaged grazing can cause soil disturbance that facilitates cheat grass establishment thereby altering plant species composition. Drought further exacerbates this effect. Although the

effects of historic livestock and high-impact grazing are undoubtedly negative, new research suggests that moderate levels of managed livestock grazing may actually increase native plant diversity while inhibiting cheatgrass colonization in some ecosystems (Loeser et al. 2006). For example, previously introduced and/or undesirable plant species may in fact become more prolific, once livestock have been completely eliminated from the system. It is possible that some ecosystems that have been subjected to cattle grazing for long periods of time may undergo biological and evolutionary shifts that favor grazing-tolerant species in order to maintain diversity. Managers must assess these complexities on a case by case basis. On the KNF, managed and un-managed grazing occurs in ponderosa pine forest. Managed livestock grazing is under an annual permit system. Grazing numbers, season, and duration are managed in accordance with annual inspection of range conditions. Un-managed grazing occurs by deer and elk (Kaibab National Forest 2008a).

Logging/Timber

Wide scale logging and timber removal began in response to the transcontinental railroad in the late 1800s. New technology in the early 1900s allowed for more widespread and efficient harvest of trees, including snags. A “pick-and-pluck” strategy (taking the best and leaving the rest) guided forest management. This philosophy continued well into the 1980s with an increasing emphasis on even-aged management and harvesting of large trees. Multiple lawsuits identifying the negative impacts of clear cutting on flora and fauna initiated a decrease in harvest levels in the 1990s (Graham and Sisk 2002). In 1992, the Forest Service adopted a policy that allowed clear cutting only if it would be of benefit to wildlife habitat or natural resource enhancement. This was largely a result of the goshawk guidelines that were incorporated into the Forest Plan in 1996, although some previously planned projects that included clear cutting and even-aged management of ponderosa pine continued into the early 1990s. Today vegetation management strategies are the opposite of those imposed in the 1900s in that tree removal is now done specifically to enhance ecosystem health.

On the KNF, the shift to uneven-aged management supports northern goshawks and their varied prey base (Youngblood 2005). Extensive railroad logging occurred on the Williams District in the early 1900s. As a result the forest is mostly a young to mature forest with occasional old to very old trees. On the Tusayan District railroad logging occurred later, and lasted for a shorter period. As a result, there is a larger component of older pines on the Tusayan District than there is on the Williams District. Railroad logging never occurred on the NKRD and instead historic harvest consisted of cutting the occasional over-mature tree. Harvest was over a widespread area, so there are numerous old to very old pine left (Figure 32). Shelterwood harvests were initiated in the mid-1970s through the early 1990s in areas with high levels of dwarf-mistletoe (Kaibab National Forest 2008a, B. Higgins *pers. comm.*).

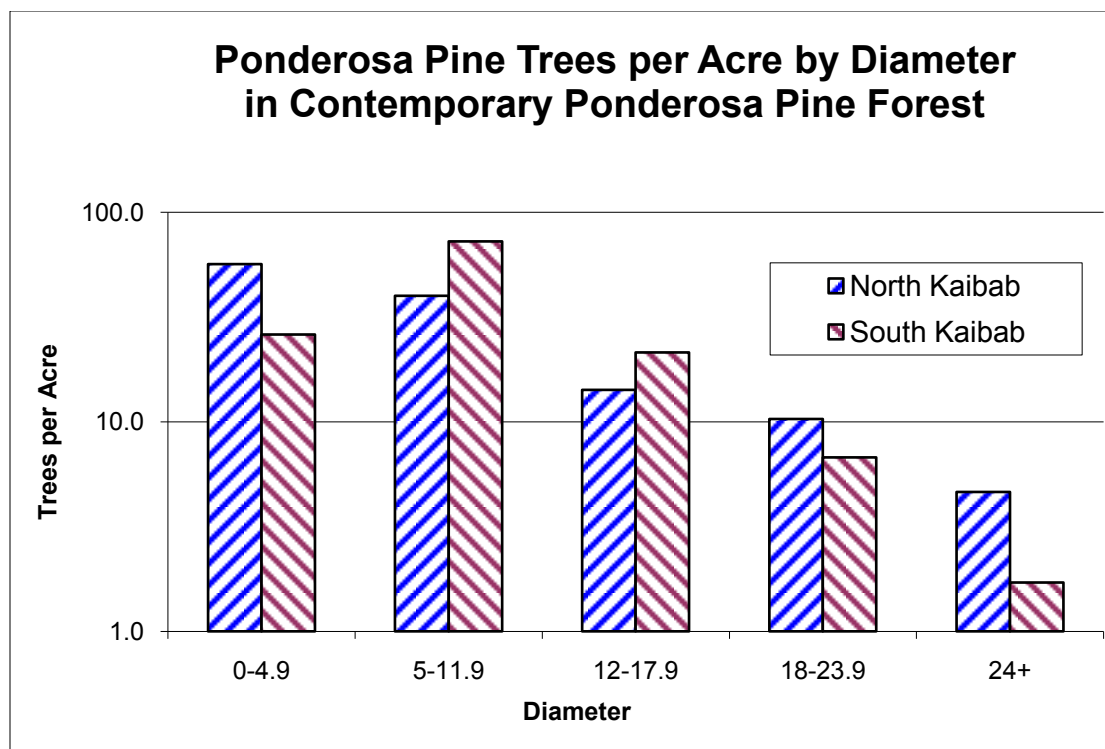


Figure 32. Tree densities by size class across the KNF. Historic to present (1990) day (Adapted from FIA data).

Before the 1996 Forest Plan Amendment, management objectives emphasized even-aged harvests that treated nearly 2.5 times more total acreage per year than the post-Forest Plan, uneven-aged harvests. Pre-1996 management largely relied on shelterwood cuts that yielded higher timber volumes (overall and also on a per acre volume) and cut many more trees per acre in the larger size classes. Treatments after 1996 switched to uneven-aged management that primarily applied commercial thinning and group selection cuts. Today, thinning projects are aimed at maintaining the large tree component across the forest and through time, i.e., as one cohort of large trees is eventually removed from the forest; neighboring groups will provide the same structure and forest characteristics. Thinning also reduces time required to achieve recruitment of trees into the larger size-classes and can reduce the risk of crown fires. Crown fires cause extensive stand mortality and have become increasingly prevalent throughout the Southwest (Covington and Moore 1994, Savage and Mast 2005). The ultimate goal of forest management is to maintain a heterogeneous forest devoid of even-aged stands. Instead, areas the size of typical stands (30-40 acres) will be a mosaic of early-, mid-, and late-seral vegetation with trees in the larger size-classes represented throughout the area. This mosaic is thought to better represent the patch dynamics of presettlement forests and should be more sustainable through time than the current forest structure.

Special Note on Gambel oak

Gambel oak, a component of the ponderosa pine cover type, is important to many wildlife species, including Mexican spotted owls. Higher species richness has been correlated with higher densities of Gambel oak, a habitat that provides critical nesting and foraging resources for many

northern Arizona birds (Griffis-Kyle and Beier 2003). In pine-oak forest, Gambel oak can provide alternate nesting sites for owls when pine snags become limited (Rosenstock 1996 as cited in Moir et al. 1997).

Currently, pine-oak habitat is managed as Restricted Habitat under the Mexican Spotted Owl Recovery Plan (USDI 1995). Years of fire suppression have permitted other woody species to out-compete Gambel oak, which is adapted to fire (Harper et. al. 1985). Many individual large oak trees as well as oak copses have become over-topped with pine trees, making their continued health on the forest problematic. In general, the management direction within pine-oak habitat is to retain the oak component. The KNF does not conduct commercial harvests of oak. Oak sprouts prolifically after treatments (Harper et. al. 1985) and may be cut or burned to stimulate new growth, maintain growth in large diameter trees, or to stimulate mast production. Large (> 8 inches DBH), live oak are not cut and small oak are only cut to initiate oak regeneration. Firewood collection is regulated so that firewood cutters are only allowed to cut standing dead or live oak less than 8 inches DBH. The Forest Plan encourages the use of ponderosa pine slash for fuel wood rather than oak or aspen. However, large oak are cut illegally on the Forest. No MIS were selected to specifically represent oak or the pine-oak association. We mention it here for reference purposes only because of its potential contribution to wildlife habitat, particularly for Mexican Spotted Owls.

Ponderosa Pine Habitat Data

According to FIA data collected from two cycles in 1995 and 2007, there has been a steady increase of trees in the small to medium size classes, with a slight decrease in the large tree component class (Figure 33). These data highlight the increase in density of smaller-sized trees, especially among seedling/saplings (Table 17). This increase in trees per acre is consistent across all Ranger Districts. Across the forest there have been minimal changes in density of trees > 5 in DBH.

Although the KNF's overall management strategy since the 1990s has been to maintain a forest with structural diversity, there have been some declines in the large tree component due to fire, insects, disease, and management activities since implementation of the Forest Plan. Overall however, the harvest rate of the larger DBH size-classes has dropped off since the goshawk management guidelines began to be incorporated into forest planning in 1992.

A major drought event occurred throughout the Southwest in 2002 (Breshears et al. 2005). Following this regional drought, Gitlin et al. (2006) found that ponderosa pine experienced a 15.4% mortality rate, with adult mortality significantly higher ($p < 0.0001$) in younger cinder soils (81% vs. 29% for trees growing in older soils). Soils on the NKRD are characterized by limestone and should contain higher levels of nutrients than the older and more acidic basalt soils of the SZ.

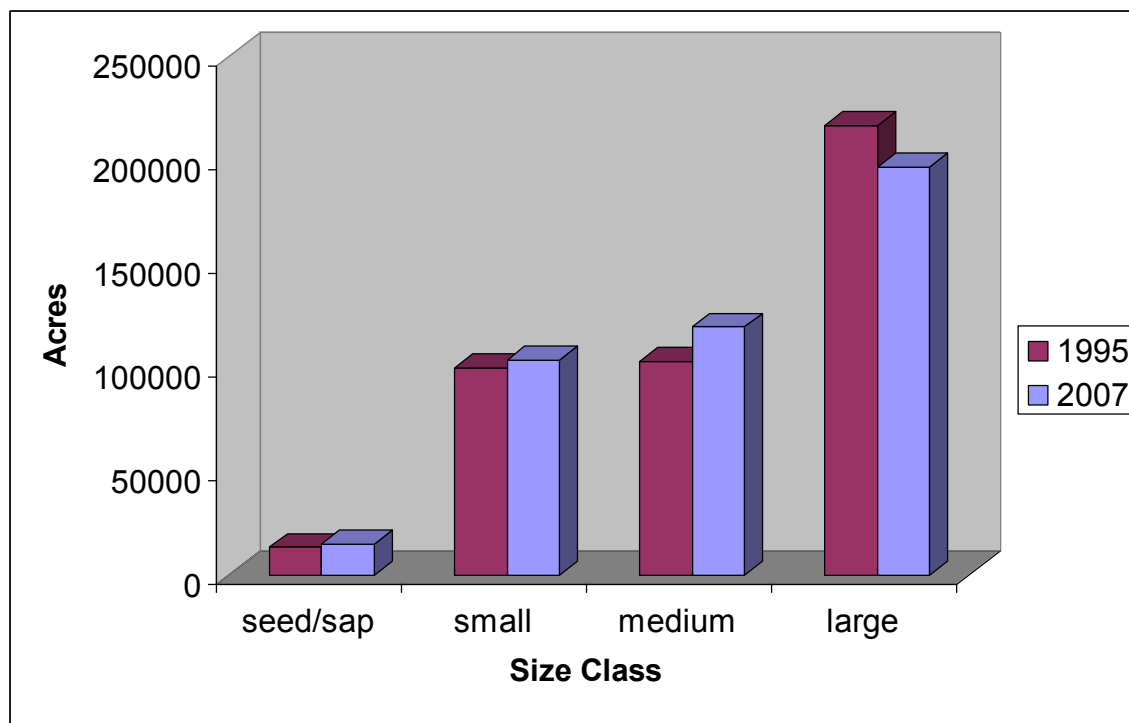


Figure 33. Acres of ponderosa pine forest by size-class across the KNF. Large (≥ 18 in DBH), medium (12-17.9 in DBH), small (5-11.9 in DBH) Seedlings/saplings (less than 5 in DBH).

Table 17. Number of live ponderosa pine trees per acre on forest land by Ranger District and diameter class, KNF (FIA unpublished data).

Ranger District	Diameter Class				
	1"-4.9"	5"-11.9"	12"-17.9"	18"-23.9"	≥ 24 "
1995					
North Kaibab	20.23	13.97	5.09	3.81	1.70
Tusayan	11.51	10.29	2.36	1.00	0.59
Williams	14.97	37.68	12.79	3.52	0.71
Totals	46.71	61.94	20.24	8.32	3.00
2007					
North Kaibab	35.80	12.74	5.08	3.54	1.53
Tusayan	67.64	18.26	4.61	1.32	0.33
Williams	21.03	27.81	13.51	3.18	0.50
Totals	124.47	58.81	23.19	8.04	2.36

Forest Service policy prior to 2000 was to fully suppress all wildland fires, resulting in increased stand densities and more frequent and severe forest fires. In 1910, only two crown fires were big enough to map in the ponderosa pine habitat. The larger of the two fires burned about 80 acres (Lang and Stewart 1910). By comparison, in 2006, the Warm Fire burned roughly 39,000 acres with 36,000 acres comprised chiefly of ponderosa pine. Of the total area approximately 7,290

acres (19%) were classified as moderate to high burn severity and 15,780 acres (40%) were classified as high burn severity (USDA 2009) in ponderosa pine (Appendices 7-8).

In total, about 135, 875 acres of ponderosa pine have burned on the KNF since 1987 (Table 18). Since 1995, about 3.7% of the ponderosa pine PNVT has experienced stand replacement fire (Kaibab National Forest 2008a). In general, there is a trend toward more frequent stand replacement, or mixed severity fires (Figure 34).

Table 18. Total acres burned and severity of wildland fires for unplanned ignitions for large fires in ponderosa pine on the KNF since 1987. Does not include acreages of fires managed for resource benefit.

Fire	Year Burned	Total Acres Burned	Severity		
			Surface	Mixed	High
Willis	1987	2078.5	345.031	172.5155	1207.609
Government	1989	77	0.7546	0.231	0
Bear	1990	185	92.5	74	18.5
Point	1993	1656.8	178.9344	178.9344	89.4672
Bridger	1996	53504	9630.72	3210.24	3210.24
Pumpkin	1996	15775	3786	2839.5	2839.5
Bachelor	1996	217	210.49	63.147	126.294
Mangum	2000	291	11.64	58.2	162.96
Outlet	2000	15427	462.81	925.62	3239.67
Oak Point	2000	216.8	10.84	54.2	151.76
Hidden	2001	463	64.82	64.82	194.46
Big	2002	110	27.5	55	27.5
Trick	2002	5568	2572.416	734.976	367.488
Lefevere	2004	221	8.2875	8.2875	149.175
Nail	2005	970	19.4	77.6	291
Warm	2006	39115	7627.425	7627.425	10169.9
TOTALS		135875.1	25049.57	16144.7	22245.52

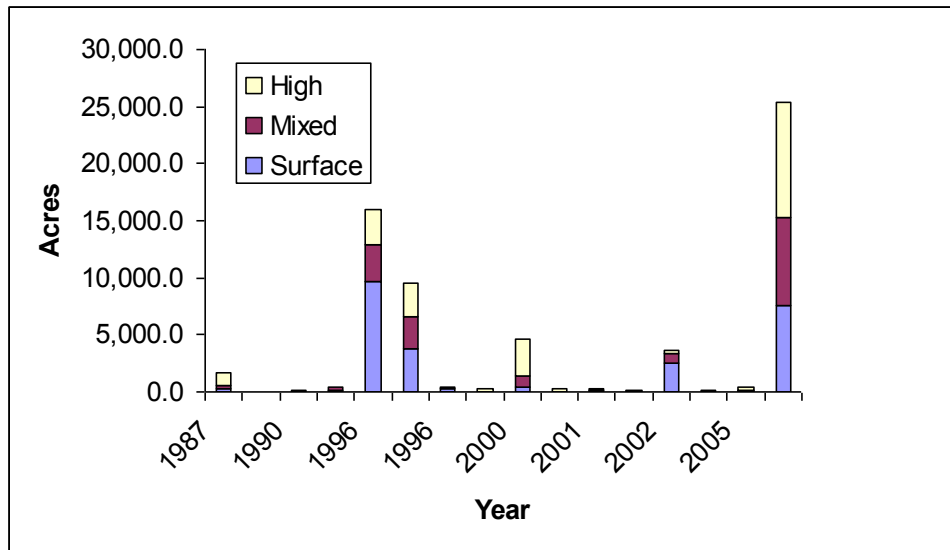


Figure 34. Acres of ponderosa pine burned by wildfire per year by fire type.

Does not include acres burned by wildland use fires (KNF unpublished data).

Habitat Trend

The current management direction, i.e., providing a continuous flow of habitat structural stages through time, an emphasis on retaining groups of large trees, and maintaining large-sized reserve trees spread throughout the other age-classes, was designed to account for the needs of goshawks and their diverse prey base. These habitat characteristics should also create the diverse habitat conditions favored by tassel-eared squirrels.

The large tree component and the designed sustainability of this element across the landscape should benefit pygmy nuthatches by creating a dispersion of trees large enough for the creation of future large DBH snags while providing foraging and roosting habitat. Similarly, this strategy should provide for turkey roost trees while also maintaining patches of foraging habitat and hiding cover.

It is therefore assumed that overall habitat trends brought about by the shift in Forest objectives and harvest techniques initiated in 1992 have been positive for each of the MIS associated with the ponderosa pine cover type with the possible exception of elk. Although the increased heterogeneity in forest structure should increase availability of graminoids and forbs throughout the forested cover types, it is unclear how this compares to the much larger openings that resulted from even-aged stand management. The former should provide for a better distribution of individual animals while the latter likely provides more food in a given area. How the disparate strategies compare when factoring optimal foraging opportunities and hiding, escape, and thermal cover for elk is unknown. It is difficult to assess changes in elk numbers and elk habitat without factoring in hunter pressure and hunter success indices. These estimates are beyond the current scope of this document.

In spite of these positive management efforts for wildlife, the long-term prognosis for ponderosa pine is less than satisfactory. On the KNF, ponderosa pine forest is characterized by very dense

conditions with 79% of the stands in a “closed” state, i.e. greater than 30% canopy cover (Kaibab National Forest 2008a). The spaces that historically occurred between clumps of trees have been greatly reduced and in many cases eliminated, leaving the forest vulnerable to stand replacing fire. Only 19% of the ponderosa pine PNVT resembles historic conditions (Kaibab National Forest 2008a). Current management strives to move the forest toward desired conditions by managing fire for resource benefit and by using fuel reduction treatments such as prescribed fire and mechanical thinning. About 20,000 acres of forest are treated in this manner each year (Kaibab National Forest 2008a).

Projected

State and transition models developed during the forest plan revision process suggest that ponderosa pine forests on the KNF are highly departed from reference conditions. At the current rate of natural and human disturbance, the ponderosa pine forest is trending toward a younger and denser state. Given the current trajectory, a decline in fine-scale stand diversity will result in more homogeneous stands. Structural heterogeneity between large forest patches (>100 acres) is expected to increase, due to stand replacing fire, encroachment of fire intolerant tree species, and a continued buildup of live and dead fuels (Kaibab National Forest 2008a). This pattern will create a continuing trend away from sustainable forests and presettlement habitat conditions.

Aspen Cover Type

The most massive individual organism in the world is a 6-million-kilogram single male aspen clone called Pando (Latin meaning “I spread”). Found in Fishlake National Forest in Utah, it covers 106.3 acres, is approximately 1 million years old, and has approximately 47,000 stems. Its mass is triple that of the largest giant sequoia (Grant et al. 1992, Mitton and Grant 1996).

Historical Info and Background

Quaking aspen, a member of the willow family (Salicaceae), is the most widely distributed tree species in North America and the second most widely distributed tree species in the world (Campbell and Bartos 2001, Rogers et al. 2007). Named for its fluttering leaves (Mitton and Grant 1996), aspen has the unique ability to reproduce by “suckering” from parent roots (Bartos and Campbell 1998). In the west, dry environmental conditions rarely allow establishment of aspen by seedlings (Patton and Jones 1977). Major disturbance events however, such as the 1988 fire in Yellowstone NP, can facilitate seedling germination (Romme et al. 2005).

More typically, aspen reproduce asexually by forming a new grove of trees that represents a clone of the original parent tree. The average clone age varies from 1000 to over a million years (Barnes 1975). Aspen is an early successional species that often requires disturbance to stimulate suckering (Mitton and Grant 1996). Prior to Euro-American settlement, Native Americans may have played a crucial role in maintaining aspen by limiting elk numbers and using low-intensity fires that stimulated regeneration (Hessl 2002). Remnant aspen may indicate a site was once dominated by aspen (O’Brien 2002). Aspen is considered mature when over 80 years old (Hart and Hart 2001) and, without additional disturbance, clones can dominate a site for about a century. In the absence of disturbance, aspen is usually successional to conifers. Conifers typically dominate aspen between 100 to 200 years although mixed conifer species can become established within a decade (reviewed in Dahms and Geils 1997).

Although common continentally, aspen are threatened regionally. As a result of increased fire suppression activities, unchecked forest succession, overgrazing and over browsing by elk, aspen stands are currently in decline in most of the southwest (Bartos 2001, Graham and Sisk 2002, Hessl 2002, Griffis-Kyle and Beier 2003, Kaye et al. 2005). In Southwest forests, aspen typically exist as smaller stands within a larger forest matrix dominated by ponderosa pine and mixed coniferous species (Patton and Jones 1977).

Aspen stands are usually moister and richer in flora and fauna than their coniferous counterparts. This makes them an integral component of southwestern forests. In fact, aspen acts as a “keystone” species (Simberloff 1998). The removal or addition of aspen may have significant impacts on community composition and structure. Aspen is second only to riparian ecosystems in biological diversity (Campbell and Bartos 2001). The soft wood of decaying stems and snags provide valuable habitat for various cavity-dependent species, while young sprouts provide forage for elk and mule deer. As a component of coniferous forests, aspen stands tend to have higher plant, invertebrate and cavity-nesting bird species than the surrounding forest (Hessl 2002). In fact, aspen support more bird species populations than other forested areas in the western U.S. (Hollenbeck and Ripple 2007). Even small aspen stands may provide refugia for certain plant and animal species (Rogers et al. 2007). Loss of aspen represents a loss of diversity in the forest that may affect avifauna and invertebrates, including pollinators like lepidopterans (Chong et al. 2001, Rumble et al. 2001, Struempf and Hayward 2001).

Red-naped sapsuckers and mule deer were selected as MIS to represent aspen. Red-naped sapsuckers were selected to represent late-seral aspen while mule deer were selected to represent early-seral stage aspen habitat.

Potential Management Impacts

In general aspen continues to be overtopped by conifers and regeneration is limited by fire suppression and ungulate grazing. More recently weather events, insects, and disease have caused widespread diebacks.

Herbivory

Ungulate browsing by wildlife (elk and deer), and livestock (cattle and sheep) is a major factor limiting aspen regeneration in the Southwest (Bartos 2007). In many places throughout the west, game management and lack of top predators has resulted in increased elk herbivory and subsequent lack of recruitment in aspen (Rogers et al. 2007). For example, removal of the gray wolf in Yellowstone NP during the 1920-1930s coincided with widespread aspen decline (Ripple and Larsen 2000). This may have been a result of increased ungulate browsing resulting from predator release. A recent assessment at Rocky Mountain NP suggests that an 80% reduction in elk is needed to allow declining aspen stands to regenerate (reviewed in Bartos 2007). According to Hessle (2002), heavy elk browsing has affected aspen regeneration in the intermountain west for the past 150 years. Kaye et al. (2005) found that mean recruitment of aspen dropped by over 50% in low and high-browsed stands. Elk can cause aspen mortality directly through herbivory and indirectly through mechanical injuries to the boles and stems that allow invasion by pathogenic fungi (Lynch 2008). Aspen is very susceptible to gnawing or having the bark stripped by ungulates. Recently, aspen die off that affects mature overstory trees has been reported at an

alarming rate throughout the west. Suspected causal agents are cankers (*Saperda calcarata*) and poplar borers (*Agilus liragus*), however other causes may yet be discovered (Bartos 2008). Ordinarily, aspen are highly resistant to fungal pathogens until wounds alter the trees condition (Hart and Hart 2001).

Over-browsing by deer may also affect aspen recruitment. Smith et al. (1972), as reviewed in Hart and Hart (2001), reported that deer herbivory alone had little effect on aspen reproduction, but when deer were present with cattle, aspen regeneration was dramatically impacted. Kay and Bartos (2000) studied the effects of herbivory and aspen regeneration. They found a significant decrease in regeneration of aspen stems [1,012 stems per acre vs. 1,812 stems per acre (2,498 per ha vs. 4,474 stems per ha)] in exclosures as a result of deer browsing. Plots that were used by wildlife and livestock regenerated at an even lower rate [410 stems/acre (1,012 stems/ha)]. Binkley et al. (2006) conclude excessive deer herbivory likely impacted age structure of aspen on the Kaibab plateau. They found that low aspen numbers coincided with high unchecked deer populations on the KNF in the mid-1900s.

The indirect effects of ungulate browsing may include limited recruitment of aspen saplings into larger size classes over time. This results in forest structural changes that may prove detrimental for some wildlife species. Hollenbeck and Ripple (2008) found that unmanaged aspen forest contained greater numbers of small diameter trees, which in turn facilitated greater overstory canopy cover. Griffis-Kyle and Beier (2003) found that the most significant habitat variable negatively affecting species richness among birds in aspen stands was high sapling densities, possibly because younger aspen stands provide decreased foraging opportunities.

Fire and Conifer Encroachment

In general, unprotected aspen suckers on the KNF are repeatedly browsed by elk which limits recruitment. The most notable exception to this occurs in the 2000 Pumpkin wildfire area. Extensive aspen regeneration may be occurring because of the overall amount of aspen post-fire regeneration, the proximity of large meadow systems, or potentially because elk are limited by the topography. In addition, elk are not a considered a desirable game species on the Kaibab Plateau. The Warm fire in 2006 affected approximately 2,609 acres of aspen (Appendices 7 and 8). This has had positive effects on aspen regeneration with extremely dense suckers observed throughout the area. Fire can be an effective method for stimulating aspen regeneration, particularly in stands affected by conifer encroachment (Hessl 2002, Bartos 2007, Rogers et al. 2007), which has become increasingly widespread across the forest and throughout the intermountain west. Aspen is replaced by coniferous species that out compete it in sites that lack fire disturbance. The Sierra Nevada is one area where past fire suppression activities have negatively impacted aspen recruitment (Rogers et al. 2007). Others have noted that initial declines in aspen coincided with the commencement of fire suppression activities (Hessl 2002). In general, aspen depend on periodic disturbance such as fire to regenerate. Fire suppression decreases recruitment and further facilitates invasion by conifers or sagebrush (Bartos 2007). These successional changes may lead to ecosystem wide effects which result in losses of water, biodiversity and understory vegetation. Evapotranspiration rates are higher in conifer dominated forests and loss of aspen resulting from conifer encroachment will likely lead to increases in net water loss (Bartos 2007). Some studies have documented water loss of as much as 5%, or 3-7 inches in forests where conifers have replaced aspen (Shepperd et al. 2006). The resulting effects

include a decrease in ground water recharge and loss in understory vegetation (Bartos 2001). A conifer canopy with just 10% - 20% cover can suppress aspen recruitment and understory production, decreasing available forage and cover habitat for wildlife (Bartos 2007, Stam et al. 2008). Kaye et al. (2005) found a mean annual reduction of over 30% aspen in mixed forests.

Insects/Disease

Disease and insects have also impacted aspen. The KNF has experienced extensive defoliation events caused by Western tent caterpillars, large aspen tortrix, melampora rust, and black rust since the 1940s, although in general mortality has been minimal until now. Recently however, the effects of these causal agents have been exacerbated by weather events such as severe frost and drought. Since the late 1990s, these abiotic agents have acted cumulatively with regard to disease and insects to cause accelerated dieback and mortality (Lynch 2008) (Figure 35). Other forests have experienced similar trends. On the neighboring Coconino National Forest, Fairweather et al. (unpublished data 2008) found that a frost event in June of 1999 followed by a long-term drought and periodic defoliation caused accelerated aspen decline. They found considerable changes in stand composition across 6000 acres of aspen forest. Stands below 7500 ft. in elevation were especially hard hit, experiencing over 95% mortality by 2007. Mid-elevation sites (7500-8500 ft) experienced 61% mortality while the highest elevations (> 8500 ft) experienced 16% mortality. Continued drought has exacerbated defoliation events. In 2007, 143,000 acres were damaged region-wide with 99,000 acres in Arizona alone (USDA 2008b). This marks a substantial increase since 2006.

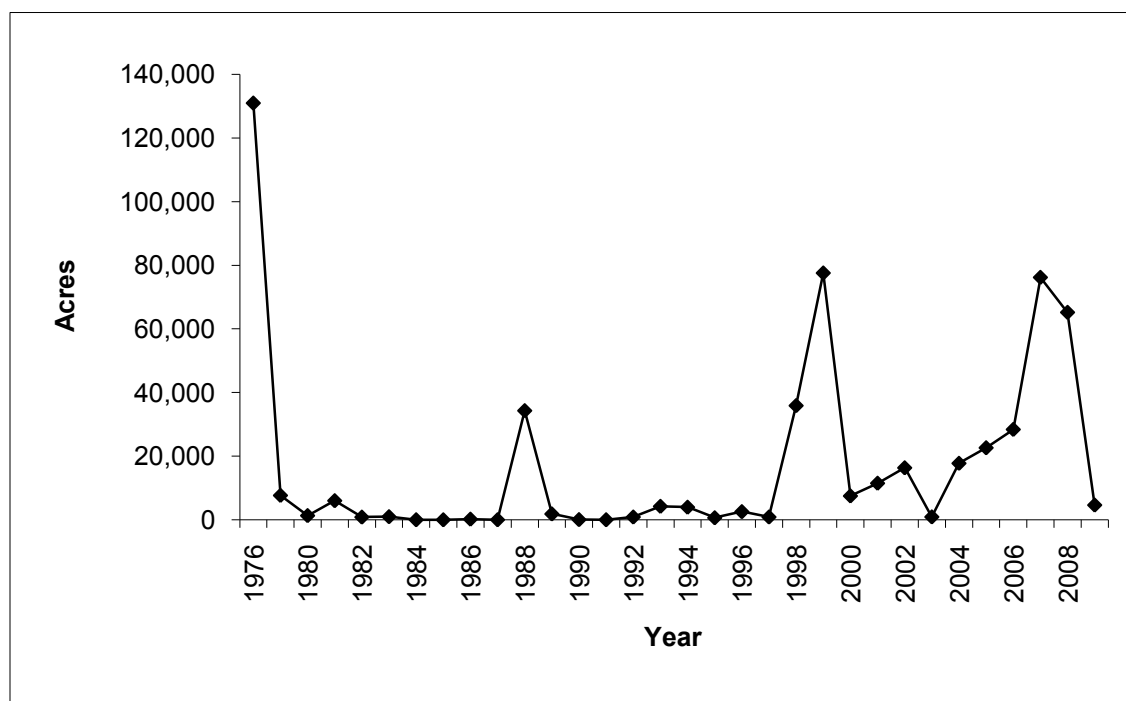


Figure 35. Aspen defoliation history on the KNF (S. Dudley *pers. comm.*, Fairweather et al., unpublished data 2009).

Management Response

Management actions that mimic disturbance events such as low intensity fires and/or thinning to reduce conifer encroachment may be necessary to facilitate aspen regeneration. There is wide

consensus that thinning, prescribed burning and fencing for elk are all important management priorities for sustaining aspen in the southwest (Patton and Jones 1977, Grifffis-Kyle and Beier 2003). Although some work has been done to improve aspen regeneration, these efforts have been limited in scope and success. On the KNF, aspen habitat is treated opportunistically, with the goal of stimulating suckering and removing over-topping conifers. Many of the even-aged cutting units on the NKRD from the 1970s have regenerated as aspen. Even-aged aspen cutting can prove beneficial to aspen regeneration (Patton and Jones 1977) and could be desirable method for improving the regeneration potential of declining stands if a market for aspen exists (Bartos 2007). It is important to note however, that treatments such as cutting and fire that stimulate aspen regeneration may only exacerbate aspen loss if ungulate herbivory is heavy and uncontrolled (Kay and Bartos 2000, Campbell and Bartos 2001). Thus it is important to consider the net benefits of various treatments with regard to browsing pressure and recruitment. Due to heavy elk browse and the resulting effects of fungal disease, fencing appears especially important to maintain aspen on the Williams RD (Lynch 2008).

Grazing and browsing effects from elk are seriously impacting aspen regeneration on the Williams District where efforts to successfully regenerate aspen are completely dependent on protective fence exclosures. Fencing of aspen on the Williams RD has been limited: 6 acres treated in 2000-2007, 68 prior to 2000, an additional 48 in 2008. However, even small pockets of aspen may provide refugia for wildlife and plant species requiring moist conditions (Rogers et al. 2007, Grifffis-Kyle and Beier 2003). Grifffis-Kyle and Beier (2003) found that small aspen stands within coniferous forests act as important reservoirs for maintaining high avian diversity and densities in the Southwest. They found that bird species diversity and density peaked in ponderosa pine forest with small inclusions of aspen. The results of their study highlight the beneficial role that even small, but numerous aspen stands may play in maintaining high bird abundance and habitat connectivity in the Southwest, particularly when viewed at larger forest wide and regional scales.

Although fencing is expensive to build, difficult to maintain, and may need to stay in place for at least 15 years before aspen persistence is likely (Rolf 2001), exclosures to protect aspen from ungulate browsing pressure are critical for aspen recruitment. On the Kaibab Plateau, Binkley et al. (2006) found that a major decline in aspen during the 1920s coincided with high unchecked deer populations during that time. Aspen recruitment was successful in areas that had been fenced or were patrolled by hunting dogs. Other researchers have found that total-exclusion exclosures have significantly positive effects on aspen regeneration, even when disturbance that typically stimulates regeneration is lacking, and/or invasion by conifers is imminent (Kay and Bartos 2000).

In 2007, the KNF initiated efforts to map and assess current conditions of aspen clones across the SZ to ensure future regeneration of aspen. A better understanding of the spatial distribution and condition of aspen stands will help to facilitate better overall management by prioritizing clones for treatment and identifying appropriate actions such as conifer removal, prescribed burning, and/or fencing. Ongoing surveys continue to locate small aspen clones throughout the Williams and Tusayan Ranger Districts. These clones vary in size from less than 1 acre to approximately 4.27 acres in size and are typically found in drainages (W. Rokala, H. McRae *pers. comm.*) Due to their small size and somewhat concealed nature, surveys to detect them are labor intensive. As

of 2008, surveys on the Williams RD had been conducted on 1,273 acres with a total of 298 clones detected with an average clone size of 4.27 acres each (W. Rokala *pers. comm.*). Preliminary results from some surveys suggest that aspen regeneration only occurs in existing elk exclosures, or on rocky faces that are inaccessible to elk. Four new clones, up to 1 acre each in size, were recently located on the Tusayan District with encroachment from surrounding conifers and/or pinyon juniper imminent (H. McRae *pers. comm.*).

Efforts to better understand aspen decline across the KNF include a collaborative project with Northern Arizona University and Rocky Mountain Research Station. Objectives include assessing factors causing aspen decline and whether fencing out ungulates and livestock is an effective strategy for improving aspen vigor and reproduction. The proposed timeline for this project is FY 2008-2011.

Habitat Data

Aspen stands are common across the Williams (2,171 acres) and NK Ranger Districts (26,333 acres) but represent only a trace presence (less than 4 acres) on the Tusayan District (Figure 36). Aspen occurs throughout much of the coniferous forest types on the NKRD. Large aspen clones occur on the taller mountains of the Williams RD and smaller patches occur within the ponderosa pine cover type. The Williams RD in particular is experiencing declines in aspen. Overstory conditions are declining due to weather and defoliation and regeneration is limited by elk browsing and further exacerbated by lack of fire and conifer encroachment.

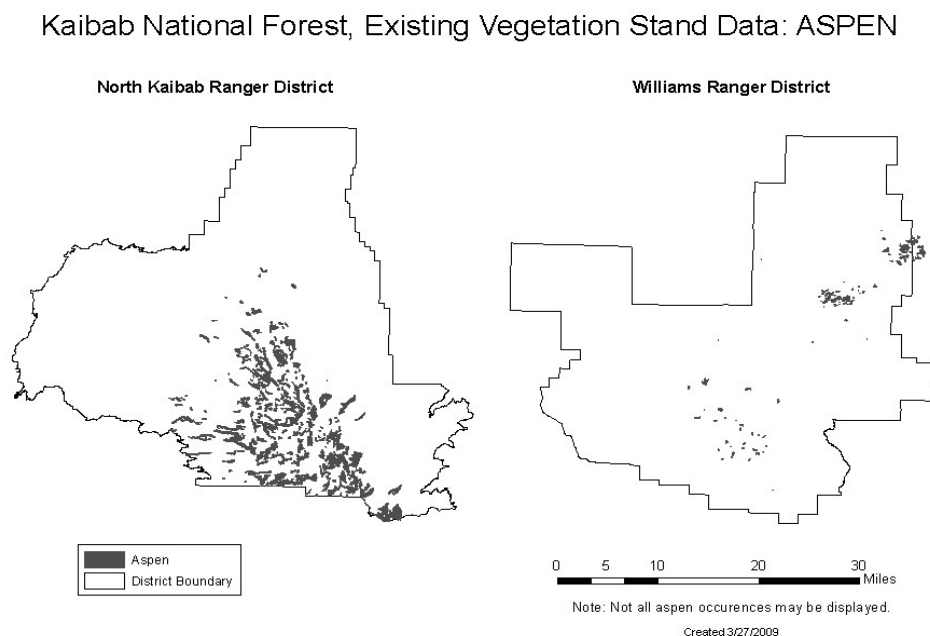


Figure 36. Aspen distribution across the KNF.

FIA data collected in 1995 and 2007 were used to assess aspen habitat on the NKRD. FIA surveys were unable to detect aspen on the South Zone (Williams and Tusayan Ranger Districts; SZ) because aspen is of limited and patchy distribution. There is an overall decreasing trend in aspen density and distribution across the NKRD in all size classes with the exception of medium sized trees (12-17.9 inches DBH) which appear to be static (Table 18, Figure 37). Decreasing trends among the smaller size classes suggest there will be limited recruitment to sustain mature forest in the future. This forest-wide risk for aspen loss is similar to other regional analyses. Between 1962 and 1986 alone, there was a 46% decline in aspen across the Southwest (Dahms and Geils 1997). FIA data from Utah show a 60% decline of aspen since European settlement (Bartos 2007). In Arizona, approximately 95% of the aspen-dominated lands have succeeded to conifers (Bartos 2001).

Table 18: Number of live aspen trees per acre on the North Kaibab Ranger District by diameter class, KNF (FIA unpublished data).

Ranger District	1"-4.9"	5"-11.9"	12"-17.9"	18"-23.9"	>=24"
1995					
North Kaibab	61.73	5.54	1.32	0.00	0.00
2007					
North Kaibab	53.70	3.45	1.44	0.00	0.00

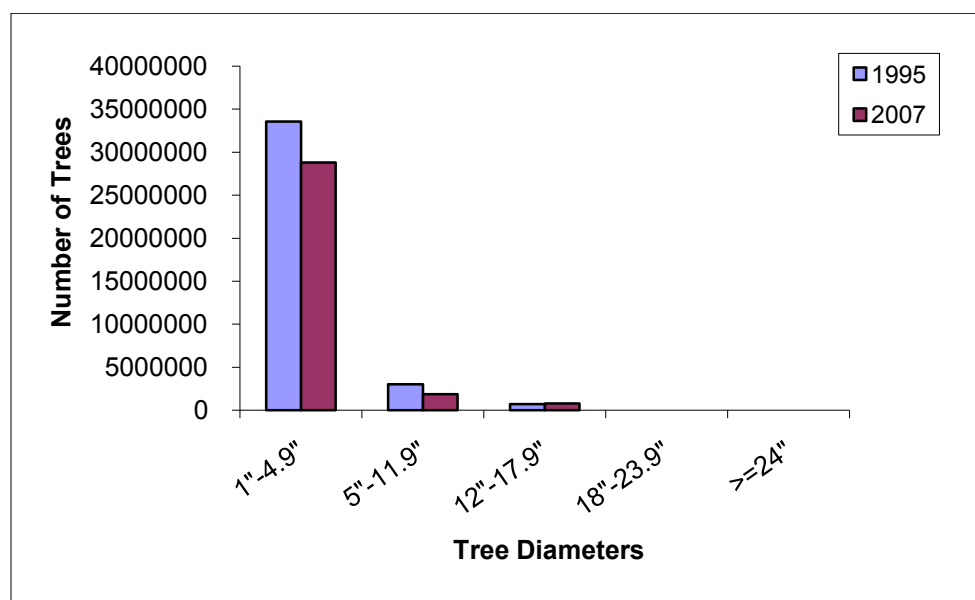


Figure 37: Number of live trees by species and diameter class, North Kaibab (FIA unpublished data).

Habitat Trend

Elk browsing represents a high severity, moderate duration threat to aspen on the Williams RD (Kaibab National Forest 2008a). The loss of regeneration means aspen are declining in the long-term from elk pressure, conifer encroachment, and perhaps climate change. Weather events, insects, and diseases create a negative trend in the short-term. While aspen is adapted to recovering from loss of overstory size classes, continued elimination of regeneration by elk are expected to prevent recovery of aspen from short-term disturbances and substantially reduce aspen presence on the Williams RD in a relatively short period (elk are absent or occur in limited numbers on the NKRd and the goal of the Arizona Game and Fish Department is prevent them from becoming established there). On the south half of the forest aspen reproduction is not growing to an intermediate or mature state because of wild ungulate grazing. Aspen on the north half of the forest is in decline from frequent tent caterpillar defoliation. In general, the trend for aspen is away from historic conditions (Kaibab National Forest 2008a). It seems unlikely such changes in aspen will negatively affect mule deer populations across the forest because mule deer are such generalists. Conversely, red-naped sapsuckers are tied closely to aspen habitat. Habitat for red-naped sapsuckers is expected to remain static forest-wide in the short-term and decrease in the long-term. It is likely that the lack of aspen regeneration and limited recruitment into larger diameter classes (> 10 in. DBH) will provide decreased nesting and foraging opportunities for red-naped sapsuckers in the future.

Snags

Snags are dead or decaying standing trees that may benefit a variety of wildlife species. Hollow cavities in standing dead wood make excellent nests for woodpeckers, while bark dwelling insects provide food. Bats, tree frogs and beetles all make their homes in crevasses in the wood and between bark and trunk. (<http://www.nwf.org/backyard/snags.cfm>). In northern Arizona, exit counts have commonly documented 100 to 300 bats roosting in a single snag. One particular snag was sheltering over 900 bats at once (C. Chambers, pers. comm.).

Background

Snags are ephemeral resources that provide vital habitat for various wildlife species and help to maintain the ecological integrity of the forest through decay and other nutrient cycling processes (Bull 1997, Chambers 2002, Ganey and Vojta 2005, Ganey 2007). In addition, snags provide indirect benefits to forest health by attracting insectivores that help to regulate insect populations (Thomas et al. 1979). Despite their transient nature (both temporally and spatially) across the landscape, snags are used by over 75 species of animals in the Southwest (Chambers 2002, Chambers and Mast 2005), and are particularly important for cavity-nesting birds (Ganey and Vojta 2004, Chambers and Mast 2005, Holden et al. 2006). Snags are especially critical in mixed-coniferous and ponderosa pine forests because of the large component of bird species that utilize those cover types (Ganey 1999). For example, ponderosa pine snags provide more nesting habitat for cavity-nesting species than any other forest type in the Southwest (Chambers and Mast 2005). Over 25 species of birds in northern Arizona use snags for nesting (Waskiewicz 2003). Animals either make their own cavities (primary excavators) or use existing cavities (secondary cavity users). Different species have different needs with regard to snag height and diameter.

A variety of factors determine snag usage and snags in differing stages of decay may appeal to a broad spectrum of wildlife (Lowe 2006). Tree species, diameter, height, density, and spatial dispersion throughout the forest are all factors that may determine what types of wildlife utilize snags (Chambers and Mast 2005). In general, the relevant importance of snags is determined both by quantity and quality, with spatial dispersion playing an equally, if not more important role than snag density (Bull et al. 1997).

The 1987 Forest Plan called for retaining not less than 200 snags per 100 acres across 65% of the landscape within ponderosa pine forest. A snag was defined as 14 inches DBH or greater and at least 15 feet in height. This definition was modified in 1996 to 18 inches DBH and at least 30 feet in height. The amended Forest Plan now calls for retaining 2 snags per acre on all acres within the ponderosa pine cover type. In mixed-conifer and spruce-fir habitat, the amended Forest Plan prescribes 3 snags per acre. The direction in woodlands (pinyon-juniper) is to provide for not less than 100 snags 9 inches and larger DRC and 12 feet in height per 100 acres over 65 percent of the forested area. At least four snags ≥ 20 in DBH per acre are to be provided for osprey near water sources. Under the Mexican Spotted Owl Recovery Plan, which covers the pine-oak portion of the pine cover type on the Williams RD and the mixed conifer cover type on the NKRD, “substantive” amounts of snags are to be retained. In practice, this becomes an effort to avoid losing snags due to fire and management activity. In the remaining ponderosa pine forest, the Recovery Plan calls for 1 snag per acre of at least 14 inches DBH, and 15 or 25 feet in height minimum, depending on site class conditions. Density of aspen snags across the forest is not determined in the current Forest Plan; however maintaining snags of at least 10 inches DBH is desirable as large-diameter trees are preferred by cavity nesting species (Bull 1997).

Management indicators for snags include hairy woodpeckers (ponderosa pine, mixed-conifer, and spruce-fir forest types), juniper titmice (pinyon-juniper woodland), and red-naped sapsuckers (aspen). In addition, pygmy nuthatches use snags for communal roosts and were selected to represent late-seral ponderosa pine.

Potential Management Implications

Snags tend to be a transitory, limited resource with a patchy distribution, making them difficult to sample. The parameters that influence insects, disease, and fire patterns across the landscape also influence the distribution of snags. Thus presence of snags on the landscape is also variable. Areas protected from wind or areas with good soil development can promote snag longevity because they are typically less prone to wind throw. Conversely, areas closer to residential communities and along forest roads are vulnerable to illegal cutting by firewood collectors. “Snag poaching” appears to be a significant issue on the NKRD, even though the human population near the Forest boundary is relatively low. The Forest Plan makes provision for commercial and personal-use fuelwood as a cost effective way to improve forest health and wildlife habitat. Most fuelwood programs emphasize the collection of dead and downed pinyon or juniper trees. However, standing dead trees may be cut under the following criteria: mixed conifer species (ponderosa pine, Douglas-fir, spruce) less than 12 inches DBH; aspen less than 12 inches DBH; and pinyon-juniper less than 10 inches DBH.

Fire and Snags

The increased intensity of forest fires in the southwest may be damaging to wildlife species that depend on snag habitat. Although wildfires can create additional snags, they may simultaneously destroy existing snags. In the short-term, slower decay rates of partially burned snags may make them less desirable for cavity-nesters because they are more difficult to excavate (Chambers and Mast 2005). In recent years, the Forest Service has increased Wildland Fire Use fires. These naturally ignited fires are allowed to burn under managed conditions for the benefit of forest health. Naturally ignited fires are critical for forest restoration efforts and for reducing fuel loads. Snag densities however, have been shown to decrease as a result of repeated, prescription burns in ponderosa pine forests in Arizona. Bagne et al (2008) found that in areas that had experienced fire suppression for long periods of time, the greatest loss of snags occurred during initial burns. In the long-term however, the rate of loss decreased and leveled off as new snags were created during each subsequent burn. Holden et al. (2006) found that low-intensity repeated burning may not kill living trees, but may threaten already weakened snags that lack a protective cambium. They found that snag densities in the Gila National Forest (New Mexico) were significantly lower in areas that had been burned 2-3 times since 1946 [range was 1.6 - 2.8 snags/acre (4 - 7 snags/ha)] compared to areas that had burned once. They also found that following multiple burns, large ponderosa pines had the highest survival rates.

Although prescribed fires may promote recruitment of large snags, treatments that maintain a mixture of both small and large diameter trees, both burned and unburned, for foraging and nesting are optimal. In northern Arizona, Chambers and Mast (2005) compared snag retention in severely burned and unburned areas on the Coconino and KNFs. They found greater densities of large diameter snags in unburned plots vs. burned plots and that greater surrounding basal area (of live and dead trees) around snags increased the likelihood of burned snags to remain standing. Unburned areas retained more nesting habitat (cavities) while burned areas maintained more foraging habitat.

Availability of usable snags can be promoted by management activities that promote the retention of large trees, both living and dead, and that select for preferred species such as ponderosa pine, Gambel oak and quaking aspen within mixed forest types (Ganey and Vojta 2004). Treatments such as thinning and burning can facilitate snag retention. For example, Waskiewicz (2003) studied snag recruitment in areas that had been thinned and burned and found lower recruitment rates in untreated areas [.73 new pine snags per acre (1.8 per ha) vs. treated areas (5.5 snags per acre (13.5 per ha)]. He also found higher densities of newly created foraging trees in treated vs. untreated areas. Snags greater than 14.8 inches (37.5 cm) DBH had more longevity than those in stands dominated by smaller diameter trees.

Fire has the potential to create snags in excess of the Forest Plan Guidelines. In controlled burns, efforts are made to avoid burning large DBH trees. Years of fire suppression have allowed the duff layer (principally dead pine needles) to build to the point that large trees, i.e., the trees most resistant to fire, are inadvertently killed. This lowers the potential for recruiting new trees into the larger size-classes but does create additional snags. Experience on the KNF indicates that about 50% of the large snags survive controlled burns with pretreatment (putting fire lines around large snags before burning). Not burning allows the duff and fuel loads to continue building, thereby risking even higher loss from future fires. Wildfire creates snags, but can also destroy most of the forest structure in the process. The resulting snags are then suitable for only a

portion of the wildlife species that use snags (e.g., hairy woodpeckers) and even then use can be limited in interior portions of high severity burns. Covert-Bratland et al. (2006) found that in high-severity burn areas in ponderosa pine forest, hairy woodpeckers preferred to forage within 82 feet (25m) of the forest edges 70% of the time. In general, the wildlife value of fire-created snags will not increase until the surrounding habitat develops. There is also a question on the longevity of fire-created snags. A preliminary look at snags marked after the Bridger-Knoll fire the NKRD indicate that, after five years, 20% of the snags had fallen and 8% were missing (stumps associated with eight snags indicate they were cut for firewood) (C. Chambers, *pers. comm.*). Standing snags averaged 20 inches DBH and percent scorch (amount of bark burned) averaged 63% (range 0 to 100%). Similarly ponderosa pine snags fell at increasing rates approximately 4-7 years following the Pumpkin Fire (Williams RD) with 14% of snags falling after 3 years of the fire. In general smaller snags (< 23 inches DBH) experienced greatest fall rates during the first 7 years post fire, suggesting limited recruitment in the future. Fire-killed snags have a relatively short life and over 40% are expected to be on the ground within 7 years (Chambers and Mast 2005). Over 80% of ponderosa pine snags created by high severity fire fall by 10-years post fire (Chambers and Mast *pers. comm.*) In summary, while wildfire has the ability to create snags, the value of such habitat for wildlife will vary depending on the size and severity of the burn, and the resulting effects on the surrounding forest.

Salvage Logging

Salvage logging may have detrimental effects on the nesting success of cavity-nesting species because it can negatively affect size class diversity, species composition and density. Russell et al. (2006) assessed ponderosa pine and Douglas-fir snags in logged and unlogged forests following two different burns in Idaho. They found that overall Douglas-fir was more resilient than ponderosa pine: Nine years post-fire, 90% of Douglas-fir was still standing as opposed to 45% of ponderosa pine in unlogged forest. For a partially logged fire, 85% of Douglas-fir remained standing while only 25% of ponderosa pine snags were still standing. This may be a result of slower decay rates in Douglas-fir heartwood than in ponderosa pine (Bull 1997). Russell et al. (2006) also found that in both logged and unlogged areas, larger snags and forests with higher snag densities had higher snag survival rates than smaller snags in exposed forests. Over an 11 year period they found that snag recruitment was lower in burned/logged forest than in areas where recruitment occurs as a result of more gradual, natural processes such as decay, insects, and disease. Chambers and Mast (2005) recommend that in the southwest, salvage logging operations in ponderosa pine should retain straight, large diameter snags in clumps to maintain desirable habitat for cavity-excavating species. This approach can create habitat for some cavity-nesting species (Saab et. al. 2008).

During early implementation of the Forest Plan, a combination of factors led to a disproportionate harvest of trees in the larger size-classes. Since the early 1990s, harvest of large ponderosa pines trees has been reduced and trees with obvious structural “defects” are no longer targeted for harvest. Large snags cannot be created without large trees first occupying the site and there is concern about retaining adequate habitat for cavity-nesting birds while the mid-sized trees grow into the larger size classes (Miller and Benedict 1994). There are indications that “partial snags,” i.e., trees with dead tops or lightning strikes, may have historically provided snag habitat for cavity nesting birds during frequent fire return intervals (B. Block, C. Chambers, and J. Ganey *pers. comm.*). Partial, or green snags, are living trees that may provide the same habitat

benefits for wildlife as fully dead snags with the advantage of having bark that is more resistant to wildfire. This may be particularly important for southwest forests as restoration treatments such as fuel-reduction and prescribed fire become more prevalent. Waskiewicz (2003) found that in forests treated with prescribed fire, partial snags were more likely to remain standing than full snags. Treated areas experienced snag recruitment rates 7.6 times more per 2.47 acres than controls providing short-term benefits.

Although there have not been surveys for this particular habitat component, discussions are underway on the KNF on how to incorporate this feature into the standard stand examinations surveys. Retaining “defect” trees, such as partial snags and lightning strike trees, rather than deliberately harvesting is commonly adopted at the project level and should directly benefit cavity-nesting birds.

Management Synopsis

Some research suggests that current Forest Plan specifications for snag retention, which are in part driven by the goshawk guidelines, may be unrealistic. As a result of their patchy distribution, Stephens (2004) suggests that management of snag densities should occur over hundreds of acres, rather than on a per acre basis. Several studies support these recommendations. Habitat models developed by (Ganey 2007), suggest that although densities of large snags in ponderosa pine and mixed-conifer forest will increase in the short-term, they will continue to remain below USFS planning thresholds. Ganey (1999) found that even on un-logged sites, only 30% of ponderosa pine plots met or exceeded USFS snag guidelines. Waskiewicz (2003) also found pine snag densities to be well below USFS recommendations even in relatively undisturbed forest in northern Arizona where management activities have been limited. Even if forests maintain the recommended snag densities, not all snags will provide suitable nesting habitat. Ganey and Vojta (2004) found that only 17% of snags in ponderosa pine forests actually contained nesting cavities. Similarly, Zack et al. (2002) cited in Lowe (2006) surveyed 1,812 snags in pine-white fir and found that while there were 6.4 snags per acre, only 2.4 snags per acre actually contained nesting cavities. In summary, there is a growing body of research that suggests factors such as quality, recruitment, and spatial distribution may be more important criteria than snag per acre criteria (Bull et al. 1997, Ganey 1999, Ganey and Vojta 2007, Bagne et al. 2008). Future management strategies might incorporate surveys of nesting cavity quality and quantity. Recent research by Ganey and Vojta (2007) suggests that modeling snag population dynamics in mixed-conifer and ponderosa pine forests may serve as an innovative tool for assessing snag dynamics at the landscape scale. SnagPRO, a new program developed by the Pacific Northwest Research Station (Bate et al. 2008) was developed to assist with effective field survey design and data analysis across various spatial scales. It can be downloaded from the following website: <http://www.fs.fed.us/pnw/publications/tools-databases.shtml>.

Habitat Data

Snags in Coniferous Forests

Several local studies have been done on snags on or near the KNF. Miller and Benedict (1994) found an average of 0.6 ponderosa pine snags (12 inches DBH or greater; range equaled 0 to 3.5) per acre on 100 randomly selected 4-acre plots. Ganey (1999) found a median of two snags per acre (range equaled 0 to 18) on the Kaibab and Coconino National Forests. The Forest Inventory Assessment found 0.6 ponderosa pine snags per acre that were 19 inches DBH and larger across

Arizona forests in 1995 (O'Brien 2002). For that same assessment, there was an average of 2.9 snags per acre greater than 11 inches DBH on the KNF, chiefly comprised of Utah juniper and two needle pinyon (Frescino 2003). By comparison, repeat FIA surveys completed in 2007 found 6.8 snags per acre across the forest.

Lang and Stewart (1910) surveyed snags on the Kaibab Plateau. They reported total biomass for dead trees and calculated that there were about 0.16 snags per acre in the 18 inch and larger DBH size classes. These data, combined with the habitat represented by partial snags, suggests the NKRD is well above historic levels of large snags (Figure 38).

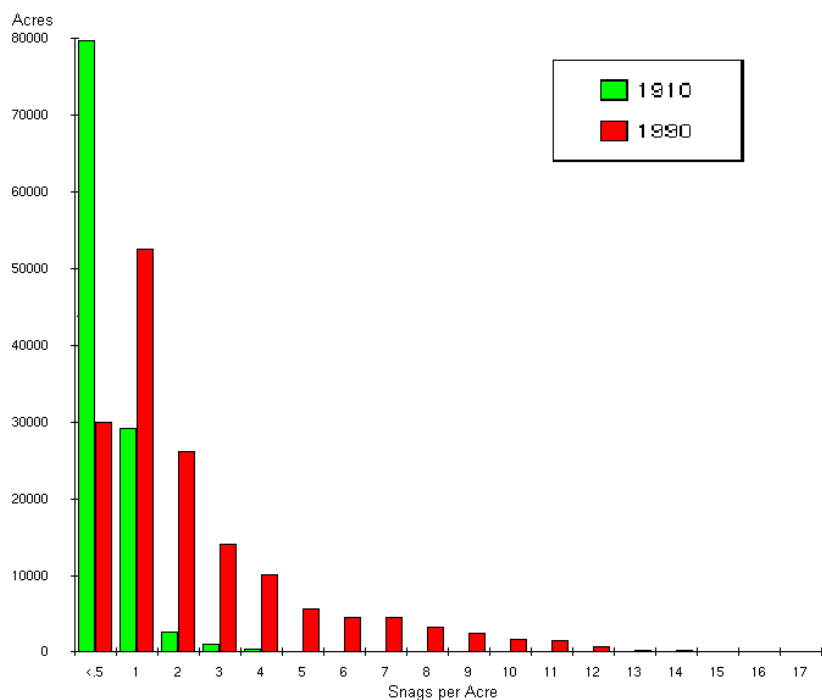


Figure 38. Snag densities from 1910 and 1990, North Kaibab Ranger District. The numbers for 1910 are inferred from inventoried snag biomass translated to trees per acre using stand tables of trees per acre by size class included in the 1910 report. Snags are more prevalent across 130,000 acres of the North Kaibab RD now than in 1910.

FIA surveys completed in 1995 and 2007 show an overall increase in ponderosa pine and mixed-coniferous forest snag density across the KNF. In general, there is an increasing trend for all size classes (Tables 19-20). However, the greatest increase is apparent among smaller sized trees.

These data are supported by the work of Ganey and Vojta (2005). They also found overall increases in snag abundance and recruitment for all size classes in both forest types with the most significant increases among smaller sized trees (< 15.7 inches DBH), particularly in mixed-coniferous forest. The snags in their study were either young trees, or older dying trees that had been suppressed. This pattern has been noted elsewhere for Douglas-fir and ponderosa pine and may likely be a result of competition for limited resources in overly dense stands. An increase of larger diameter trees is more evident in spruce-fir forest (Table 21). This difference is perhaps a result of the differing fire regimes and management strategies that characterize these forest types.

Table 19. Number of dead Ponderosa pine trees per acre on forest land by Ranger District and diameter class, Kaibab National Forest (FIA unpublished data).

Diameter class			
Ranger District	5"-10.9"	11"-14.9"	>=15"
1995			
North Kaibab	1.12	0.00	0.14
Tusayan	0.39	0.00	0.11
Williams	0.99	0.00	0.24
<i>Totals</i>	2.49	0.00	.49
2007			
North Kaibab	2.49	0.29	0.00
Tusayan	0.33	0.16	0.33
Williams	2.18	0.60	0.79
<i>Totals</i>	5.00	1.50	1.2

Table 20. Number of dead mixed conifer trees per acre on forest land by Ranger District and diameter class, KNF. Mixed conifer component includes white fir, Douglas-fir and blue spruce (FIA unpublished data).

Diameter class			
Ranger District	5"-10.9"	11"-14.9"	>=15"
1995			
North Kaibab	1.73	0.48	0.28
Tusayan	0.00	0.00	0.00
Williams	0.00	0.00	0.00
<i>Totals</i>	1.73	0.48	0.28
2007			
North Kaibab	1.72	1.15	0.57
Tusayan	0.00	0.00	0.00
Williams	1.49	0.10	0.40
<i>Totals</i>	3.21	1.26	.97

Table 21. Number of dead Spruce-fir trees per acre on forest land by ranger district and diameter class, KNF. Spruce-fir forest consists of Engelmann spruce and subalpine fir (corkbark; FIA, unpublished data).

Diameter class			
Ranger District	5"-10.9"	11"-14.9"	>=15"
1995			
North Kaibab	1.11	0.07	0.07
Tusayan	0.00	0.00	0.00
Williams	0.00	0.00	0.00
<i>Totals</i>	1.11	0.07	0.07
2007			
North Kaibab	0.38	0.00	0.10
Tusayan	0.00	0.00	0.00
Williams	0.00	0.00	0.00
<i>Totals</i>	0.38	0.00	0.10

Snags in Aspen

Longevity of aspen snags is relatively limited. Aspen decay and topple much quicker than coniferous snags and firewood gathering contributes to the loss of snags. The amount of decay in a stand increases with stand age, but stands tend to begin deteriorating after 100 years with most snags standing an average of 10.7 years after death. (reviewed *in* Hart and Hart 2001). The potential for future large DBH aspen snags is limited given the current status of aspen health. A review of primary cavity nesters in aspen forests found that the most successful nests were located in living trees (Struempf et. al. 2001). Hart and Hart (2001) found that, although the percentage of living aspen with heart rot was low in the stands they studied, most cavity excavators seemed to select for trees with heart rot. Conks of *Phellinus tremulae* were present in 71% of all cavity trees, but were present on only 9.6% of the total aspen examined (Hart and Hart 2001). Their results suggest that heart rot may be more critical than aspen snags. Some researchers have shown that red-naped sap suckers for example, use live aspen more often than dead for nesting in Oregon and Colorado (cited *in* Hollenbeck and Ripple 2008).

Current research suggests that reducing elk numbers could facilitate recruitment of larger diameter aspen snags. Hollenbeck and Ripple (2007) found that aspen stands with large diameter snags (≥ 7.9 inches DBH) were 80% greater within Yellowstone NP than aspen stands outside the park that were subject to more browsing pressure from elk. Inventory data for the KNF suggest that in general, aspen snags are declining across the forest (Table 22). This is not surprising given the overall decline of aspen forest on the KNF (refer to the section on aspen cover type for more details).

Potential conflicts may arise particularly as a result of aspen harvest, due to the low recruitment rate and paucity of aspen snags in larger size classes.

Table 22. Number of dead aspen trees per acre on forest land by Ranger District and diameter class, KNF (FIA, unpublished data).

Diameter class			
Ranger District	5"-10.9"	11"-14.9"	>=15"
1995			
North Kaibab	3.12	0.49	0.21
Tusayan	0.00	0.00	0.00
Williams	0.00	0.00	0.00
<i>Totals</i>	3.12	0.49	0.21
2007			
North Kaibab	2.01	0.29	0.00
Tusayan	0.00	0.00	0.00
Williams	0.00	0.00	0.00
<i>Totals</i>	2.01	0.29	0.00

Snags in Pinyon-Juniper Woodlands

FIA data from two survey periods on the KNF show an increasing trend in snag density in pinyon-juniper habitat for all size classes across the KNF. The most notable increase is for trees between 5 and 10 inches in diameter (Table 23). This trend far exceeds the FS plan components for old growth habitat that specifies retaining between 0.5 and 1 snag per acre.

Table 23. Number of dead pinyon-juniper trees per acre on forest land by Ranger District and diameter class, KNF (FIA, unpublished data).

Diameter class			
Ranger District	5"-10.9"	11"-14.9"	>=15"
1995			
North Kaibab	3.27	0.87	0.62
Tusayan	5.72	3.13	0.82
Williams	1.82	0.74	0.77
<i>Totals</i>	10.81	4.74	2.21
2007			
North Kaibab	12.07	2.87	1.53
Tusayan	3.45	1.48	0.99
Williams	2.99	0.80	1.00
<i>Totals</i>	18.71	5.15	3.52

Examination of these data by species highlights the loss of the pinyon component within the pinyon-juniper woodland cover type (Figure 39). Common pinyon snags increased by about 6,332,500 trees from 1995 to 2007 (5 snags per acre to 14.7 snags per acre). A total of approximately 205,600 (0.31 snags per acre) snags of dead, single leaf pinyon were detected in 2007 with none detected in 1995. This wide-spread die off of pinyon has been attributed to increased bark beetle epidemics, discussed in greater detail in the Pinyon-Juniper Woodlands Cover Type and PNVN section.

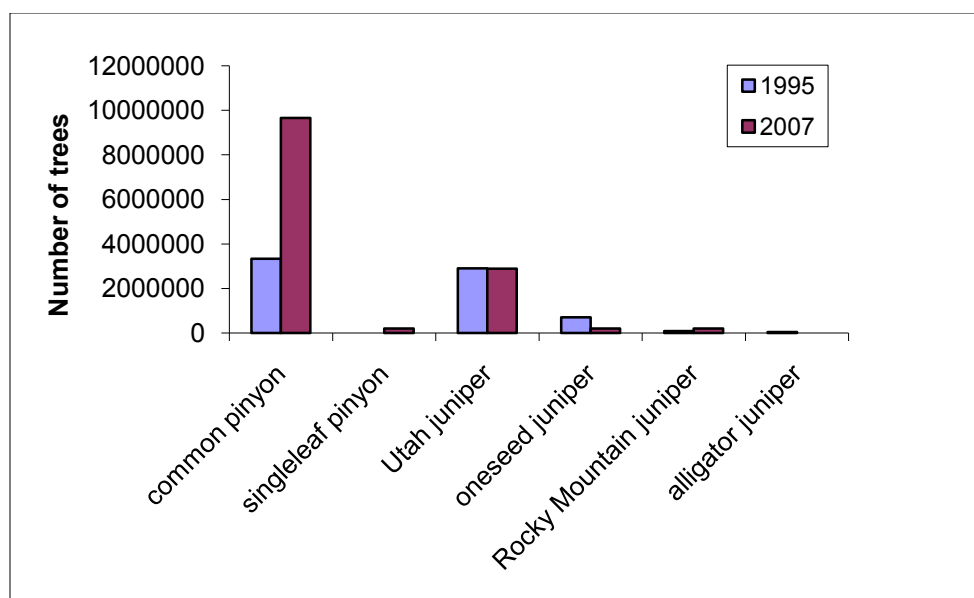


Figure 39. Number of dead trees by species in pinyon-juniper woodlands across the KNF (FIA, unpublished data).

Habitat Trend Assessment

Snags should continue to increase throughout the Southwest given increasing drought and insect outbreaks (Breshears et al. 2005, Ganey and Vojta 2005). The outlook for snags in mixed coniferous forests, ponderosa pine and pinyon-juniper is generally favorable, although achieving Forest Plan standards may be unrealistic in all forest types. In the long-term, climate change and increased insect damage should all favor snag recruitment. However, stand replacing fires and increases in prescribed burning could prove detrimental to retaining persistent, high quality snags for wildlife.

The long-term prognosis for aspen is poor due to low levels of recruitment resulting from overstory dieback browsing pressure preventing clonal regeneration. The potential for recruitment of large aspen snags is limited. Retaining living trees with heart rot however, may be more critical than retaining existing aspen snags. Whether or not cavity nesters will be able to utilize live aspen in stands that lack suitable snags will depend on the plasticity of such species.

Drought and beetle activity are dramatically increasing pinyon snag availability across pinyon-juniper woodlands. The resulting pulse of pinyon snags will be at the cost of future pinyon trees and pinyon snag availability. Although increased mortality of pinyon is creating an abundance of snags in the short-term, the long-term expectation is for lower levels of pine snags in the pinyon-juniper woodlands.

In summary, we expect the increasing trend in snag numbers to continue in the pinyon-juniper, ponderosa pine and mixed-conifer cover types. The trend for spruce-fir forest is less clear, but may suggest limited recruitment in the smaller size classes. While aspen and pinyon snag numbers may increase in the short-term, we expect a decreasing trend in the long-term. A caveat

is that uncharacteristic, large scale, high severity fires and insect epidemics resulting from unnaturally dense forest stands and climate change can alter these predictions.

Mixed-coniferous Forest Cover Type and PNV

This forest type is more lush and diverse than ponderosa pine forests. Historically, as a result of increased moisture levels at higher elevations, mixed-coniferous forests were relatively open and contained a diversity of tree species, as well as understory shrubs, forbs, and grasses. Past fire regimes in mixed-conifer have been variable due to diversity in forest composition, topography, and human intervention. Recent fire suppression efforts have greatly altered the structure and composition of this forest type throughout the West (<http://cpluhna.nau.edu/Biota/mixedconifer.htm>).

Background

Like spruce-fir, mixed-coniferous forest is a minor component of the southwestern landscape. It comprises approximately 3% of the total land cover in Arizona (Moir and Ludwig 1979, TNC 2006). On the Colorado plateau, mixed-conifer forests typically occur from 8,000-10,000 feet in elevation. Douglas-fir (*Pseudotsuga menziesii* var. *glauca*) is typically co-dominant with white fir (*Abies concolor*) blue spruce (*Picea pungens*) and white (limber) pine (*Pinus strobiformus*). Mixed-coniferous forest is bordered by ponderosa pine forest at its lower limits, and spruce-fir forest at its upper limits. Ponderosa pine, quaking aspen, and gambel oak may also be present (Graham and Sisk 2002, Battaglia and Shepperd 2007). As a result, mixed-conifer forests typically mark the transition zone between surface fire (in ponderosa pine) to stand-replacement (in spruce-fir) fire regimes (Mast and Wolf 2006). Characterized by mixed severity fire regimes, the temporal and spatial variability of fire in mixed-coniferous forests may largely be a function of the diverse topography they inhabit, years of livestock grazing, and fire suppression (Beaty and Taylor 2007, Sakulich and Taylor 2007). On the Kaibab Plateau, several studies have concluded that pre-settlement mixed-coniferous forests were subjected to numerous mixed severity fires, creating both open and dense forest stands (Lang and Stewart 1910, Fulé et al. 2002, Fulé et al. 2003a).

On the KNF, mixed-conifer (~39,122 acres) occurs at high elevations on the Williams and NK Ranger Districts. The central portion of the NKRD accounts for



Figure 40. Mixed-conifer PNV.

the majority of the mixed-conifer cover type on the KNF. The mixed-conifer PNVNT encompasses 128,000 acres of forest (about 8% of the forest) and includes dry mixed-conifer and mixed-conifer with aspen (Figure 40). The higher acreage of the PNVNT reflects both dry and mesic forest types which were too difficult to differentiate based upon the sampling and modeling methods used during the forest plan revision process. Despite the difference in acreage totals, threats for both cover type and PNVNT are similar and have been treated the same in this discussion.

Mexican spotted owls are the MIS associated with late seral mixed-conifer forest and elk and tassel-eared squirrels represent early seral forest.

Potential Management Implications

Fire

Fire exclusion has been the most dramatic factor affecting the integrity of high elevation mixed-conifer forests throughout the Southwest. On the Kaibab Plateau, historic conditions suggest that dry mixed-conifer forests behaved similarly to ponderosa pine, characterized by low intensity ground fires and few stand replacing events (Fulé et al. 2004). The average area covered by fires in mixed-conifer forest was approximately 15 acres, with stand replacement events ranging from 100 to 240 acres (Fulé et al. 2003b). Like ponderosa pine forests, fire exclusion has increased fuel loads and primed conditions for mortality by drought and insect epidemics (Mast and Wolf 2004, Beaty and Taylor 2007, Sakulich and Taylor 2007). As with ponderosa pine, stand replacing fires became prevalent in the late 1900s with ~25,408 acres of mixed conifer forest having burned on the KNF since 1993.

Insects and Disease

In addition to fire, insects and disease are playing an increasingly influential role in mixed-coniferous forest. White pine blister rust results from years of fire suppression and is prolific in mixed-coniferous forests (Parker et al. 2006). Host-specific insects such as Douglas-fir beetles (*Dendroctonus pseudotsugae*), fir engravers (*Scolytus ventralis*), and western spruce budworm (*Choristoneura occidentalis*) can drastically influence forest structure and composition (reviewed in Battaglia and Shepperd 2007) and are the greatest insect threats to mixed-conifer forests on the KNF (Lynch 2008). In the Southwest, years of fire suppression has led to multistoried forests dominated by shade tolerant white fir and shade intolerant Douglas-fir, two tree species most susceptible to attacks by western spruce budworm (Lynch et al. 2008). Douglas-fir in particular may suffer severe mortality from beetle attack. Parker et al. (2006) found Douglas-fir can be severely affected in even moderately burned areas. If beetle populations reach high enough levels, they may attack unburned trees, posing significant risks to forest health.

Dwarf mistletoe, or “witches’ broom” (*Arceuthobium douglasii*), common in mixed-conifer forests throughout the southwest, is frequently targeted for removal because it increases fuel loads and decreases tree quality for timber production. Mistletoe may be well above historic levels in Southwest forests due to more than a century of fire exclusion, but studies have shown that brooms may provide important microhabitat for wildlife, including MIS species such as Mexican spotted owls, northern goshawks, and red squirrels (Garnett et al. 2004, Garnett et al. 2006, Hedwall and Mathiasen 2006). Hedwall et al. (2006) studied red squirrel use of witches’

brooms in Arizona and New Mexico and found that approximately 67% of brooms examined ($n = 706$) were used by wildlife. Squirrels used brooms more than birds or other mammals. Brooms located close to the trunk were most often selected for nesting and cover.

Management Synopsis

A program that combines prescribed burning and thinning of fire intolerant species such as white fir may help to reverse the current trend toward stand replacement fires (Mast and Wolf 2004). Fulé and Laughlin (2007) found that prescribed wildfire can move ecosystems closer to their historic reference conditions even when they have experienced a century of fire suppression. Prescribed fire may also help to increase understory plant diversity, particularly in forests with limited (anthropogenic) disturbance history. In a little grazed and never harvested forest in Grand Canyon NP, Huisinga et al. (2005) found that high intensity prescribed fire resulted in decreased overstory canopy cover and significantly greater herbaceous plant cover (mean change = + 13% cover) in burned plots vs. unburned plots (mean change = -5%). Species richness of native plants was also greater in burned plots, with a minimal increase (+ 1%) of exotics.

Silvicultural prescriptions that promote a diversity of successional stages should also help to decrease the likelihood of stand replacing crown fires. Reduction in shade tolerant species such as white fir and retention of fire tolerant species such as Douglas-fir and ponderosa pine should prove beneficial to this cover type (Battaglia and Shepperd 2007). These changes should also help to offset the severity of western spruce budworm outbreaks which are most prevalent in fir dominated forests (Parker et al. 2006). However, timing of such treatments should be carefully planned as to minimize short-term increases of bark beetles and wood boring invertebrates that react favorably to slash removal activities (reviewed in Pilliod et al. 2006). In Douglas-fir and ponderosa pine forests, Machmer (2002, *in* Pilliod et al. 2006), found that following prescribed fire, bark beetles and wood borers increased by 170%. This number decreased to 54% for treatments which combined thinning and burning, and was only 45% for thin only treatments. Current knowledge about the effects of such restorative treatments on beetle activity is still evolving, but is generally perceived as favorable for long-term stand health. Treatments that result in grouping and clumping of trees should also slow the spread of mistletoe.

Current management does seek to move toward desired conditions by thinning small trees or using group selection cuts, typically 0.5 to 2 acres in size, to promote structural regeneration and greater structural diversity. Despite these efforts, landscape level processes may ultimately exert more influence on fire regimes in mixed-coniferous forest than local factors. Beaty and Taylor (2007) suggest that spatial variation in fuel may control fire occurrence at the stand scale, but that regional climate variation caused by landscape level phenomenon (e.g. drought, El Nino) affects the synchrony of fires among stands and overall forest structure.

Ongoing management programs that include thinning and prescribed fire should improve foraging habitat for elk (Pilliod et al. 2006). The noise and disturbance associated with such treatments could negatively impact some habitat (e.g. food caches) for red squirrels in the short-term (Hedwall et al. 2006, Pilliod et al. 2006), but are not expected to overtly impact squirrel nest trees and should in fact increase the sustainability of squirrel habitat in the long-term.

Habitat Data

Since the signing of the Forest Plan, timber harvest in the mixed-conifer forest type has been minimal (no more than 200 acres) and limited to noncommercial thinning on the SZ (M. Herron, *pers. comm.*). Tree recruitment rates changed with the shift in Forest Plan objectives in 1996. Trees greater than 24 inches DBH are still cut when site specific evaluations indicate an abundance of trees in that size-class, but the focus of most harvests since the 1990s has been trees less than 24 inches DBH. Silvicultural prescriptions are based on group selection cuts typically applied to areas of one-half to two acres in size. The goal is to create heterogeneous habitat that mimics natural forest patch dynamics that occurred in these forests during the presettlement era.

There continues to be an increasing trend with regard to forest density and number of trees in this cover type. These increases are most notable in the smaller diameter classes (< 5 inches DBH; Table 24, Figure 41). Reference conditions for the mixed-conifer PNVT are uneven-aged forest with an open canopy (< 32%) and a mix of size classes and developmental stages (Kaibab National Forest 2008a). Mixed-conifer forest is trending toward younger more dense forest than occurred during historic times with 72% of the canopy closure greater than 30%. This tracks well with studies by Fulé et al. (2002, 2003, and 2004). On the North Rim of the Grand Canyon, they found that since 1880, tree densities in mixed-conifer forest have increased from 98 trees/acre (242 trees/ha) to 354 trees/acre (873 trees/ha) and that basal areas have increased from 76.5ft²/acre (17.6 m²/ha) to 169.1ft²/acre (38.8 m²/ha). Individual species increased 743±993% (ponderosa pine) to nearly 1500% (white fir) due to fire exclusion on the Kaibab Plateau (Fulé et al. 2002). Given the current trajectory, they also project that bulk canopy density would change from 0.003 lb/ft³ (0.041 kg/m³) in 1880 to 0.006 lb/ft³ (0.094 kg/m³) by the year 2040, a 129% increase.

Table 24. Number of live mixed-conifer trees per acre on forest land by Ranger District and diameter class, Kaibab National Forest (FIA unpublished data).

Ranger District	Diameter class				
	1"-4.9"	5"-11.9"	12"-17.9"	18"-23.9"	>=24"
1995					
North Kaibab	21.78	9.58	2.15	0.98	0.35
Tusayan	0.00	0.00	0.00	0.00	0.00
Williams	0.85	1.71	0.27	0.34	0.00
<i>Totals</i>	<i>22.63</i>	<i>11.29</i>	<i>2.42</i>	<i>1.32</i>	<i>0.35</i>
2007					
North Kaibab	26.25	10.73	3.64	1.53	0.77
Tusayan	0.00	0.00	0.00	0.00	0.00
Williams	1.24	0.89	0.20	0.20	0.10
<i>Totals</i>	<i>27.49</i>	<i>11.62</i>	<i>3.84</i>	<i>1.73</i>	<i>0.87</i>

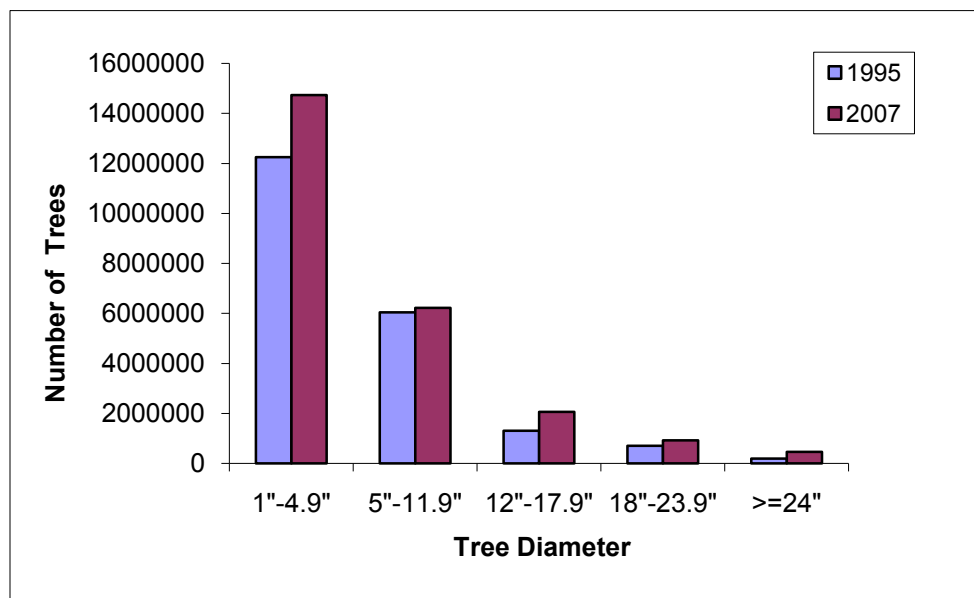


Figure 41. Number of live trees in mixed-conifer forest by size class across the KNF. Includes white fir, Douglas-fir, white pine, and blue spruce (FIA, unpublished data).

FIA data also show changes in species composition with white fir increasing and Douglas-fir decreasing. Fire tolerant white pine, which occurs in low numbers, appears static (Figure 42). Several recent studies support these trends. Mast and Wolf (2004, 2006) looked at historic spatial patterns of mixed-conifer forest on the north rim of the Grand Canyon. They also found increases in density among smaller diameter classes particularly among white fir. Their data suggest that after 1920, there was a compositional shift from ponderosa pine to white fir with 81% of white fir on the north rim established since that time. Most recently, Sesnie et al. (2009) compared historical inventory data from 1909 to that of 1990 on the NKRD and found that tree densities have become 10 times greater than during historic time. They also found that basal areas have doubled and that shade tolerant species such as spruce and true fir have experienced the largest increases: 17- 41 ft²/ac for true-fir and 10 - 42 ft²/ac for spruce fir.

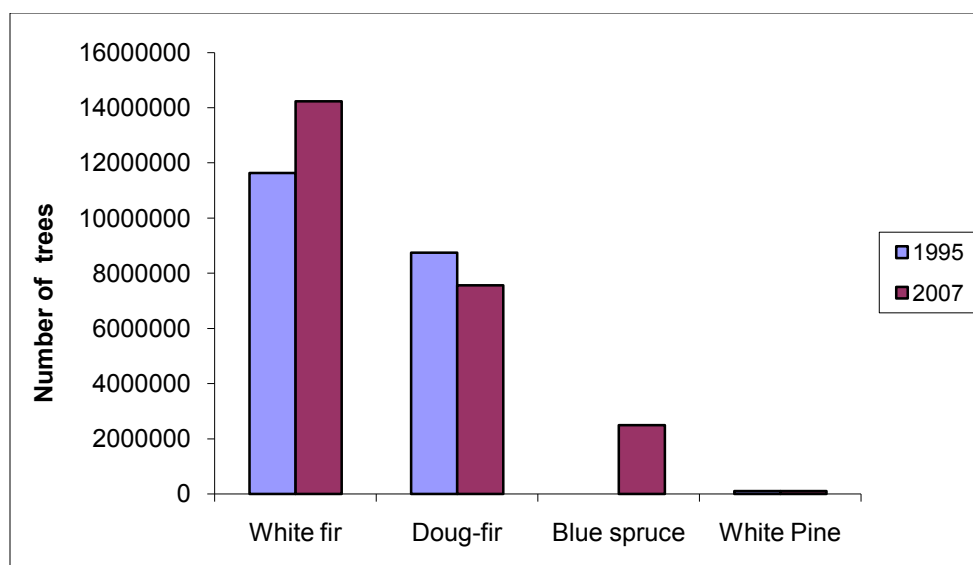


Figure 42. Species composition in mixed-coniferous forest across the KNF (FIA, unpublished data).

Wildfire will likely continue to be an influence on mixed-conifer/spruce-fir forests on the KNF. Recently in 2006, the Warm Fire affected approximately 7,433 acres of mixed-conifer, with approximately 18% being high severity burn (Appendices 7 and 8). The Point and Pumpkin fires in 1993 and 2000 included about 16,800 acres of mixed-conifer/spruce-fir forest (Table 25). About 3,450 of these acres were stand-replacing events. These acres are now broad areas of early seral stage communities. The Point and Pumpkin fires included the loss of all existing vegetation on steep slopes which led to mass-wasting events, reducing soil depth by up to three inches across many acres where forest once stood. It will take many centuries for soil to rebuild on these steep, rocky slopes and subsequent centuries before mature forests can again occupy the sites.

Table 25. Acres burned by wildfire, including acres of crown fire, within the mixed-conifer/spruce-fir cover types, Kaibab National Forest, 1987 through 2009.

Fire	Year Burned	Acres Burned	Acres of Crown Fire
Point	1993	1,800	1,200
Pumpkin	2000	15,000	2,250
Trick	2002	1,174	n/a
Warm	2006	7,434	4,041
Total Acres Burned		25,408	7,491

The Pumpkin fire burned through 4 active Mexican spotted owl Protected Activity Centers (PACs), the areas surrounding nest sites. PACs are intended to secure these vital areas on the landscape and help ensure successful reproduction of spotted owls. There are eight PACs that occur completely or partially on the KNF (Table 26). Only six of the PACs are actually administered by the KNF. Management within these areas is largely restricted to attaining forest

health objectives. All but two of the PACs on the KNF are on isolated peaks and consist of dense forests with high fuel loads, making them at risk to loss from stand-replacement fires.

Table 26. Mexican spotted owl PACS on the Kaibab National Forest.

PAC Name	PAC Number	Administrative Authority
Pumpkin	070201	KNF
Kendrick	070210	KNF
Stock tank	040230	CNF ¹
Sitgreaves	070205	KNF
Bill Williams	070120	KNF
Big Springs	070125	KNF
Tule	070115	KNF
LO Pocket	040230	CNF ¹

¹*Monitored by the Coconino National Forest.*

The Pumpkin Fire may have improved the habitat in two of the Kendrick Mountain PACs by creating understory burns (Figure 43). However, fire intensity was severe in two PACs (1 of which occurs on the KNF), causing nearly complete mortality of the overstory. The Pumpkin Fire started in late-May and burned into early June, the time when owlets are still in the nest or beginning to fledge, a time when they are incapable of flight. There is no data on potential juvenile mortality from fire, smoke or disturbance from fire-fighting activities.

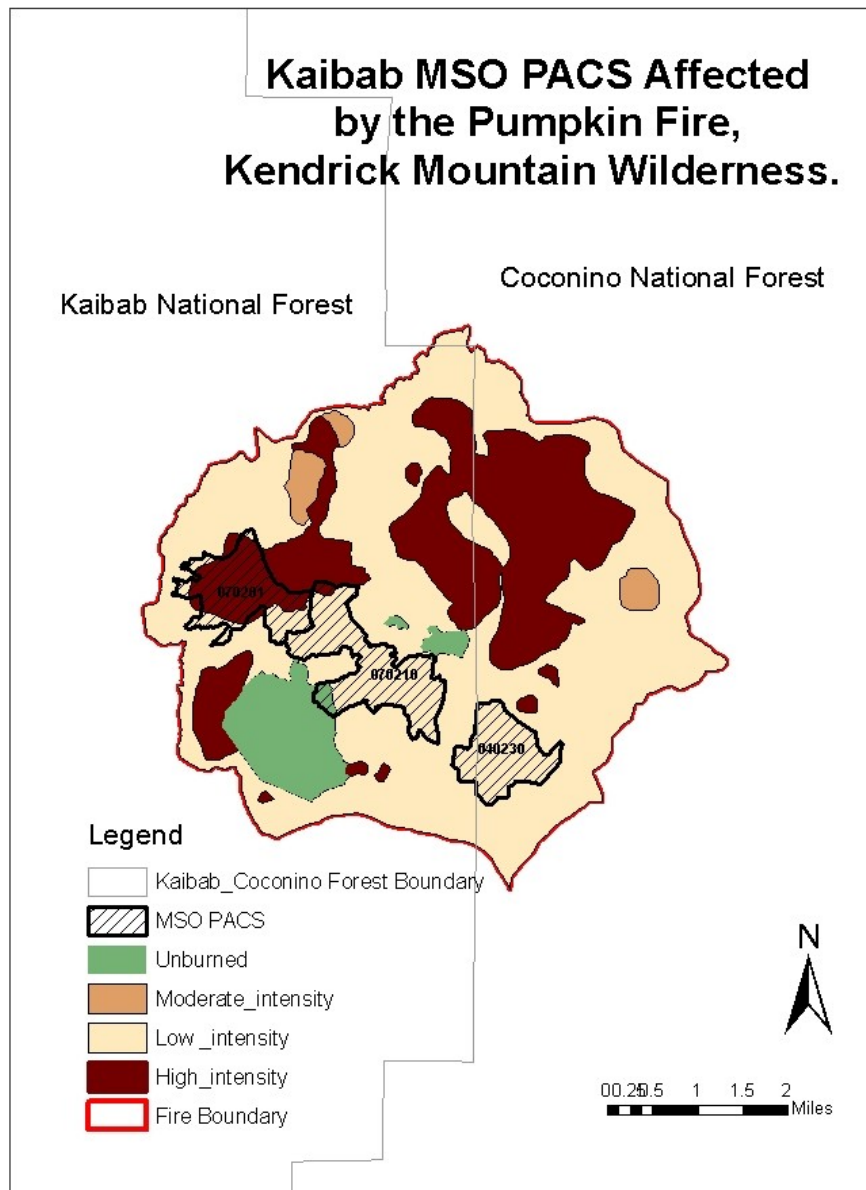


Figure 43. Impacts of the Pumpkin Fire on Mexican spotted owl Protected Activity Centers, Kendrick Mountain Wilderness.

Lynch et al (2008) synthesized insect activity on the KNF and Grand Canyon NP from 1918-2006. On the KNF, two spruce budworm outbreaks occurred during the 1950s and 1970s, damaging approximately 200,000 acres of forest. Occasional damage from Douglas-fir beetles has also occurred. Although the amount of total acreage affected has been relatively small,

preferential use of larger diameter trees can have far reaching affects by further increasing additional outbreaks. High mortality among white fir trees caused by fir engravers (true fir beetles) has occurred from 1990-2006. This appears to be a departure from historic insect activity.

Recent increases in Douglas-fir beetle in Arizona have especially affected the KNF, Apache Sitgreaves and Coronado National Forests. Damage peaked during 2003-2005 where nearly 200,000 acres were affected. Damage has tapered off over the last several years but remains relatively high (with regard to historic levels) across the region with 14,000 acres of damage observed in 2007 (USDA 2008b). Increased forest density and changes in forest structure combined with periods of high moisture levels followed by drought have lead to more extensive outbreaks of Western spruce budworm (a defoliator) in Utah forests as well (reviewed *in* Battaglia and Shepherd 2007).

Habitat Trend

In the short-term, increases in stand density should provide adequate thermal cover for elk, although decreases in understory diversity resulting from a closed canopy state may decrease available forage. Dense canopies should also provide habitat for red squirrels in the short-term. Current harvesting regimes should maintain adequate structural diversity for Mexican spotted owls. In the long-term however, the risk of habitat loss from catastrophic fire may negate positive benefits, particularly for Mexican spotted owls and red squirrels.

Under current management, including direction from the USFWS (USDI 1995), old open forest is not expected to be achieved (Kaibab National Forest 2008a). The federal code of regulations, as amended in 2004 (50 CFR 17.95), states that Mexican spotted owl habitat shall contain a diversity of tree species, with > 45% of trees greater than 12 inches DBH, and > 40% overstory canopy cover. Given these federal mandates, the KNF is not likely to manage mixed-conifer forest within its historic range of variability.

Projected

The VDDT model developed during the forest plan revision process suggests that mixed-coniferous forest is highly departed from reference conditions and is projected to trend further away given current management conditions. At present, there is an increased likelihood of stand replacement fire with over half of the forest in a dense state. Changes in forest composition from fire tolerant to intolerant species such as white fir further increase the likelihood of stand replacing fire events and undermines the ecology of this ecosystem.

Spruce Fir Cover Type and PNVT

Subalpine conifers, which includes spruce and fir species, occur in high elevation windswept forest and are particularly suited to the strong winds and colder temperatures that occur in those areas. With their narrow pointed crowns, they are especially adapted to shed snow. Short, brittle, condensed branches allow them to grow close together which further protects them from windthrow. The subalpine- fir forest represents a climax ecosystem. Other species have difficulty competing because they cannot reproduce in the shade at these altitudes, hence spruce and fir dominate.

Background and Historical Info

Also known as sub-alpine conifer forests, spruce-fir forests range in elevation from 8,400 to 10,418 feet along a variety of gradients which include gentle to very steep mountain slopes. In Arizona, this cover type occurs on less than .5% of the state (TNC 2006). Spruce-fir forests primarily consist of Engelmann spruce (*Picea engelmannii*) and cork bark fir (*Abies lasiocarpa* var. *lasiocarpa*) and gradually grade into mixed-coniferous forest between 8,000 and 10,000 ft. The big difference between spruce-fir and lower elevation pine forests is that spruce-fir forests are characterized by moister and more vegetatively diverse conditions than lower elevation ponderosa pine forests (Graham and Sisk 2002). Spruce-fir typically occurs in areas where snow is present 6-8 months out of the year, generally areas that are colder and wetter. They may persist 500-600 years and are highly susceptible to disturbance events such as fire, insects, and wind as a result of the harsh climates they inhabit (Battaglia and Shepperd 2007).

Spruce-fir forests differ from lower elevation forests such as mixed conifer and ponderosa pine because frequent, low intensity fire regimes were historically less common (Dahms and Geils 1997). As a result, fire suppression activities have had less of an impact on this forest type (in a general sense) than on fire dependent systems such as ponderosa pine. On the KNF, the spruce-fir cover type encompasses 28,635 acres of forest. Similarly, there are approximately 29,100 acres of the spruce-fir PNVT, roughly 2% of the forest (Figure 44). Spruce-fir occurs mostly on the NKRD and

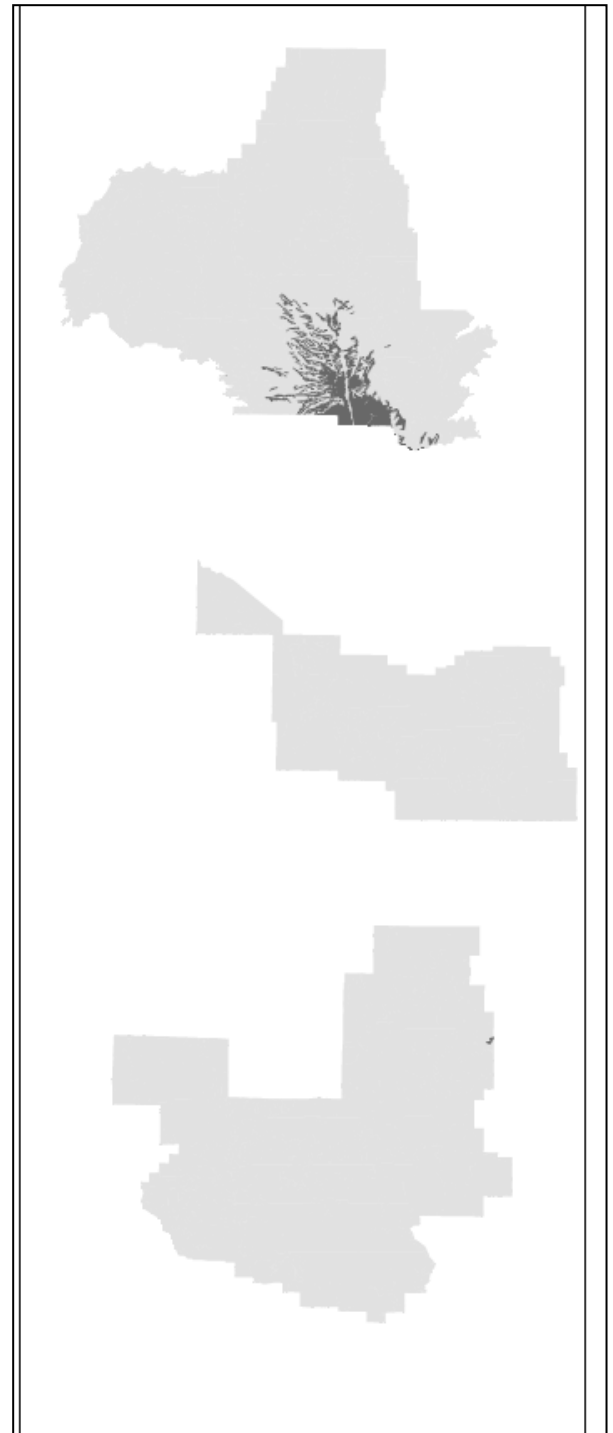


Figure 44. Spruce-fir PNVT.

the north facing aspects of Kendrick Mountain on the Williams RD. It occurs in patches that are generally surrounded by mixed-conifer and ponderosa pine forests or montane grasslands (Kaibab National Forest 2008a).

Mexican spotted owls were selected to represent late-seral habitat structure and elk and red squirrels represent early-seral forests.

Potential Management Implications

Because spruce-fir occurs at higher, more inhospitable elevations, a primary disturbance is windthrow. As a result, logging in spruce-fir forest can be particularly detrimental because it may create openings, which in turn expose trees to wind stress.

(http://cpluhna.nau.edu/Biota/subalpine_conifer.htm). Additionally, increased woody debris resulting from windthrow may subsequently increase fuel loads, increasing the risk of high severity fires and setting the stage for insect outbreaks. In areas with widespread burns, weakened trees are more susceptible to epidemic insect attacks, especially from spruce beetles (Battaglia and Shepperd 2007).

Fire

Spruce-fir forest is generally characterized by a mixed fire regime that varies depending on site specific conditions such as variable topography and moisture (Fulé et al. 2003a). Fires in subalpine forests such as spruce-fir are typically high-severity crown fires

(http://cpluhna.nau.edu/Biota/subalpine_conifer.htm). Unlike ponderosa pine and most mixed-conifer forest types, spruce-fir forests did evolve with stand replacing fires, thus the fire suppression effects that have proven detrimental to other fire dependent forest types have been less influential in this forest type (McKenzie et al. 2004).

Steep elevation gradients as well as spatial orientation within the larger forest matrix may create variable fire regimes from forest type to forest type. In Arizona, Grissino-Mayer et al. (1995) as cited by Fulé (2003a) found that fire frequency varied greatly between spruce-fir forests above and adjacent to mixed-conifer. The mean fire return interval (MFRI) for mixed-conifer was 6.2 years, while the MFRI for spruce-fir was greater than 300 years. This highlights the differences in fire regimes (mixed-severity) which pose management dilemmas for both forest health and wildlife benefit.

In general, MFRIs for spruce-fir forest across the Southwest vary ranging from 40-400 years with the MFRI for the north rim of the Grand Canyon ranging from 70-150 years (Battaglia and Shepperd 2007 and references therein). Longer fire return intervals result from cooler temperatures and greater precipitation throughout the year. Because spruce-fir species have such thin bark, severe fires can convert forests to meadows or aspen stands for hundreds of years (Battaglia and Shepperd 2007).

On the Kaibab Plateau however, fire return intervals appear to be shorter than for other high-elevation forests throughout the Southwest. Fulé et al. (2003a) reported a historic fire return interval of approximately 2-32 years. Stand-replacing fires are believed to have occurred at small scales (Lang and Stewart 1910, Fulé et al. 2003b). Fulé (2004) found that fire dates in spruce-fir often coincided with fire dates for other types of forest on the Plateau, suggesting that fires may

have started in ponderosa pine or mixed conifer forests and spread into the spruce-fir. Sub-stand patches were initiated both by fire and by other means. In general the resulting forest structure appears to have been highly variable over small spatial and temporal scales (Fulé et al. 2003a). The mixed severity nature of the Outlet fire that burned on the KNF in 2000 tracks well with historical accounts from fire scar data from the later 1800s, suggesting severe burning in high elevation forest was common. The main difference now is that shorter fire return intervals coupled with an increase in density of young spruce and fir can lead to burns over greater portions of the landscape (Fulé et al. 2003a, Kaibab National Forest 2008a).

Insects

Stand conditions that influence outbreaks of spruce beetle and western balsam bark beetle include high stand density index, prior mortality, tree diameter and basal area (McMillin et al. 2003, Fettig et al. 2007, Lynch 2008). Disturbance events such as logging slash, warm temperatures, and windthrow also facilitate spruce beetle outbreaks (Lynch et al. 2008). Past research suggests that large diameter trees seem to be the desired hosts for spruce beetles and Douglas-fir beetles (reviewed in Fettig et al. 2007). Risk and hazard models by Weatherby and Their (1993) and Schmid and Frye (1976) (as cited in Fettig et al. 2007) suggest that trees with diameters greater than 16 inches DBH are most susceptible to beetle attack. Battaglia and Shepperd (2007) concluded many trees killed by bark beetles, particularly large diameter trees, will remain standing years after they have been attacked. However, current research on the KNF indicates that beetle-killed trees may be as ephemeral as fire-killed trees (C. Chambers *pers. comm.*). In general, spruce-fir is less prone to large-scale insect outbreaks because it occurs in such limited amounts, in colder environments, and because fire suppression has not had an overt impact on this particular forest type. Minor outbreaks may occur every 2-4 decades (Lynch et al. 2008). When populations are at endemic levels, spruce beetle are found in individual diseased or damaged spruce trees. When outbreaks occur, the impacts can convert spruce-fir stands to fir-dominated stands. This has not occurred across large areas on the KNF; however some forests in Utah have experienced entire diebacks of spruce as a result of spruce budworm outbreaks (Battaglia and Shepperd 2007, Lynch et al. 2008).

Thinning, if timed appropriately, can be an effective method for preventing bark beetle attacks. In the Rocky Mountains, single tree and group selection cuts have stimulated englemann spruce regeneration by promoting greater spatial dispersion and heterogeneity throughout the stand. However, some *Ips* beetles may be attracted to the resultant slash from such projects because there is little or no host resistance from dead and/or dying material (Fettig et al. 2007). Scattering of slash in the late summer may minimize these effects because flight activity is greatly reduced during that time, lower temperature may induce additional beetle mortality (Fettig et al. 2007). In the short-term, some bark and wood boring beetles respond favorably to forest treatments, during which tree defenses are weakened. It is not uncommon for beetle populations to increase the year following management efforts such as prescribed fire and thinning (Pilliod et al. 2006 and references therein). On the KNF, treatments mostly thin smaller trees adjacent to or under older trees. In some cases, group selection of larger trees is used to achieve desired conditions that include a consistent flow of trees into larger size-classes in areas where young age classes are lacking (Kaibab National Forest 2008a). Related to insect epidemics are root diseases. In the Southwest root diseases are most prevalent in high elevation forests. Conifers typically affected include spruce and fir because they lack protective resin. Fungi typically invade root systems of

cut and/or killed trees and can survive for decades by feeding on stumps and snags. Root fungi cause death and decay of both heartwood and sapwood. Trees that have been infected by root disease will typically experience decreases in vigor eventually leading to death. Overall decline however may take years before it becomes evident at the stand level. Root fungi are most aggressive in stands less than 30 years of age and facilitate bark beetle epidemics, particularly during times of drought. Fungi causing mortality include *Armillaria* spp. and *Inonotus tomentosus* (spruce) *Heterobasidion annosum* (subalpine fir) and *Phaeolus schweinitzii* (Douglas-fir) (unpublished report Fairweather 2004, 2007).

Habitat Data

Several studies have found significant increases in spruce-fir forest densities from historic conditions. Fulé et al. (2003a) found high elevation forests on the North Rim of the Grand Canyon to be significantly denser with significantly higher basal areas compared to 1880. Historically, trees were 16-24% less dense with 36-46% lower basal areas than existing conditions. In spruce-fir forest they found tree densities increased from 60.7 trees per acre (149.8 per hectare) to 393 trees per acre (946 per hectare) and that basal area increased from 42.3 ft²/acre (9.7 m²/ha) to 121.1 ft²/acre (27.8 m²/ha). Similar trends have been found by Heyerdahl et al (2006). They modeled forests on the Fishlake and Dixie National Forests and found that trees showed an increasing trend toward smaller diameters and height than pre-settlement conditions. White and Vankat (1993) in Fulé et al (2003a) also found increasing densities of spruce-fir forest at high elevations on the North Rim since 1880. Some studies have shown up to a 371% increase in bulk canopy density in spruce-fir forest (Fulé et al. 2004). They found that in 1880, canopy bulk density was 0.001 lb/ft³ (0.017 kg/m³), but increased to 0.004 lb/ft³ (0.072 kg/m³) by the year 2000 and increased to 0.005 lb/ft³ (0.080 kg/m³) by 2040 with major increases in canopy biomass for both subalpine fir and spruce species.

Spruce beetle outbreaks have been minimal on the KNF. The most significant outbreak affected approximately 1000 acres in the 1990s. Defoliator activity continues to be low due to limited host availability (USDA 2008b). Regionally however, there has been a general increase in Western balsam bark beetle (*Dryocoetes confuses*) activity from 1997-2007. Most of this damage has occurred in New Mexico with minimal effects in the subalpine forests of Arizona (USDA 2008b). On the KNF, there is no evidence of western balsam bark beetle attacks that primarily affect corkbark and subalpine fir (Lynch 2008). On the NKRD, root disease has caused continued mortality since 1991 at the De Motte Campground (Fairweather 2004 unpublished report).

FIA data for two cycles suggest that densities of smaller sized trees (1-4.9 in DBH) have decreased on the KNF since 1995. This may reflect recent damage from the Outlet Fire that burned over 5,260 acres of forest on the NKRD and Grand Canyon forests in 2000. Since the 1995 cycle, there has been minimal change in tree densities among middle aged trees and a slight increase among older aged trees (Table 27, Figure 45).

Table 27. Number of live spruce-fir trees (includes Engelmann spruce and subalpine fir) per acre on forest land by Ranger District and diameter class (FIA, unpublished data).

Ranger District	Diameter class				
	1"-4.9"	5"-11.9"	12"-17.9"	18"-23.9"	>=24"
1995					
North Kaibab	12.11	4.81	1.11	0.35	0.07
Tusayan	0.00	0.00	0.00	0.00	0.00
Williams	0.00	0.00	0.00	0.00	0.00
2007					
North Kaibab	4.77	4.79	1.05	0.38	0.19
Tusayan	0.00	0.00	0.00	0.00	0.00
Williams	0.00	0.00	0.00	0.00	0.00

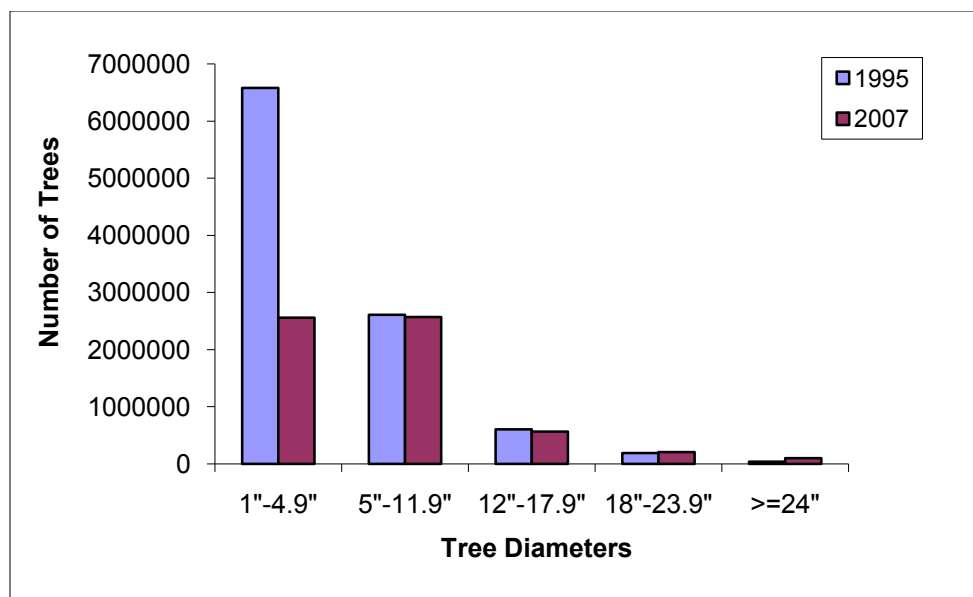


Figure 45. Number of spruce-fir trees across the forest by diameter class. Englemann spruce is the dominant species.

Habitat Trend

Although spruce-fir is a limited cover type on the KNF, it is important for wildlife in general and for MIS purposes specifically. No spotted owls have ever been confirmed nesting, roosting, or foraging in either spruce-fir or mixed conifer forests on the NKR. The limited spruce-fir forests occurring on Kendrick Peak (Williams RD) supports the only spotted owl PAC occurring in this forest type on the KNF. Similarly, elk only occur on the NK as colonizing individuals that likely travel south from Utah. The policy of the Arizona Game and Fish Department is to prevent them

from becoming established on the Kaibab Plateau. There has essentially been no forest management within this cover type. Therefore, population trends of MSO and elk are likely unaffected by management activities within this habitat. The threat of lightning strikes and crown fires spreading from the surrounding ponderosa pine forests makes it difficult to predict future trends for these species, relative to the spruce-fir cover type.

In general, spruce-fir forests are highly departed from historic conditions with higher overall tree densities and canopy cover than historic times (Fulé et al 2002, 2003a, 2004). The forest plan revision team found the lack of characteristic fire disturbance to be the primary threat to the Spruce-fir PNV. Although this is unusual for spruce-fir across other parts of the region (i.e. Grand Canyon NP), the spruce-fir forests on the Kaibab Plateau were likely influenced by fire disturbance from adjacent ponderosa pine and mixed conifer forests. State and transition models indicate a rapid shift from mid-aged closed forest to an older closed state. In 50 years, little change is expected from the current condition, with 75% of the area in a mature closed canopy states versus the reference condition of 34%. Under current management, including direction from the USFWS (USDI 1995), a return to the reference condition of old open forest is not expected (Kaibab National Forest 2008a). Furthermore, the federal code of regulations, as amended in 2004 (50 CFR 17.95), states that Mexican spotted owl habitat shall contain a diversity of tree species, with > 45% of trees greater than 12 inches DBH, and > 40% overstory canopy cover.

Pinyon-Juniper Woodlands Cover Type and PNVT

Native American Indians relied heavily on pinyon-juniper for at least two millennia. Plant fibers were used for fuelwood, baskets, sandals, dyes, textiles, and ceremonies while edible fruits were used for cosmetic and medicinal purposes (<http://www.santafe.edu/~pth/pj.html>).

Background

Pinyon-juniper (*Pinus edulis/Juniperus monosperma*) is the most abundant forest type in Arizona, covering about 7.7 million acres (56%) of the State's forested lands (O'Brien 2002) and 100 million acres of the western U.S. (Romme et al. 2007). Pinyon is mostly found on lower slopes of mountains and in upland rolling hills from 4,500 to 7,500 feet in elevation. The most common pinyon pine is the Colorado pinyon (*Pinus edulis*), with singleleaf pinyon (*Pinus monophylla*) occurring in limited areas. One-seed juniper (*Juniperus monosperma*) is most common in Arizona and New Mexico; however, Utah juniper (*J. osteosperma*), alligator juniper (*J. depenna*) and Rocky Mountain juniper (*J. scopulorum*) also occur across the Southwest.

About 657, 849 acres of pinyon-juniper habitat occurs on the KNF covering more area than any other habitat type. The comparable PNVT covers approximately 638,000 acres (Figure 46). Pinyon-juniper habitat is highly variable and comprised of three sub-vegetation types: pinyon-juniper grasslands, pinyon-juniper shrubland, and persistent pinyon-juniper woodlands (Romme et al. 2007). All three types exist on the KNF. The pinyon-juniper grassland PNVT has a canopy cover of less than 30% with an expansive herbaceous understory and a fire return interval of less than 25 years. A pinyon-juniper shrubland

PNVT is found in cooler climates and on less productive soils with a fire return interval of approximately 46 years. The persistent pinyon-juniper woodland PNVT is found in areas of fire refugia, where the historic fire return interval is significantly longer than the surrounding woodlands.

This occurs due to topographic features that prohibit the natural spread of fire, or because of very poor soils that will not sustain herbaceous growth. The historic fire return interval was greater than 250 years (Huffman et al. 2006, Kaibab National Forest 2008a) with no evidence of severe



Figure 45. Pinyon-juniper PNVT.

fires having occurred in the last 300-400 years (Huffman et al. 2008). Thus the fire suppression efforts that have largely impacted other forest types have been less influential with regard to pinyon-juniper (Romme et al. 2007, Huffman et al. 2009 and references therein). Unlike ponderosa pine forests, low-intensity surface fires were uncommon, while patchy, occasional, wind driven crown fires were typical (Romme et al. 2007).

The NKRD has over 258,103 acres of pinyon-juniper woodland and the balance of 441,545 acres occurs on the SZ. McCulloch and Smith (1991) report evidence of rain shadow effects on the NKRD, with available moisture increasing from east to west. This pattern is reflected in the vegetation, with xeric plant communities extending to higher elevations on the East Kaibab Monocline than they do on the western escarpments.

Historically, pinyon-juniper habitat was used for grazing, fuelwood collection (including large-scale charcoal production during the mining boom of the 1800s), posts, poles, beams, and pinyon nut collection (Evans 1988). Annual precipitation in pinyon-juniper woodlands is sparse and occurs bimodally as winter snowfall and summer monsoon rains. Most precipitation falls during winter; summer rainstorms are typically intense and spottily distributed across the landscape.

On a regional scale, pinyon-juniper forests are believed to have increased in both total land cover and stem density. When viewed at the millennia scale, juniper has been expanding at the upper elevations of its range while decreasing in the lower elevations (Davis 1987, Wells 1987). Historic range management practices and changes in the fire history are thought to have contributed to the expansion of pinyon-juniper woodlands into grassland ecosystems (Ronco 1986, Dahms and Geils 1997). Prior to this expansion, a greater portion of pinyon-juniper woodland may have once been dominated by juniper savanna having only 4.1- 6.1 trees/acre (10-15 trees/ha) (Brockway et al. 2002). In Arizona and New Mexico, many former types of grassland have been converted to juniper or pinyon-juniper woodlands of moderate to high canopy cover. The conversion of these areas has resulted in less heterogeneity across the landscape and decreases in biodiversity (Romme et al. 2007). These effects are discussed further in the Grassland portion of this document.

Pinyon-juniper is used as winter and spring range by both wild and domestic animals. Radio-marked and un-marked Mexican spotted owls have been documented over-wintering in pinyon-juniper habitat; radio-marked juvenile spotted owls used pinyon-juniper during both winter and dispersal movements (J. Ganey, *pers. comm.*). Northern goshawks may use pinyon-juniper habitat in a similar way (R. Reynolds, *pers. comm.*). On the NKRD, pinyon-juniper habitat provides primary and critical winter range and transitional habitat during migration for the Kaibab deer herd (McCulloch and Smith 1991). Similarly, pinyon-juniper habitat on the SZ provides winter and transitional range for elk. Pinyon-juniper woodlands, typically occupying the lower elevations of the KNF, have traditionally provided important spring grazing for livestock.

On the KNF, juniper counts of 405 stems per acre (1,000 stems per hectare) were documented in stands that were devoid of trees in 1880 (Eddleman 1987). Pollen analysis from Pecks Lake, Prescott National Forest, indicates a steady increase of juniper over the last 2600 years, but the trend also reveals a sudden increase in juniper pollen as exotic weeds and indications of livestock become discernable in the sediments (Davis 1987). Tree densities within existing stands have

also increased (Eddleman 1987, Ffolliott and Gottfried 2002). Junipers tend to establish under existing trees or shrubs. The absence of fire has increased the presence of woody vegetation which, if occurring near mature trees, can enhance seedling survival for juniper (Eddleman 1987). Ffolliott and Gottfried (2002) calculated an annual increase of 1.2 trees per acre per year in a 2-acre pinyon-juniper stand near the KNF. In those areas where extrapolating this estimate is valid, stem densities in pinyon-juniper habitat have increased an average of 120 trees per acre in the last century. On a nearby site in Northwestern Arizona, Huffman et al. unpublished data (2005) as cited by Stoddard et al. (2008) note that tree densities are 4 times greater than those in 1870.

On the KNF, Mule deer and juniper titmice are the Management Indicator Species for early-seral and late-seral pinyon-juniper woodlands respectively.

Potential Management Implications

Increases in tree density, canopy cover and loss of understory plant cover and diversity were noted as the primary threats to the pinyon juniper PNVT during the forest plan revision process (Kaibab National Forest 2008a). Similar trends have been observed in the Great Basin (Tausch and Hood 2007). Miller et al. (2008) emphasize that in the absence of disturbance and active pinyon-juniper removal, woodlands will reach a dense closed state in the next 40-50 years. Several studies have shown that density dependent factors are especially impacting the pinyon pine component of the pinyon-juniper system. Areas with high tree density experience higher levels of competition. Subsequently, trees may become weakened and less resilient to increasing environmental stressors (Negron and Wilson 2003, Greenwood and Weisberg 2008). In the past, fire has been the primary disturbance agent affecting pinyon-juniper, but insects, drought, and disease are becoming more influential (Greenwood and Weisberg 2008 and references therein). Increasing levels of mortality caused by the pinyon *Ips* beetle may be attributable to increased levels of dwarf mistletoe infection, competition from higher densities of large diameter trees, and stressors inherent in drought and higher temperatures (Allen 2007, Lynch 2008). Climate change is not likely to help the situation.

Drought has the potential to impact pinyon-juniper both directly through water induced stress and indirectly by increasing susceptibility to insect attack. Breshears et al. (2005) found that during the southwestern region's most recent drought episode (2000-2003), the *minimum* annual temp had risen ($p < 0.001$) since the last major drought event in the 1950s. With these rising temperatures, Seager et al. (2007) predict that the Southwestern region is on a rapid trajectory toward an increasingly arid climate. Rapid, regional mortality of pinyon pine has occurred as a result of this ongoing drought and extreme temperature increase (Breshears et al. 2005). The results of an 8-year study (1997-2004) in central New Mexico suggest that both oak and pinyon mast production has decreased as a result of such drought. On Sevilleta Nation Wildlife Refuge, Zlotin and Parmenter (2008) determined that air temperature, air relative humidity, wind speed, and precipitation significantly impacted mast production which, depending on site, decreased from as much as 18.2 lbs/acre (20.4 kg/ha) of dry weight to as little as 2.5 lbs/acre (2.8 kg/ha) from 1998-2004. Decreases in pinyon production were associated not only with cone production, but also an increase in the number of unfilled seeds. Unfilled seeds increased in average number by 35% in 1998 to 80% by 2004.

These temperature increases and associated water stress also increase the susceptibility of pinyon trees to bark beetle infestations. *Ips* beetles, which burrow into the cambium of living and dying trees, infest pinyon through mass attacks. Having 2 generations per year, populations could increase by two orders of magnitude on an annual basis (Allen et al. 2007). The magnitude of beetle outbreaks varies in spatial scale. Widespread drought can cause regional outbreaks and mortality, while slash resulting from range improvement projects may cause localized episodes. Although mortality of pinyon can approach 100% in a given stand, outbreaks rarely last more than 1-2 years (Lynch et al 2008). The loss of pinyon in pinyon-juniper woodlands results in structural changes that may have large-scale community consequences decreasing habitat for a variety of wildlife species that nest, roost, and forage on pinyon trees. Over 1000 species of wildlife are associated with pinyon pine, and the vegetational shift that would accompany a large-scale loss of pinyon would likely be detrimental for many species (Mueller et al. 2005). For example Christenson et al. (1991 in Mueller et al. 2005) found that avian seed dispersers abandoned individual pinyon trees and stands when insects caused a 57% reduction in cone crops. This resulting loss of avian seed dispersers in the system creates a negative feedback loop that could further decrease recruitment, as pinyon pines are largely dispersed by birds and small rodents (Chambers 2001, Pearson and Theimer 2004).

Additional ramifications resulting from widespread loss of pinyon include increases in flammability resulting from high levels of dead wood and the opportunity for noxious plants such as cheatgrass to invade into open areas (Greenwood and Weisberg 2008). Having the potential to replace native species, some studies have also shown that as cheatgrass cover increases in native herbaceous woodlands, so does fire size and frequency (James 2007 and references therein). Subsequent effects on wildlife may result from substantial changes to local fire regime and soil and plant diversity.

Juniper tend to be more resilient to drought than pinyon and subsequently less vulnerable to insect outbreaks (Mueller et al. 2005 and references therein). Woodborers such as Cypress/juniper bark beetles (*Phloesinus* spp.) do occur in areas with poor site conditions (Mueller et al. 2005). However, recent survey results suggest this insect is not a threat on the KNF and is in decline compared to previous years. During aerial surveys in 2007, only 70 acres of damage were mapped in Arizona and New Mexico with mortality patchy and scattered (USDA 2008b). Gitlin et al. (2006) found that juniper mortality from *Phloesinus* spp. averaged around 3.3% within an 49.7 mile (80 km) radius of Flagstaff. They found mortality to be higher on grassland than non-grassland sites. Greenwood and Weisburg (2008) found that stand density had the greatest influence on tree canopy mortality and the prevalence of *Ips* and dwarf mistletoe in pinyon-juniper woodlands.

Grazing has also contributed to the interruption in the fire sequence by reducing grass cover to the point where understory vegetation could no longer sustain fires. While high levels of herbivory can decrease the biotic and genetic diversity of some native species, and can contribute to the introduction and establishment of exotic species (Dahms and Geils 1997), recent studies have shown that moderate levels of herbivory may actually increase species diversity and prevent conversion to exotic monocultures (e.g. cheatgrass) in systems which have been subject to long term disturbance from ungulates (Loeser et al. 2006). Livestock can also have negative impacts on soil. Erosion rates are greater and soil infiltration lower in pinyon-juniper habitats, relative to historic levels, across the Southwest (Dahms and Geils 1997). Soil compaction

resulting from livestock use is cumulative and so prolongs restoration of infiltration capacity (Evans 1988). Excessive soil impacts from past management may prevent the recovery of understory vegetation in some sites.

Long-term effects from past grazing and fire suppression have favored the development and expansion of pinyon-juniper woodlands. Pinyon-juniper habitat is likely more extensive and woodland forests denser than pre-settlement conditions. The development of pinyon-juniper habitat can lead to a substantial redistribution of nutrients. Pinyons and junipers send out lateral roots to collect water and nutrients that are translocated into the canopy. Litter fall beneath the canopy retains most of the nutrients for the tree while the roots “mine” the neighboring intercanopy areas (Evans 1988). Tree encroachment into surrounding meadows and grasslands shifts the storage of nutrients from a soil-litter-duff system to living plants. The greater biomass of the trees occurs at the expense of the surrounding shrubs, grasses, and forbs (Evans 1988). Removing the trees, whether by dropping (i.e., cutting, chaining, bulldozing, etc.) and burning or from fuelwood harvest can locally decrease available nutrients. Burning slash and litter can decrease total nutrients in the plant-soil system by 13% (Evans 1988). In a nutrient-limited system, this can result in lower biomass production, alter successional patterns, and encourage invasion by annual, noxious, or other weedy species (Evans 1988, Neal 2007, Owen et al. 2009).

Wildlife species composition and richness changes as woodlands expand into grasslands. Bird community composition changes dramatically along the grassland-woodland gradient, with ground-nesting species decreasing sharply as tree density increases (Rosenstock and van Riper 2001). The proportion of shrub-nesting species does not appear to vary significantly as shrubland changes to woodland. Predictably, tree- and cavity-nesting species increase with increased tree densities. Changes in avifaunal species appear proportional to the change in the tree component, with densities of about ten or more juniper trees per ha being the approximate threshold at which suitability for grassland species declines (Rosenstock and van Riper 2001). Rosenstock and van Riper (2001) also discuss anecdotal observations suggesting pinyon-juniper expansion may be detrimental to Gunnison’s prairie dogs.

When pinyon-juniper initially encroaches into neighboring grassland and shrub steppe communities, the plant community becomes more diverse. In the short-term this may be of benefit to some species such as mule deer and elk. These short-term gains however, may be negated in the long-term as pinyon-juniper eventually out compete herbaceous plants thereby decreasing available forage (Mule Dee Working Group 2003). Ongoing woodland expansion is best mediated by tree removal, either through mechanical means or prescribed burning. If not used carefully however, these methods may actually place additional stress upon the ecosystem. The resulting slash from range improvement projects has the potential to facilitate bark beetle outbreaks (USDA 2008b, Lynch et al. 2008). Prescribed fire carries with it the risk for additional establishment of non-native grasses such as cheatgrass, and dangerously high burning conditions that may decrease soil quality and future germination conditions for herbaceous plants (reviewed in James 2007, Tausch and Hood 2007). Nevertheless, several studies have shown that fuel wood harvest, prescribed burning, and mechanical thinning in pinyon-juniper woodlands can reduce encroachment into adjacent grasslands and improve herbaceous plant diversity and forage quality and quantity (Brockway et al. 2002, Albert et al. 2004, Sheley and Bates 2008). Jacobs and Gatewood (2002) found that prescribed fire may be an effective tool to control regeneration of

woody shrubs in woodland savanna that was previously thinned by mechanical means. They note that the timing of such restorative treatments is critical however, such that burn intensity does not negate positive benefits by increasing mortality of herbaceous grasses and forbs.

Several studies have shown that mechanical thinning in particular can be an especially effective means to restore herbaceous vegetation and increase biodiversity in pinyon-juniper woodlands (Brockway et al. 2002, Albert et al. 2004, Sheley and Bates 2008, Owen et al. 2009). In New Mexico, Albert et al. (2004) found that thinning pinyon and juniper trees to basal area densities (BA) of 30-57 ft² increased herbaceous plant growth and species diversity. In their study they found deer, elk, and small mammal use also increased. In the Cibola National Forest, Brockway et al. (2002) found that plots mechanically thinned of juniper to 6.1 trees/acre (15 trees/ha) resulted in approximately 250% increases in grass cover as well as species richness. In that same study, overstory cover was reduced from as much as 25% to < 2% creating a more open community structure. Within just 2 growing seasons, grasses increased from a low of 9% to as much as 38% cover. Because mechanical thinning does have the potential to cause system wide affects, clear objectives are imperative from the outset. Crowe and van Riper (unpublished 2009) have found that mechanical thinning can have negative implications for some bird species, particularly for pinyon-juniper obligates such as the gray vireo, a species of conservation concern on the Colorado Plateau.

Locally on the Tusayan district of the KNF, Huffman et al. (2009), found that treatments that included thinning and/or thinning and burning were significantly more effective for reducing crown fuel loads in pinyon-juniper than prescribed burning alone. This is supported by similar findings in ponderosa pine (Fulé et al. 2002) and mixed conifer forests (Stephens and Moghaddas 2005). In Northwest Arizona, Stoddard et al (2008), found that scattering slash in areas that had been seeded with native grasses, increased grass cover more than sites that had been seeded, or slash treated only. Grass cover and density was significantly higher in the seed-and-slash treatments ($p < 0.0001$). In addition, scattered slash decreased soil movement, increased litter cover and improved soil chemistry and microclimate. Mastication of woody debris can also be an effective means for disposing of thinned trees. Owen et al. (2009) found that at least in the short-term, mastication used during thinning operations may be more beneficial to soil properties and plant species composition than pile burning of the resulting slash. In their study, masticated areas had greater plant cover, less bare ground, and higher grass and litter cover than untreated areas while pile burns had 80% more bare ground and very little plant cover.

When planning restoration efforts, Miller et al. (2008) urged managers to consider historical (pre- vs. post-settlement) distribution and density of trees, placing emphasis on treatments in areas where they will be most effective and to the old-growth component. For example, work conducted in woodlands that are closer to historic conditions may be more effective and economically feasible than ambitious large-scale projects in highly altered habitat.

Management Response

Prescription burns in pinyon-juniper woodlands are still uncommon as knowledge about fire behavior as a restoration tool is evolving. About 1000 acres of pinyon-juniper are thinned

annually to improve understory conditions; very little prescribed burning is done (Kaibab National Forest 2008a). The forest is currently experimenting with different burning treatments. Since 2005, approximately 11,000 acres have been managed as wildland fire use in pinyon-juniper, primarily on the Williams and Tusayan RDs (H. Kleindienst *pers. comm.*)

Although labor intensive, mechanical treatments may show more promise as a management tool because they are easier to control than fire. These treatments have been ongoing on the KNF since the 1980s. A query of the FACTS database shows that approximately 25,514 acres of pinyon-juniper have been treated for the benefit of wildlife habitat since 1990 (Figure 46). Past projects have included prescribed burning, seeding, and “pushes” that utilized bulldozers to push back encroaching pinyon-juniper. More recent treatments use the agra axe (Bobcat mounted with hydraulic shears) to cut down or limb small diameter trees. These clipping treatments range from roughly 65-600 acres in size and emphasize removal of the juniper component. The resulting slash from such projects is spread across the ground to decrease soil erosion and improve microsite conditions. On the North Kaibab, hand removal (lop and drop) has recently been used to improve wildlife habitat with 21 acres treated to date. Another 2,475 acres have been “pushed”. In general, fuelwood removal has occurred in spot treatments of a ¼ acre or less.

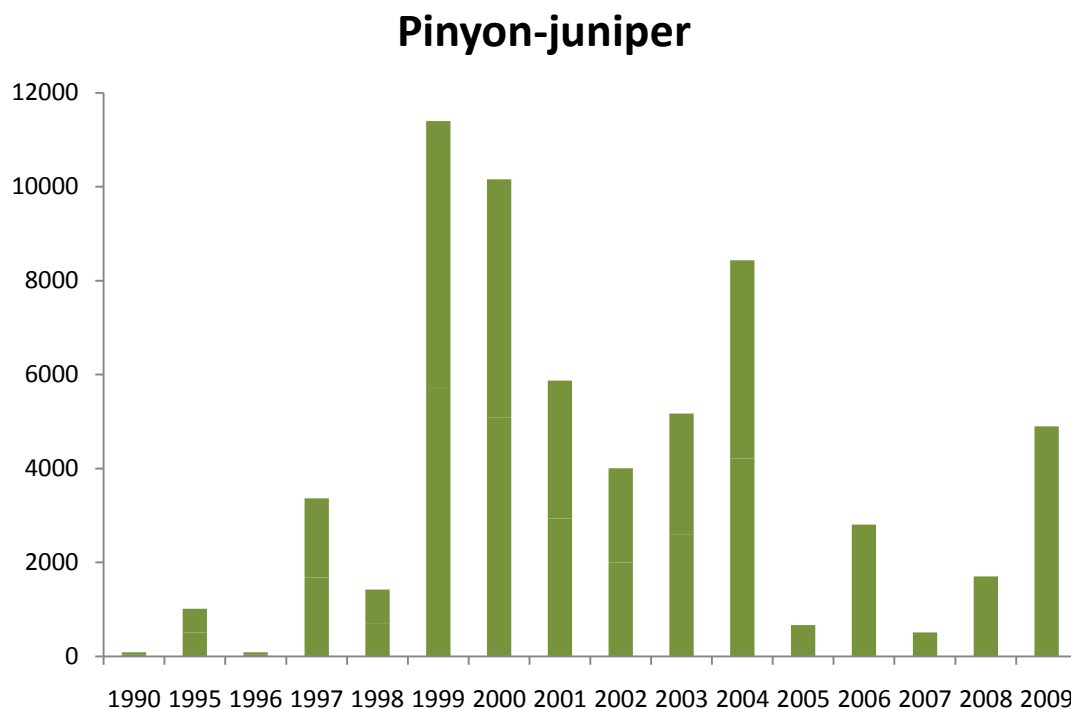


Figure 46. Acres treated in pinyon-juniper habitat for wildlife benefit. Includes activities such as thinning and tree encroachment control, (pushes, chaining, agra axe), and prescribed burning.

Habitat Data

FIA data for two survey periods show a decrease in all size classes of living trees across the forest with the exception of those greater than 24 inches DBH. According to these data, the greatest decrease on the KNF has occurred among the smaller size classes (Table 28, Figure 47). Even so, the number of live trees per acre still exceeds forest service plan requirements for old growth forest that specify retaining between 12 and 30 live canopy trees per acre with a DBH of

9-12 inches. Despite the decreases experienced since the 1995 FIA inventory, the pinyon-juniper woodland continues to maintain high density levels across all size classes, highlighting the tendency of the system to persist even amidst drought conditions. However, the pervasiveness of this system may be attributed to the resilient nature of juniper, as mortality of pinyon has exceeded growth (Figure 48).

Table 28. Number of live pinyon-juniper trees per acre on forest land by diameter class, KNF (FIA, unpublished data).

Diameter class					
Ranger District	1"-4.9"	5"-11.9"	12"-17.9"	18"-23.9"	>=24"
1995					
North Kaibab	64.70	50.96	12.50	4.15	1.59
Tusayan	97.30	98.23	19.42	5.10	0.80
Williams	62.73	36.66	10.14	2.79	1.06
<i>Totals</i>	<i>224.73</i>	<i>185.85</i>	<i>42.06</i>	<i>12.04</i>	<i>3.45</i>
2007					
North Kaibab	52.55	36.91	11.02	2.87	0.96
Tusayan	86.09	69.14	17.77	5.43	1.48
Williams	50.85	33.51	7.96	2.19	1.09
<i>Totals</i>	<i>189.49</i>	<i>139.56</i>	<i>36.75</i>	<i>10.49</i>	<i>3.53</i>

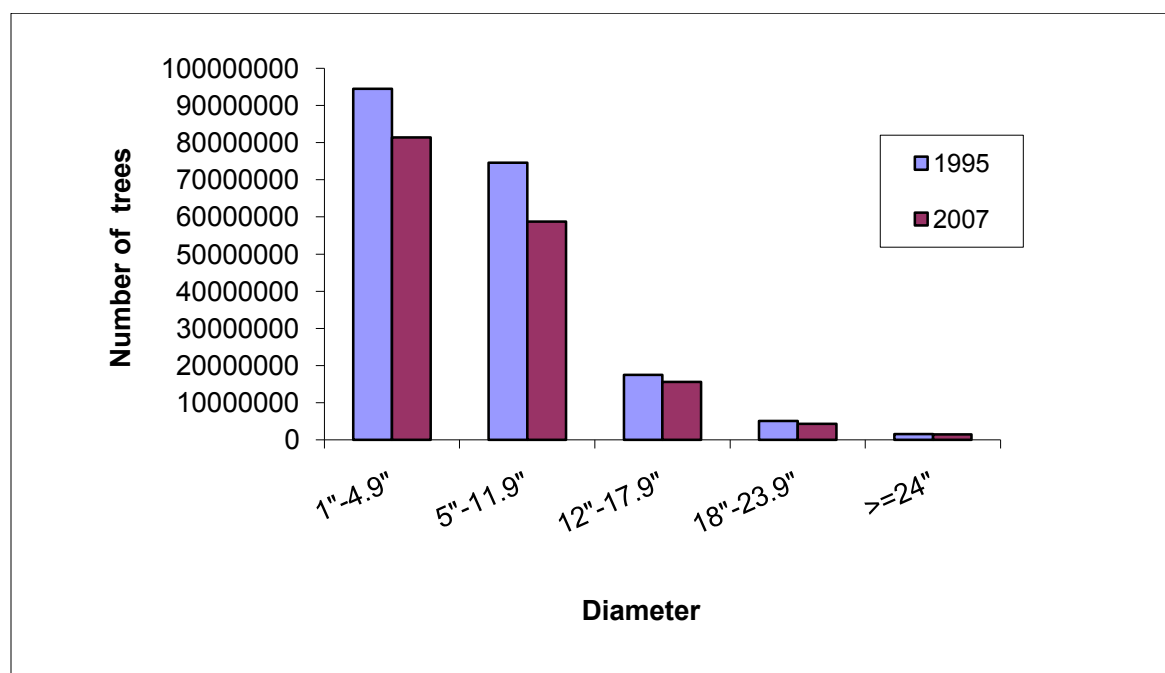


Figure 47. Number of live pinyon-juniper trees by size class across the KNF (FIA, unpublished data.)

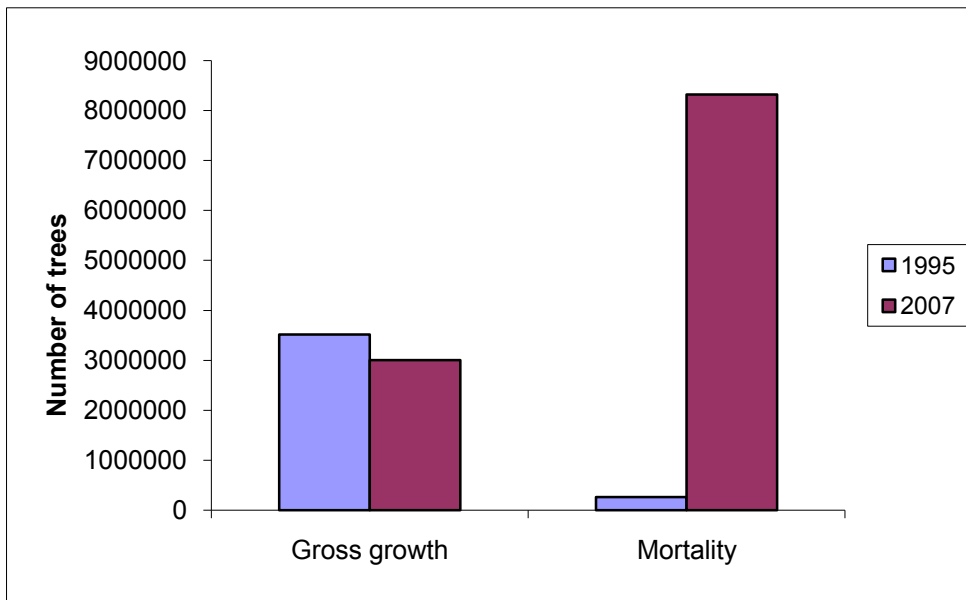


Figure 48. Mortality vs. growth, pinyon trees KNF (FIA, unpublished data).

Insects, fire, and disease have been the primary causal agents affecting these trends, all intricately tied to drought. In the 1950s and early 2000s, two major drought events caused outbreaks of pinyon *Ips* on the KNF and Grand Canyon NP. In surveys flown from 2001-2003 the USDA Forest Service Forest Health Protection office detected high levels of pinyon mortality. Two million acres of forest were affected across the region and 160,000 acres across the KNF and Grand Canyon NP. The attack was likely perpetuated by drought induced stress of pinyon (USDA 2008b, Lynch et al. 2008). On the KNF, the associated mortality resulted in more open canopy cover and juniper dominated woodland further resulting in an increase of deadwood and forests more susceptible to severe fire (Kaibab National Forest 2008a). In Northern Arizona, the current die-off of pinyon is unprecedented and 100 times greater than previously recorded damage for the region (Mueller et al. 2005, Allen 2007, Lynch et al. 2008). The 1950s drought and associated *Ips* induced mortality affected less than 4,000 acres while damage incurred in 2003 approximated 150,000 acres (Lynch et al 2008). Others have documented differential loss of pinyon. In the Coconino NF, Mueller et al. (2005) found pinyon mortality to be significantly greater than juniper ($t = 4.40$, $p < 0.0001$) during recent drought years with the larger tree component most affected. Gitlin (2006) quantified drought-induced mortality across the dominant plant communities within an 80 km radius of Flagstaff Arizona. They found pinyon pine mortality was among the highest at 41.4%. Conversely, they found that one-seeded juniper had the lowest mortality at 3.3% ($p < 0.0001$).

Prior to 1996, wild fires on the NKRND burned over 97,000 acres of pinyon-juniper woodland. Invasion of exotic and noxious plant species is a concern on these vast tracts of disturbed ground. Since 1996, approximately 33,700 acres (5%) of the pinyon-juniper woodland PNVT have experienced uncharacteristic stand replacing fire. The Bridger fire in 1996, which also affected mixed conifer forest, burned approximately 37,000 acres. Since that time, small fires ranging from 50-500 acres in size continue to erupt in that area (Kaibab National Forest 2008a). In 2006, the Warm Fire burned approximately 9000 acres in the pinyon-juniper woodland cover type with

approximately 52% of the stand experiencing high severity burn (Appendices 7 and 8). Historically, stand replacement fires did occur in pinyon-juniper woodlands but were infrequent and common in only certain ecotypes (e.g., true woodland vs. pinyon-juniper shrubland or grassland) with long fire return intervals of approximately 300-400 years (Kaibab National Forest 2008a, Greenwood and Weisberg 2008, Huffman et al. 2008).

Herbicide treatments and drill seeding have recently been used on the NKRd to improve wildlife habitat in areas that have been affected by wildfire. Preliminary data suggest that there has been a substantial increase of litter, forb and grass cover from 2007 to 2008, although the results were not statistically significant. In addition, no statistically significant decreases in cheatgrass were found as a result of the initial work (C. Albano *pers. comm.*, Grand Canyon Trust, unpublished data 2008). It may take several years before significant improvements to wildlife habitat are detected. Predictive models developed for that same project, suggest that continued cheatgrass invasion on the NKRd is imminent (Grand Canyon Trust, unpublished report 2008).

Habitat Trend Estimate

Changes currently occurring in pinyon-juniper habitat (e.g., current management practices, drought) are likely at a scale that will only incrementally affect mule deer. The one exception to this generalization is fire (e.g., the acreage of pinyon-juniper burned on the NKRd). In this regard, efforts to reduce the basal area of pinyon-juniper stands and to return manageable fire to the landscape are expected to improve mule deer habitat. Given the mix of positive and negative influences, the overall trends in pinyon-juniper habitat for mule deer on the KNF appear to be relatively stable since 1987.

Except for actions aimed at restoring grasslands and savanna, the low levels of active management of pinyon-juniper woodland have allowed stands to increase in area, density, and have allowed seral succession to continue. Results above suggest increases in drought severity and frequency will lead to decreases in pinyon and expansion by more drought tolerant juniper. While a decrease in pinyon may have dramatic effects on wildlife, overall there has been an increasing trend in juniper titmouse habitat quantity and quality since the signing of the KNF Land Management Plan.

Projected Trend

The VDDT model developed during the revision process found the pinyon-juniper PNVt to be moderately departed from reference conditions and trending away. Under current disturbances and management, there will be a shift to younger and denser woodlands, chiefly a result of uncharacteristic stand replacing fire. With this added risk of increased severe fire disturbance is the potential for invasion by non-native plants, further shortening the fire return interval and altering plant community structure and composition (Kaibab National Forest 2008a).

Grasslands and Grasslands PNTs

Intermediate to forests and deserts, grasslands are located in every continent except Antarctica. Fragile but dynamic ecosystems, grasslands are home to many uniquely adapted animals and plants. Grasslands go by a variety of names including prairies in the U.S. Midwest, pampas in South America, steppes in Central Eurasia and savannas in Africa. Although they all have different names, grass is the dominant vegetation type (<http://environment.nationalgeographic.com/environment/habitats/grassland-profile.htm>).

Background

Grasslands, which cover approximately $\frac{1}{4}$ of the earth's land mass, are diminishing in size as land is converted for urban and agricultural purposes. Although they occupy approximately 40% of the land area, they comprise 30-35% of terrestrial net primary productivity (Bailey 1996, Field et al. 1998 cited in Browning et al. 2008). Noss et al. (1995) note that grasslands (and savannas) are among the most endangered ecosystems in the world. The loss of grasslands has important global implications as they are a key component of the carbon, water, and nitrogen cycles (Browning et al. 2008).

In the Southwest, grassland types vary based on topography, rainfall, soil type and precipitation (TNC 2006). Major grassland categories found throughout the region include Desert Grasslands, Great Basin Grasslands, Colorado Plateau Grasslands, Plains Grasslands, and Montane Grasslands (Robbie 2004).

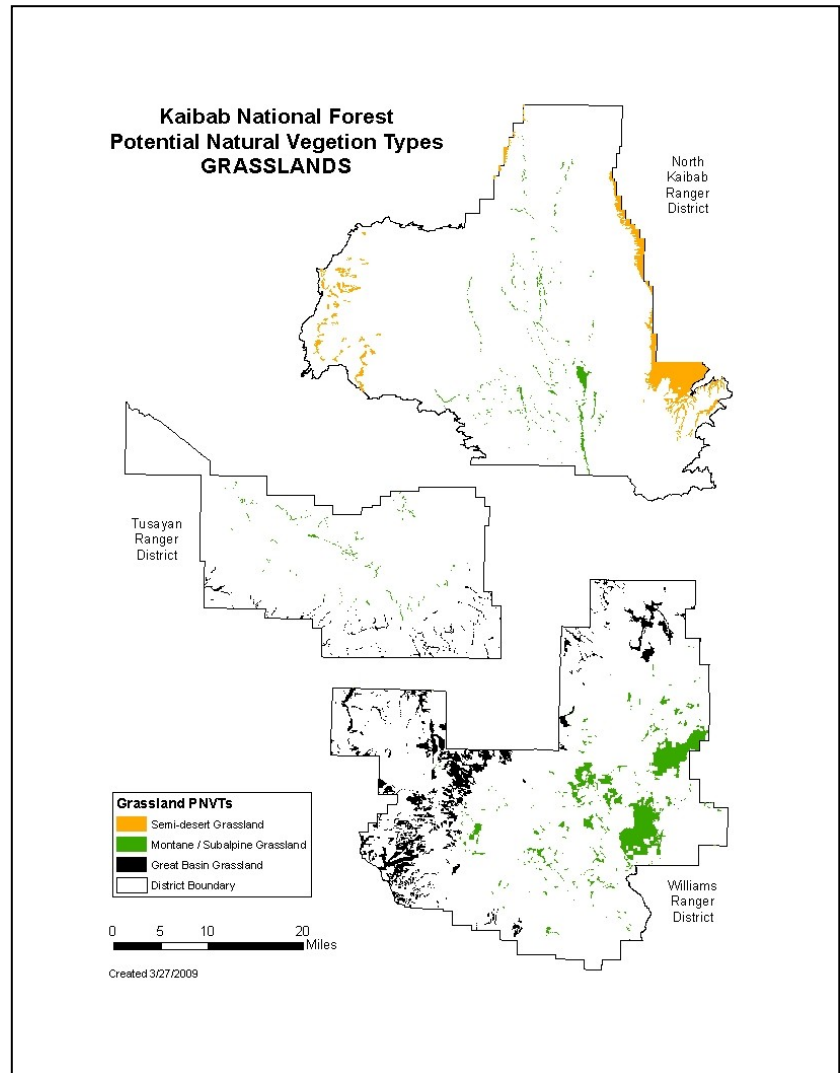


Figure 49. Grassland PNTs.

They are further characterized by intermediate levels of rainfall, long dry seasons, and extreme seasonal shifts from long warm summers to short cold winters, a dominance of grasses, and large grazing and burrowing mammals. Although they can provide important habitat for various wildlife, semiarid or desert grasslands like those found in southwestern North America have the lowest productivity of grasslands on the continent (Van Auken 2000).

The KNF is considered one of the driest forests in the National Forest system and the lack of permanent water in historic times likely limited the ability of some large herbivores to take advantage of its graminoids resources. The current proliferation of stock tanks, wildlife waters, and reservoirs associated with human development however, has likely changed the carrying capacity of grassland ecosystems for some large herbivore species.

Three major grassland PNVTs were identified for the KNF during the ongoing forest plan revision process; Montane/Subalpine (40,900 acres), Colorado Plateau/Great Basin (44,300 acres) and Semi-Desert Grasslands (25,000 acres). According to KNF Geodatabase files there are approximately 216,157 acres of the grassland cover type. Nevertheless, these grasslands, savannas, and mountain meadows continue to border every forest type on the KNF (Figure 49).

The only MIS for grasslands on the KNF is pronghorn antelope which use grasslands for both cover and forage (O'Gara and Yoakum 2004). Diminished habitat quality as a result of overgrazing and barriers to movement are cited among the top threats for the species (O'Gara and Yoakum 2004, Clark 2006 and references therein, Brown and Ockenfels 2007). Although cattle do not directly compete with pronghorn for forage (Kitchen and O'Gara 1982), they may affect fawn mortality through reductions in hiding cover.

Potential Management Implications

Threats to northern Arizona grasslands are many and include disrupted fire regimes, improper range use, urban development resulting in habitat fragmentation, and invasive species. Juniper encroachment is likely the greatest ecological threat facing grasslands today (Knapp et al. 2008).

Woody Encroachment in Great Basin/Semi-Desert Grasslands

Woody encroachment, which has been noted as a threat for all the grassland PNVTs on the KNF (Kaibab National Forest 2008a), has had adverse effects on grassland and savanna ecosystems in the interior West for the past 150 years (Van Auken 2000, Clark 2006, Browning et al. 2008). Recently, dramatic increases in areal density of juniper resulting from chronic domestic ungulate herbivory and the subsequent reduction of fine fuels and fire regime have led to the conversion of grasslands to savanna and woodlands throughout the central and western U.S. (Van Auken and Smeins 2008). On the neighboring Coconino National Forest, the woodlands and forests on Anderson Mesa have increased by about 12% since 1912. (Loeser and Miller unpublished data 2005 in Clark 2006)

Forest encroachment and subsequent grassland restoration is a dynamic process. West and Van Pelt (1987) referred to this as an intercommunity cycle. Past management practices have played a role in changing the balance of this cycle in favor of the advance of trees into grassland habitats by limiting and controlling fire on the landscape. An increase in fire frequency can create a pure grassland community whereas fire exclusion favors forest development. These fire-driven intercommunity cycles appear to be particularly prevalent in pinyon-juniper forests (West and Van Pelt 1987).

Overstory development affects understory cover through multiple pathways. In summary, the issues include: the ability of trees to use physical resources more efficiently than grasses or

forbs; once established, pinyon and juniper have allelopathic characteristics that inhibit grass and forb growth; trees slow nutrient cycling by exploiting essential and frequently scarce nutrients for long time periods; and the availability of soil moisture steadily decreases through direct competition and the interception of precipitation as tree crowns develop. Combined, these factors cause a proportional decrease in understory production as the overstory develops. The decrease in understory biomass can cause a decrease in the soil seed bank, limiting or delaying understory response after tree removal (Bedell 1987, Doughty 1987, Tiedemann 1987, Vaitkus and Eddleman 1987). On average, Arizona grasslands can produce about 600 pounds of forage per acre, but production drops to about 300 pounds per acre by the time canopy cover reaches 20% (Arnold et. al. 1964 *in* Doughty 1987).

Pronghorn generally utilize grassland and savanna habitat with < 20% canopy cover (Brown and Ockenfels 2007). Historically on the KNF, both the Great Basin and Semi-desert PNVTs were dominated by grass and forbs with canopy coverage less than 9% (Kaibab National Forest 2008a). Alteration of compositional and structural attributes of grasslands by woody encroachment can negatively affect pronghorn habitat by decreasing available forage, sighting distance to predators and disrupting migration corridors.

Fire helps maintain the integrity of grasslands by minimizing the establishment of encroaching trees and promoting nutrient cycling and decomposition; however drought conditions can significantly alter those effects (Ford et al. 2004a). The loss of the natural fire regime in southwestern grasslands, either directly through suppression activities, or indirectly through domestic livestock grazing and the subsequent loss of fine fuel loads, has led to woody plant encroachment and conversion to woodland systems that drastically alter habitat conditions for wildlife (Merola-Zwartjes 2004). Historically, the fire return interval for grasslands on the KNF ranged from 2.5 (semi-desert) to 35 years (Great Basin and Montane) (Kaibab National Forest 2008a).

Impacts from grazing, logging, and fire suppression practices that started in the late 1800s are still discernible on the landscape today (Covington and Moore 1992, Eddleman 1987). These practices reduced or eliminated the vegetation necessary to carry low intensity ground fires across the landscape, thereby altering the natural fire regimes. The development of roads, railroads, and trails further dissected the landscape and contributed to a change in the size of the average fire and an interruption in fire frequency (Covington and Moore 1994).

These activities and recent prolonged drought have led to a decrease in the overall diversity of grasses and forbs and have allowed for increases in the introduction and establishment of woody and exotic species (Dahms and Geils 1997, Allen and Breshears 1998, Van Auken 2000, Clark 2006). Overstory reduction of pinyon-juniper through mechanical means can be a highly effective method for improving grass cover and diversity. In New Mexico, Brockway et al. (2002) found that mechanical reduction of pinyon-juniper density to 6.1 trees/acre (15 trees/ha) resulted in savanna-like conditions reducing overstory cover from as much as 25% to less than 2%. They also found that grass cover in thinned plots increased by 250%, expanding from 9-38% cover with productivity increasing from 267.7-356.9 lbs/acre to 803-892.2 lbs/acre (300-400 kg/ha to 900-1000 kg/ha) in just two growing seasons. Blue grama and black grama responded most favorably to treatments.

On Anderson Mesa (Coconino NF), a collaborative effort between multiple agencies targeting woody overstory reduction has resulted in the treatment of approximately 22,000 acres of woody species (Clark 2006). Sheley and Bates (2008) found that following mechanical thinning, moderate to severe levels of prescribed fire in conjunction with seeding will recover grasses more quickly than natural recovery alone. They found that the density of recovering grass species in burned areas that had been seeded was 3 to 10 times greater than the nonseeded controls. Broadcast seeding following severe wildfire is routinely employed by many national forests, including the KNF. However, the efficacy of this method has been questioned in the recent literature, since many seeds can be washed away during seasonal monsoon rains. Daniels et al. (2008) found that the timing of broadcast seeding is critical to achieve successful results. They suggest that seeding in the fall, may be more effective in northern Arizona than during summer monsoons and encourage natural resource managers to coordinate restorative broadcast seeding efforts with climate patterns relevant to the region of interest. Furthermore Stella (2009), found that seeding following high-severity fire had minimal positive effects on increasing groundcover and preventing the establishment of non-native species. Managers on the KNF have learned to base seeding on soil types and that aerial seeding and seeding over snow improve success rates (D. Burger, *pers. comm.*).

Plant community response to pinyon-juniper control will vary depending on a variety of pretreatment conditions including; weather, grazing management, seed banks and plant composition. The reestablishment of displaced forbs and grasses following juniper control is necessary to recover the original plant community (Miller et al. 2005 cited *in* Sheley and Bates 2008). However, colonization of treatment sites by exotic annuals such as cheatgrass is a typical concern during restoration projects and efforts to restore native plants in weed infested areas may fail because of poor germination conditions and competition. As cheatgrass becomes established, it depletes soil moisture so that native perennials cannot establish (Young and Clements 2007). It eventually becomes so abundant that it increases ignition rates resulting in altered fire regimes and successional pathways (Adair et al. 2008). Although cheatgrass does appear vulnerable to high levels of heat stress and inadequate nitrogen availability (Adair et al. 2008) recent studies have shown that it may be particularly invasive at the edge of burned areas (Getz and Baker 2008, Grand Canyon trust unpublished data). Furthermore, seed mixtures typically broadcast in burned-over areas to prevent erosion and non-native plant invasion can inadvertently enhance cheatgrass germination if the mix contains inferior competitors (Getz and Baker 2008). Managers can mediate this effect by considering which plants enhance or prevent cheatgrass establishment when conducting broadcast seeding. For example, mixtures that contain a diversity of herbaceous plants are better able to adapt to varying site conditions over time than monocultures (Sheley and Half 2006). Cheatgrass currently occupies less than 1% of the Grasslands PNVT on the KNF (Kaibab National Forest 2008a) and efforts to mitigate increases of this invasive plant are ongoing.

Woody Encroachment in Montane/Subalpine Grasslands

Tree encroachment also poses threats to subalpine grasslands, although to a lesser extent. Approximately 8% of the Montane/subalpine grassland PNVT on the KNF has been invaded by conifers with encroachment happening at the rate of 12 to 16 feet per decade (Kaibab National Forest 2008a). On the North Rim of the Grand Canyon, Moore and Huffman (2004) found that

tree encroachment of meadows started in the early 1900s. They identified release from ungulate browsing pressure (Binkley et al. 2006) and fire exclusion (Wolf and Mast 1998, Fulé et al. 2002) as likely causes. Woodland encroachment into adjacent alpine grasslands may happen at varying spatial scales. Sankey (2008) found that aspen suckers may establish themselves at high densities, but in close proximity of the parent tree (tens of meters) while Douglas-fir seedlings establish themselves at variable densities and at significantly greater distances from the adult trees (0.6-1.2 miles or 1-2 km.). This gives them an evolutionary advantage with regard to forest expansion, particularly in areas of limited to no canopy cover. In that same study, Sankey (2008) found that Douglas-fir seedlings established themselves in adjacent grasslands at distances of 275 feet (84m) from the parent trees while aspen suckers established themselves up to 82 feet or 25m ($p < 0.01$). Douglas-fir seedling distances increased with increases in percent canopy, although seedling densities tended to decrease with increases in aspen canopy cover. He further found that aspen encroachment was happening in small spatially distinct pulses, while Douglas-fir encroachment was a gradual continuous process.

Tree removal and prescribed fire have great potential as restoration tools, particularly since frequent fire may have maintained coniferous alpine forests during historic times (Fulé et al. 2002). However, even when encroaching conifers have been removed from montane grasslands, additional restoration methods may be necessary to reestablish grass species. In a chronosequence study of open meadow, young (< 75 years), and old (95 to > 200 years) forest, Lang and Halpern (2007) found that over 70% of meadow species were absent from the seed bank which was of limited diversity. Their data suggest that most species will not reestablish themselves via the seed bank through tree removal alone. This phenomenon has been found to limit restoration of other grassland systems as well (Lett and Knapp 2005, Knapp et al. 2008). Targeting small areas or those areas that likely retain original meadow soils and seeds during the initial stages of conifer encroachment may be the most effective restoration strategy for invaded areas (Lang and Halpern 2007).

Woody tree removal has been an ongoing emphasis of restoration projects across the KNF with particular emphasis on the Williams District. In 2006, Forest Service and Arizona Game and Fish Department specialists identified approximately 179,195 acres of historic grasslands on the Tusayan RD and 177,827 on the Williams RD as priority areas for grassland restoration treatments (Kaibab National Forest 2007a)(Appendix 9). Identification of these areas supports the Kaibab Forest Plan, as amended and the ongoing forest plan revision process by identifying historic grassland cover types to be removed from the suitable timber base and identifies areas for future restoration work that will include thinning encroaching trees. This should enhance understory development by opening up the canopy while improving pronghorn travel and sight-distance to predators. Restorative actions that open up overstory and reduce tree density should improve understory plant diversity especially for cool-season grasses such as bunch grasses (Moore and Dieter 1992, and references therein, Moir et al. 1997, Kaibab National Forest 2007a). Two areas, Government and Garland Prairie, are particularly important because historically they have provided important grassland habitat for pronghorn and other wildlife species. Past research has shown that Garland Prairie has significantly better forage for pronghorn than other locally available habitat. In 2002, (Miller and Drake 2002) evaluated diet of both pronghorn herds and found that the mean species diversity of forbs was significantly higher ($p \leq 0.01$) for the Garland Prairie herd than that on Anderson Mesa. They also found that

usage of forbs was higher on Garland Prairie; 66.2 - 83% vs. 25.4 - 61.7% for Anderson Mesa. Garland Prairie is threatened by encroaching conifers and other woody species (Kaibab National Forest 2008a). The Williams RD continues to work with the Arizona Game and Fish Department during project planning to identify modifications that benefit pronghorn (Kaibab National Forest 2007a). Habitat models that have been developed to assist Arizona Game and Fish with managing woody encroachment on Anderson Mesa (Clark 2006) could help to better inform future management strategies on the KNF.

Ganey et al. (2009) conducted a baseline assessment of the small mammal and vegetative community in Garland and Government Prairies. Median ground cover across all transects ranged from 55-75% with most transects dominated by grasses. Each prairie supported 2-3 small mammal species: deer mice (*Peromyscus maniculatus*) and Mogollon voles (*Microtus mogollonensis*) were common and spotted ground squirrels (*Spermophilus spilosoma*) present. The limited spatial and temporal extent of the study however, precludes its ability to draw inferences about causal relationships between the vegetation and small mammal community at this time. This work should however, provide a platform from which to develop future assessments in those areas.

Ungulate grazing

Livestock grazing has had a long and contentious history with approximately 70% of the western U.S. grazed (Fleischner 1994, Brown and McDonald 1995). Historically, much of the Nation's attention shifted West after the Civil War and by the 1890s close to a million cattle grazed throughout what is now Arizona. Drought conditions between 1891 and 1894 led to the starvation and death of about 250,000 head of cattle, but heavy grazing pressure continued into the 1900s. Nearly five million head of livestock grazed the state of Arizona by that time (<http://www.emaprogram.com/ImportanceGrasslands.asp>). Across the Southwest, grazing in the pinyon-juniper habitat was estimated to have caused a 60 to nearly 100% decrease in available forage (Stoddart et. al. 1975). Overgrazing was being addressed by the KNF in the 1950s, but substantial progress started in the 1970s. Since that time, all grazing allotments on the KNF have been through the NEPA process and are continually updated and reviewed for Forest Plan consistency.

Where consistent with other multiple use goals and objectives, Congress directs the Forest Service to allow livestock grazing on suitable lands (Multiple Use Sustained Yield Act of 1960, Forest and Rangeland Renewable Planning Act of 1974, Federal Land Policy and Management Act of 1976, National Forest Management Act of 1976). Accordingly, the KNF is directed by its forest plan to provide forage for livestock grazing.

Frequently cited ecological effects of grazing include alteration of species composition and communities, disruption of ecosystem functioning, and alteration of ecosystem structure (Fleischner 1994). Grazing also reduces the integrity of soil crusts (a soil surface community of mosses, lichens, cyanobacteria, and other organisms) an important regulator of proper ecosystem function in semiarid regions. Loss of soil crusts can lead to accelerated range and grassland degradation through increased erosion, and decreased productivity. Recently, predictive models have been used to improve management practices of this important ecosystem component (Bowker et al. 2006).

Grazing management is based on controlling the intensity, timing, frequency, and selectivity of livestock. In general, the KNF continues to keep grazing at conservative use levels (30–40%). This grazing intensity, based on % use of forage by weight, should provide for plant integrity, density, diversity and regeneration over time (Holechek and Galt 2000). Because grassland systems have been grazed for so long, moderate levels of grazing intensity may facilitate native plant diversity better than complete cattle removal or high intensity short-term grazing. Loeser et al. (2005) found that past cattle grazing led to decreases in perennial forbs and increases in annuals and exotic grasses, particularly cheatgrass, in both high-impact grazing sites as well as sites where cattle had been completely removed ($p < 0.01$). Valone et al. (2002) note that it may take 20 years or more for perennial grasses to respond favorably to cattle removal in some grassland systems. Thus active revegetation programs may be necessary to improve plant diversity.

To mediate the negative effects of ungulate grazing and drought, the KNF implements deferred-rotation grazing with a special emphasis of deferment during the spring (C. Hydock *pers comm.*). This should help to facilitate growth of cool season grasses that have been particularly impacted by lack of precipitation and drought.

In addition to direct threats to the plant community, indirect effects of grazing include a reduction of the fine fuels necessary to maintain periodic fires. Climate change and prolonged drought further exacerbate this condition. Loss of fine fuel loads can in turn lead to woody plant encroachment in semiarid grassland systems. Ordinarily fire prevents establishment and growth of woody plants in grassland systems, however fire has decreased in grasslands over the past 150 years, (Van Auken 2000, 2008 and references therein).

Related to grazing and rangeland management are fences. Within a decade of erecting barbed wire fences on western rangelands, Caton (1877; as cited by the North Dakota Game and Fish Department [gf.nd.gov/multimedia/pubs/prong-mgmt-guide-pt7.html]) was reporting 4-foot (1.2m) high fences restricting pronghorn movements:

“ This inability to leap over high objects may no doubt be attributable to the fact that they live upon the plains, where they rarely meet with such obstructions, and so they and their ancestors for untold generations have had no occasion to overleap high obstructions, and thus from disuse they do not know how to do so, and never attempt it when they do meet them ”.

Fences can restrict the movement of pronghorn, cause changes in herd distribution, deaths, and fragment or isolate habitat. In Utah and Colorado, Harrington and Conover (2006) estimated wire fences caused 0.16 ungulate mortalities/mile of fence. Pronghorn mortality was higher than both elk and mule deer and significantly more deaths resulted from woven-wire fences topped with single strands of barbed wire as opposed to multiple strands ($p < 0.01$). Sheldon (2005) found that 64% ($n = 28$) of monitored pronghorn utilized areas with the lowest fence densities as migration routes in Wyoming. O’Gara and Yoakum (2004) list management recommendations for fencing that include reduced fencing and raising the bottom wire and replacing it with smooth wire to allow pronghorn to pass beneath the fence line. Pasture fencing is evaluated annually on the KNF and changes are made specifically to benefit pronghorn antelope. In recent years the

Williams RD has targeted areas important to pronghorn for fence modifications, e.g., grazing practices were changed in Davenport Lake, a key forage area, to include the installation of 82 pronghorn crossings along the fence lines. The majority of woven wire on the KNF has been removed. A barb-less bottom wire 18 inches from the ground is the KNF standard for new range fences.

Another manmade barrier that continues to be one of the most significant factors affecting pronghorn populations is increasing urbanization (Brown and Ockenfels 2007). “Exurban” development is becoming increasingly prevalent throughout the Southwest and some researchers believe that habitat conversion for human use has been far more detrimental to habitat than grazing pressure (Brown and McDonald 1995).

Since this document was last updated The Forest Service Plan incorporated two amendments that should benefit grassland and pronghorn habitat. In 2003, amendment 5 set aside 1,355 acres of suitable pine as pronghorn habitat (Frenchy Ecosystem Management EA). In 2004, amendment 7 established forest wide standards for the treatment of invasive weeds (Kaibab National Forest Plan as amended). As of 2009, approximately 4,000 acres have been treated for noxious weeds in grassland systems (M. Hannemann *pers. comm.*).

Habitat Data

The Nature Conservancy statewide grassland assessment has found that approximately 1,647,356 acres in northern Arizona (out of 2,685,266 open habitat statewide) are still in an open state, with at least 3,320,667 (out of 5,705, 160) acres in restorable condition (only 10-35% shrub cover). Approximately 2,400 acres of grasslands (not including montane grasslands) in northern Arizona are in a non-native state dominated by perennial non-native grasses, while 1,217,668 acres have been converted to shrublands (<http://www.emaprogram.com/ImportanceGrasslands.asp>).

Woody Encroachment

On the KNF, 87% of historic grasslands have succeeded to forest or savanna (Kaibab National Forest 2009b). Several analyses have been completed with regard to woody encroachment and grassland/savanna invasion on the KNF. All offer similar results. A 2003 assessment found that pinyon-juniper woodland, ponderosa pine and juniper woodland were the primary invaders of grasslands and savannas systems on the SZ. This invasion by woody species caused a concurrent decrease in understory productivity. Many grasslands capable of producing 550-4500 lbs per acre annually were found to produce less than 10% of that range equaling a total loss of roughly 90% grass and forb production across 13,000 acres. In that same study there was an estimated loss of 9500 tons of understory productivity (Kaibab National Forest 2003a). A subsequent analysis on the William’s RD looked at historic vegetative structure (grassland vs. open/closed forest) and ponderosa pine-dominated Mollisols, or those soils that likely supported grasslands historically (Kaibab National Forest 2008b). That analysis concluded that approximately 69% of ponderosa pine forests occurring on Mollisols were classified as “open” and approximately 15% of the area classified as grassland. Currently, 9% of ponderosa pine stand structure on Mollisols exists in an “open” state. About 7% of Mollisols are classified as grassland and the remaining habitat is classified as “closed forest” (Kaibab National Forest 2008b).

A more recent forest-wide analysis that examined soil type information collected during Terrestrial and Ecological Surveys (TES), and stand density indices (SDI) calculated from stand exam data across the forest, concluded that the most heavily altered system has been savanna (Figures 50-51). Eighty-three percent of what was historically savanna is now forest and only 15% is still savanna. Additionally, about 50% of historic grasslands have also succeeded to forest (Table 29). Savannas are currently dominated by ponderosa pine and pinyon-juniper, while grasslands are dominated by pinyon-juniper and ponderosa pine respectively (Figure 52). This analysis could only be completed for areas that had existing stand exams, therefore the extent of grasslands may be underrepresented (Kaibab National Forest 2009b). The SDI indices used during these analyses correlated strongly (0.88) with accepted forest-wide canopy cover (CC) thresholds for delineating grasslands and savanna from forest. An SDI value of 100 was equivalent to 9.5% canopy cover (10% or less CC is typically used to delineate grasslands) and an SDI of 400 was equivalent to 27.2 which corresponds well with the 30% or less threshold used to delineate savanna.

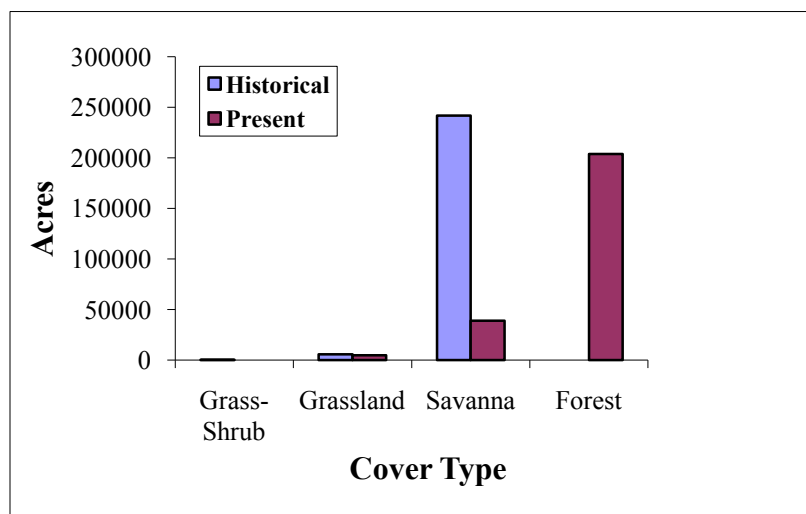


Figure 50. Changes in historical cover type, based on Mollisol soil presence (pre-1980) and current (2009) stand density indices. *Note: Historic forest was not analyzed, only those areas identified as historic grassland, savanna, and shrubland were analyzed.*

Table 29. Comparison of historic grassland and savanna conditions to current (2009) conditions.
Note: Grass-Shrubland was dropped from additional analyses due to its limited occurrence (≤ 52 acres).

Historic to Current Conditions	Historic to Current Ratio
Grassland to Forest	0.50
Grassland to Grassland	0.12
Grassland to Savanna	0.37
<i>Total</i>	<i>1</i>
Savanna to Forest	0.83
Savanna to Grassland	0.02
Savanna to Savanna	0.15
<i>Total</i>	<i>1</i>

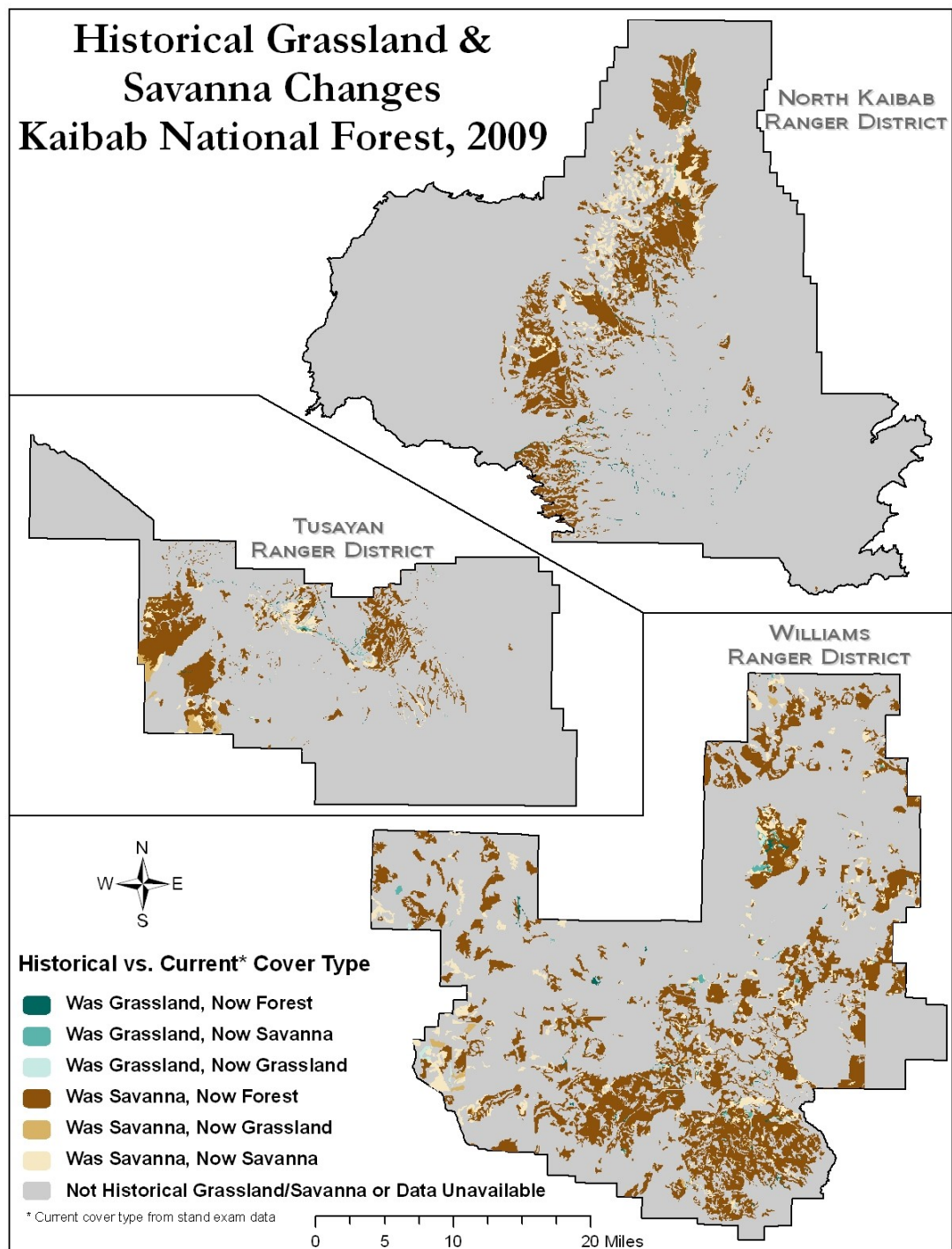


Figure 51. Forest-wide overview of historical changes in grassland and savanna habitat, KNF .

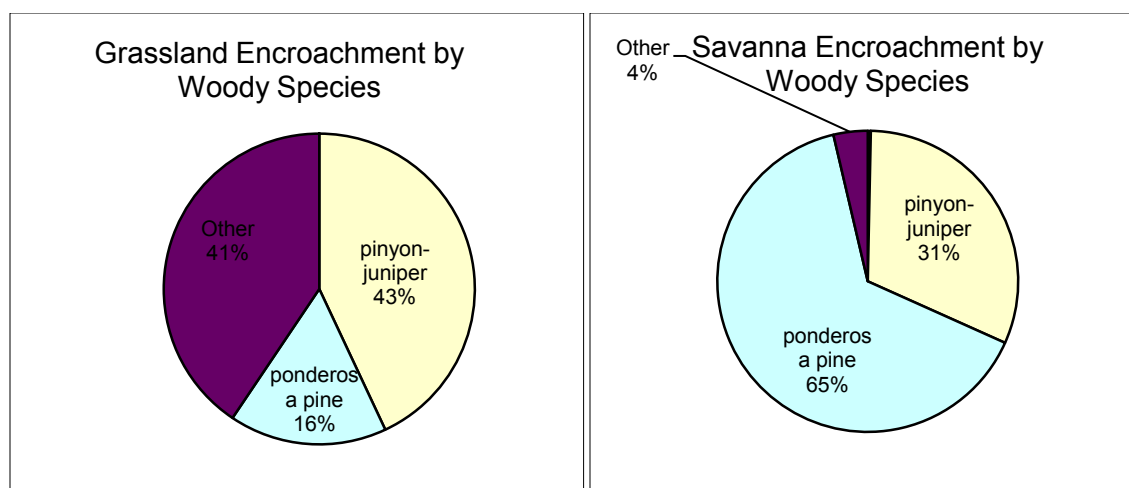


Figure 52. Encroachment of historic grassland and savanna by woody species. Pinyon-juniper includes juniper woodland. “Other” includes various woody trees/shrubs including gambel oak, mixed hardwood, sagebrush, oak woodland, sagebrush, shrubland, treeland, and other hardwood.

Ungulate Use

The KNF has used Parker three-step surveys to evaluate range trends and conditions since the 1950s. Although the technique is biased towards evaluating forage for domestic herbivores, the data from these surveys represent a long-term and consistent record of vegetation conditions on the Forest. An earlier version of this document summarized Parker Survey data for the KNF from 1950-2001 (Kaibab National Forest 2003b). Those analyses showed that in general, across the forest, there was an increasing trend with regard to the ratio of cool to warm season grasses during the 1950s to 1980s with the trend becoming static from 1980-2001. Plant cover essentially remained unchanged across the forest during that time with increases in litter cover attributed to decreased livestock grazing pressure. A recent review of pertinent range documents that presented summary and trend data from Parker Surveys since 2001 suggest that in general, cool season grasses are now declining across the forest while warm season grasses have remained static or have increased. The decline in cool season grasses over the last 10 years coincides with ongoing drought and decreased precipitation noted throughout the region (Breshears et al. 2005). In general there has been a static to positive increase in overall plant numbers and litter cover (Kaibab National Forest Range files). These recent changes on the KNF are a result of a climatic shift that favors warm season growing conditions over cool season growing conditions. This line of evidence suggests climate change as the primary factor influencing cool season grasses, although improper grazing management can further exacerbate these effects. Grazing management is based on controlling the intensity, timing, frequency, and selectivity of livestock. To that end, KNF Range Management have been proactive in mediating drought effects by keeping grazing at conservative use levels (30-40%), this includes wildlife use. Conservative grazing intensity which provides for plant vigor, density, diversity and regeneration over time is associated with approximately 35% use levels (Holechek and Galt 2000). Galt et al. (2000) also recommends a 25 percent utilization guideline for livestock, with 25 percent allocated for wildlife and natural disturbance, and the remaining 50 percent for site protection.

Since 2003, the total number of grazing allotments on the KNF has decreased from 41- 38. Twenty seven allotments are currently managed on the Williams RD, 8 on the NKRD, and 3 on the Tusayan RD. Summary information for allowable stocking levels from 1971, 1983, 2002 and 2009 is presented below (Table 30). Since 1971, the maximum allowable stocking levels have decreased by approximately 45,194 acres across the forest.

Table 30. Changes in the total number of allowable Animal Unit Months by Ranger District, Kaibab National Forest.

Ranger District	Animal Unit Months			
	1971	1983	2002	2009
Williams	61,295	41,255	41,547	40,193
North Kaibab	16,720	14,590	12,508	11,532
Tusayan	30,530	30,530	19,486	12,626
Forest Total	108,545	86,375	73,541	63,351

Management Response

Grassland restoration continues to be a consistent goal of the KNF. It is expected that this trend will continue in the long-term. Available summary data show that approximately 14,000 acres of grasslands were treated as of 1995. An additional 12, 455 acres were treated for wildlife benefit. Such activities include prescribed fire, creating openings to allow for better understory vegetation development, seeding, and control of encroaching trees (Figure 53). It is expected that additional management activities such as herbicide treatments that target noxious weeds will play an increasing role in grassland management, as new areas to be treated are identified. An environmental impact statement was completed in 2003 for 3 northern Arizona forests. Highlights of that analysis can be found in the ponderosa pine section of this document.

Grasslands

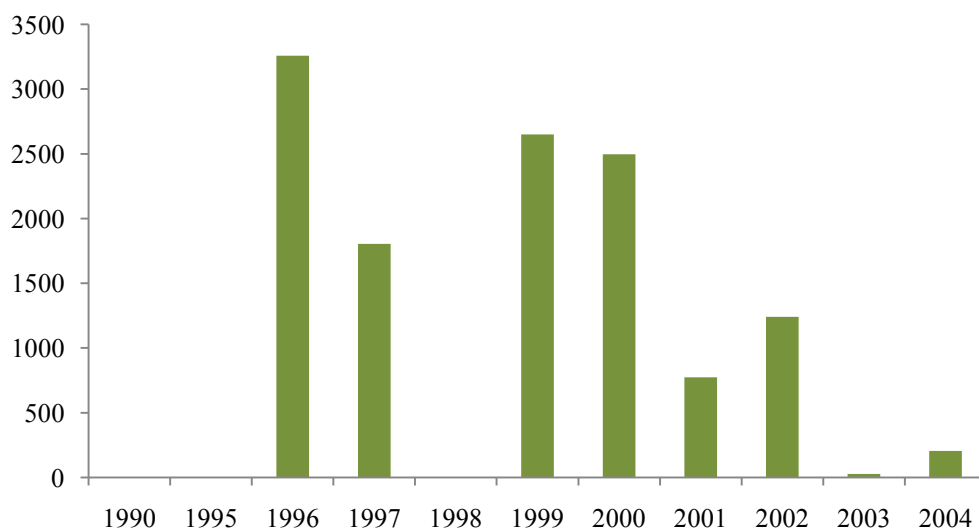


Figure 53. Grassland acres treated on the KNF for wildlife benefit (FACTS database).

Fire can have both positive and negative effects on grassland restoration depending on the intensity with which it burns. Low intensity, slow moving fires facilitate native grass regeneration while high intensity fire can facilitate the establishment of non-natives such as cheatgrass and other weeds. For example, on the NKRD, the Bridger Knoll Fire burned approximately 50,000 acres on the west side of the Kaibab Plateau in 1996. Old Parker 3 Steps surveys conducted in the area, show those areas that burned as high severity fire in pinyon-juniper woodland were also the densest and have subsequently become infested with cheatgrass. Conversely, Great Basin Grassland/shrub communities typically burned with low intensity and native grasses and shrubs have responded favorably to the fire in those areas (D. Burger *pers. comm.*). Other fires that have affected grasslands include the Warm Fire which burned about 1,376 acres of grassland. The majority of those acres were classified as low severity (Appendix 7 and 8). In 2007, the Slide Fire burned several thousand acres on the west central portion of the District, within the Bridger-Knoll perimeter; however the severity of that fire on grasslands was moderate to low. Several small fires have been managed for resource benefits in montane grasslands since 2003.

In general, low intensity, slow moving fire, and more gradual mechanical approaches, such as thinning should be better for native perennial plants. Current thinning projects on the KNF utilize an agra axe (Bobcat mounted with hydraulic sheers) and/or chainsaws to remove encroaching trees. The resultant slash is usually dropped on site to improve ground conditions. Ongoing research suggests that these methods can improve soil, understory and wildlife habitat conditions (Clark 2006 and references therein). Although this method may be labor intensive, it may be more effective than fire, and causes fewer disturbances than methods such as chaining which may improve propagation conditions for noxious weeds. Disturbance facilitates invasion by changing the integrity of the original plant community. Increases in resource and nutrient availability, alterations in the soil microclimate and its associated microbes and increased substrates promote germination conditions for non native plants (Adair et al. 2008).

Habitat Trend Estimate

At this time, pronghorn habitat on the KNF appears to be stable in the short-term. In the long-term however, state and transition models developed during the forest plan revision process suggest that all grasslands on the KNF are trending away from historic reference conditions. The trend away for Great Basin Grasslands and Semi-Desert Grasslands was found to be low to moderate while the trend for Montane Grasslands was high. Conifer encroachment is expected to continue to negatively affect Montane grasslands, while pinyon-juniper encroachment is expected to reduce Great Basin and Semi-Desert Grasslands (Kaibab National Forest 2008a).

Current management should help to move grasslands toward desired conditions, although the effects of reoccurring drought or catastrophic fire could easily lead to a declining trend.

Riparian Habitat and Associated PNVTs

The U.S. Fish and Wildlife Service (USFWS) estimates that 70% of the riparian habitat nationwide has been lost or altered. Eighty percent of all vertebrate wildlife in the Southwest depends on riparian areas for at least half of their life. Of the 1200 species in the United States listed as threatened or endangered, 50% depend on rivers and streams. (<http://www.americanrivers.org/library/river-facts/river-facts.html>).

Background

Southwestern riparian ecosystems, which include ephemeral, intermittent, and perennial streams and rivers, are ecologically dynamic habitats characterized by linear patches of vegetation (Knopf et al. 1988). Riparian systems have decreased in size over the past 100 years, largely a result of human development (Perriman and Kelly 2000, Sada 2008). In the west, factors such as livestock grazing, beaver extirpation, and road development are commonly attributed to the loss of riparian habitat (Lucas et al. 2004 and references therein). Nonetheless, riparian areas are considered one of the most important habitat types for Arizona and the Southwest (The Arizona Wildlife Linkage's Workgroup, 2006). Activities such as channelization and river diversion, domestic livestock grazing, timber, invasive species, recreation and infrastructure development have led to a 90% reduction of this habitat type in Arizona and New Mexico compared to historic (presettlement) conditions (Graham and Sisk, 2002).

In Arizona, The Riparian Habitat Task Force (1990) defines riparian areas as “vegetation, habitats, or ecosystems that are associated with bodies of water (streams or lakes) or are dependent on the existence of perennial, intermittent, or ephemeral surface or subsurface water drainage”. These riparian areas act as water, cover and food sources and as migration corridors for a wide range of wildlife species (Latta et al. 1999). In addition, they may act as a buffer zone between upland fires and the in-stream environment (Pettit and Naiman 2007).

Perhaps the most heavily impacted habitat type in Arizona, millions of people depend on and recreate in riparian habitats. About 65% of the Southwestern animal species use riparian habitats during all or part of their life cycle (Dahms and Geils 1997). Most species on the Region 3 Regional Forester's Sensitive Species list are riparian dependent or thrive in healthy riparian habitats (Dahms and Geils 1997). Although less than 1% of the western U.S. is covered by riparian vegetation (Knopf et al. 1988), approximately 50% of all southwestern bird species breed primarily or exclusively in deciduous riparian habitat (Tewksbury et al. 2002 and references therein). In northern Arizona, there are only three perennial rivers, but hundreds of ephemeral streams (Shaw and Cooper 2008).

In recognition of their unique attributes, riparian systems are currently protected by the Federal government through two executive orders. Order No. 11988 (Floodplain Management) requires agencies to minimize adverse impacts to floodplains, while Order No. 11990 (Protection of Wetlands) mitigates impacts to, and enhances the natural resource value of such systems. Furthermore, the USFS is specifically directed under the National Forest Management Act (NFMA) of 1976 to manage riparian vegetation and minimize negative impacts to such areas (36 CFR Part 219). To that end, the USFS recognizes the importance of riparian areas during project planning and implementation and emphasizes their protection while managing them within

multiple use guidelines. In 2006, a new federal bill was signed to facilitate riparian restoration in the Southwest due to the extreme value of riparian ecosystems and their high risk for exotic species invasion (Salt Cedar and Russian Olive Control Demonstration Act: United States Public Law 109-320 2006).

Although the current forest plan for the KNF manages riparian systems within this broad context, it offers little management direction with regard to some of the more unique aquatic features that occur on the Forest. These include seeps, springs, ephemeral wetlands and one perennial stream, North Canyon Creek. The NKRD has the highest density of seeps and springs in Arizona, approximately 129 out of 709 total statewide, comprising about 18% of all Forest 4th code watersheds in Arizona (Kaibab National Forest 2008a). The high density of seeps and springs found on the Kaibab Plateau make this area significant in terms of ensuring the sustainability of these rare features in the southwest (Kaibab National Forest 2009).

Using the general descriptions provided by the Arizona Partners in Flight (AZPIF), riparian associations occur in or adjacent to drainage ways and/or floodplains, and are characterized by species and/or life forms that are distinctly different from the immediately surrounding non-riparian habitat (Latta et al. 1999). These characteristics typically include different soil types that are generally deeper and contain higher soil moisture than the surrounding areas (Latta et al. 1999).

The AZPIF Bird Conservation Plan defines four kinds of riparian features (Latta et al. 1999). Open water and freshwater marshes are the most common riparian habitats occurring on the KNF. Low elevation riparian (generally less than 4,000 ft) and high elevation riparian (generally 4,000–11,000 ft.) associations are also present on the Forest, but both are limited in distribution. The most typical forms of riparian habitat on the KNF are developed stock tanks and guzzlers and trick tanks developed as wildlife waters. These structures typically provide water, but do not necessarily support riparian vegetation.

Although the KNF is limited in riparian habitats, their very scarcity underscores their value. Marshlands and open waters are particularly important for waterfowl, wading birds, and resident landbirds. Migrating birds have been documented using riparian habitats on the KNF. Flocks of brown pelicans (*Pelecanus occidentalis*), white-faced ibis (*Plegadis chihi*), and gulls (*Saundersilarus spp.*) have been reported resting at Davenport Lake on the Williams RD when this ephemeral feature is holding water. However, the limited size, distribution, and ephemeral nature of riparian features on the KNF make them unlikely to be critical in supporting populations of any given avian species. Research reviewed by Hutto (2000) identified general patterns in habitat use during migration: migrating birds stopping en route do not use available habitat in a random fashion; seasonal patterns appear consistent year after year; the configuration of habitat types in the broader landscape may influence the probability of use; habitat patches are used by some migratory species only when they exceed a certain minimum size; and the value of the habitat used by migratory birds appears closely related to food production at those sites.

MIS for riparian habitat on the KNF include aquatic macroinvertebrates, Lincoln's sparrow, Lucy's warbler, Yellow-breasted chat, and Cinnamon teal.

Kaibab National Forest, Riparian, Wetland, & Stream Habitats

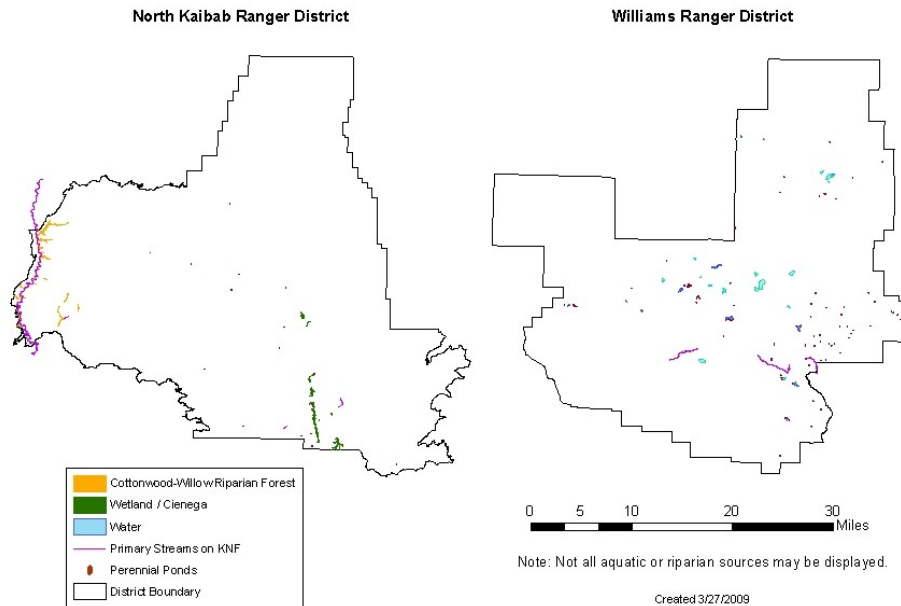


Figure 54. Distribution of aquatic habitat across the KNF. Does not include trick tanks and guzzlers.

Low Elevation Riparian Habitat and Cottonwood-Willow Riparian Forest PNV

Low elevation riparian habitat includes perennial, ephemeral, or sub-surface water that support woody shrubs and deciduous trees (Latta et al. 1999). This kind of habitat is typically found below the Mogollon Rim. The only low elevation riparian habitat on the KNF occurs in Kanab Creek, beneath the western edge of the Kaibab Plateau and covers about 1,200 acres. Historically Kanab Creek was a perennial stream with 29% of the stream distance within the forest (Kaibab National Forest 2008a). Elevations range from 3,500-6,000 feet and about 80% of the area has slopes greater than 40%. Unlike the Kaibab Plateau, the climate is semi-arid with the mean annual precipitation ranging from 8-12 inches. The entire Forest Service portion of the drainage is a designated Wilderness Area and has ephemeral flow due to the diversions north of the KNF. Cottonwood-Willow Riparian Forest is the comparable PNV designation for Kanab Creek (Steinke 2007, Kaibab National Forest 2008a).

Management Considerations

Currently the riparian corridor on Kanab Creek is dominated by salt cedar (*Tamarisk spp.*). There are only 5-10 large cottonwoods per mile of stream with few willows and herbaceous cover. This deviates widely from the reference conditions. Historically the Cottonwood-willow PNV consisted of 55% mid-aged to mature cottonwood and willow trees, 25% younger trees and 20% in grass, shrubs, and tree sprouts (Kaibab National Forest 2008a). Grazing has not occurred on the KNF portion of Kanab Creek since 1996 and a Decision Notice formalizing the no grazing

policy was signed in 2001. Furthermore, Kanab Creek is also designated as a Wilderness Area. The KNF forest plan revision team has discussed adding another possible designation (Wild and Scenic River Status) to the Kanab Creek area in the new KNF land management plan. However, this designation would offer little additional protection to the area beyond what currently exists (C. Minor, *pers. comm.*). Although these policies are intended to protect the habitat from further degradation, a hands-off management policy can be somewhat problematic in an area that has suffered from historic disturbance. For example, one negative outcome from these well intended policies is that the KNF no longer conducts Parker surveys in the area, because there is no need to monitor range conditions. Lack of monitoring diminishes the KNF's ability to detect change, positive or negative.

Unlike many other areas on the forest, fire has played a limited role in shaping this ecosystem. The primary disturbance agent for this habitat is springtime flooding that creates germination conditions for seedlings. Restricted water flow by diversions upstream of the Forest boundary has greatly diminished flooding disturbance since the 1900s (Kaibab National Forest 2008a). Future management to mitigate such conditions could include experimental flooding. Currently, the lack of flooding in Kanab Creek makes it difficult to achieve forest plan objectives that specify at least 3 age classes of woody riparian species with 10% of woody plant cover in sprouts, suckers, seedlings, and saplings. Experimental flooding has been used on the Rio Grande to stimulate cottonwood suckering after mechanical removal of invasive species. However, negative outcomes of such work have included increases in salt cedar that germinate in disturbed soil following mechanical removal of other invasive species (Sprenger et al. 2002).

In addition to inadequate germination conditions, invasive species such as salt cedar and russian olive (*Elaeagnus angustifoli*) are out-competing and displacing native woody vegetation in Kanab Creek (Steinke 2007). Current estimates suggest that tamarisk occupies approximately 440 acres of the Kanab Creek Wilderness area with cheatgrass present in limited areas (Phillips 2008). Tamarisk control could extend to about 565 acres and would include mechanical, cultural, herbicidal, and biological treatments (Kaibab National Forest 2008a). Removal is the first step towards control of tamarisk. However restoration of riparian ecosystems involving little active revegetation has resulted in limited recovery of native riparian vegetation (Harms and Heibert 2006, Bay and Sher 2008). On the Colorado plateau, Harms and Heibert (2006) found that cutting and burning reduced salt cedar cover by 82-95%, but native plant cover and diversity did not increase over time. Soil substrate can further affect such restoration efforts. Bay and Sher (2008) surveyed 28 restoration sites in the southwest and found that native plants were positively associated with sites containing coarse soils and gravel while non-natives experienced negative associations with those same soil types. Furthermore, tamarisk removal alone has limited effects on water loss through evapotranspiration rates. Nagler et al. (2008) found that only 1% of the mean annual river flow along the Lower Colorado was lost to tamarisk. They suggest habitat improvement projects that incorporate a 10-15% mix of native species, such as cottonwood, willow, or mesquite among tamarisk, may be a more cost effective way to improve riparian habitat. This strategy may be especially beneficial for enhancing habitat for birds (Sogge et al. 2005, 2008).

In general, tamarisk has been considered negative for ecosystems throughout the western U.S. because of its high water consumption, and low habitat value. New research however, questions

this existing paradigm. Some researchers have shown that under *some* circumstances, tamarisk may be more important to both migrating and resident birds than was once thought (Fleishman et al. 2003, Owen et al. 2005, van Riper 2008). Recent studies suggest that the southwestern willow flycatcher (*Empidonax traillii extimus*: flycatcher) a federally-listed endangered species, may depend on tamarisk at least in part, for nesting. The current breeding range of the flycatcher overlaps greatly with the core range of salt cedar in the southwest, and flycatchers have been known to nest in tamarisk habitat (Sogge et al. 2005, 2008). Although Kanab Creek may have potential flycatcher habitat, the area is small in extent and previous surveys have failed to detect this endangered subspecies. However, surveys have been limited and additional work should be conducted prior to extensive habitat manipulation in the Kanab Creek area.

Future monitoring and assessments of invasive vegetation may become increasingly important to insure the future integrity of the Kanab Creek ecosystem. A recent survey of invasive plant taxa in western riparian ecosystems identified 12 invasive plant species that can be used to indicate ecological change as a result of anthropogenic disturbance (Ringold et al. 2008). At least 4 of the species; cheatgrass, leafy spurge (*Euphorbia esula*), russian olive, and salt cedar are known to occur on the KNF (Kaibab National Forest 2008a, Phillips 2008) and could be incorporated in future stream surveys as a proxy for stream condition. KNF range specialists have also noted an increase of cheatgrass along Kanab Creek (D. Burger, *pers. comm.*).

High Elevation Riparian Habitat and Wetland Cienega PNV

AZPIF characterizes high elevation riparian as occurring in steep, narrow canyons, drainages, or in mountain meadows at elevations above 4,000 feet with frequent or permanent water (Latta et al. 1999). Douglas-fir, aspen, cottonwood, willow, and oak are some of the tree species indicative of high elevation riparian habitat. North Canyon Creek on the NKRD and Big Spring on the Williams RD are the only high elevation riparian associations on the KNF. In general, flows and stream lengths can vary dramatically between years and seasons.

The portion of the North Canyon watershed occurring on National Forest lands is nearly entirely within the Saddle Mountain Wilderness Area, located within the extreme southeast part of the NKRD. The entire surface flow of North Canyon Creek is generally restricted to National Forest lands. Surface flow originates at the head of North Canyon. A recent aquatic assessment concluded that North Canyon Creek is at or near historic water flow conditions, and runs for approximately 1-6 miles (Kaibab National Forest 2008). The length of the creek varies, depending on winter snow pack and seasonal temperature and precipitation patterns. A survey in June 1990 found 4-4.5 miles of perennial water, but flow was measured at 1-4 cubic feet per second. Slopes vary tremendously, from about 60% near the springs, to 4-5% in some of the middle and lower reaches. The upper drainage is at about 8,000 feet elevation and the steep, narrow canyon creates a mix of microsite conditions. The isolation and mix of microsite conditions combine to support 167 invertebrate taxa, including 4 endemic invertebrates and 5 rare plants (Stevens 2009). Ponderosa pine can be found interspersed with white fir, Douglas-fir, Gambel oak, New Mexico locust, and juniper. The north-facing slopes support a dense forest of true fir, Douglas-fir, and ponderosa pine. The south-facing slopes blend from dense ponderosa pine to open slopes with exposed rock, bare soils, and scattered pine and juniper trees. Lower portions of North Canyon have a limited shrub layer, including maple and Arizona cypress (*Cupressus arizonica*), that likely tap into the subsurface water flow. The only fish species in

North Canyon Creek is Apache trout. Apache trout are native to Arizona and listed as Threatened under the Endangered Species Act but were introduced into North Canyon Creek in 1963. Subsequent fish introductions into native Apache trout waters made the North Canyon stock valued for their genetic purity. Arizona Game and Fish Department conducts annual surveys for Apache trout. Maintaining an isolated, genetically pure population of Apache trout in North Canyon Creek allows added options in future management decisions.

Management Considerations

Management activities in the high elevation riparian areas are minor and have been further restricted since the signing of the Forest Plan. Active monitoring and management occurs within North Canyon Creek. About 3/4s of the Saddle Mountain Wilderness is part of the Central Summer Allotment (South Summer Pasture). The allotment is on a rest rotation system and is grazed every other year. Grazing is light in the Wilderness portion due to: steep terrain; dense forests on the north aspects; and the lack of forage on the dry south-facing slopes. Because livestock do occasionally find their way into the Wilderness Area, 2 drift fences were installed near North Canyon Creek in 2003 to deter livestock use. One is in the upper portion of the drainage and one is in the lower reaches. Field sign of livestock in North Canyon Creek is rare.

Fencing of livestock can be an effective strategy for managing wildlife habitat, potentially benefiting fish, birds, small mammals, and herpetofauna. Livestock grazing may cause habitat fragmentation that includes alteration of vegetation composition and structure in riparian zones, changes that may have significant impacts on small mammals and birds. For example, Giuliano and Homyack (2004) found that areas that had been excluded from livestock grazing experienced greater species richness and higher abundance of meadow voles (*Microtus pennsylvanicus*) and meadow jumping mice (*Zapus hudsonius*) than areas that were grazed. They attributed this difference to the increased litter cover and vertical vegetation that resulted from livestock exclusion. Plant species richness was 1.7 times greater and small mammal abundance was 2.2 times greater in ungrazed sites. Tewksbury et al. (2002) looked at the effects of grazing on riparian systems and found that avian abundance and richness was significantly lower in grazed sites. Red-naped sapsuckers were among the species that responded most negatively to human habitat alteration and livestock grazing in riparian habitat.

Fencing for livestock exclusion may improve riparian vegetation and stream morphology on short isolated stream reaches; however management at upstream watershed-scale levels may be necessary to significantly improve conditions for instream biota such as benthic macroinvertebrates. One long-term study conducted in an area of short isolated stream reaches found positive changes in stream depth, vegetation and water chemistry ($p = 0.031$) as a result of livestock exclusion. However, no significant response by macroinvertebrates was observed (Ranganath et al. 2009). Their data suggest that although fencing of small isolated stream reaches may improve the physical integrity of the stream itself, it may be an ineffective means for improving habitat for invertebrates.

Another study conducted in montane areas of western New Mexico, assessed the effects of seasonal use and grazing intensity on narrowleaf cottonwood (*Populus angustifolia*) saplings and other herbaceous vegetation. They found that cattle grazing at light and moderate intensities during all three grazing seasons (cool, warm, and dormant) did not have negative impacts on

vegetation. In fact, young cottonwoods in grazed areas experienced increased vigor, possibly a result of disturbance. In addition, herbaceous species richness was significantly greater in enclosures that had been grazed during the cool season (Lucas et al. 2004). The results of these studies highlight the differing response of riparian systems to grazing and the differing needs that result from the inherent complexities of riparian ecosystems. Grazing may be compatible with management of riparian systems, but the needs of differing systems need to be assessed on a case by case basis to implement effective management strategies for the entire ecosystem.

Riparian ecosystems are increasingly at risk of stochastic fire events resulting from ongoing drought, water diversion and climate change. In general, fire regimes in riparian zones are of low intensity and frequency. However, these areas can act as fire corridors and under certain circumstances, have the potential to carry more severe fires than upland areas (Pettit and Naiman 2007). The ecological consequences of severe fire in riparian systems are many and can include fluxes in nutrient cycling, increased sediment loads, and erosion (Petit and Naiman 2007). Some fire effects are immediate, such as short-term pulses of increased nutrient levels (Beche et al. 2005). Long-term effects resulting from changes in channel morphology and overstory canopy cover include decreases in wood mobility and elevated stream temperature (Zelt and Wohl 2004, Moore et al. 2005). The effects of low severity fire on nutrient availability may be positive, however severe fires can result in loss of nutrients through erosion, denitrification, and leaching. These effects can trickle down the watershed, as discharge from higher order streams is mobilized downslope during rainstorms. As surface runoff increases, nearby vegetation cover, vital to the maintenance of the systems integrity, decreases (Pettit and Naiman 2007). Living and dead plant material provide runoff control, recycle water throughout the system and into the atmosphere, and help to improve water quality by acting as nutrient filters (Tabacchi et al. 2000).

North Canyon Creek is currently surrounded by dense mixed-conifer forest, putting it at risk for catastrophic wildfire. A crown fire burning through the drainage would likely eliminate the Apache trout population due to extremely high fuel loading along and above North Canyon Creek, the small width of the creek, and its shallow depths.

In perennial waters such as North Canyon Creek isolated pool communities may become more frequent as a result of increasing temperatures and the resulting fragmentation that occurs as riffles dry out (Magoulick and Kobza 2003). Love et al. (2008) found that upstream community assemblages may be particularly vulnerable because they are typically associated with smaller and more isolated habitats than the larger pools that typically occur downstream. They found extinction rates of fishes increased with decline of pool size and that declines in pool size during summer drought influenced immigration and emigration of the species within aquatic communities. In another study on a high elevation riparian system in northern Arizona, Martin (2007) found that decreases in the surrounding deciduous vegetation were correlated with a long-term decline (20 years) in winter snowfall. This further led to complex trophic interactions including local extinction and severe population declines in previously common bird species.

The only perennial water on the SZ is located in the southeast quadrant of the Williams RD. Big Spring forms a small, shallow pool at the upper portion of Big Spring Canyon and the flow creates a linear feature that, including the entire riparian zone, is generally 2-3 feet wide. Actual open, flowing water is typically less than one foot in width and is often only inches across.

Riparian plants are primarily sedges with upland vegetation in immediate proximity. The overstory consists of ponderosa pine or is lacking entirely. One small patch of decadent willows (most of the plants' structure consists of dead wood) occurs along the riparian strip and some of the few cottonwoods growing on the SZ of the KNF are located several miles down the drainage where Forest Service road 109 crosses Big Spring Canyon. Flows are absent or very limited through most of the drainage and scouring rarely occurs, leaving little opportunity for natural cottonwood regeneration. Big Spring Canyon is within an active sheep allotment. Since the mid-1990s the permittee has agreed to avoid camping in Big Spring Canyon. A herder stays with the flock and only allows the sheep to graze lightly in a single pass through the area. Impacts from the sheep are monitored and subsequent site inspections after the sheep have passed through have found little to no effect on the springs, pool, and riparian vegetation.

Big Springs is a relatively unique perennial water on the NKRD that flows into two created ponds before going subsurface and serving as ground water recharge. Big Springs is within an Administrative Work Site that includes bunkhouses, mess hall and a field office. The development and associated activity likely reduces the wildlife value associated with the waters. However, the ponds receive use by birds, bats, and supports submergent vegetation. An introduced population of rainbow trout has been established in these ponds since the 1970s.

Wetland/Cienega PNV

Wetland habitat exists on both the NKRD and the SZ, although mostly as isolated areas (typically measured in 10s of acres), fragments of habitat (i.e., measured in feet rather than acres), or ephemeral habitat dependent upon annual and/or seasonal weather patterns. Wetlands are an anomaly on the KNF, reputed to be the driest forest in the National Forest system. Brown (1982 cited in Latta et al. 1999) attributes this phenomenon, in part, to the lack of recent glaciation and high evaporation rates. The scarcity of wetlands places even more emphasis on this already important habitat. AZPIF defines marshes as areas of permanent to semi-permanent fresh water characterized by relatively shallow depths and extensive coverage of submergent and emergent vegetation. Wetland habitat on the KNF tends to be a result of dams, diversions, and/or dependent on annual weather cycles. Ephemeral wetlands might only produce the defining vegetation once every several years. Most are limited in size and occur as isolated features on a xeric landscape. Few reports of Lucy's warblers, Lincoln sparrows, or yellow-breasted chats exist for the KNF. Sightings of cinnamon teal are not uncommon, but the KNF appears to support individuals or pairs of birds, but not actual populations of teal. Ephemeral marsh habitat is a key habitat in maintaining resident, riparian-dependent avifauna, but there is not sufficient habitat to support viable *populations*. Using species of riparian-dependent avifauna to monitor riparian health is ineffective under these conditions. Cinnamon teal habitat is present but limited both spatially and temporally.

There are 88 wetlands on the forest (Steinke 2007) that primarily occur on the Williams RD. The Tusayan RD does not have any wetlands.

Seventeen natural wetlands occurring on the NKRD Central Summer Allotment were described in the Kane Ranch Environmental Assessment (USDA 2001). A primary objective for the NKRD is to fence these areas for the improvement of riparian conditions. Nine of the natural sink holes have never been mechanically manipulated or used by livestock (D. Burger, *pers. comm.*).

Franks Lake Geologic-Botanical Area and Three Lakes provide key wetland habitat on the NKRD. Franks Lake is perennial whereas Three Lakes is an ephemeral feature. Franks Lake is a series of five limestone sinks with a mosaic of wet and dry meadows, wetland, and floating sedge bog. The area was designated a Geologic-Botanical Area in 1988 and was fenced to exclude livestock in 1990. A vegetation survey of the area was completed in cooperation with The Nature Conservancy in 1991. The impacts of grazing were still evident at the time of the survey, including vegetation trampling, limited plant diversity in some zones, and the floating bog was limited to those regions of the pond too deep for cattle grazing. The area was resurveyed in September 1992. Vegetative cover increased from 35-90% in the meadow sample plots and the species composition consisted nearly entirely of native plants. The shoreline had filled in with native vegetation, the bog had expanded in size, and the emergent vegetation had recovered in areas previously trampled by cattle. Before the special area designation, the bog consisted of a small island of vegetation in the middle of the lake, occupying the area beyond the reach from cattle. Today most of the lake supports bog vegetation with only limited channels of open water still exposed.

Three Lakes has an artificial island created in the middle of the pond to provide nesting habitat. Emergent vegetation occupies the center portion of the lake. Although other examples of this habitat exist on the NKRD, they tend to be limited in size, e.g., Crane Lake (1.5 acres), Deer Lake (1.3 acres), and Indian Lake (0.3 acres).

The Williams RD has a number of wetland areas, including small patches or limited linear strips of wetland vegetation such as Big Spring and Keyhole Sink, to broad wetland and wet meadow complexes such as Coleman Lake. Many of these areas are microsites that are only fractions of an acre in size and support small patches of wetland vegetation, such as Dow Springs, Bear Springs, and Pomeroy Tanks. These perennial springs are more prevalent along the Mogollon Rim where melted snowpack comes to the surface. Coleman Lake, JD Dam, and Scholz Lake are the most extensive perennial wetlands on the Forest (30-80 acres of water). The amount of hydrophytic vegetation and water are highly variable, depending on seasonal weather patterns. Winter snows are the primary contributor and monsoon rains secondarily provide moisture for these features.

Ephemeral wetlands support a number of migrant and resident species that could not otherwise exist across most of the KNF. Ephemeral wetlands such as Davenport, Duck, Depot, and Dry Lakes occur when adequate amounts of winter snows accumulate. These precipitation patterns are unpredictable and may occur once every three to ten years. Even rarer is having consistent marsh habitat in successive years. The degree of flooding is variable and can result in broad shallows or accumulate to several feet deep. During dry conditions, the characteristic marshland vegetation is absent and the clay soils tend to dry and crack open. When flooded, these sites attract migratory waterfowl. However, use by resident waterfowl is limited even in wet years (Table 32). Many of the major wetlands have been fenced to exclude livestock and most exclude off-road vehicles.

Table 31. Average number of waterfowl spotted at Davenport Lake during wetter than normal years (1987, 1988, and 1995) from records at the Williams Ranger Station District Office.

Month	Totals
February-May	~3400
June	10
July	1
August-October	0
November	8
December-January	0

Management Considerations

The primary threat identified for the Wetland/Cienega PNVN during the forest plan revision process was a lack of characteristic fire disturbance and associated nutrient cycling. Decreased water input and flow resulting from encroaching conifers was identified as an additional stressor, with approximately 7% of the PNVN lost to conifer encroachment. Invasive plants were also identified, but the rate of establishment is low, about 1% or 15 acres per year (Kaibab National Forest 2008a). Nevertheless, Lecain et al. (2006) found that spotted knapweed and Canada thistle can grow exponentially once established.

Controlled fire may also offer restoration potential. Some research has shown that early-season burning in high-elevation areas can be an effective means to stimulate germination of plants such as Bebb willow (*Salix bebbiana*) by increasing soil water content. Working in Hart Prairie on the Coconino National Forest, Mullen et al. (2006) found that burning the surrounding herbaceous plant community resulted in a decrease in above ground biomass and subsequent decrease in evapotranspiration rates. This resulted in short-term increases of soil water content. They found the optimal timing of such treatments was in the spring when soil water content was highest due to snow melt and when germination conditions were optimal. However, in riparian areas where invasive weeds are present, Lecain et al. (2006) caution against those practices that increase nitrogen availability in the soil.

In general, riparian habitat protection is largely dependent on fencing because grazing can directly affect nesting cover for birds. Although problematic, the overwhelming majority of the fence lines are intact at any given time and fence maintenance is conducted annually. The KNF has begun tracking fence conditions and maintenance visits on an electronic database and this is expected to improve overall conditions. Existing wetland enclosure fences with heavy yearly maintenance are currently being improved with a top wire ¼" steel cable to reduce maintenance costs over time.

In general, the primary proximate factor affecting wetland size and condition on the KNF is lack of precipitation (Steinke 2007). The ephemeral lakes and wetlands of the KNF cannot support viable populations of riparian obligates if they cannot provide consistent habitat. The temporary presence of an otherwise absent habitat provides benefits to individual birds, but this is very different from habitat that can consistently support viable populations of a given species. Nesting

habitat near open water, such as that required by cinnamon teal, is not present in sufficient volume either spatially or temporally to support viable populations. In these instances, much of the teal habitat is so limited and isolated that it may serve as sink habitat due to the limitations in spatial extent and vulnerability to nest depredation. Alternately, successive years of abundant precipitation may allow a “pulse” of reproductive success that benefits the overall metapopulation on the Coconino Plateau (R. Miller *pers. comm.*).

Open Water Habitat

The most common example of open water habitat on the KNF is the ubiquitous stock tank created for livestock and wildlife starting in the 1930s. There are 492 reservoir and stock tank claims on the Forest. Recent surveys suggest that these manmade features have increased the surrounding riparian vegetation (Steinke 2007).

Most reservoirs, such as Cataract, Dogtown, JD, Kaibab, and Whitehorse support developed recreation sites. The impacts of human disturbance and the associated changes in vegetation lower the wildlife values of these reservoirs, although the reservoirs can still provide key habitat for waterfowl, wading birds, and foraging raptors.

When flooded, wetlands such as Duck, Dry, Depot, and Davenport can provide excellent habitat, as do the natural wetlands on the North Kaibab. As a group, they provide habitat for resident, wintering, and migrating passerines, waterfowl, raptors (including osprey, bald eagles, peregrine falcons, goshawks, and harriers) and multiple great blue heron rookies. In addition, natural lakes occurring on the KNF also contribute to open water habitat.

Stock tanks tend to be created by constructing dams to collect overland flow. The value of individual tanks to wildlife varies considerably, with some tanks generally holding water, some tanks generally dry, and many tanks going through seasonal variations in water levels based on precipitation patterns. Tanks with consistent water occur across the KNF, although the Tusayan RD is the most limited in terms of available water. The berms creating the stock tanks are typically compacted earth with little or no vegetation due to impacts from livestock and wildlife. When vegetation does grow on the berms, it typically consists of terrestrial species. Most stock tanks have little or no riparian habitat and fluctuations in water availability at many of the tanks further complicates assessing potential contributions to local fauna. Invasive and noxious plant species are common near tanks due to the impacts of concentrating people, cattle, elk and, to a lesser degree, deer. The drier the habitat, the less ground cover there tends to be in the vicinity of the tanks.

Habitat for ground nesting species immediately surrounding the tanks is generally poor, but the availability of the water itself is important to a wide range of wildlife. In Arizona, some research has shown that developed waters may be of primary importance to nongame species. O’Brien et al. (2006) found that the use of such waters by bats, raptors, mammalian predators and rodents exceeded use by game species such as mule deer, doves and Gambel’s quail. Lynn et al. (2006) assessed the value of developed waters on habitat selection by migratory birds. Their data suggest that developed waters are of little importance to migratory birds, however resident birds use developed waters frequently. Migratory bird use accounted for less than 0.1% of their total

observations. In contrast, they recorded > 69,000 observations of local resident birds using artificial water sources, mostly doves and house finches.

The KNF supports a diverse assemblage of bats and even ephemeral stock tanks can provide critical water needs for bats in arid environments (Mollhagen and Bogan 1997). A bat project on the NKR D netted isolated waters in August for two successive years and recorded over 20 species of bats (there are 28 total species of bats occurring in Arizona), capturing a total of 2,282 individual bats (Painter 2003). Five nights of netting on the Tusayan RD resulted in 900 bat captures (B. Solvesky, pers. comm.).

Management Considerations

The value of developed waters for wildlife in the southwest is often controversial. Some would argue their importance for both game and nongame species. Merriam's turkey populations for example, have been known to increase with the construction of such waters (Shaw and Mollohan 1992 *in* Rosenstock et al. 1999). Opponents however, speculate that such developments increase predation, competition, and disease transmission among wildlife (see reviews *in* Rosenstock et al. 1999, Krausman et al. 2006). For example, placement of fencing around smaller tanks may expose bats to a high rate of injury and negatively affect drinking behavior (Tuttle et al. 2006).

Mosquitoes carrying West Nile virus were identified for the first time in Arizona in 2003 and considered a threat to avifauna. However, the data collected by the Arizona Department of Health Services show that since 2003, there has been a decreasing trend with regard to West Nile virus in mosquitoes and birds. The prevalence of avian malaria in Coconino County is among the lowest for the state (http://www.azdhs.gov/phs/oids/westnile/wnv_update08.htm) (Table 32).

Table 32. Positive samples with avian malaria in Coconino County (Arizona Department of Health Services data).

Year	Mosquitoes	Birds
2004	250	98
2005	350	19
2006	0	1
2007	2	0

Until additional data on the use of developed waters as a wildlife management tool can be produced, the debate over the value of these artificial structures for wildlife will likely continue.

Habitat Data

Fencing projects have been ongoing since the 1920s and typically focus on range improvement although sporadic fencing for the benefit of wildlife has occurred (Figure 55). According to the KNF INFRA database, approximately 18 acres out of about 221 total acres have been fenced specifically to enhance wildlife habitat. Fencing projects typically range from ¼ acres-2 acres in size with a few exceptions for wildlife; Moritz Lake Wetland exclosure (4 acres), Pine Creek and Sitgreaves (2 acres each), and Scholz Lake (340 acres), JD Dam (110 acres), Davenport Lake (110 acres), Moritz Lake (100 acres), Perkins Lake (30 acres), Coleman Lake (160 acres), Duck Lake (150 acres), Dry Lake (975 acres), Mineral Lake (40 acres) . Range exclosures exceeding 1

acre include the Bellemont (3 acres) and Hat (10 acres) allotments. On the NKR D, the largest fence projects are for wildlife habitat protection, including Franks Lake (237 acres) and Three Lakes (30 acres). The NKR D continues to fence 1-2 lakes per year as identified in the Kane Ranch EA. Approximately half of the 17 natural lakes identified in the EA have been fenced to date (D. Burger *pers. comm.*)

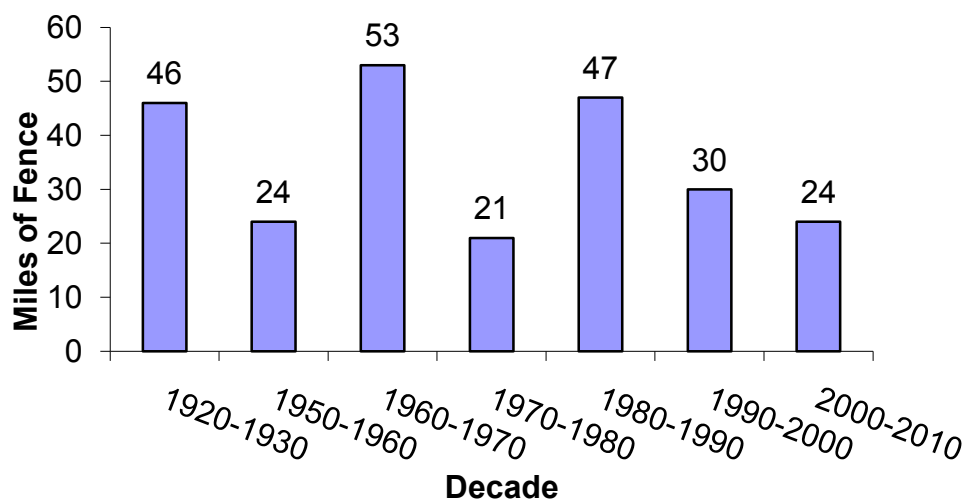


Figure 55. Total miles of fence by decade across the KNF (INFRA data).

The KNF began a cyclic repair and replacement of existing wetland exclosures in fall 2009. A total of 9 wetland areas have been prioritized for this work and considerable funds (\$52,000) have been allocated toward accomplishing this task. This project will repair, improve, and replace existing fence lines surrounding 4 separate wetland areas that have deteriorated over the years. Davenport Lake (110 acres), Moritz Lake (100 acres), JD Dam (110 acres) and Coleman Lake (160 acres) will be fenced first because of their high value for wildlife. New and improved fencing will include top steel cables and wildlife jumps to increase fence longevity and decrease maintenance costs over the long-term. It is expected that these fences will incur less damage from elk and be more resistant to damaging environmental agents such as tree fall. Because water is such an uncommon resource across the KNF, maintaining the integrity of riparian habitat is even more critical for winter and summer resident birds that depend on riparian areas for nesting and foraging. Fencing these areas will also benefit bird species at a broader landscape scale, because these riparian oases provide vital stopover habitat for migrating avifauna across the KNF. These improved exclosures should also benefit numerous other terrestrial and aquatic vertebrates, invertebrates, and plants. Because these areas are still in proper functioning condition, the positive effects from this project should be realized within a short amount of time.

The amended forest plan (June 2008) requires an inventory of riparian areas and that current management should provide for satisfactory or better conditions. A review of KNF aquatic conditions was conducted in support of the forest plan revision process (Steinke 2007). Several different sources were used in this review including the National Hydrologic Dataset, 1990 on-site survey data for riparian and wetland areas across the Forest, and water quality data obtained through the Arizona Department of Environmental Quality. For 88 wetland and open water

features on the KNF, 31 were found in poor condition, 45 were in fair condition, and 12 were in good condition. Grazing from livestock and elk, and impacts from recreation were noted to cause poor conditions (Steinke 2007). In general, those areas experiencing downward trends in habitat quality are typically areas where fences have not been constructed or adequately maintained.

Recent repeat surveys of these areas, as well as additional areas, were conducted in 2008 and the results were compared to survey data collected in 1990. A total of 95 riparian areas were assessed which included perennial and ephemeral springs, stock tanks, reservoirs, intermittent streams and seasonal wetlands. Out of the riparian features surveyed, 14 exhibited a downward trend, 63 were static, and 17 were improving in habitat quality since 1990 (Figure 56). Twenty-two of the total riparian areas surveyed were considered functional but at risk, 18 were considered non-functional and 55 were considered to be in proper functioning condition (Table 33).

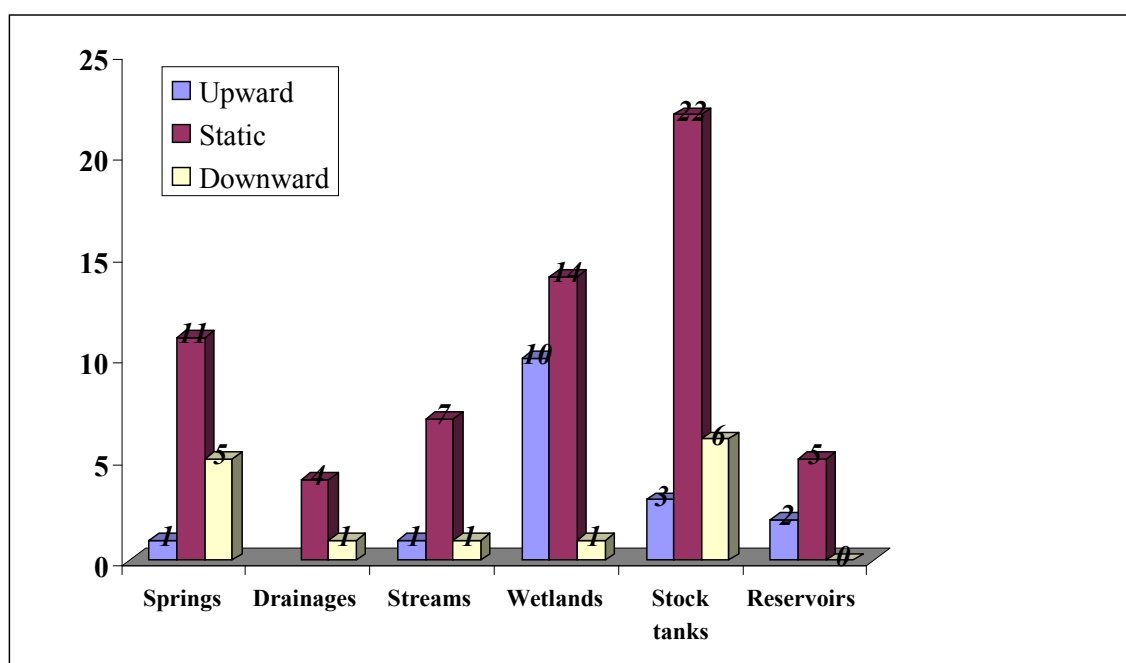


Figure 56. Riparian Trends from 1990-2008 across the KNF (KNF, unpublished data).

Table 33. Current (2008) riparian conditions across the KNF.

Riparian Category	PFC ¹	Functional at Risk	Non-functional	Total
Springs	8	3	6	17
Streams	7	2	0	9
Wetlands	22	1	2	25
Stock tanks	9	13	10	32
Reservoirs	6	1	0	7
Drainages	3	2	0	5
<i>Total</i>	55	22	18	95

¹Proper Functioning Condition

Those habitat types experiencing downward trends included springs (5), stock tanks (6) and 1 ephemeral stream, drainage and wetland. Tree encroachment, grazing by livestock and elk, and drought were commonly cited factors leading to the decline in habitat quality. One stock tank with little or no riparian vegetation (Joe's Mud Hole) was negatively impacted by the Warm Fire of 2006. Glen Lake, a seasonal wetland was noted as non-functional, with ash flow from the Warm fire eliminating riparian vegetation. The negative impacts of elk use can be seen at Dutch Kid, a functional but at risk tank that was developed for livestock and wildlife. The positive benefits associated with fence improvement projects can be seen at Mud and Whitehorse Lakes (Appendix 10).

Seventeen features were associated with upward trends, with wetlands accounting for a little over half of that total. Streams (1), springs (1), stock tanks (3) and reservoirs (2) were also noted as improved. Fencing for ungulates was noted as a factor contributing to these positive trends. Conditions of the remaining features were similar to that of 1990 and noted as static.

Although the current MIS are not applicable to the Kanab Creek drainage, habitat changes are still evident. A week of small mammal trapping in 2002 yielded capture rates of 56 animals (six different rodent species) per 100 trap nights, a rate that is an order of magnitude above average small mammal trap success rates. Additional wildlife sightings and sign included gray fox, ringtail cat, and bighorn sheep. The only beaver known to reside on the KNF has been detected in Kanab Creek each year from 2001 through 2003. Although there are no previous records to compare to, the abundance of small mammals suggests the current vegetation is providing food and cover.

This assumption is supported by vegetation changes detected by reading the Kanab Creek Parker Transect Cluster (Figure 57). One way to evaluate range conditions is to categorize plants by their response to grazing (Table 34). Native perennial vegetation normally present in climax communities is classified as increasers (plants that increase under heavy grazing pressure), decreaseers (plants that decrease under heavy use), and invaders (species not present in native vegetation assemblages but appear in response to grazing) (Stoddart et al. 1975). Generally, increasers are less palatable than decreaseers.

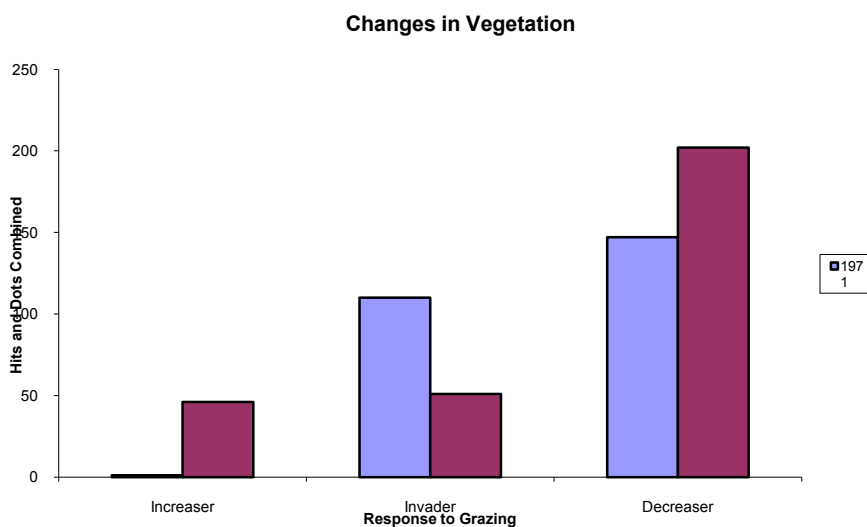


Figure 57. Changes detected in perennial species in Kanab Creek using both hits and dots along a permanent Parker Transect Cluster. Increasers and decreasers refer to the plants response to grazing; invaders are non-native species.

Table 34. Perennial plants encountered along the Parker Transect Cluster in Kanab Creek, KNF. Dots and Hits represent techniques used to record plants along established transects.

SPECIES	COMMON NAME	GRAZING RATING	Survey Year 1971	Survey Year 1999
<i>Atriplex canescens</i>	4-wing saltbrush	Decreaser	0	1
<i>Bouteloua eriopoda</i>	Black grama	Decreaser	85	104
<i>Ephedra spp</i>	Mormon tea	Invader	1	32
<i>Gutierrezia spp</i>	Snakeweed	Invader	3	19
<i>Hilaria jamesii</i>	Galleta	Increaser	1	39
<i>Kochia americana</i>	Greenmolly	Increaser	0	7
<i>Poa fendleriana</i>	Muttongrass	Decreaser	1	0
<i>Spartina spp</i>	Cordgrass	Invader	59	0
<i>Sporobolus cryptandrus</i>	Sand dropseed	Decreaser	61	97
<i>Tridens pulchella</i>	Fluffgrass	Invader	24	0
<i>Zygadenus spp</i>	Deathcamas	Invader	23	0

Although no Parker surveys have been conducted in Kanab Creek since 1999, KNF range specialists have noticed invasive cheatgrass and tamarisk becoming more prevalent throughout the area (D. Burger, pers. comm.). The NKRD, in collaboration with the Grand Canyon Trust, began surveys to monitor invasive tamarisk in 2007. This effort should provide future trend data to evaluate the status of this noxious species in riparian areas.

Habitat Trend

As a whole, riparian habitat conditions across the KNF are stable to improving and can be summarized as follows:

Low Elevation Riparian (Kanab Creek): Management changes since the signing of the Forest Plan have resulted in an improving trend, although this system is still at risk in the long run, see discussion below.

High Elevation Riparian (North Canyon Creek and Big Spring): Habitat trends have been stable to improving for both high elevation riparian areas due to changes in the grazing programs.

Wetlands: Habitat conditions are stable to improving as a result of fencing projects since the signing of the Forest Plan.

Open water: It is estimated that open water habitat is stable across the KNF. This takes into account ongoing efforts to maintain existing fencing and that new areas are periodically fenced to protect additional portions of habitat.

Projected

Despite the positive effects resulting from current management strategies of riparian habitat on the KNF, these systems are at future long-term risk. State and transition models developed during forest plan revision suggest that the Cottonwood-Willow Riparian Forest PNVF (i.e. Kanab Creek area) is highly departed from reference conditions. Lack of year round water flow, flooding disturbance, and invasion of tamarisk and, to a lesser extent, Russian olive continue to put this system at high risk. Wetlands and cienegas (i.e. marshlands and open water) are minimally departed from reference conditions and are slowly trending away from historic conditions. Tree encroachment, drought, and lack of characteristic fire disturbance and limited nutrient cycling are all contributing factors (Kaibab National Forest 2008a).

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Appendix 1. Common northern goshawk prey in the Southwest (Reynolds et al. 1992)

Common Name	Scientific Name	Common Name	Scientific Name
American robin	<i>Turdus migratorius</i>	Stellar's jay	<i>Cyanocitta stelleri</i>
Blue grouse	<i>Dendragapus obscurus</i>	Williamson's sapsucker	<i>Sphyrapicus thyroideus</i>
Band-tailed pigeon	<i>Columba fasciata</i>	Chipmunks	<i>Tamias spp</i>
Hairy woodpecker	<i>Picoides villosus</i>	Cottontail rabbits	<i>Sylvilagus spp</i>
Mourning dove	<i>Zenaida macroura</i>	Mantled ground squirrel	<i>Citellus lateralis</i>
Northern flicker	<i>Colaptes auratus</i>	Red squirrel	<i>Tamiasciurus hudsonicus</i>
Red-naped sapsucker	<i>Sphyrapicus nuchalis</i>	Tassel-eared squirrel	<i>Sciurus aberti</i>

Appendix 2. Population trend analyses conducted on Kaibab National Forest's Management Indicator Species using the Breeding Bird Survey (BBS) Analysis Website (Sauer et al. 2005). Summaries are by State and Bird Conservation Region. Trend values, the probability of this value being due to chance alone (p-value), and the number of BBS routes where the species was detected are given for each of the three time periods. Ninety-five percent confidence intervals and relative abundance are given for the 1966-2005 time period.

Species	Analysis Engine	BBS Region or BCR	Group Qualifier	Data Quality	Time Period											
					1966 to 2005						1966 to 1978			1980 to 2005		
					Trend	p	n	- C. I.	+ C. I.	R.A.	Trend	p	n	Trend	p	n
Cinnamon Teal	Region	Arizona	-	Red	-11.500	0.350	4	-30.0	7.0	0.24	-12.6	0.84	2	-8.100	0.150	2
		Southern Rockies	-	Red	0.600	0.920	12	10.4	11.5	0.19				1.200	0.900	11
	BCR	Sierra Madre Occidental	-	No Data												
		Southern Rockies	-	No Data												
	BCR & Group	Sierra Madre Occidental	Wetland breeder	No Data												
		Southern Rockies	Wetland breeder	No Data												
		Arizona	Wetland breeder	No Data												

Data Quality	Blue	This category reflects data with at least 14 samples in the long term, of moderate precision, and of moderate abundance on routes.
	Yellow	This category reflects data with a deficiency, including: <ol style="list-style-type: none"> 1. The regional abundance is less than 1.0 birds/route (low abundance), 2. The sample is based on less than 14 routes for the long term (small sample size), 3. The results are so imprecise that a 3%/year change would not be detected over the long-term (quite imprecise), or 4. The sub-interval trends are significantly different from each other (P less than 0.05, based on a z-test). This suggests inconsistency in trend over time).
	Red	This category reflects data with an important deficiency. In particular: <ol style="list-style-type: none"> 1. The regional abundance is less than 0.1 birds/route (very low abundance), 2. The sample is based on less than 5 routes for the long term, or is based on less than 3 routes for either subinterval (very small samples), or 3. The results are so imprecise that a 5%/year change would not be detected over the long-term (very imprecise).

Species	Analysis Engine	BBS Region or BCR	Group Qualifier	Data Quality	Time Period											
					1996 to 2005						1966 to 1978			1980 to 2005		
					<u>Trend</u>	<u>p</u>	<u>N</u>	<u>- C. I.</u>	<u>+ C. I.</u>	<u>R.A.</u>	<u>Trend</u>	<u>p</u>	<u>n</u>	<u>Trend</u>	<u>p</u>	<u>n</u>
Hairy Woodpecker	Region	Arizona	-	Yellow	2.700	0.440	20	-4.0	9.4	0.81	16.1	0.19	3	3.800	0.250	18
		Southern Rockies	-	Yellow	2.200	0.450	49	-3.5	7.9	0.66	2.1	0.83	4	0.100	0.980	48
	BCR	Sierra Madre Occidental	-	Blue	1.300	0.510	18	-2.6	5.2	0.12	2.3	0.47	19	1.500	0.630	19
		Southern Rockies	-	Blue	0.100	0.950	67	-3.6	3.8	0.27	0.6	0.75	93	0.400	0.820	88
	BCR & Group	Sierra Madre Occidental	Cavity nester	≥14 routes	2.872	0.474	19							-0.195	0.272	17
			Woodland breeder	≥14 routes	-2.441	0.474	19							-1.578	0.272	17
			Permanent resident	≥14 routes	-0.497	0.474	19							0.023	0.272	17
		Southern Rockies	Cavity nester	≥14 routes	0.698	0.454	49							0.675	0.980	48
			Woodland breeder	≥14 routes	1.111	0.454	49							0.435	0.980	48
			Permanent resident	≥14 routes	1.542	0.454	49							0.629	0.980	48
		Arizona	Cavity nester	≥14 routes	0.017	0.443	20							0.902	0.251	18
			Woodland breeder	≥14 routes	0.481	0.443	20							0.433	0.251	18
			Permanent resident	≥14 routes	0.128	0.459	20							0.258	0.265	18

Species	Analysis Engine	BBS Region or BCR	Group Qualifier	Data Quality	Time Period											
					1996 to 2005						1966 to 1978			1980 to 2005		
					Trend	p	N	- C. I.	+ C. I.	R.A.	Trend	p	n	Trend	p	n
Juniper Titmouse	Region	Arizona	-	Yellow	-4.100	0.160	22	-9.6	1.4	0.85	-6.4	0.20	6	-4.600	0.140	18
		Southern Rockies	-	Red	-1.200	0.870	8	-15.7	13.3	0.22				-1.100	0.860	8
	BCR	Sierra Madre Occidental	-	No Data												
		Southern Rockies	-	No Data												
	BCR & Group	Sierra Madre Occidental	Successional or scrub breeder	≥14 routes	-2.373	0.036	18							-3.084	0.022	15
			Permanent resident	≥14 routes	-0.150	0.036	18							-2.399	0.022	15
			Cavity nester	≥14 routes	8.689	0.036	18							-2.542	0.022	15
			Mid-story canopy nesting	≥14 routes	-1.615	0.036	18							-2.871	0.022	15
		Southern Rockies	Successional or scrub breeder	≥14 routes	-2.679	0.000	61							-2.690	0.003	58
			Permanent resident	No Data												
			Cavity nester	No Data												
			Mid-story canopy nesting	No Data												
		Arizona	Successional or scrub breeder	≥14 routes	-1.419	0.162	22							-2.123	0.140	18
			Permanent resident	≥14 routes	-1.391	0.097	22							-2.066	0.125	18
			Cavity nester	≥14 routes	-0.131	0.162	22							-1.720	0.140	18
			Mid-story canopy nesting	≥14 routes	-1.740	0.097	22							-2.010	0.125	18

Species	Analysis Engine	BBS Region or BCR	Group Qualifier	Data Quality	Time Period												
					1996 to 2005					1966 to 1978			1980 to 2005				
					Trend	p	N	- C. I.	+ C. I.	R.A.	Trend	p	n	Trend	p	n	
Lincoln's Sparrow	Region	Arizona	-	No Data													
		Southern Rockies	-	Red	1.600	0.140	57	-0.5	3.6	5.89				1.700	0.130	57	
	BCR	Sierra Madre Occidental	-	No Data													
		Southern Rockies	-	No Data													
	BCR & Group	Sierra Madre Occidental	Successional/scrub breeder	No Data													
			Open cup nester	No Data													
			Neotropical migrant	No Data													
			Ground or low nester	No Data													
		Southern Rockies	Successional/scrub breeder	≥14 routes	1.349	0.136	57							1.349	0.127	57	
			Open cup nester	≥14 routes	1.288	0.136	57							1.577	0.127	57	
			Neotropical migrant	≥14 routes	1.385	0.127	57							1.385	0.127	57	
			Ground or low nester	≥14 routes	1.192	0.136	57							1.416	0.127	57	
		Arizona	Successional/scrub breeder	No Data													
			Open cup nester	No Data													
			Neotropical migrant	No Data													
			Ground or low nester	No Data													

Species	Analysis Engine	BBS Region or BCR	Group Qualifier	Data Quality	Time Period											
					1996 to 2005						1966 to 1978			1980 to 2005		
					<u>Trend</u>	<u>p</u>	<u>N</u>	<u>- C. I.</u>	<u>+ C. I.</u>	<u>R.A.</u>	<u>Trend</u>	<u>p</u>	<u>n</u>	<u>Trend</u>	<u>p</u>	<u>n</u>
Lucy's Warbler	Region	Arizona	-	Blue	-0.300	0.710	36	-2.0	1.4	11.26	-0.5	0.94	9	-0.300	0.790	33
		Southern Rockies	-	No Data												
	BCR	Sierra Madre Occidental	-	No Data												
		Southern Rockies	-	No Data												
	BCR & Group	Sierra Madre Occidental	Successional/scrub breeder	≥14 routes	-0.317	0.827	17							-0.414	0.959	16
			Cavity nester	≥14 routes	-2.497	0.827	17							-0.550	0.959	16
			Short distance migrant	≥14 routes	-0.293	0.827	17							-0.468	0.959	16
			Mid-story canopy nesting	≥14 routes	-0.342	0.827	17							-0.468	0.959	16
		Southern Rockies	Successional/scrub breeder	No data												
			Cavity nester	No data												
			Short distance migrant	No data												
			Mid-story canopy nesting	No data												
		Arizona	Successional/scrub breeder	≥14 routes	-0.405	0.714	36							-0.706	0.794	33
			Cavity nester	≥14 routes	-0.091	0.714	36							-0.328	0.794	33
			Short distance migrant	≥14 routes	-0.493	0.714	36							-0.672	0.843	33
			Mid-story canopy nesting	≥14 routes	-0.194	0.824	36							-0.445	0.843	33

Species	Analysis Engine	BBS Region or BCR	Group Qualifier	Data Quality	Time Period											
					1996 to 2005						1966 to 1978			1980 to 2005		
					Trend	p	N	- C. I.	+ C. I.	R.A.	Trend	p	n	Trend	p	n
Northern Goshawk	Region	Arizona	-	Red	13.000	0.030	5	6.8	19.3	0.06				11.800	0.040	5
		Southern Rockies	-	Red	-3.000	0.690	6	-16.6	10.5	0.20				-2.900	0.690	6
	BCR	Sierra Madre Occidental	-	No Data												
		Southern Rockies	-	No Data												
Spotted Owl	Region	Arizona	-	No Data												
		Southern Rockies	-	No Data												
	BCR	Sierra Madre Occidental	-	No Data												
		Southern Rockies	-	No Data												

Species	Analysis Engine	BBS Region or BCR	Group Qualifier	Data Quality	Time Period											
					1996 to 2005						1966 to 1978			1980 to 2005		
					<u>Trend</u>	<u>p</u>	<u>N</u>	<u>- C. I.</u>	<u>+ C. I.</u>	<u>R.A.</u>	<u>Trend</u>	<u>p</u>	<u>n</u>	<u>Trend</u>	<u>p</u>	<u>n</u>
Pygmy Nuthatch	Region	Arizona	-	Blue	-1.500	0.530	15	-5.9	3.0	5.34	12.2	0.51	3	-2.200	0.500	13
		Southern Rockies	-	Yellow	-3.100	0.080	21	-6.3	0.2	0.72	-8.0	0.07	3	-3.300	0.040	21
	BCR	Sierra Madre Occidental	-	No Data												
		Southern Rockies	-	No Data												
	BCR & Group	Sierra Madre Occidental	Woodland breeder	≥14 routes	-2.249	0.564	16							-1.496	0.538	14
			Cavity nester	≥14 routes	4.085	0.564	16							-1.251	0.538	14
			Permanent resident	≥14 routes	-0.394	0.564	16							-1.030	0.538	14
			Mid-story canopy nesting	≥14 routes	-0.795	0.564	16							-1.335	0.538	14
		Southern Rockies	Woodland breeder	≥14 routes	-1.869	0.078	21							-2.765	0.035	21
			Cavity nester	≥14 routes	0.599	0.078	21							-2.910	0.035	21
			Permanent resident	≥14 routes	-2.051	0.078	21							-2.648	0.035	21
			Mid-story canopy nesting	≥14 routes	-2.082	0.078	21							-2.850	0.035	21
		Arizona	Woodland breeder	≥14 routes, for 1966-2005 only	-0.467	0.531	15									
			Cavity nester	≥14 routes, for 1966-2005 only	-0.832	0.531	15									
			Permanent resident	≥14 routes, for 1966-2005 only	-0.621	0.567	15									
			Mid-story canopy nesting	≥14 routes, for 1966-2005 only	-0.651	0.567	15									

Species	Analysis Engine	BBS Region or BCR	Group Qualifier	Data Quality	Time Period											
					1996 to 2005						1966 to 1978			1980 to 2005		
					<u>Trend</u>	<u>p</u>	<u>N</u>	<u>- C. I.</u>	<u>+ C. I.</u>	<u>R.A.</u>	<u>Trend</u>	<u>p</u>	<u>n</u>	<u>Trend</u>	<u>p</u>	<u>n</u>
Red-naped Sapsucker	Region	Arizona	-	Red	-14.700	0.070	3	-22.6	-6.8	0.15				-14.800	0.070	3
		Southern Rockies	-	Blue	7.400	0.000	61	3.1	11.6	1.30	16.2	0.21	7	10.400	0.000	60
	BCR	Sierra Madre Occidental	-	No Data												
	BCR & Group	Sierra Madre Occidental	Cavity nester	No Data												
			Woodland breeder	No Data												
			Short distance migrant	No Data												
		Southern Rockies	Cavity nester	≥14 routes	0.778	0.001	61							7.262	0.005	60
			Woodland breeder	≥14 routes	3.840	0.001	61							5.734	0.005	60
			Short distance migrant	≥14 routes	4.959	0.001	61							6.770	0.005	60
		Arizona	Cavity nester	No Data												
			Woodland breeder	No Data												
			Short distance migrant	No Data												
Sapsuckers Sp.	BCR	Southern Rockies	-	Red	24.400	0.010	6	15.1	33.8	0.05	4.8	0.00	84	4.100	0.010	81

Species	Analysis Engine	BBS Region or BCR	Group Qualifier	Data Quality	Time Period											
					1996 to 2005						1966 to 1978			1980 to 2005		
					Trend	p	N	-C.I.	+C.I.	R.A.	Trend	p	n	Trend	p	n
Yellow Breasted Chat	Region	Arizona	-	Red	0.900	0.550	16	-2.0	3.8	2.74	16.3	0.56	2	2.400	0.030	16
		Southern Rockies	-	Red	0.600	0.950	9	-20.0	21.2	0.08				21.400	0.060	8
	BCR	Sierra Madre Occidental	-	Blue	-15.800	0.470	7	-55.8	24.2	0.05	1.3	0.48	14	1.400	0.480	14
		Southern Rockies	-	Blue	-3.000	0.530	24	-12.1	6.2	0.06	1.8	0.39	42	2.200	0.420	40
	BCR & Group	Sierra Madre Occidental	Neotropical migrant	≥14 routes	-0.995	0.479	14							1.719	0.018	14
			Successional/scrub breeder	≥14 routes	0.118	0.479	14							2.082	0.018	14
			Open cup nester	≥14 routes	0.061	0.479	14							1.987	0.018	14
			Ground or low nester	≥14 routes	-0.255	0.479	14							1.776	0.018	14
		Southern Rockies	Neotropical migrant	No data												
			Successional/scrub breeder	No data												
			Open cup nester	No data												
			Ground or low nester	No data												
		Arizona	Neotropical migrant	≥14 routes	-0.115	0.517	16							1.771	0.030	16
			Successional/scrub breeder	≥14 routes	0.122	0.552	16							1.419	0.031	16
			Open cup nester	≥14 routes	0.104	0.552	16							1.612	0.031	16
			Ground or low nester	≥14 routes	58.732	0.517	16							-2.739	0.030	16

Appendix 3. Species detections per habitat type and forest-wide for all avian species heard or seen during the 2005 and 2006 Kaibab National Forest Landbird Surveys, Kaibab National Forest, Coconino County, AZ.

National Forest, Conejito County, AZ.							
Year	Species	Habitat					Forest Wide
		Aspen	Mixed Conifer	Montane Grassland	Ponderosa Pine	Woodland / Grassland	
2005	Acorn woodpecker				4		4
	American robin	4	8	1	49		62
	Ash-throated flycatcher				16	20	36
	Audubon's warbler	2	56	4	75		137
	Bewick's wren					7	7
	Blue gray gnatcatcher					1	1
	Brown headed cowbird				10	1	11
	Black headed grosbeak		10		21	3	34
	Brewer's blackbird			3			3
	Brown creeper		8		6		14
	Broad-tailed hummingbird				5		5
	Ban-tailed pigeon		3				3
	Black-throated gray warbler				6	12	18
	Bushtit				1	3	4
	Cassin's finch				3		3
	Cactus wren		1				1
	Chipping sparrow	3	17	13	49	25	107
	Cordilleran flycatcher		7		4		11
	Common raven	2	2		11	6	21
	Dark-eyed junco	4	15	5	105		129
	Downy woodpecker		1				1
	Dusky flycatcher				8	1	9
	Evening grosbeak		2				2
	Gambel's quail					1	1
	Great blue heron				1		1
	Gray flycatcher				8	14	22
	Grace's warbler	14	14		100		128
	Hammond's flycatcher				1		1
	Hairy woodpecker	1	5		23	1	30
	Hermit thrush	7	40	2	21		70

House finch				1		1
House wren	3	10	1	9		23
Juniper titmouse				1	13	14
Lark sparrow					12	12
Lesser goldfinch				4		4
Lesser nighthawk					1	1
Mountain bluebird			1	7		8
Mountain chickadee	4	10	5	77	3	99
Mourning dove				1	2	3
Northern flicker	3	9	1	36		49
Pinyon jay				8	5	13
Pine siskin	2	32		11		45
Plumbeous vireo			1	42	1	44
Pygmy nuthatch	1	6	2	98	1	108
Red-breasted nuthatch		15		1		16
Ruby-crowned kinglet	3	27		11		41
Red crossbill		1		5		6
Red-faced warbler		12		3		15
Red-winged blackbird				2		2
Spotted towhee		1		1	7	9
Stellar's jay	1	12		38		51
Townsend's solitaire		2		6		8
Vesper sparrow					2	2
Violet-green swallow			2	8		10
Virginia's warbler	1	8		17		26
Warbling vireo	18	41	5	18	1	83
White-breasted nuthatch	1	7	2	50	1	61
Western bluebird	1		1	41		43
Western kingbird					1	1
Western meadowlark				1		1
Western tanager	2	28	4	75	2	111
Williamson's sapsucker		1		2		3
Wild turkey				1		1
Western Scrub jay				3	1	4

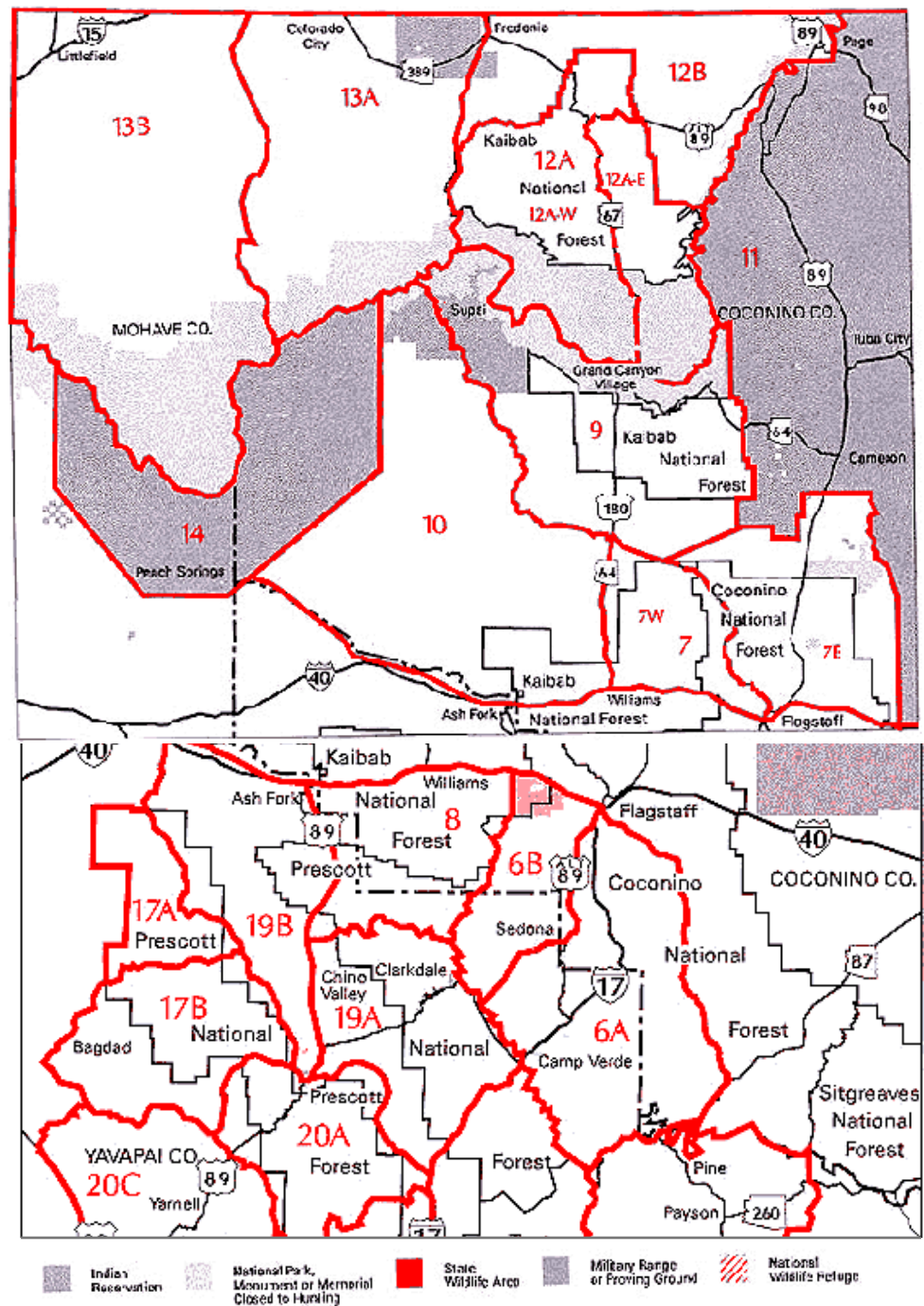
2006

Western wood-peewee	4	4	70	1	79
Abert's towhee			1		1
American cow	1	2	1	8	13
American robin		28	4	34	68
Ash-throated flycatcher			17	59	76
Audubon's warbler	1	257	20	115	395
Black-chinned hummingbird				3	3
Bewick's wren		1	6	72	79
Blue gray gnatcatcher				21	21
Brown headed cowbird	2	1	12	12	27
Black headed grosbeak		19	34	26	79
Brown creeper	2	17	6	12	37
Brewer's sparrow				1	1
Broad-tailed hummingbird			5	5	10
Ban-tailed pigeon		3	1		4
Black-throated sparrow				9	9
Black-throated gray warbler			16	36	52
Bushtit	9	8	2	9	28
Cassin's finch		3	1		4
Cassin's kingbird			4	15	19
Canyon wren			2	1	3
Canyon towhee				1	1
Chipping sparrow	4	74	12	77	235
Clark's nutcracker	1	10	2	4	17
Cordilleran flycatcher		4	9		13
Cooper's hawk				2	2
Common raven	5	9	2	23	73
Dark-eyed junco	5	96	25	157	290
Downy woodpecker		8		1	9
Dusky flycatcher			1		1
Evening grosbeak	1	5	1		7
Gambel's quail	1			3	4
Gray flycatcher			19	58	77
Gray vireo			1	13	14

Grace's warbler		29	12	108	6	155
Green-tailed towhee					4	4
Hairy woodpecker		24	2	23	6	55
Hepatic tanager					2	2
Hermit thrush		143	23	51		217
House finch	1	4			13	18
Horned lark					3	3
Hooded oriole					1	1
House wren		22	6	10	1	39
Hutton's vireo					1	1
Juniper titmouse		14		4	36	54
Lark sparrow					28	28
Lesser goldfinch	1			3		4
Mountain bluebird			3		2	5
Mountain chickadee	5	126	28	91	24	274
Mourning dove		29	4	80	48	161
Northern flicker	1	50	14	66	11	142
Northern mockingbird					74	74
Northern pygmy owl				1	1	2
Orange-crowned warbler		1		1		2
Olive warbler		1		7		8
Olive-sided flycatcher		2		3	1	6
Pinyon jay	8	9		7	12	36
Pine siskin	10	25	1	11	2	49
Plumbeous vireo		1		47	8	56
Purple martin	1	1				2
Pygmy nuthatch	30	32	3	56	3	124
Red-breasted nuthatch	4	51	8	7	3	73
Ruby-crowned kinglet		129	17	26		172
Red crossbill	17	7	1	7	2	34
Red-faced warbler		7		4		11
Red-naped Sapsucker		1		5		6
Rock wren		1		3	1	5
Red-tailed hawk					1	1

Red-winged blackbird				1		1
Say's phoebe					1	1
Scott's oriole					6	6
Spotted towhee				7	54	61
Stellar's jay	3	30	3	36	6	78
Townsend's solitaire		2		6		8
Three-toed woodpecker		7				7
Turkey vulture					1	1
Vesper sparrow					4	4
Violet-green swallow	9	9	4	6	1	29
Virginia's warbler		1		15	4	20
Warbling vireo		139	27	45		211
White-breasted nuthatch		16	7	84	19	126
Western bluebird	7	5	6	46	3	67
Western kingbird					3	3
Western meadowlark					1	1
Western scrub jay	1			1	19	21
Western tanager		127	21	103	3	254
Western wood-peewee		50	3	49	3	105
Williamson's sapsucker		32	5	15		52
Wild turkey				1		1

Appendix 4. Arizona Game and Fish Department Game Management Units on and near the Kaibab National Forest.



Appendix 5. Glossary of Terms

CLEARCUTTING Harvesting of all trees in one cut on an area for the purpose of creating a new, even-aged stand. The area harvested may be a patch, stand, or strip large enough to be mapped or recorded as a separate age class in planning. Regeneration is obtained through natural seeding, or through planting or direct seeding.

COMMERCIAL THINNING Cutting for the sale of products (poles, posts, pulpwood, etc.) in immature stands to improve the quality and growth of the remaining stand.

DBH: Diameter at breast height. Diameter of a tree approximately 4 1/2 feet above the ground.

EVEN-AGED SILVICULTURE The combination of actions that results in the creation of stands in which trees of essentially the same age grow together. Managed even-aged forests are characterized by a distribution of stands of varying ages (and therefore tree sizes) throughout the forest area. Regeneration short period at or near the time that the stand has reached the desired age or size and is harvested. Clearcutting, shelterwood cutting, seed tree cutting, and their many variations are the cutting methods used to harvest the existing stand and regenerate a new one. In even-aged stands, thinnings, weedings, cleanings, and other cultural treatments between regeneration cuts are often beneficial. Cutting is normally regulated by scheduling the area of harvest cutting to provide for a forest that contains stands having a planned distribution of age classes. (36 CFR 211.3[k] NFMA Regulations.)

GROUP SELECTION CUTTING Involves the removal of small groups of trees to meet a predetermined goal of age distribution and species in the remaining stand. The distance across an opening created by removal of a group of trees is usually no more than one to two mature tree heights.

PRECOMMERCIAL THINNING The selective felling, killing, or removal of trees in a young stand primarily to accelerate diameter growth on the remaining stems, maintain a specific stocking or stand density range, and improve the vigor and quality of the trees that remain. A thinning done in a stand of trees not yet of commercial size.

SALVAGE CUTTING Done to harvest trees in imminent danger of being killed or damaged by injurious agents. Dead and dying trees are included in salvage cutting.

SEED CUT A cutting that leaves adequately spaced seed-bearing trees and opens the stand to provide conditions for restocking and establishment of a new stand of trees. Under a shelterwood system, the first of the shelterwood cuttings.

SELECTION CUTTING The annual or periodic harvest of trees, usually the oldest or

largest trees, individually or in small groups from an uneven-aged forest, as a means for encouraging continuous natural reproduction and maintenance of an uneven-aged condition.

SHELTERWOOD CUT An even-age regeneration system designed to establish a new crop under a portion of the old stand, which provides both the seed source and shelter for the new seedlings. The mature trees are removed in two or more cuts. (1) The preparatory cut removes a portion of the mature trees and is intended to make the remaining trees more wind firm; preparatory cuts may be omitted where windfall is not a major concern. (2) The seed cut removes additional trees with the intent of allowing additional sunlight to reach the forest floor. The new trees become established following the seed cut. (3) The removal cut removes the last of the mature trees.

THINNING Cutting made in an immature crop or stand, primarily to accelerate the diameter increment (annual growth) of the residual trees, but also by suitable selection, to improve the average form of the trees that remain.

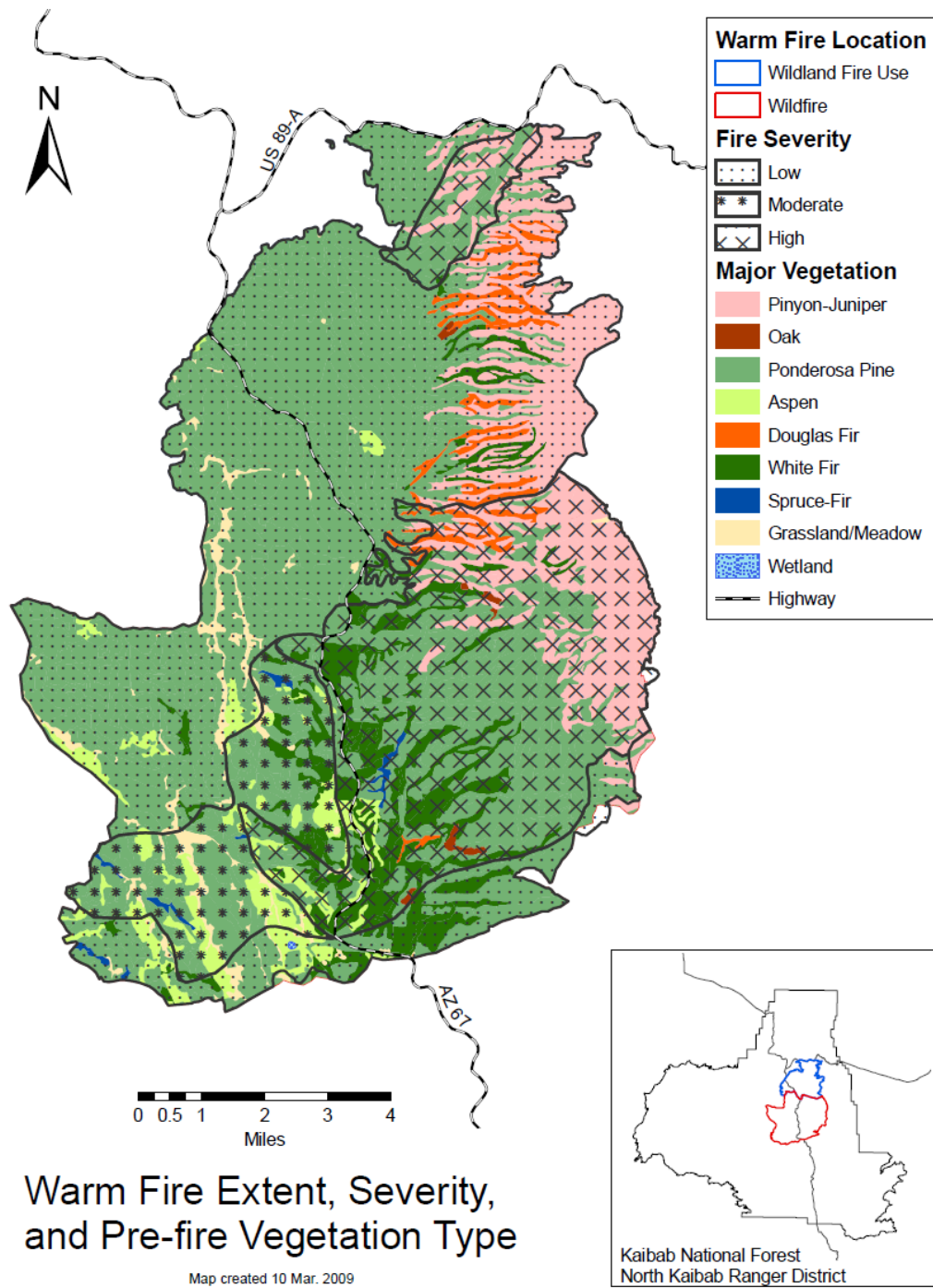
UNEVEN-AGED SILVICULTURE SYSTEMS The combination of action that results in the creation of forests or stands of trees, in which trees of several or many ages grow together. Cutting methods that develop and maintain uneven-aged stands are individual tree and group selection cutting methods.

Appendix 6: Distribution of PNVTs and habitat/cover type across the KNF by district
(Kaibab National Forest 2008a).

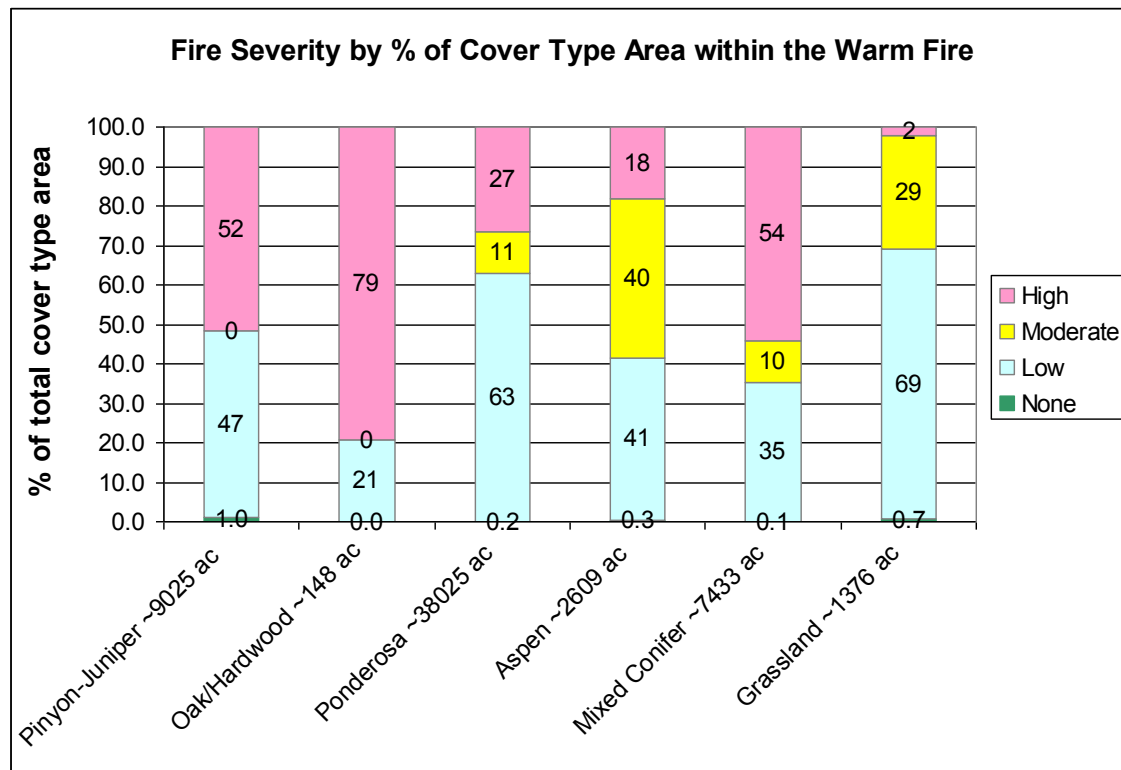
Habitat or Cover Type Equivalent	PNVT	Acres of PNVT on KNF	Percent of KNF	Acres on North Kaibab District	Percent of North Kaibab	Acres on Tusayan District	Percent of Tusayan	Acres on Williams District	Percent of Williams
Pinyon-juniper Woodland	Pinyon-juniper Woodland	647,604	40.5	248,242	37.9	188,961	57.0	210,4010	34.3
Ponderosa Pine	Ponderosa Pine	553,310	34.6	155,209	23.7	104,881	31.6	293,219	47.8
Dry Mixed Conifer Forest	Mixed Conifer Forests ¹	127,848	8.0	113,620	17.3	0	0.0	14,228	2.3
Grasslands	Montane/Subalpine Grasslands	48,584	3.0	6,545	1.0	2,211	0.7	39,828	6.5
Grasslands	Great Basin Grassland	44,181	2.86	0	0.0	3,761	1.1	40,419	6.6
Spruce-fir Forest	Spruce-fir Forest	29,146	1.8	29,002	4.4	0	0.0	144	0.0
Grasslands	Semi-desert Grasslands	25,115	1.6	25,115	3.8	0	0.0	0	0.0
Riparian Associations	Wetland/Cienega	1,479	0.1	608	0.1	0	0.0	871	0.1
Riparian Associations	Cottonwood Willow Riparian Forest	1,197	0.1	1,197	0.2	0	0.0	0	0.0

¹ Mixed Conifer includes aspen.

Appendix 7. Map of Warm Fire severity and habitat types affected.



Appendix 8. Warm Fire effects on percent cover.



Appendix 9. Priority grassland restoration areas based on the South Zone Grassland Restoration Assessment, KNF 2007.

Tusayan Ranger District Priority Areas	Acres	Location Description
Donaldson	17,890	SW corner of Tusayan RD in patchy grassland openings, savannah, pinyon-juniper woodland and ponderosa pine stringers
Heather	28,230	NW corner of Tusayan RD in open grass-sagebrush, savannah, pinyon-juniper woodland, and ponderosa pine stringers
New Dent	91,860	SE boundary area of district in open lowland grass-sagebrush, savannah, and pinyon-juniper woodland
Upper Basin (includes Lower Basin)	41,215 (Ongoing Grassland and Savannah Restoration Area)	NW corner of district in open lowland grass-sagebrush and savannah east and west of State Highway 64
Williams Ranger District Priority Areas	Acres	Location Description
Clark	28,730	NE boundary of Williams Ranger District and east of ST Hwy 64
Ida	45,345	North boundary of district and east of ST Hwy 64
Wash Tub	57,640	SW part of district and adjacent to and south of I-40
Corridor (Joins Garland Prairie and Wash Tub Priority Area)	21,112	South of City of Williams and south of I-40 connecting Wash Tub and Garland Prairie
Garland Prairie	15,000 Ongoing Grassland Restoration Area	SE part of district; mostly south of I-40 with a small section north of I-40
Government Prairie	10,000 Ongoing Grassland Restoration Area	NE part of district and just north of I-40

Appendix 10. Trends in riparian conditions across the KNF.



Figure A: Downward trend associated with Warm Fire effects on Joe's Mudhole

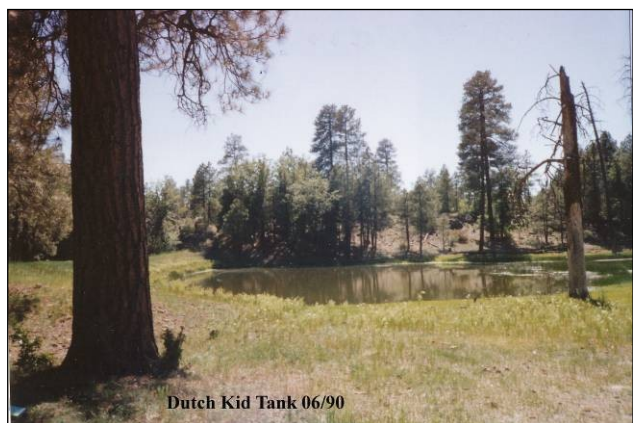


Figure B: Downward trend associated with elk impacts on Dutch Kid Tank.



Figure C: Upward trend and positive effects of fencing on Mud Lake.



**Figure D: Increasing trend associated
fencing and livestock exclusion, proper
functioning condition**