Management Indicator Species of the Kaibab National Forest: Population Status and Trends

Version 2.0

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Historic Ponderosa Pine Forest , 1914
Photo by Gus Pearson

Pronghorn (*Antilocapra americana*)
Photo by Corbis

Hairy Woodpecker (*Picoides villosus*)
Photo by J. R. Woodward

Red Squirrel (*Sciurus vulgaris*)
Photo by John Koprowski

Sonefly (*Pteronarcyidae sp.*)
Photo by Milton Rand/Tom Stack & Associates

Current Ponderosa Pine Forest , 1999
Photo by Tom Bean

Prepared by: Kristin Bratland, Wildlife Biologist, Kaibab National Forest, Supervisor's Office

Bill Noble, Forest Biologist, Kaibab National Forest, Supervisor's Office

Roger Joos, Wildlife Biologist, Kaibab National Forest, Supervisor's Office

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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. It is expected that the habitat portion will soon be updated as well.

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 Arizona Game and Fish Department (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 incorporate 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.

	Management Indicator Species	**Habitat or Habitat Component**
Insects	Aquatic macroinvertebrates	Riparian
Birds	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
	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
Mammals	Elk	Early-seral ponderosa pine, mixed conifer, spruce-fir
	Mule deer	Early-seral aspen and pinyon-juniper

	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. 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.

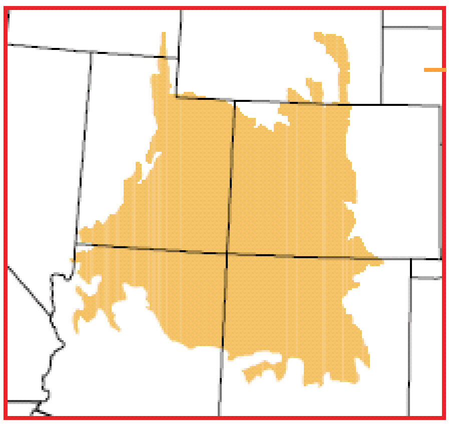
There are 37 BCRs across North America (Sauer et. al. 2005) two of which occur on the KNF: the Northern Kaibab Ranger District is in the Southern Rockies/Colorado Plateau BCR and the Williams and Tusayan Ranger Districts occur in the Sierra Madre Occidental BCR (Table 2). Historically, only species occurring 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.

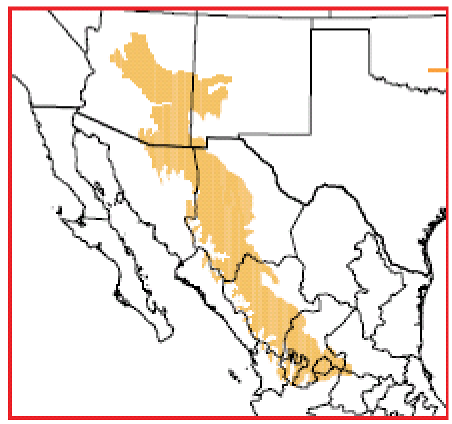
Table 2. Bird Conservation Regions overlapping the Kaibab National Forest, Coconino County, AZ.

Bird Conservation Region	**Total # of BBS Routes in BCR**[1]	**Overlapping Kaibab Ranger District(s)**
Southern Rockies/Colorado Plateau	215	North Kaibab
Sierra Madre Occidental	35	Williams and Tusayan

[1]The 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	
	Woodland	Successional Scrub	Wetland Open Water	Cavity	Open cup	Permanent Resident	Short Distance	Neotropical	Ground/ Low	Mid-Story/ Canopy
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 Ranger District) commonly measures inches wide and does not sustain fish. The short stream portion of Big Springs (North Kaibab Ranger District [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

[1]The 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 wildlife, 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 (*Callibaetus 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 m^2 (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. Steinke (2007) identified 566 acres of actual mapped wetlands, based on the 1990 KNF Forest-wide Riparian Survey. During the survey, 129 springs and seeps and 492 stocktanks 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 from the Forest-wide survey. Of these, 31 wetland areas or 35% are in poor condition, 45 or 51% are in fair condition, and 12 or 14% are in good condition. Wetland areas in fair and poor condition do not provide optimum riparian habitat conditions. Of the wetland and riparian areas in good condition (14%), 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 interspersion 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

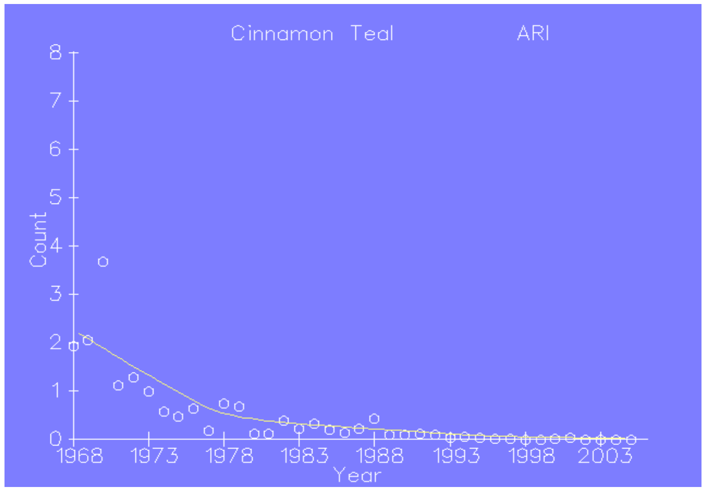


Figure 1. Cinnamon teal population trend data for Arizona from 1966-2005, BBS data (Sauer et al., 2005).

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 were 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 respond 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 ha ±

500 SD (4344.1 ± 1235.5 ac, Bright-Smith and Mannan 1994) and the average distance between neighboring nests was 3.0 km ± 0.83 SD (1.86 ± 0.52 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. 1993b, 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 NKRD. Virtually no part of the NKRD 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 ± 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.

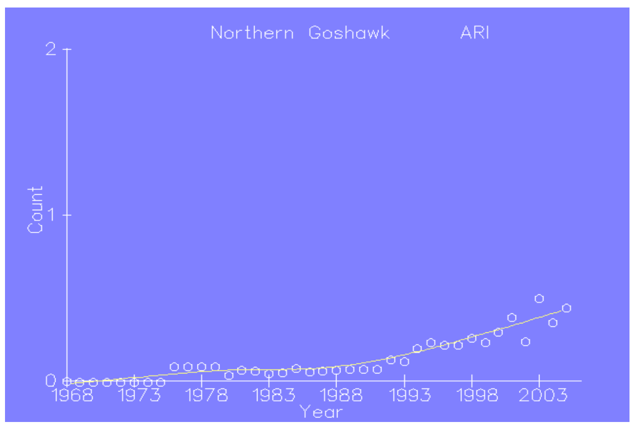


Figure 2. Goshawk population trend data for Arizona from 1966-2005, BBS data (Sauer et al., 2005).

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.

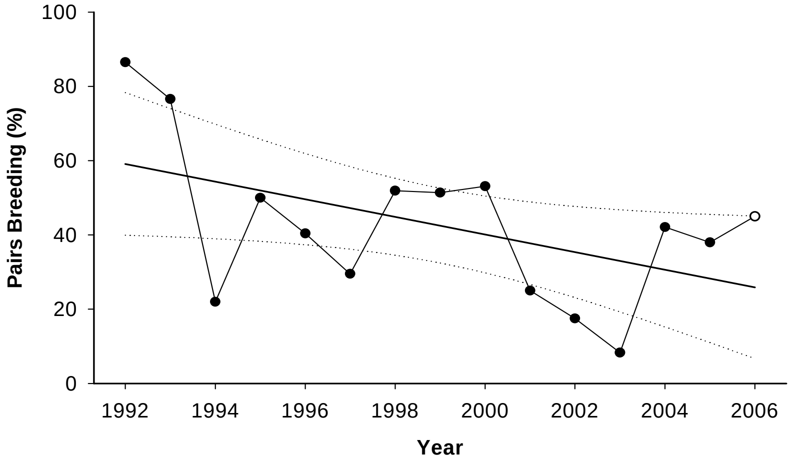


Figure 3. The annual percent of territorial goshawks breeding (laid eggs) on the Kaibab Plateau, between 1992 and 2005 (●) and the *minimum* percent of goshawks breeding in 2006 (○) needed to cause the current significant decline in reproduction (P = 0.030) to become non-significant (P = 0.053). From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

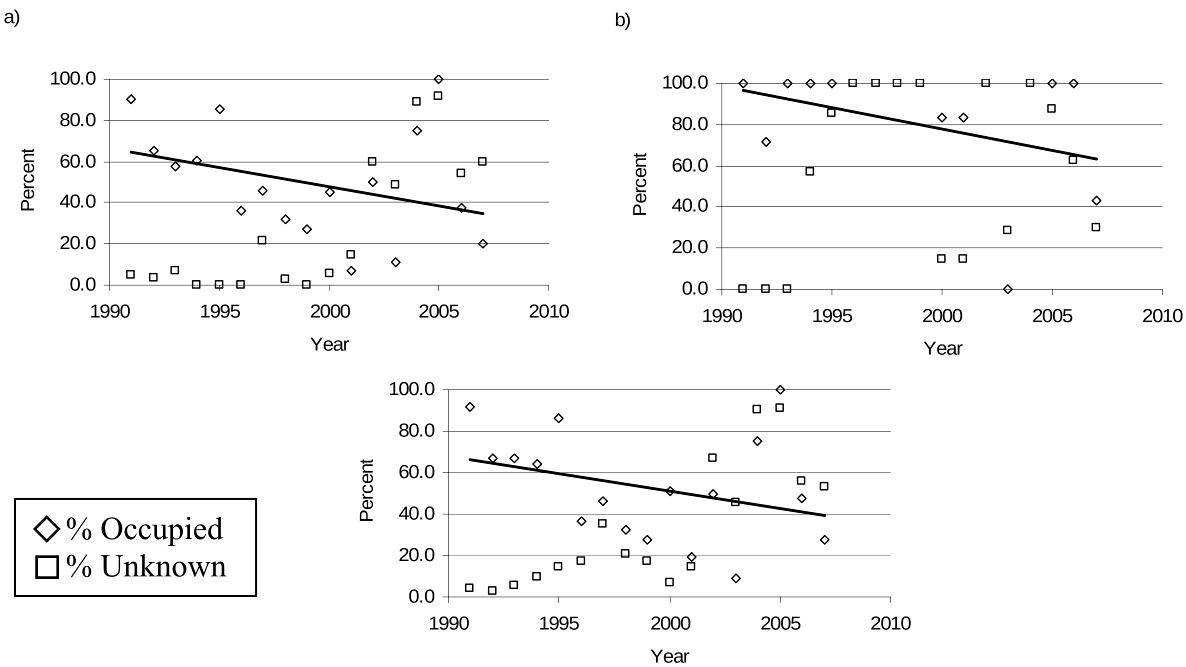


Figure 4. Percent of surveyed and occupied territories present on the Kaibab National Forest, Coconino County, AZ. a) William's Ranger District, b) Tucsyan Ranger District, and c) South Zone (William's and Tucsyan Ranger Districts combined).

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, McPeek 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 the 541,097 acres of ponderosa pine, 127,687 acres dry mixed conifer, 127 acres mixed conifer with aspen, and 29,146 spruce fir habitat. Hairy woodpeckers prefer ponderosa pine and mixed conifer in Northern Arizona, giving a potential habitat availability of 668,784 acres (270,647 ha). 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 lighting 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

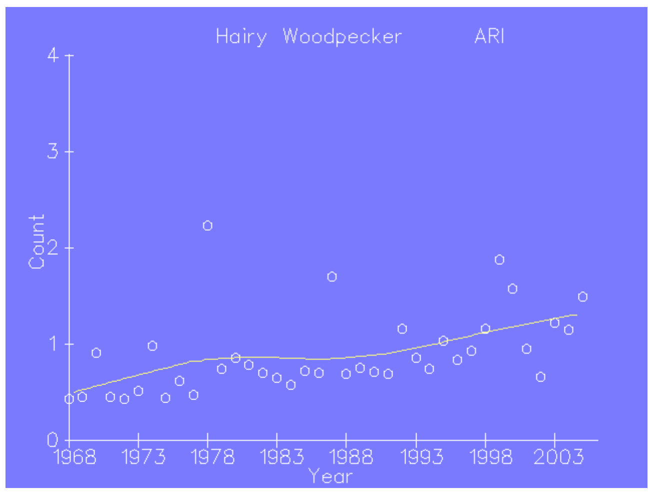


Figure 5. Hairy woodpecker population trend data for Arizona from 1966-2005, BBS data (Sauer et al., 2005).

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).

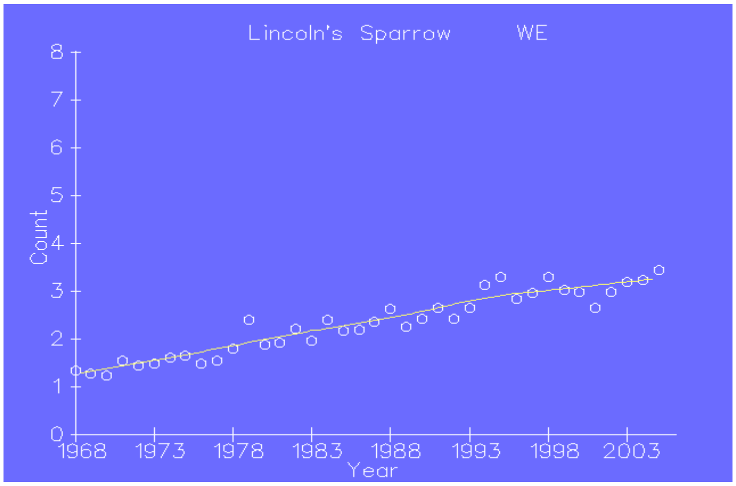


Figure 6. Lincoln's sparrow population trend data for the Western BBS region from 1966-2005, BBS data (Sauer et al., 2005).

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 course 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 it’s 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.

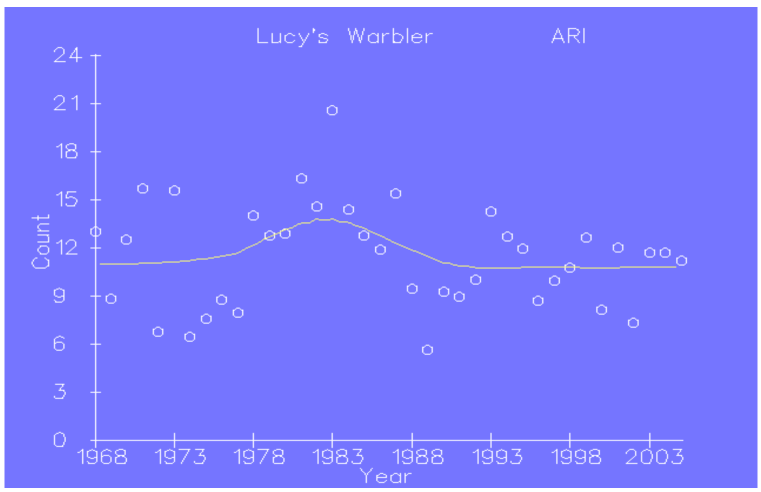


Figure 7. Lucy's warbler population trend data for Arizona from 1966-2005, BBS data (Sauer et al. 2005).

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 Metropolus (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). Grinell 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 Paridea, 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:
On the KNF, 638,475 acres are classified as pinyon-juniper woodland. This only represents habitat for the juniper titmouse if the required habitat structure and function is present. 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.

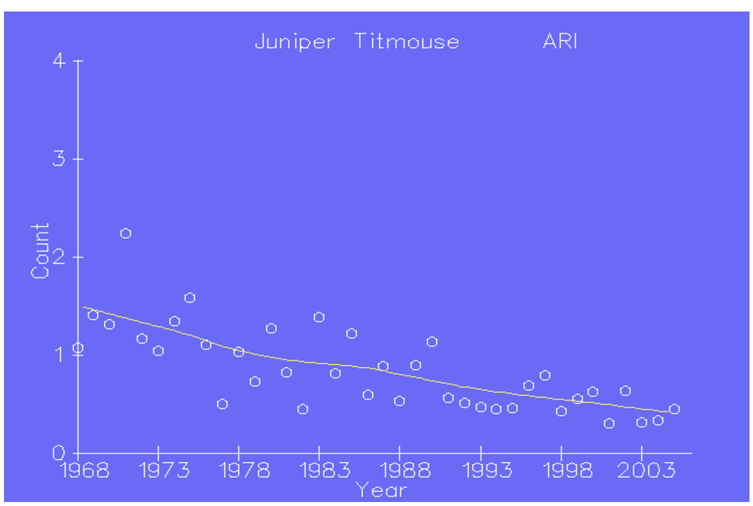


Figure 8. Juniper titmouse population trend data for Arizona from 1966–2005, BBS data (Sauer et al. 2005).

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 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 on 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, Kingerny 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 (*Poecile 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:
The KNF contains 555,100 acres of ponderosa pine habitat (Vander Lee at al. 2006). 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

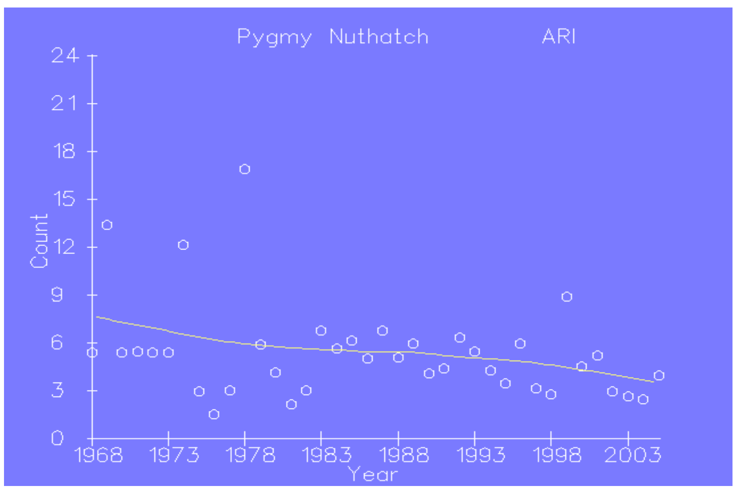


Figure 9. Pygmy nuthatch population trend data for Arizona from 1966-2005, BBS data (Sauer et al. 2005).

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 (≥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 protected activity centers (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 affect 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°) 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 lackof 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

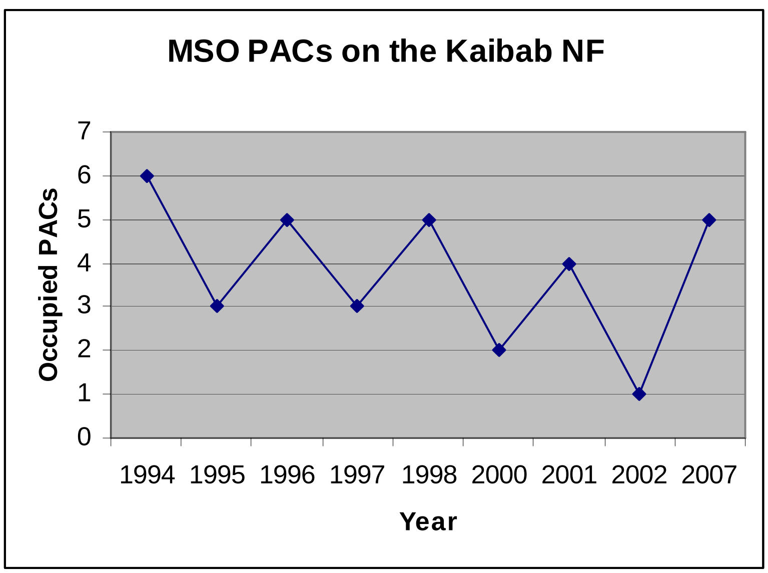


Figure 10. Number of occupied Mexican spotted owl protected activity centers (PACs) during years when all known PACs were monitored on the Kaibab National Forest, Coconino County, AZ.

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.

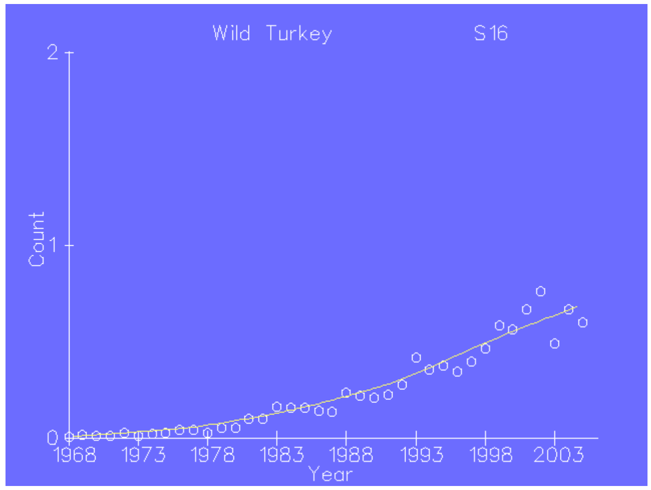


Figure 11. Wild turkey population trend data for the Southern Rockies/Colorado Plateau region, BBS Data (Sauer et al. 2005).

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

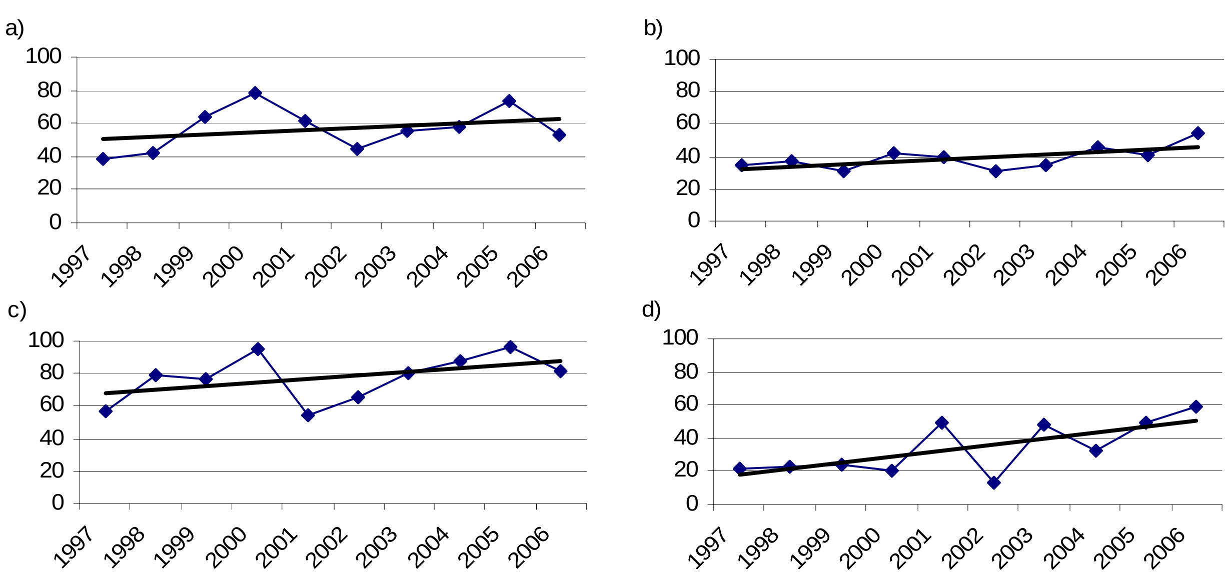


Figure 12. Percent of hunters per day seeing turkeys in hunt units a) unit 6B, b) unit 7, c) unit 8, and d) unit 9 on the Kaibab National Forest, Coconino County, AZ (AGFD).

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 rubber*). 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. Here it typically occurs as small (typically < 25 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 signigicant 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.

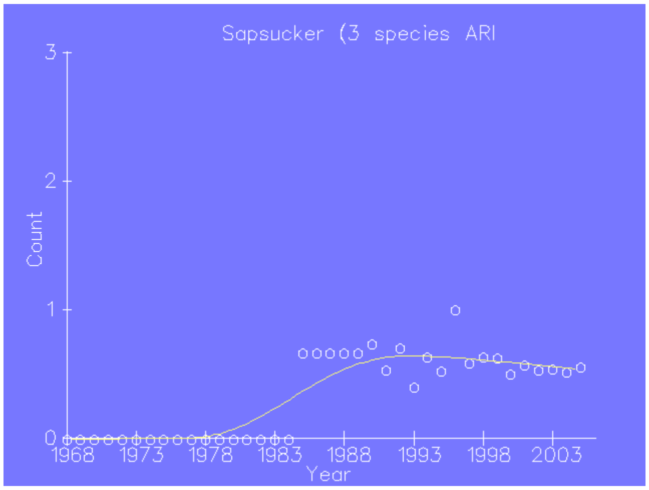


Figure 13. Sapsucker population trend data for Arizona from 1966-2005, BBS data (Sauer et al. 2005).

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, work in spotted owl habitat is frequently cumbersome when working with the U.S. Fish and Wildlife Service in regards to policy and regulation and the KNF has been advised to work outside of Mexican spotted owl habitat but 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 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 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 angustifloia*), 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 quinqurfolia*), 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 ranger, 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.

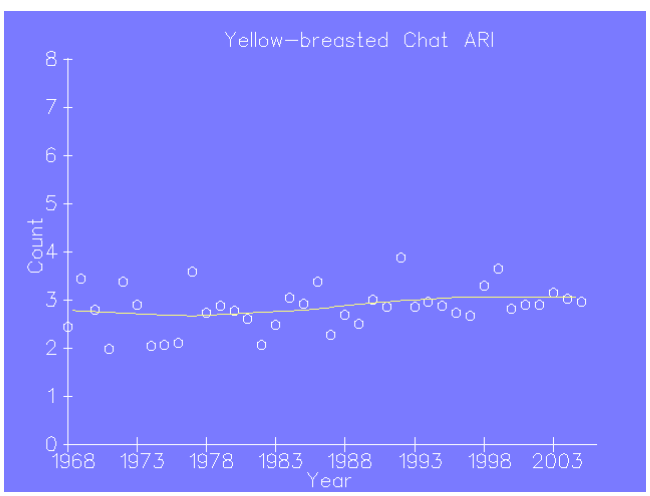


Figure 14. Yellow-breasted chat population trend data for Arizona from 1966-2005, BBSdata (Sauer et al. 2005).

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 affective 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 interspersion 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). Grazing and browsing effects from elk are seriously impacting aspen regeneration on the Williams District and vigor of key shrub species and Gambel's oak on the Tusayan District. Efforts to successfully regenerate aspen on the Williams Ranger District are completely dependent on fencing clones off to prevent access by elk. Arizona Game and Fish Department manage the wildlife populations and they currently have no plans for reducing elk numbers on the South Zone of the KNF.

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				

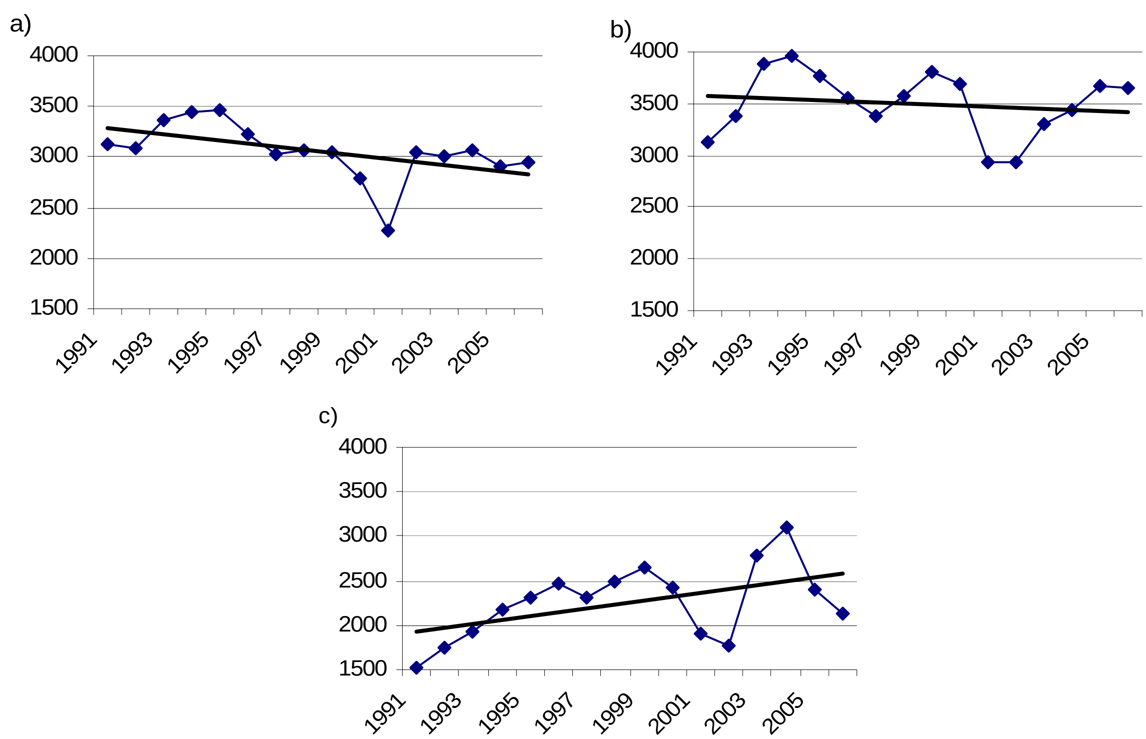


Figure 15. Estimated elk populations by hunt units a) 6B, CN, and 8, b) 7, and c) 9, on the Kaibab National Forest, Coconino County, AZ (AGFD).

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 (Hoffmiester 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.

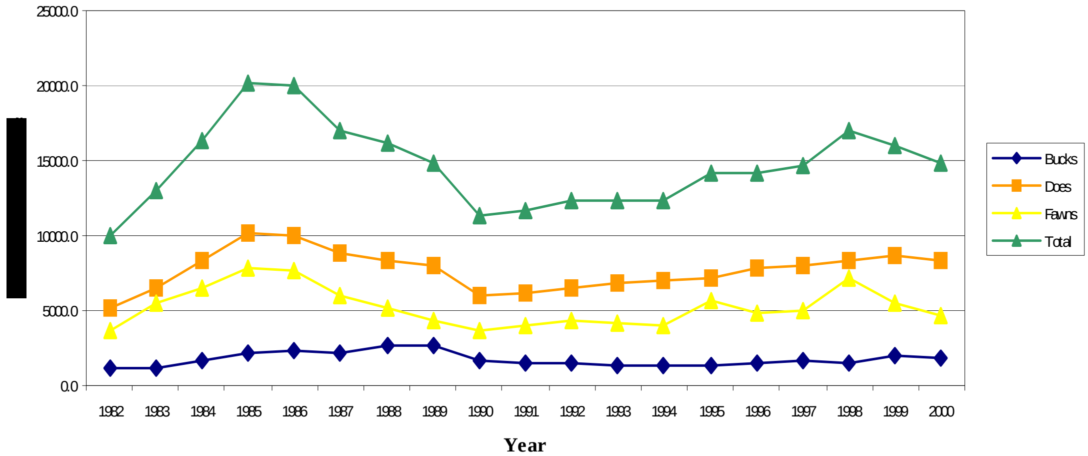
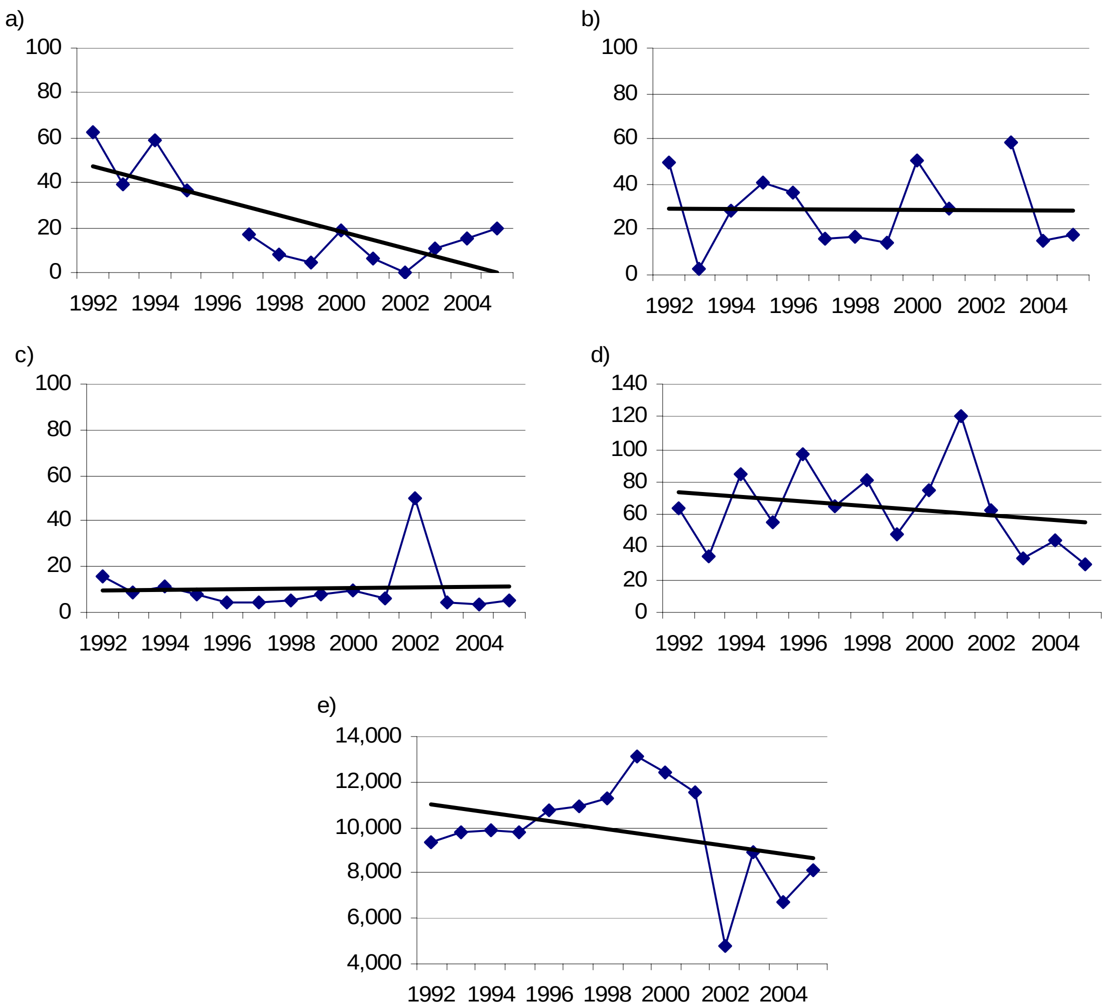


Figure 16. Mule Deer Population Estimates for the North Kaibab Ranger District, Kaibab National Forest, AZ (Buck 2002).



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).

Year	South Zone				North Kaibab RD
	Unit 6B	Unit 7	Unit 8	Unit 9	Unit 12A
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.

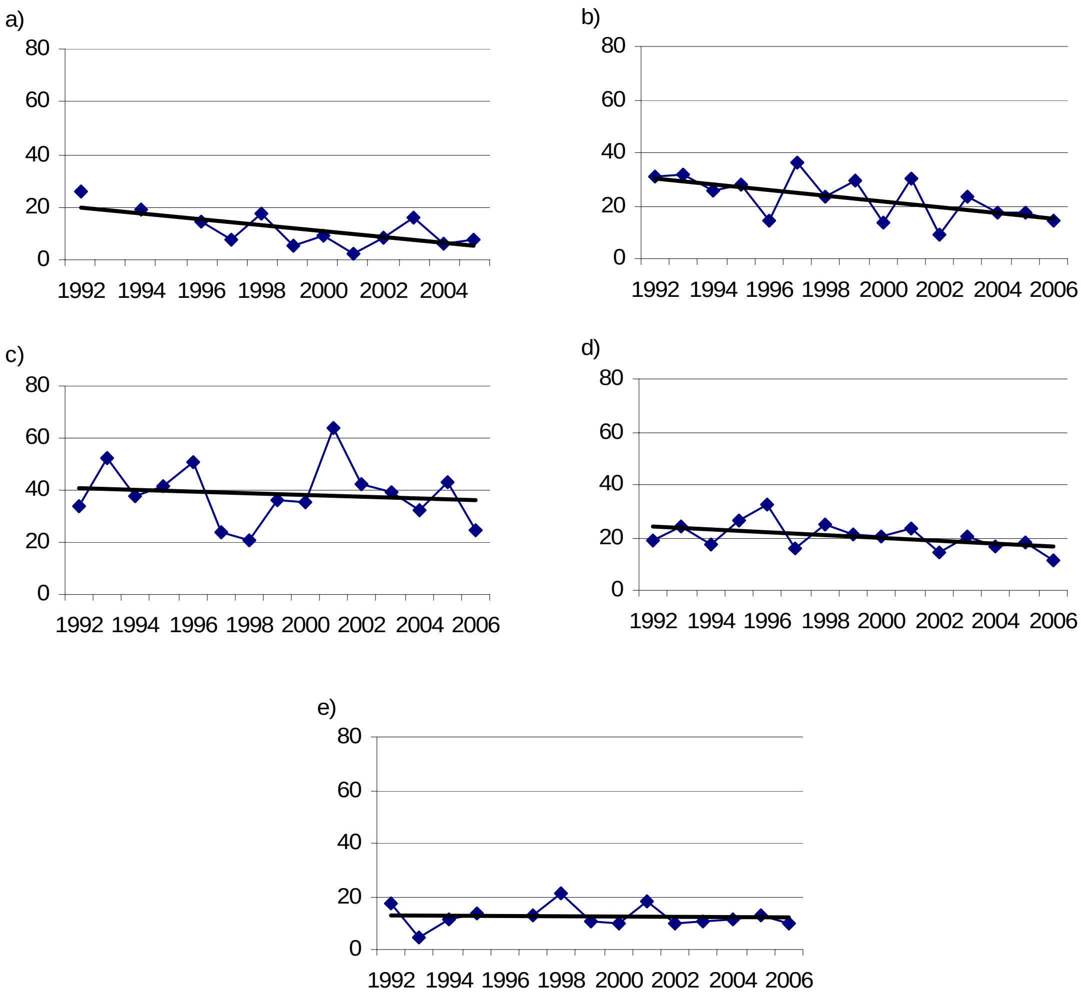


Figure 18. Pronghorn seen by air per hour by hunt unit a) 6B, b) 7, c) 8, d) 9, e) 12A, on the Kaibab National Forest, Coconino County, AZ (AGFD).

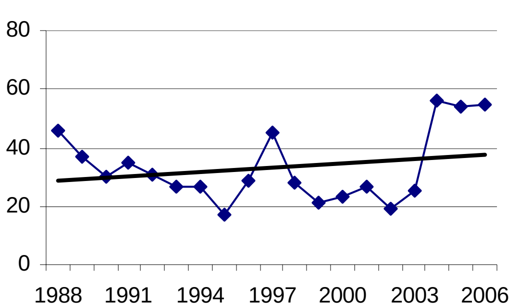
Table 12. Pronghorn fawns per 100 does by Hunt Unit on the Kaibab National Forest, Coconino County, AZ (AGFD, 1993, 1998, 2000, 2006).

	Coconino NF[1]		South Kaibab		North Kaibab
Year	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

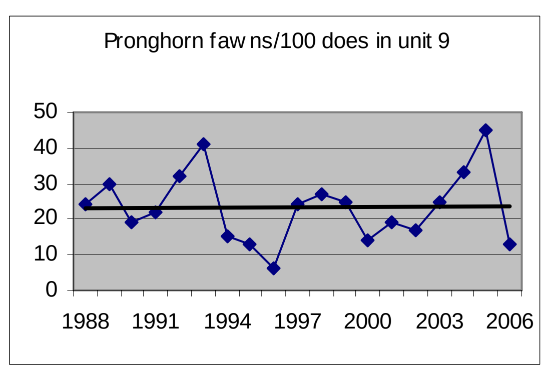
[1]Coconino National Forest

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.

a)



b)



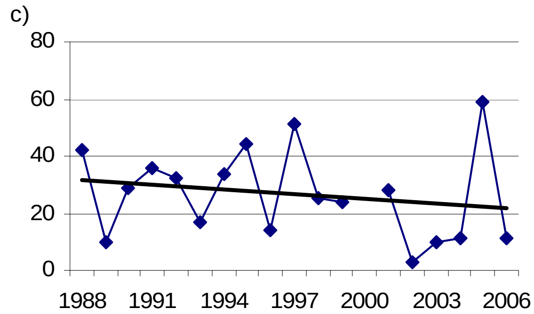


Figure 19. Pronghorn fawns per 100 does on hunt units a) 8, b) 9, and c) 12 on the Kaibab National Forest, Coconino County, AZ (AGFD, 1993, 1998, 2000, and 2006).

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.

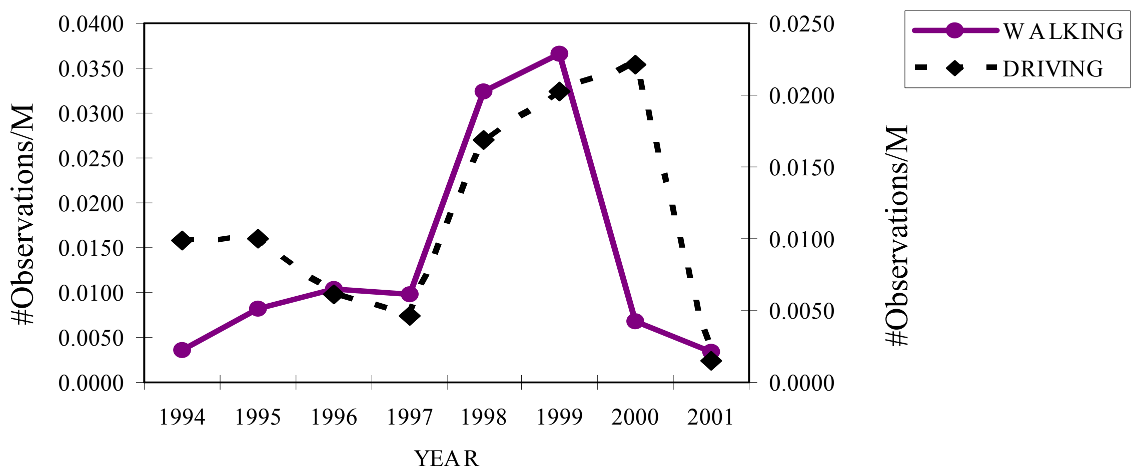


Figure 20. Comparison of walking and driving counts for estimating red squirrel abundance for the Kaibab Plateau, North Kaibab Ranger District, Kaibab National Forest, Coconino National Forest, AZ (Salafsky 2002).

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	Mixed Conifer	
	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 interspersion of 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

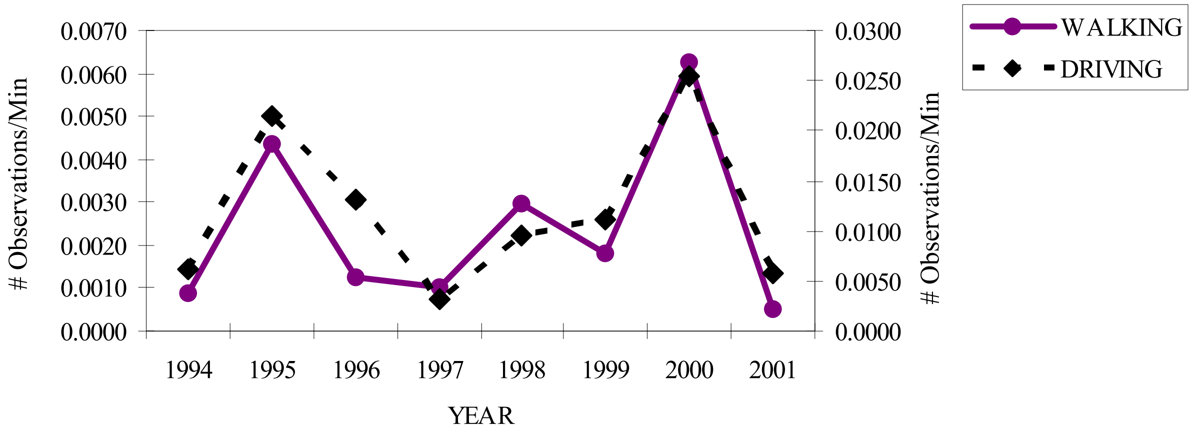


Figure 21. Comparison of walking and driving counts for estimating Kaibab squirrel abundance for the Kaibab Plateau, North Kaibab Ranger District, Kaibab National Forest, Coconino National Forest, AZ (Salafsky 2002).

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	Ponderosa Pine	
	Number	Density
1999	19	0.0485
2000	44	0.1123
2001	15	0.0383
Average Density per ha		**0.0664**

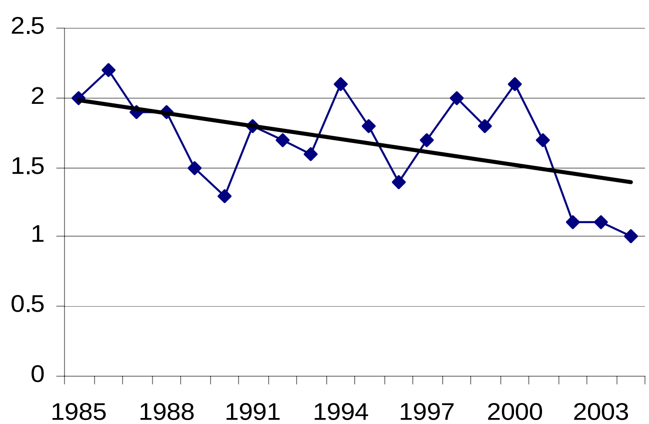


Figure 22. Tree squirrels harvested by hunters in Arizona (AGFD 1993, 1998, 2000, and 2006).

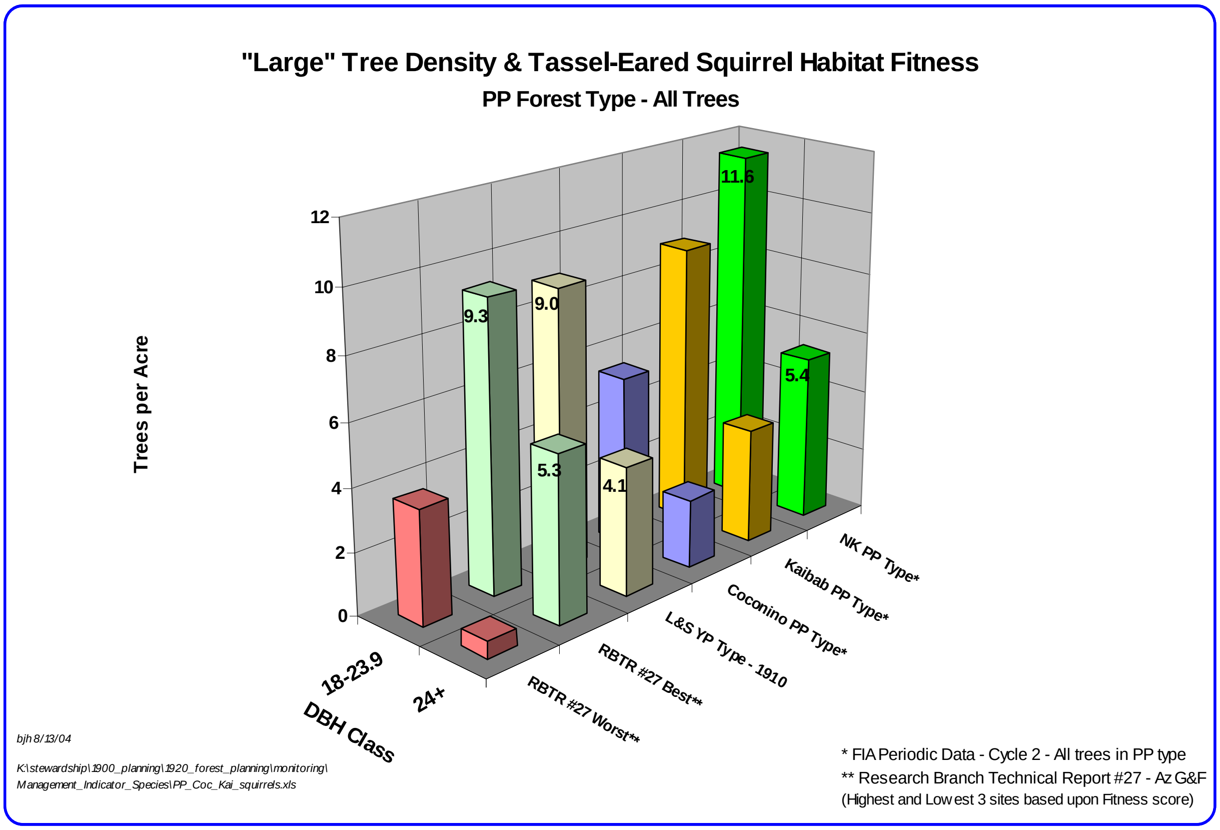


Figure 23. Numbers and density of large ponderosa pine trees as described for tassel-eared squirrels by Dodd et al. 1998 (RBTR #27 Best and Worst), recorded by Lange and Stewart during thier 1910 survey of the Kaibab Plateau (L&S YP ["Yellow Pine"]Type – 1910), the Coconino National Forest (where Dodd 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 to50 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.

Habitat Trends on the Kaibab National Forest

INTRODUCTION

The first Forest Plan for the KNF was completed in 1987 and included a description of the baseline habitat conditions at that time. To complete the review of MIS trends, it was our intention to analyze trends in habitat changes since 1987. It became apparent at the outset that habitat changes would be difficult to quantify because of differences in methodology for assessing vegetative conditions at different points in time.

When the KNF was tasked with summarizing vegetative cover types across the National Forest in 1987, very little quantitative data existed. At the time, a major emphasis of forest planning was setting and achieving timber harvest goals. Accordingly, the Forest was described in terms of Potential Natural Vegetation (PNV), i.e., "the land's potential for vegetation development" absent disturbances such as fire and management actions (USDA 1986). What can potentially grow on a given site can be very different from what is actually there at any particular point in time. For example, classification differences are apparent with current grasslands and savannahs that have a PNV of piñon-juniper or ponderosa pine. The potential vegetation is linked to the soil type and does not change within typical management, or even human generational, timeframes. In short, the point in time that should initiate the trend analysis was characterized in a manner that is not sensitive to change over time. One exception to this is extreme site degradation. Anecdotal observations suggest this may have occurred when the Bridger-Knoll, Point, and Pumpkin Fires seared steep, forested slopes. The PNV for those sites has likely been changed until new soils develop and trees once again develop on site. Nevertheless, when evaluated across the Forest, PNV is insensitive to changes since 1987.

Other means of classifying vegetative cover types have been used, but each has fundamental differences that make comparisons a produce exercise, i.e., comparing apples to oranges. The Continuous Forest Inventory (CFI) system is a long-term inventory of National Forest lands that was initiated in the 1950s. Permanent plots were established across all National Forest lands and the vegetation was sampled about once every decade. However, CFI plot locations were marked on the ground and were frequently discovered and subsequently avoided as Forest projects were implemented. Avoiding the plots undermined the usefulness of CFI as an indicator of forest conditions. The CFI system was discontinued in the late-1980s and replaced by another national sampling effort called the Forest Inventory and Analysis system. There are less than 250 of these plots on the KNF. Using such relatively few samples to characterize over 1.5 million acres makes this is a coarse system with high sampling errors at the Forest scale, i.e., the error rate is about 50 percent of the mean values when applied to individual cover types on the KNF. The error rate increases as comparisons are made at smaller scales, such as structural stages within cover types. The high error rates render the mean values meaningless for the purpose of this analysis.

In order to sample the Forest in an appropriate manner, surveys need to be conducted in adequate numbers and carefully (e.g., randomly) located across the various vegetation types. In 1987, the KNF began conducting stand examination surveys primarily in areas where management activities were being planned. Although stand examination surveys provide a precise estimate of overall forest structure, they are not suited for summarizing trends in habitat development because each survey represents a single point in time. The surveys are averaged together for a single view of the forest, but repeat surveys at the same location have seldom been done, so trends cannot be described. Sample locations were chosen for reasons other than generating a sound forest survey (i.e., problems exist with pseudoreplication) and to achieve adequate sample numbers too many years would need to be combined to define a "discreet moment" in time to compare to 1987. The end result is similar to the national efforts described above: the numbers generated have unknown reliability due to their non-random nature and inflated error terms relative to the scale of change we are trying to track. For the sake of display purposes, some of the different datasets, including projections into the future, are presented in Tables 15 and 16.

Table 15. Sources of Vegetation Types for the Kaibab National Forest.

Vegetation Cover Type	Diversity Technical Report	Forest Plan EIS	Potential Natural Vegetation	Existing Cover Type	Difference Between EIS and Existing Cover (%)[1]
Seral Grassland	25404	25404	76404	249293	11.6%
Shrub grassland	0	46402	76676		
Ponderosa pine	548025	532465	403299	521235	-0.7%
Spruce-fir	28518			29546	
Mixed conifer	129632	142590	107557	33260	-3.4%
Aspen	15842			26887	
Woodland	786036	786036	699324	682539	-6.7%
Riparian	985	0	0	113	-0.0%
Other	0	1556	181470	213	
Total	1534442	1534453	1544730	1543086	-0.6%

[1]Percent difference based on total acres presented in Forest Plan EIS.

Table 16. Comparisons of estimates of coniferous forest structural conditions from: the time of the 1987 Forest Plan analysis; projections from the 1996 Plan Amendment; and current conditions.

Age Class	1986 Conditions	%	1996 Data Projected for 2003		2002 Existing Conditions	%
Seedling-Sapling	77580	11%	63094	10%	28955	5%
Poles	226227	34%	4487	1%	159113	26%
Immature Sawtimber	81619	12%	281275	44%	188603	31%
Mature Sawtimber	289625	43%	282085	45%	234256	38%
Total	**675051**	**100%**	**630942**	**100%**	**610927**	**100%**

The KNF completed an inventory of forested lands in 1990, based on about 260 randomly selected sites with over 2800 individual sample points. Although this does not yield the data necessary to complete a trend analysis, it does provide the opportunity to replicate the samples and get meaningful information in the future. A record has been retained of each stand location and the KNF is planning on re-sampling the plots in 2003. This should create a database that allows for relatively rigorous comparisons of changes across the Forest over the last 13 years.

Changes in Management Direction

The 1987 Forest Plan describes even-aged harvest strategies for timber management on suitable timberlands. Suitable 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 to 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 being 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 one-half to four acres in size. Rather than creating even-aged stands across the landscape, the desired future condition was a mix of varying age-classes such that:

- -10 percent of the planning area is non-stocked or seedlings;
- -10 percent of the planning area consists of saplings (1-4.9 inches dbh);
- -20 percent of the planning area consists of young trees (5-11.9 inches dbh);
- -20 percent of the planning area consists of mid-aged trees (12-17.9 inches dbh);
- -20 percent of the planning area consists of mature trees (18-23.9 inches dbh);
- -20 percent of the planning area consists of old trees (> 23.9 inches dbh).

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. The 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 other plants, i.e., grasses, forbs and shrubs. The Forest Plan now manages more by forest ecology rather than timber management. Harvest objectives have become recreating patterns of natural forest patch dynamics. The average treatment now cuts about 25 percent of the total board feet per acre than was cut under the original Forest Plan. Total acres treated per year also decreased with the implementation of these guidelines (Figure 24). However, 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. Nevertheless, managing for this level of heterogeneity is expected to move the KNF much closer to pre-settlement conditions than today’s forests.

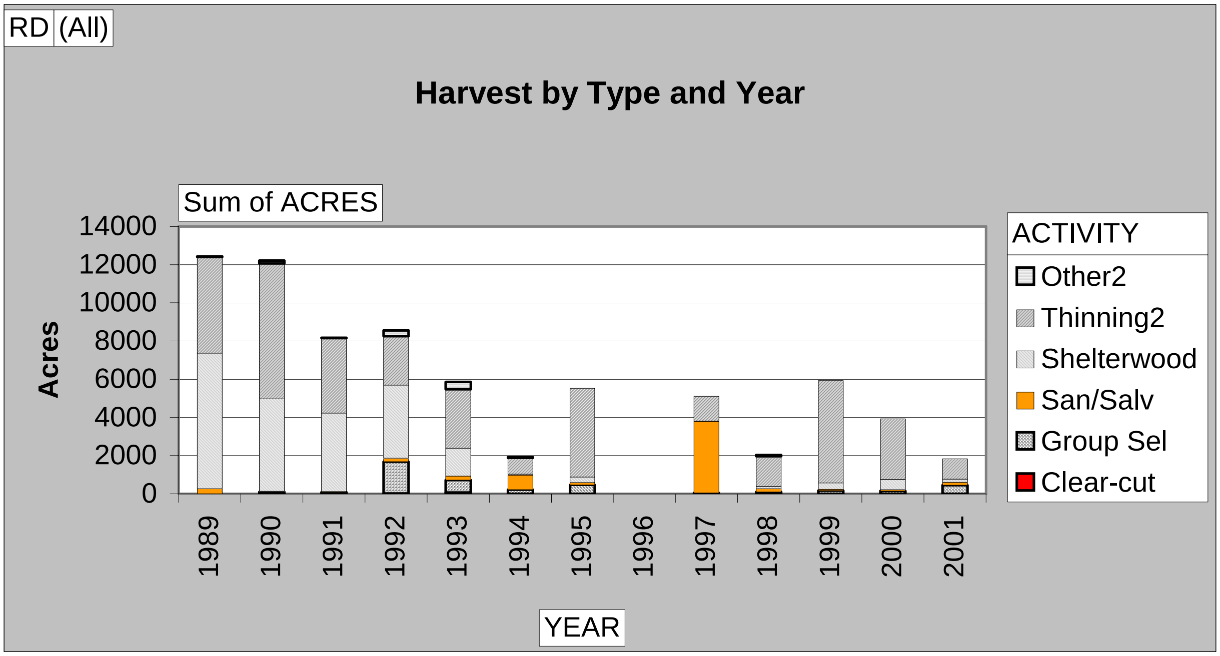


Figure 24. Harvest by year and acres since implementation of the Kaibab Forest Plan.

Restoring the grouped nature of trees and the interspersion 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., undated). 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.

A 1990 amendment to the 1987 Forest Plan called for 15 percent of the suitable timber base to be reserved as blocks of old-growth. Old-growth is here defined as sites dominated by trees greater than 120 years old and greater than 18 inches dbh. In 1996, acreage occupied by old-growth was increased from 15 to 20 percent of the suitable timber base. Old-growth structure now occurs within most forested areas over ten acres in size. An additional 20 percent 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 percent 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 three to six large, residual trees. In actual practice, three to six residual trees have also been left in some regeneration harvest patches less than an acre in size. With the addition of three to six 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 five- to ten-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 numbers. Desired numbers are based upon the desired conditions of tree size and canopy density in the goshawk management recommendations previously cited. Harvest will occur at the one to four acre scale with the objective of providing timber while enhancing forest health and growth. For example, on the Kaibab Plateau there are more ponderosa pine in the 18 to 24 inch dbh size-class then there were historically but fewer trees greater than 24 inches dbh. One strategy now is to thin the trees in the 18 to 24 inch dbh size-class to stimulate recruitment into the larger dbh size-classes. The maximum managed age- and size-classes for trees was increased from 120 years/18 inches dbh to 200-250 years/and about 27 inches dbh. The number of trees greater than 18 inches per acre is expected to increase over time. The intent over time 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 vertical diversity in forest structure. About 40 percent of any one stand 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 to the KNF.

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 should be at 20 year or greater intervals. Exceptions would be prescribed burning, small-scale operations such as planting, or addressing outbreaks of insects or disease. It is expected that uneven-aged management will.

Because of the problems related to not having comparable datasets to explore habitat trends, confounded by the change in management objectives since implementing the 1987 Forest Plan, the intent of this report is to document current habitat conditions to facilitate future trend analyses and to make inferences about the change in the baseline since 1987.

Ponderosa Pine Cover Type

Covering about 3 million acres statewide, ponderosa pine (*Pinus ponderosa*) is the third most abundant forest type in Arizona (O'Brien 2002). The KNF has about 521,000 acres of ponderosa pine forest (about 34percent of the total Forest acreage), principally occurring on the Williams and NKRDs (Table 17). The much drier Tusayan RD is dominated by piñon-pine and juniper forest. Several years of good seed crops in the early 1900s, combined with favorable conditions for germination, resulted in a dominance of about 90-year-old trees in today's ponderosa pine forests. The Williams Ranger District has historically been much more readily accessible than the relatively isolated NKRD. Timber harvest, especially railroad logging during the first half of the 20th Century, changed the forest structure on the Williams District. Even with accelerated timber harvest on the NKRD during the 1980s, growth and development of large ponderosa pine trees (greater than 18 inches dbh) has been greater than the mortality rate, including loss from logging, since the establishment of the National Forest.

Table 17. Acres of ponderosa pine forest by size-class across the Kaibab National Forest.

Ponderosa Pine	Williams		Tusayan		North Kaibab		Kaibab NF	
Age Class	**Acres**	**Mass Ft3/Ac**	**Acres**	**Mass Ft3/Ac**	**Acres**	**Mass Ft3/Ac**	**Tot Acres**	**Avg Mass Ft3/Ac**
<5" dbh	4,595	652	2,001	842	7,020	91	13,616	391
5-8.9" dbh	16,590	1,291	8,025	878	0	0	24,615	1,156
>9" dbh	132,188	1,521	35,870	1,013	181,085	2,538	349,143	1,996
Sparse[1]	35,135	1,093	16,418	977	0	0	51,553	1,056
Total	188,508	4,557	62,314	3,710	188,105	2,629	438,927	4,599

[1]**Refers to savannah-like conditions less than 30 square feet basal area.**

Annual recruitment of ponderosa pine trees can be evaluated from stand examination surveys. Because this information is an average from about 15 years of surveys, it does not provide trend information. However, the data do provide growth and recruitment estimates from which trends may be inferred when assembled with losses. Recruitment into any one size-class largely depends on the number of trees in the previous size-class minus trees that grow out of the size class. Recruitment is relatively low for trees greater than 30 inches dbh on the Williams RD (Table 18), but the numbers of trees in the 12 to 30 inch dbh classes indicates that the growing stock exists for trees to grow into the future large dbh size-classes. Most of the volume coming off of the Forest consists of small diameter trees. Most sales in recent years harvested relatively high amounts of board feet of timber, but included few large trees. On the Williams and Tusayan Districts, trees greater than 30 inches dbh are typically only cut when they present a hazard to humans (e.g., along roads or campgrounds), or have high mistletoe infection rates (e.g., the Red Mud sale with 67 trees cut in this size class). Following the Red Mud sale (prepared in the early 1990s) very few trees of this size have been designated for harvest. The next most "intensive harvest" of this size-class was likely the Saginaw-Kennedy sale, completed in 1994, where six of 9,647 trees greater than 12 inches dbh harvested were greater than 30 inches dbh. More typical now is killing trees with high dwarf-mistletoe infection rates and leaving them in place to provide structural attributes.

Table 18. Numbers of ponderosa pine trees recruited per acre per decade by Ranger District, Kaibab National Forest.

Ranger District	TPA	TPA	TPA	TPA	TPA	Trees Per Acre recruited per decade			
	>=30"	24-30"	18-24"	12-18"	5-12"	>=30"	24-30"	18-24"	12-18"
Williams	0.02	0.17	0.9	5.2	17.3	0.001	0.018	0.305	0.822
Chalender	0.03	0.22	1	4.9	14	0.002	0.029	0.345	0.609
North Kaibab	0.23	0.84	1.7	2.7	8.2	0.025	0.041	0.060	0.309
Tusayan	0.04	0.31	0.8	1.8	12.9	0.004	0.017	0.059	1.276

The NKRD is fairly unique in that a systematic timber inventory was completed on the Plateau in 1909 (Lang and Stewart 1910). If we assume that the 1909 survey data reflects a forest that is sustainable through time, then comparisons to today's forest can help evaluate management strategies. Currently, there are about 4.5 more trees per acre in the 18 to 30 inch dbh size-class than there were before logging was initiated (Figure 25). Logging before the 1990s focused on removing the larger dbh size-classes. Although annual recruitment in the larger size-classes is relatively high on the NKRD, harvest appears to have exceeded recruitment (Table 19). These results are tentative and will be reviewed. The tentative nature of this conclusion is due to combining data from two different spatial scales (site-specific timber sale areas and ecosystem management area averages) and from two different methods (timber sale cruises and stand examination surveys). It appears that today there are about 0.2 trees per acre less than there were in 1909 for trees greater than 32 inches dbh. It may be that the landscapes in which the sales were planned have a surplus of large trees relative to the desired condition. This would lead to the harvest of trees greater than 18 inches dbh even though other areas without sales are deficit in large trees. For example, some goshawk PFAs may consist of overly-dense patches of large dbh trees. Leaving artificially dense forests in those PFAs does nothing to compensate for a paucity of large trees in other territories.

The decline in the large tree component is due to management activities and the loss of large trees from fire, insects, and disease since implementation of the Forest Plan. Drought conditions since the mid-1990s only exacerbate the effects of fire, insects, disease, and overstocking. Large trees that are crowded by cohorts of young trees resulting from fire suppression do not have the same physiologic responses to stress that more open grown trees might have. These losses compound those that result from management activities. 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. It is obvious the timber harvest objectives stated in the1987 Forest Plan were being pursued in 1988 through 1991, and that, once the objectives became related more to ecosystem health, this aspect of forest structure was being retained in areas where management activity was occurring.

Table 19. Comparison of projected net change from harvest and annual growth in large trees, by Forest Plan direction, estimated over a 20-year period, 1987-2007.

	>=30				24-30				18-24			
	Trees	Recruit	Cut/Ac	Change	Trees	Recruit	Cut	Change	Trees	Recruit	Cut	Change
North Kaibab RD												
All	47495	5233	0.161	-0.110	173462	2048	0.475	-0.456	351053	325475	0.942	2.211
post MRNG	47495	5233	0.037	0.014	173462	2048	0.139	-0.119	351053	325475	0.263	2.889
South Kaibab RDs												
All	9427	767	0.007	-0.002	72709	7062	0.069	-0.024	288023	79719	0.226	0.281
post MRNG	9427	767	0.002	0.003	72709	7062	0.013	0.032	288023	79719	0.130	0.376

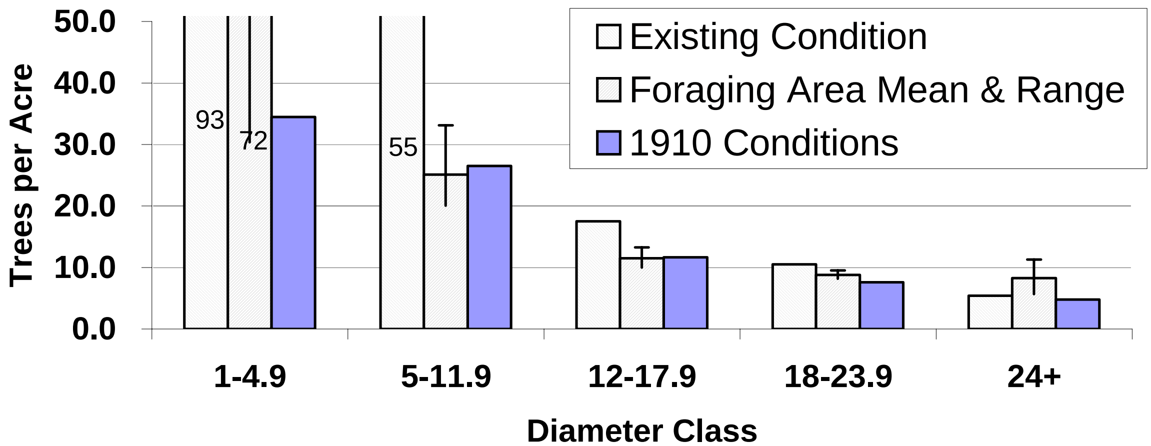


Figure 25. Comparison of trees per acre by size-class in ponderosa pine cover type between 1910 surveys and existing conditions, NKRD. Existing conditions include areas inside and outside goshawk foraging areas.

The shift in management direction 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 (Figure 26). 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 (Table 20), although some shelterwood cuts that were planned before 1992 were not harvested until later in the 1990s and even 2001. The thinning projects and group selection cuts planned since 1992 were intended to improve the distribution of ponderosa pine size-classes and reduce the fuel loads that resulted from the decades of fire suppression. Although the reduction in acres treated per year is a result of shifting the emphasis from fiber production to wildlife habitat enhancement, the fewer the number of acres treated is also inversely related to risk of loss from catastrophic fire.

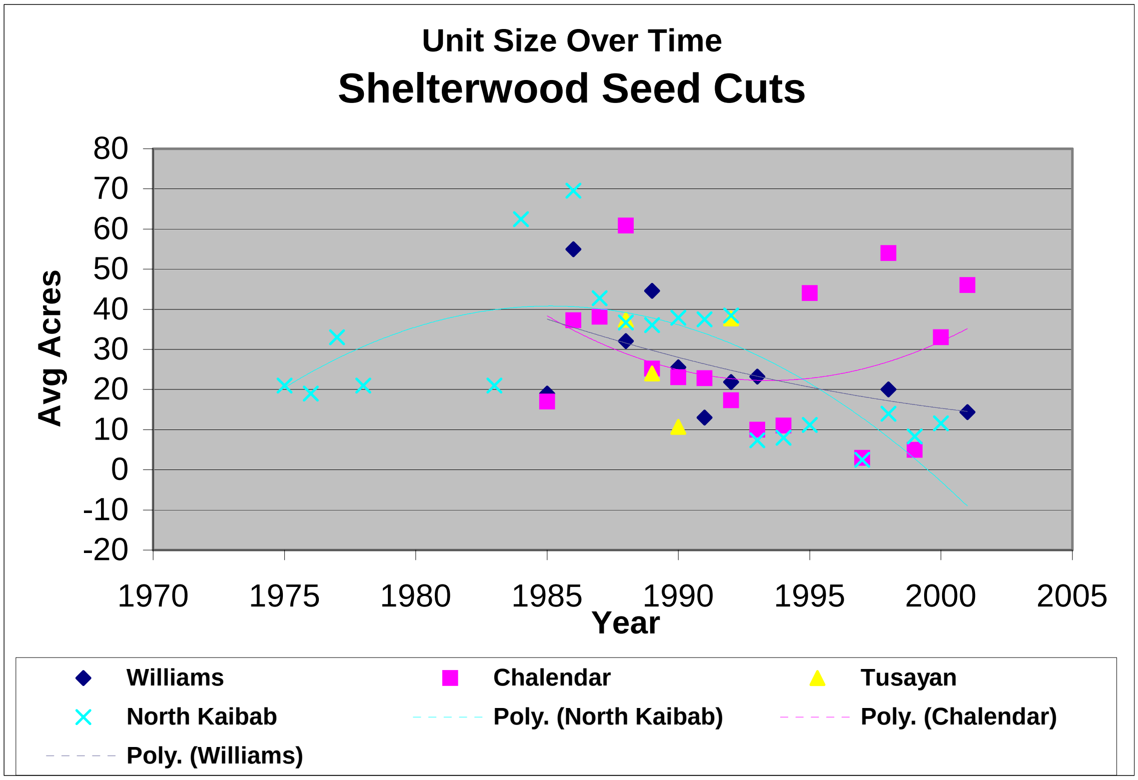


Figure 26. Average shelterwood unit size (acres) by Ranger District by year, Kaibab National Forest. Note the four squares indicating 30 to 55 acre averages between 1995 and 2001 total six stands (243 acres) and are all located in areas with levels of dwarf-mistletoe too high to carry under uneven-aged management with the desired structure identified in the Forest Plan.

Table 20. Timber harvest treatments, Kaibab National Forest, 1987 through 2001.

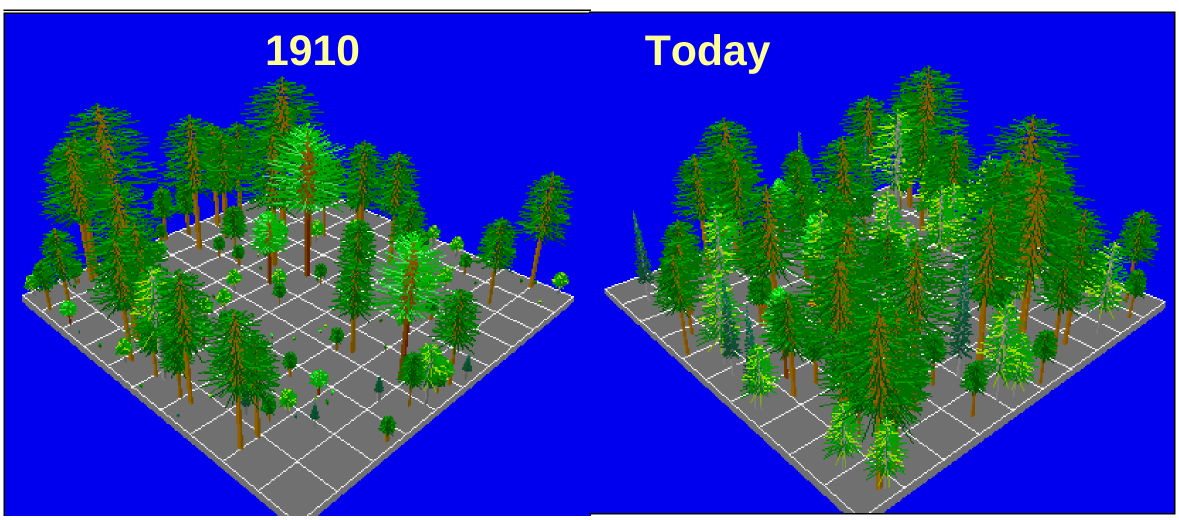
Treatment	**Acres Cut 1989-1991**[1]	**Acres Cut 1992-2001**[2]
Clearcut	95	74
Shelterwood Cuts	16,049	6,734
Sanitation/Salvage	343	5,490
Commercial Thinning	16,040	23,600
Group Selection	74	3,929
Other[3]	0	769
Total Treatment Acres	**32,601**	**40,596**
Average Acres Cut/Year	**10,867**	**4,511**

[1] *Timber harvest under the 1987 Forest Plan.*
[2] *Timber harvest following the shift to uneven-aged management.*
[3] *Typically light treatments, e.g., attaining visual quality prescriptions.*

There are about 46 percent more trees per acre now (6"dbh and over) than there were before logging began on the North Kaibab RD in pine (Figure 27). Even in trees 18" dbh and over, there is an increase in the number per acre on the North Kaibab RD of about 34percent since

1910. On the southern ranger districts, there appears to be a ten-fold increase in the number of trees 6" dbh and over (Figure 28). Directly related to this increase in stand density is the frequency and severity of 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. About 86,500 acres of wildfire have burned on the KNF since 1987. (Table 21) shows the largest and most severe fires, totaling 72,800 acres. Nearly 25,000 of these acres occurred in ponderosa pine habitat and almost 40 percent of that burned as crown fires. Each of these fires created broad areas of early seral stage communities. In some extreme cases, such as the Point and Pumpkin fires, excessive 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 25 percent of the total acres converted to early seral stage communities since 1987 (38,435) were a result of crown fire in the ponderosa pine cover type.

Figure 27. Forest schematic based on stand survey inventories for the North Kaibab Ranger District.



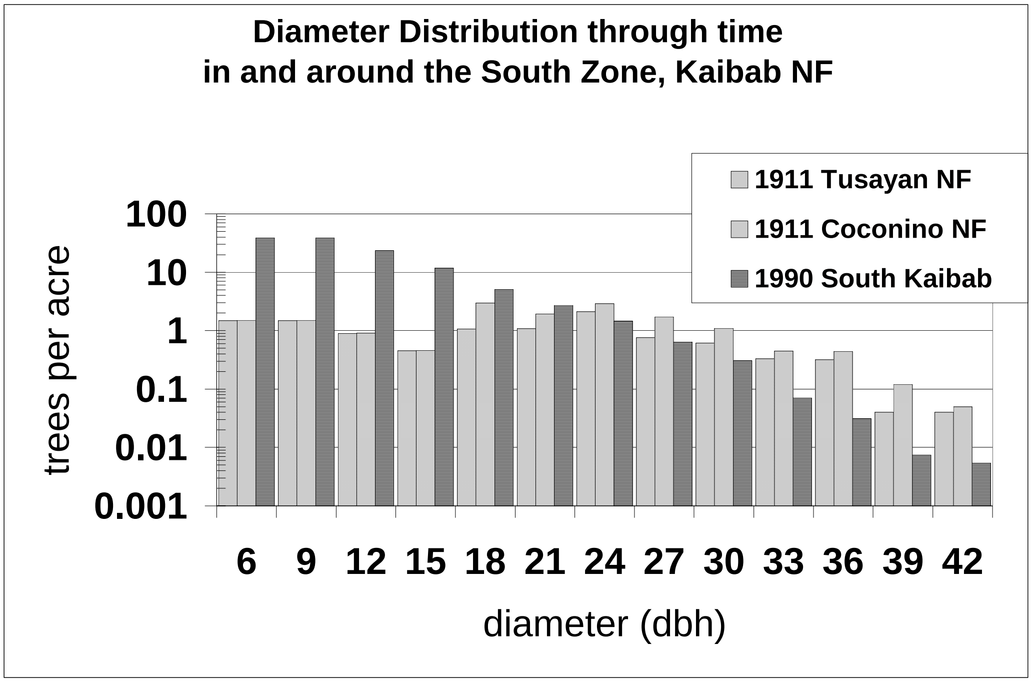


Figure 28. Trees per acre by diameter class for areas in and around the current Williams and Tusayan RDs (after Woolsey 1911) compared to ca 1990. Note the vertical axis is a log scale. There are more trees now in every size class through 21 inches dbh and less in every size class greater than 21 inches dbh than in the 1911 samples.

Table 21. Acres burned by wildfire, including acres of crown fire, within the ponderosa pine cover type, Kaibab National Forest, 1987 through 2001.

Fire	Year Burned	Acres Burned	Acres of Crown Fire
Willis	1987	2,000	2,000
Point	1993	1,800	600
Bridger-Knoll/Jump	1996	54,000	4,800
Pumpkin[1]	2000	15,000	2,250
Total Acres Burned		**72,800**	**9,650**

[1] About 7,000 acres of this fire burned on the Coconino NF. However most of that portion (the Kendrick Mountain Wilderness) is administered by the Kaibab NF and is therefore reported here

Before the 1992 Forest Plan Amendment, management objectives emphasized even-aged harvests that treated nearly 2.5 times more total acreage per year then the post-Forest Plan, uneven-aged harvests. Pre-1992 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 1992 switched to uneven-aged management that primarily applied commercial thinnings and group selection cuts. There appears to have been a net decrease in the distribution and number of ponderosa pine trees greater than 24 inches dbh since the signing of the Forest Plan in 1987. Although the data used to project growth and harvest rates is problematic, a decease in this size-class could have occurred due to: five years of pursuing 1987 harvest objectives; treatments under the 1992 guidelines that targeted mistletoe infections; and the fact that large trees may occur above target levels at specific sites and be cut even if they are below target thresholds at the landscape scale. The 1992 changes to the Forest Plan was, in effect, a change from timber management to forest management. The majority of the commercial thinning projects are aimed at securing the large tree component across the forest and maintaining them 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 will also speed the recruitment of trees into the larger size-classes and reduce the risk of crown fires. The ultimate goal 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.

Goshawks, pygmy nuthatches, turkeys, tassel-eared squirrels and elk were selected to represent the ponderosa pine cover type. Goshawks, pygmy nuthatches, and turkeys were selected to represent late-seral ponderosa pine forest. Tassle-eared squirrels and elk were selected to represent early-seral ponderosa pine forest. 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 lagre-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 much larger created 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.

A caveat to the estimated trends in ponderosa pine habitat is in regards to fire. The intent of the Amended Forest Plan is to create a forest that is sustainable through time. However, historic management goals, current forest conditions, and recent and projected weather patterns create a high risk of loss to crown fires. The scale of this risk is historically unprecedented and difficult to account for through standard forest planning. It is hoped that management can proceed at a pace that will decrease the fire threat before landscape-scale fires have the opportunity to reduce significant portions of existing forest to early-seral communities.

Gambel Oak

Gambel oak (*Quercus gambelii*) is a component of the ponderosa pine cover type. 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 presence in 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 (greater than 8 inches dbh), live oak are not cut and small oak are only cut to initiate oak regeneration. Firewood cutters are only allowed to cut dead oak 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. Forest management has been ineffective at limiting known problems with fuel wood cutters poaching large oak on the Coconino NF and this issue probably occurs on the Kaibab as well, although the distance to Flagstaff may reduce the incidence of "oak poaching." Because no MIS were selected to specifically represent oak or the pine-oak association, this cover type will not be addressed at this time. It is expected that future iterations of this report will cover this important forest habitat feature.

Mixed Conifer/Spruce Fir Cover Types

Mixed conifer, including spruce (*Picea sp*), fir (*Abies Sp),* and Douglas-fir (*Psseudotsuga menziesii*), is a minor component of Arizona forest land (O'Brien 2002). Similarly, mixed conifer/spruce fir forests account for about four percent of the KNF. Mixed conifer occurs at the higher elevations on the Williams and North Kaibab, but not the Tusayan Ranger Districts. Its range is limited to the higher elevations of the Williams District, occupying about 8600 acres. The portion of tentatively suitable forest was too small (less than 700 acres) to design an effective sampling stratum, and therefore does not show up on some of the Forest level surveys. Also, there were few areas dominated by mixed conifer trees five to nine inches dbh, so there is no stratum for that size class. Spruce-fir occurs only on the NKRD except for the Kendrick Mountain Wilderness on the Williams RD. Much of the central portion of the NKRD supports mixed conifer/spruce-fir and accounts for the majority of the mixed conifer cover type on the KNF (Table 22).

There has been no harvest in the mixed conifer forest type in the South Zone since 1988. Recruitment rates changed with the shift in Forest Plan objectives in 1992 (Table 23). Although trees greater than 24 inches dbh are still cut when site specific evaluations indicate an excess number of trees within that size-class at the group scale, the focus of most harvests in the 1990s has been trees less than 24 inches dbh. The discussion about "surpluses" of large trees for the pine cover type applies to the mixed conifer forests as well. Harvests regimes are the same as in the ponderosa pine cover type where group selection is applied to areas of one-half to four acres (two acres in spruce-fir) in size. Again, the goal is to create heterogeneous habitat that better mimics the natural forest patch dynamics that are thought to have occurred in these forests during the presettlement era.

Table 22. Existing trees and recruitment by size-class for each species of conifer surveyed in the mixed conifer/spruce-fir forest cover types, Kaibab National Forest.

Ranger District	Existing (trees per acre)						Recruitment (trees per acre per decade)			
	Cover Type[1]	>=30"	24-30"	18-24"	12-18"	5-12"	>=30"	24-30"	18-24"	12-18"
Williams	**TDF**			1.2	8.3	20.7	-	-	0.412	0.485
Williams	**TWF**	0.16	0.5	1.4	4.1	23.3	0.005	0.019	0.130	1.256
Chalender[2]	**TDF**	0.11	0.06	0.6	3.9	32.9	0.000	0.008	0.157	2.656
Chalender	**TWF**	0.28	0.37	2.9	2.3	11.1	0.003	0.103	-0.010	0.554
North Kaibab	**TBS**	0.19	0.51	1.9	4.8	22.4	0.005	0.047	0.147	1.133
North Kaibab	**TDF**	0.1	0.5	2	5.2	10	0.009	0.043	0.242	0.035
North Kaibab	**TES**	0.06	0.22	2.5	6.5	13.6	0.001	0.134	0.260	0.646
North Kaibab	**TSF**	0.11	0.37	1.5	5.4	22.8	0.003	0.033	0.208	1.145
North Kaibab	**TWF**	0.18	0.7	2	5.6	15.3	0.011	0.050	0.215	0.384

[1]DF = Douglas-fir; WF = white fir; BS = blue spruce; ES = Engelmann spruce; and SF = spruce-fir.

[2] The Chalender RD is now combined with the Williams RD.

Table 23. Comparison of projected net change from harvest and annual growth in large trees, by Forest Plan direction, estimated over a 20-year period, 1987-2007.

	>=30				24-30				18-24			
	Trees	Recruit	Cut/Ac	Change	Trees	Recruit	Cut	Change	Trees	Recruit	Cut	Change
North Kaibab RD												
All	13065	497	0.084	-0.072	40347	3129	0.237	-0.160	131284	16177	0.644	-0.245
post 1992	13065	497	0.050	-0.037	40347	3129	0.139	-0.061	131284	16177	0.977	-0.578
South Kaibab RDs												
All	886	16	0.000	0.004	1661	158	0.000	0.037	12204	2579	0.000	0.598
post 1992	886	16	0.000	0.004	1661	158	0.000	0.037	12204	2579	0.000	0.598

Wildfire has will likely continue to be an influence on mixed conifer/spruce-fir forests on the KNF. About 86,500 acres of wildfire have burned on the KNF since 1987. Two of the larger fires included about 16,800 acres of conifer/spruce-fir forest (Table 24). 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 24. Acres burned by wildfire, including acres of crown fire, within the mixed conifer/spruce-fir cover types, Kaibab National Forest, 1987 through 2001.

Fire	**Year Burned**	**Acres Burned**	**Acres of Crown Fire**
Point	1993	1,800	1,200
Pumpkin	2000	15,000	2,250
Total Acres Burned		16,800	3,450

The Pumpkin fire burned through four active Mexican spotted owl Protected Activity Centers (PACs), the areas surrounding nest sites. PACs are intended to secure critical areas on the landscape to help ensure successful reproduction of spotted owls. There are nine PACs that occur on or overlap the KNF. Only six of the PACs are actually administered by the Kaibab. Management within these areas is largely restricted to attaining forest health objectives. Many 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. The Pumpkin Fire affected four PACs in the Kendrick Mountain Wilderness (Figure 29). Two of the PACs had understory burns that may have improved the habitat in the long term. However, 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 and before they are capable of flight. There is no data on how the smoke and the disturbance from fire-fighting activities may have affected owl behavior and survival. The fire intensity was severe in two of the PACs, causing nearly complete mortality of the overstory.

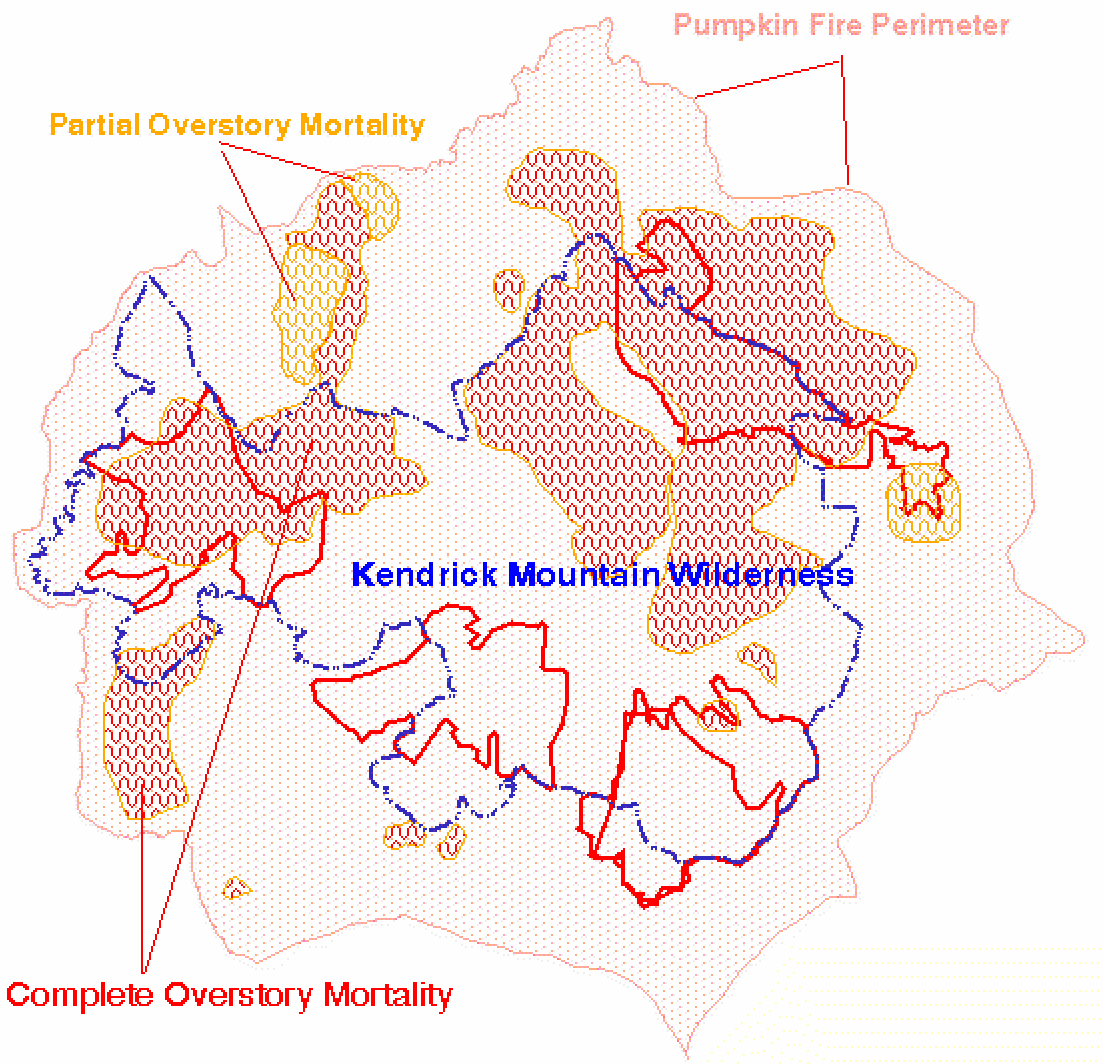


Figure29. Impacts of the Pumpkin Fire on Mexican spotted owl Protected Activity Centers, Kendrick Mountain Wilderness. Four Protected Activity Centers are outlined in red.

Although mixed conifer/spruce-fir is a limited cover type on the KNF, it is important for MIS purposes. Mexican spotted owls and red squirrels were selected to represent late-seral habitat structure and elk represent early-seral mixed conifer/spruce-fir forests. No spotted owls have ever been confirmed nesting, roosting, or foraging in the mixed conifer/spruce-fir forests on the NKRD. The patches of mixed conifer/spruce-fir forests occurring primarily on isolated cinder cone mountains in the South Zone, e.g., Kendrick Peak and Sitgreaves Mountain, support the only spotted owl PACs administered by the KNF. Due to the constraints associated with the spotted owl recovery guidelines, there has essentially been no forest management within this cover type, despite the experiences with the Pumpkin Fire on Kendrick Peak. Therefore, population trends of MSO and elk are likely unaffected by the limited activities that have occurred in this cover type on the South Zone. 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 mixed-conifer cover type on the South Zone. On the NKRD, management entries into the mixed conifer/spruce-fir cover type that follow the Amended Forest Plan should have benefited red squirrels by diversifying the forest structure while retaining the large tree component.

Snags in Coniferous Forests

The 1987 Forest Plan called for retaining not less than 200 snags per 100 acres across 65percent 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 was modified in 1996 to retaining two snags per acre on all acres within the ponderosa pine cover type. In mixed-conifer habitat, the Amended Forest Plan calls for three snags per acre and in general, snags are to be left in the pinyon-juniper cover type. Snags were also redefined in the 1996 Amendment to equal 18 inches dbh and greater than or equal to 30 feet in height. When the plan was revised in 1996, snag prescriptions became much more site specific: the goshawk guidelines, which cover most of forested areas on the KNF, call for two snags per acre; 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 on the NKRD, "substantive" amounts of snags are to be retained. In practice, this becomes an effort to avoid losing snags due to management activity. In the remaining ponderosa pine forest, the Recovery Plan calls for one snag per acre of at least 14 inches dbh, and 15 or 25 feet in height minimum, depending on site class conditions.

The USFS Forest Inventory Assessment (FIA) found 0.6 ponderosa pine snags per acre that were 19 inches dbh and larger across Arizona forests (O'Brien 2002). Several snag studies have been done 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. Stand examination surveys on the KNF that included snags 18 inches dbh or larger have recorded means of: 2.1 snags per acre for the Williams District; 0.6 snags per acre on the Tusayan District; and the NKRD averages 1.6 snags per acre.

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 influence the distribution of snags. Because the creation of snags typically occurs in a nonrandom pattern, differences in snag densities reported in the studies above are at least partly due to differences in sampling methods. Retention of snags on the landscape is also variable. Areas protected from wind or in areas with good soil development can promote snag longevity because they are typically less prone to wind throw. In addition, 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. Nevertheless, the Forest Plan, in general, calls for two or more large snags on every acre of forest.

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 percent 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). 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 percent of the snags had fallen and 8 percent were missing (stumps associated with eight snags indicate they were cut for firewood) (Dr. Carol Chambers, personal communication, 2002). Standing snags averaged 20 inches dbh and the percent scorch (amount of bark burned) averaged 63percent (range 0 to 100percent).

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 “defect” 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 some of the snag habitat for cavity nesting birds (Dr. William Block, personal communication, 2002, Dr. Carol Chambers, personal communication, 2002, and Ganey 1999). 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 rather than deliberately harvesting them should directly benefit cavity-nesting birds.

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. Ganey (1999) reported mean numbers of large snags in ponderosa pine forests were well below the KNF target density. The 0.6 to 2.1 snags per acre estimates reported above (Miller and Benedict 1994, Ganey 1999, and the KNF), combined with the habitat represented by partial snags, suggests the NKRD is well above historic levels of large snags (Figure 30). There is concern whether large snags will remain available in the long-term, given the past years of harvest levels above the rate of recruitment into the large size-classes and the long-term effects of crown fires. In the short term, the combination of management direction under the Amended Forest Plan, unsurveyed habitat structure (i.e., partial snags), and the inter-related results of fire, drought, insects, and disease should maintain snags well above historic levels within the coniferous forest vegetation types.

Hairy woodpeckers, plain titmice, and yellow-bellied sapsuckers were selected to represent the snag component in the different coniferous forest cover types. In addition, pygmy nuthatches, a MIS selected to represent late-seral ponderosa pine, use snags for communal as roosts.

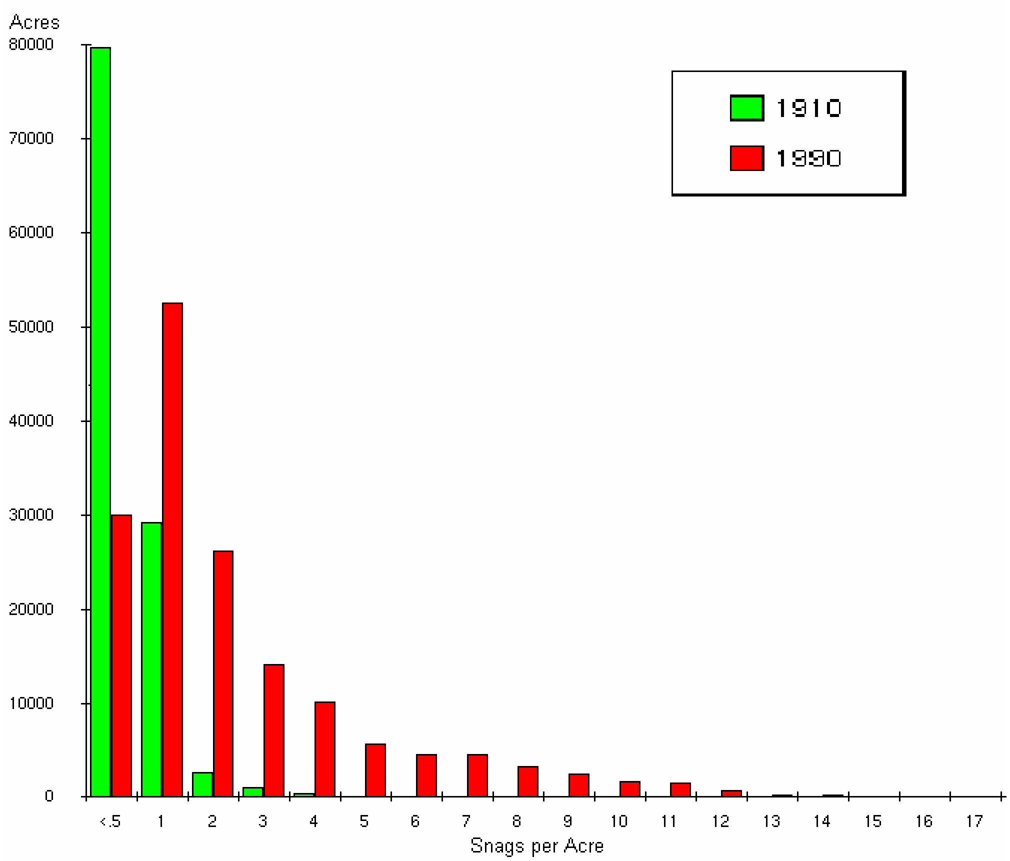


Figure 30. Snag densities from 1910 and 1990, North Kaibab Ranger District. The numbers for 1910 are inferred from inventoried snag biomass translated to trees per acre using stand tables of trees per acre by size class included in the 1910 report. Multiple snags are more prevalent across 130,000 acres of the NKRD now than in 1910.

Aspen Cover Type

Aspen (*Populus tremuloides*) dominated stands are a minor component of Arizona forests and is declining in much of its western range (O'Brien 2002). Aspen reproduce primarily by sprouting from the parent root system and often require a disturbance to stimulate suckering (Bartos 2001). It appears that aspen root systems can persist for thousands of years and that remnant aspen in an area indicates the site may have once been dominated by aspen (O'Brien 2002). In the absence of disturbance, aspen is usually successional to conifers. Aspen is considered mature when over 80 years old (Hart and Hart 2001) and clones can dominate a site for about a century. Conifers typically succeed aspen between 100 to 200 years although, in mixed stands, conifers can become established within a decade (reviewed in Dahms and Geils 1997).

About 95percent of the aspen-dominated lands in Arizona have succeeded to conifers since European settlement (Bartos 2001). There has been a 46 percent decline in aspen across the Southwest between 1962 and 1986 (Dahms and Geils 1997). Fire suppression, unchecked forest succession, and ungulate grazing have all been contributing factors to the decline (Bartos 2001, Rolf 2001). The habitat diversity represented by aspen in Southwestern forests makes them important to wildlife. Loss of aspen represents a loss of diversity in the forest that may be affecting avifauna and invertebrates, including pollinators like lepidopterans (Chong et al. 2001, Rumble et al. 2001, Struempf and Hayward 2001).

Aspen stands are common across the Williams (1,658 acres) and North Kaibab (25,229 acres) Ranger Districts, but is only present as a single clone on the Tusayan District (10 acres or less). 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. Early-seral aspen are less common on the Williams RD where regeneration is hampered by elk browsing. Elk can cause mortality directly through herbivory and through mechanical injuries to the boles and stems that allow invasion by pathogenic fungi (Hart and Hart 2001). Aspen is very susceptible to gnawing or having the bark stripped by ungulates. Aspen are highly resistant to fungi pathogens until wounds alter the trees condition (Hart and Hart 2001). Repeated fire probably contributed to wounds that facilitated the infection of live trees. This aspect of forest structure may be more limited now than it was historically due to fire suppression. In general, aspen continues to decline across the Forest. Aspen is being overtopped by conifers and regeneration is limited by fire suppression and ungulate grazing. Regeneration is more successful on the NKRD, although if current trends continue, the future status of aspen is expected to be as problematic there as it currently is on the Williams RD (Dr. W. D. Shepperd, personal communication, 1997). Smith et al. (1972), as reviewed in Hart and Hart (2001), reported that deer alone had little effect on the development of aspen reproduction, but when deer were present with cattle, aspen regeneration was dramatically impacted.

Some work has been done to improve aspen regeneration, but these efforts have been limited in scope and success. 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). Conversely, many of the "checkerboard" even-aged cutting units on the NKRD from the 1970s have regrown as aspen. More work is needed to better document the current distribution and conditions of aspen clones on the KNF so the most important areas for restoration treatments can be identified.

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 (reviewed in Hart and Hart 2001). Snags are being created as older aspen die, but the potential for future large dbh aspen snags is limited. A review of primary cavity nesters in aspen forests found that 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 percent of all cavity trees, but were present on only 9.6 percent of the total aspen examined (Hart and Hart 2001). They concluded that living trees with heart rot may be more critical than aspen snags. They measured 45 trees containing 73 cavities that averaged about 11 inches dbh (Hart and Hart 2001). Tables 25 and 26 display available living and dead aspen on the KNF.

Table 25. Number of live aspen trees per acre, Kaibab National Forest.

	Aspen							
	>=18"		>=12"		>=10"		.1-9.9"	
Ranger District	**TPA**	**SDI**	**TPA**	**SDI**	**TPA**	**SDI**	**TPA**	**SDI**
Williams	2.17	6.18	66.61	121.44	83.35	140.85	893.57	76.78
Chalender[1]	1.44	4.41	19.06	32.22	36.00	50.39	1369.46	162.46
North Kaibab	0.98	3.03	15.31	27.18	31.89	46.08	1395.84	65.07

[1]The Chalender District is now combined with the Williams Ranger District.

Table 26. Number of dead aspen trees per acre, Kaibab National Forest.

	Aspen							
	>=18"		>=12"		>=10"		.1-9.9"	
Ranger District	**TPA**	**SDI**	**TPA**	**SDI**	**TPA**	**SDI**	**TPA**	**SDI**
Williams	0.19	0.49	2.04	3.88	3.54	5.49	14.26	9.84
Chalender[1]	0.00	0.00	1.75	3.01	1.75	3.01	27.50	9.04
North Kaibab	0.34	1.00	3.68	6.28	7.20	9.92	22.68	12.30

[1]The Chalender District is now combined with the Williams Ranger District.

The habitat diversity associated with aspen in Southwestern forests makes them an important component of wildlife habitat. There has been a 46 percent decline in aspen across the Southwest between 1962 and 1986 (Dahms and Geils 1997). Fire suppression, leading to unchecked forest succession, and ungulate grazing all contribute to the decline (Bartos 2001, Rolf 2001). Aspen decay and topple much quicker than coniferous snags and firewood gathering also contributes to the loss of snags. The potential for future large dbh aspen snags is limited, but retaining living trees with heart rot may be more critical than retaining existing aspen snags. Aspen habitat is treated/protected opportunistically, with the goal stimulating suckering and removing over-topping conifers. Nevertheless, aspen continues to be overtopped by conifers and regeneration is limited by fire suppression and ungulate grazing. Some fencing has been done on the KNF to protect aspen, primarily on the Williams RD, but the acres treated are very limited, likely totaling 50 acres or less. In general, aspen continue to decline across the Forest.

Red-naped sapsuckers and mule deer were selected as MIS to represent aspen. Red-naped sapsuckers were selected to represent late-seral aspen and aspen snags while mule deer were selected to represent early-seral stage aspen habitat. Mule deer are such generalists and the NKRD consists of good deer habitat, that it is unlikely whether the gradual changes in the condition of aspen could lead to detectable changes in deer numbers, i.e., mule deer may be a poor MIS for early seral-stage aspen. Conversely, red-naped sapsuckers are tied closely to aspen habitat. Habitat for red-naped sapsuckers is expected to remain static in the short-term and decrease in the long-term.

Grasslands

The 1987 Forest Plan assessed acreage by the potential vegetation that could grow on a given site/soil type. Many grasslands are seral stages that, if the disturbance cycle is interrupted, can support different communities. In 1987, it was estimated that about 72,000 acres should be classified as grassland and the remaining acreage, i.e., acreage that currently supported grassland vegetation, was termed suitable for supporting merchantable timber. However, the grassland ecosystem evolved with a frequent fire return interval that, in the absence of active management, prevented trees from achieving dominance over the grass and forb species. Today there are over 249,000 acres of grassland habitat on the KNF (see Table 15).

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 has 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). However, grasslands, savannahs, and mountain meadows border every forest type on the KNF.

Grazing by domestic ungulates in the Southwest began with Spanish colonization (Ronco 1987). A Spanish expedition headed by Villalobos came through Arizona in 1521 with 1,000 people, 1,000 horses, 500 cattle, and 5,000 sheep; escaped and abandoned animals from these herds survived to begin stocking the country (Dr. E. E. Willard, University of Montana, personal communication). Much of the Nation's attention shifted to the West after the Civil War and by the 1890s, close to a million cattle were being grazed in Arizona. Drought conditions between 1891 and 1894 led to the starvation and death of about 250,000 head of cattle (Dr. E. E. Willard, University of Montana, personal communication). Impacts from overgrazing, logging, and fire suppression practices that started in the late1800s are still discernible today (Covington and Moore 1992, Eddleman 1987). Heavy grazing pressure continued into the 1900s. Across the Southwest, grazing in the pinyon-juniper habitat was estimated to have caused a 60 to nearly 100 percent decrease in available forage (Stoddart et. al. 1975). Overgrazing was being addressed by the KNF in the 1950s, but substantial progress was not attained until the 1970s.

In addition to livestock grazing, wildlife managers have consistently strived to increase deer and elk numbers. The indigenous elk of Arizona are classified as the now extinct Merriam subspecies. The decline in elk numbers may have started as early as the 1500s, but Merriam elk were still present at the start of the twentieth century (Bryant and Maser 1982). Arizona was considered the Merriam elk's last stronghold, but excessive hunting and overgrazing by domestic livestock led to their extinction by about 1906. In 1913, eighty-four Rocky Mountain elk were introduced on the Sitgreaves National Forest and, by the 1970s stable populations of about 12,500 animals were reported for Arizona (Bryant and Maser 1982). In 1995 alone, 10,000 elk were harvested statewide in Arizona (Dahms and Geils 1997). Elk densities in northern Arizona are thought to be higher today than they were historically due to increased water developments and grass seeding as part of pasture improvements (Chambers and Holthausen 2000). The grasslands on the South Zone support elk that are managed by the Arizona Game and Fish Department. The herds in these management units (Figures 15) provide a world-class hunting opportunity, including the first, second, and third place world archery records according to Pope and Young (Williams-Grand Canyon News 2002). Elk numbers on the KNF peaked in the 1990s when numbers for the herds that migrate across the Coconino NF and the South Zone of the KNF varied between 7,000 and about 9,500 animals (see Table 9).

Past grazing pressure reduced or eliminated the vegetation necessary to carry low intensity ground fires and developing transportation networks dissected the landscape. These actions contributed to a change in the size of the average fire and an interruption in the fire frequency (Covington and Moore 1994). Fire suppression, combined with grazing pressure, has resulted in a decrease in the overall diversity of grasses and forbs and allowed increases in the introduction and establishment of woody and exotic species (Dahms and Geils 1997).

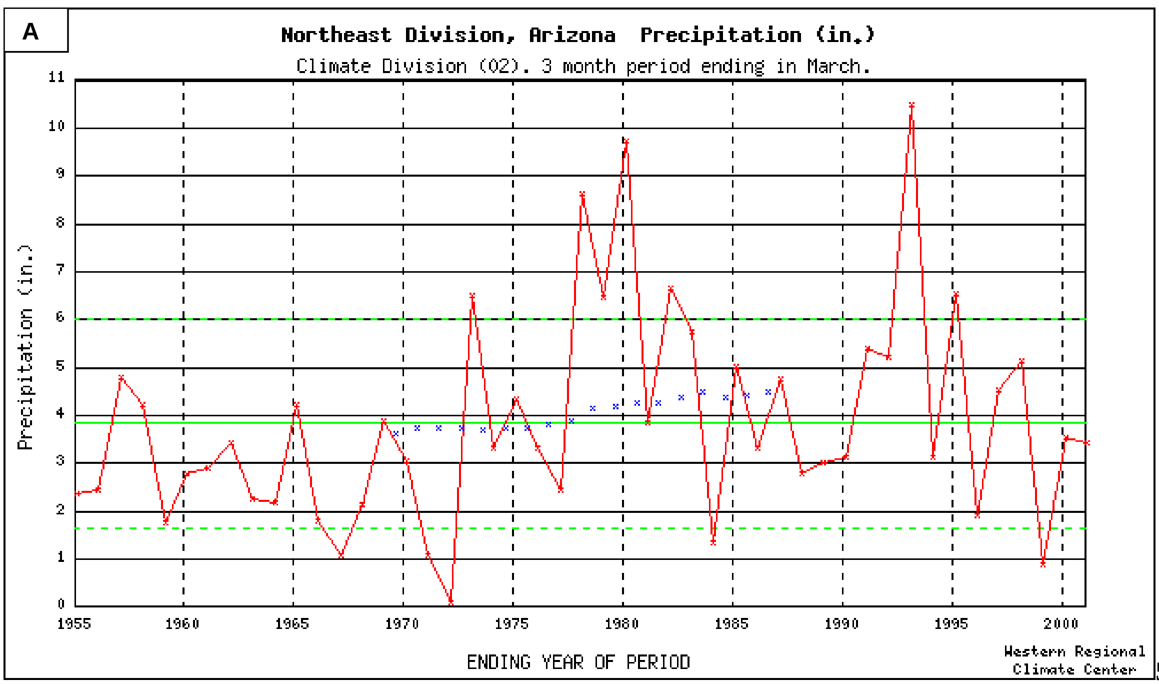
Overstory development affects understory cover through multiple pathways. In summary, the issues include: the ability of trees to use physical resources more efficiently than grasses or forbs; once established, pinyon and juniper have allelopathic characteristics that inhibit grass and forb growth; trees slow nutrient cycling by tying-up 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). 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 percent (Arnold et. al. 1964 *in* Doughty 1987).

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. Unlike the pseudoreplication issues with stand examination surveys (see "Introduction" above), trends in grassland conditions can be estimated from Parker three-step surveys because the same points were re-measured over time. However, because the Parker three-step survey technique was developed to sample range conditions, not ecological conditions, there are inherent biases, e.g., areas that are sampled within groups of trees have a lower ranking, regardless of the overall health of the stand, because the overstory will, by definition, reduce forage availability for grazing. Similarly, in heterogeneous habitat, an adequate number of samples need to be randomly located to achieve an adequate sample. However, with the Parker method, key areas are identified where livestock are expected to focus their foraging efforts. Key areas are not randomly placed and they do not necessarily represent the overall ecological conditions of the area (U.S.D.A. 1988). The focus of the Parker method is to eventually determine the range resource value ranking which may or may not correlate with ecological conditions.

The emphasis on forage value to livestock limits the interpretation of data generated from the Parker three-step survey, making it less suited for use as an ecologically based technique. In evaluating allotment conditions with the Parker system, perennial plant species are classified as increasers, decreasers, or invaders, according to the manner in which they respond to grazing. Increasers have lower palatability and are released by grazing. Decreasers are typically the most palatable plants and tend to be the first to respond to overgrazing. Invaders have little or no forage value and frequently include poisonous or noxious species. It is important to note that improvements in range conditions do not necessarily mean a return to historic conditions. For example, Parker transect data indicate that blue grama is a plant that has commonly increased over time. An increase in the sod forming blue grama could reduce the amount of bare ground, thereby breaking-up any vesicular soil crusts, improving moisture penetration and retention, and stabilizing and rebuilding the soil. However, blue grama is low growing and will not provide hiding cover for medium or large mammals, such as pronghorn antelope fawns, and may prevent other native species from becoming reestablished. Under this example, the ecology of the system is still out of balance even while positive improvements have occurred. Nevertheless, the history of overgrazing in Arizona is such that improvements in range conditions should, in general, benefit many wildlife species. A strength of the Parker three-step surveys is that the National Forests have been sampling the same locations using the same methodology for nearly 50 years.

Another method of classifying vegetation is by their season of primary growth. In Arizona, most precipitation occurs in winter and summer; spring (April through June) is the driest time of year (Fletcher and Hickey 1981). Plants can be grouped as: cool season, or those plants whose primary growth occurs in autumn, early winter, or early spring; warm season plants that produce most of their growth in late spring and summer; or generalists that can grow in either cool or warm seasons. Cool season plants tend to flower before or during June. Warm season plants may still be in bloom in or after September (Fletcher and Hickey 1981). By producing green forage during the annual dry season, cool season plants are the most vulnerable to overgrazing. Cool season grasses are frequently decreasers and warm season grasses are generally increasers. A primary goal of range management in the Southwest is to enhance production of any perennial grass capable of providing spring forage (Fletcher and Hickey 1981). Grazing pressure has, in large part, affected species composition and frequency values through reductions of cool season grasses. Overgrazing cool season plants can: reduce production later in the season; delay growth initiation the following spring, thereby amplifying drought effects; and decrease summer forage production (Fletcher and Hickey 1981).

A major influence on vegetative conditions and general ecosystem health are weather and climatic conditions. Precipitation patterns in the Southwest form the canvas upon which management activities are painted. Weather patterns can mask or emphasize management impacts, making interpretations of trend data even more speculative. Typically, precipitation patterns in Arizona include winter rain and snows and a summer “monsoon” season. Spring and fall are generally dry seasons. Drought conditions only need to occur during one of the typically wet seasons to affect plant growth and development for an array of species that may not be able to respond even if the next wet season meets or exceeds average precipitation levels. Winter precipitation patterns since the 1950s show much drier than normal years in the 1950s and 60s (Figure 31A). The early 1970s, late 1980s, and the mid-1990s through 2002 were also drier than average. Conversely, the mid-1970s through the mid-1980s and the early 1990s were years of abundant winter precipitation. Summer monsoon rainfall was generally at or below average from the 1950s through 1980. Higher than average summer rainfall occurred in the 1980s through the early 1990s and again in the late 1990s. A comparison of Figures 31A and 31B can reveal years where one season received above normal precipitation while the other season during that same year was relatively dry. This dichotomy in seasons is presented again in Figure 31C, which displays the averaged accumulated departure from normal precipitation. For example, over the last three years winter precipitation has been low while summer rains have been up, but the averaged accumulated departure is over 10 inches below normal. The interrelationship between weather, climate, and plant growth patterns adds uncertainly to interpreting grassland trends and conditions.



51A.

Figure 31A. Winter precipitation (total precipitation for the 3-month period ending in March), 1955 to 2001: Red = 3 month period; blue = 30 year running mean; green = average (solid), ± sigma (dashed). Average = 3.83; sigma = 2.17; coefficient of variation = 0.57; skewness = 1.05; median = 3.30; maximum value = 10.45; minimum value = 0.07.

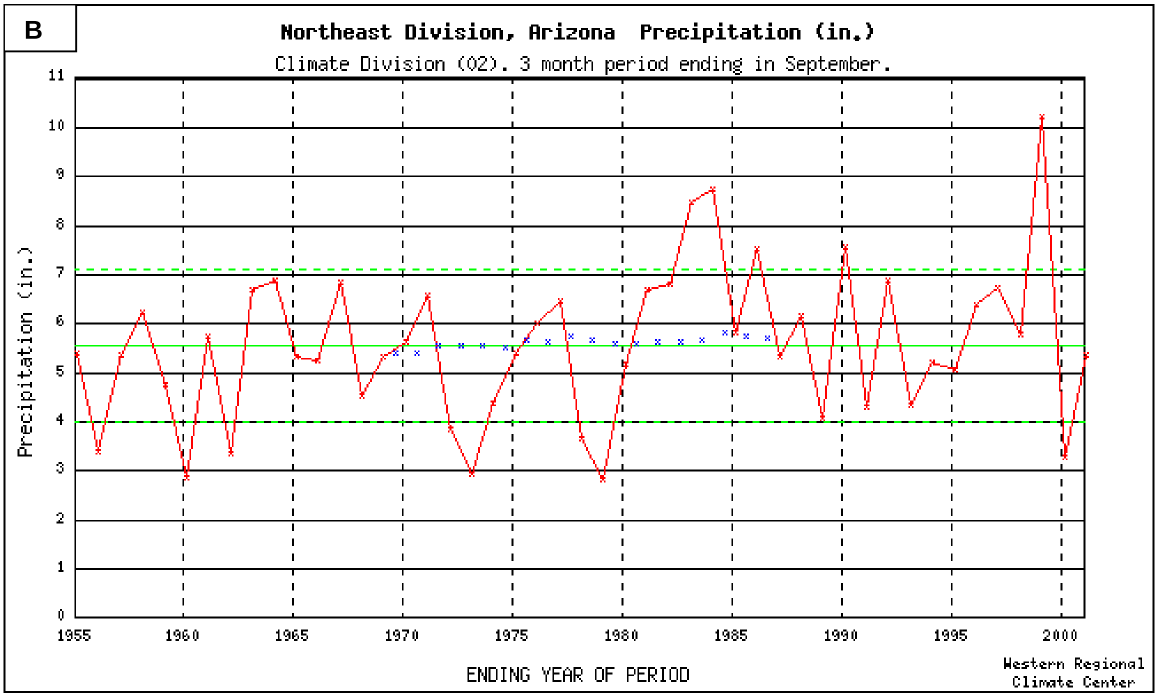


Figure 31B. "Monsoon precipitation" (total precipitation for the 3-month period ending in September), 1955 to 2001: Red = 3 month period; blue = 30 year running mean; green = average (solid), ± sigma (dashed). Average precipitation = 5.55 (Sigma =.565; Coefficent of Variation =.28; Skewness = 0.41; Median = 5.38; Maximum = 10.21; Minimum = 2.81).

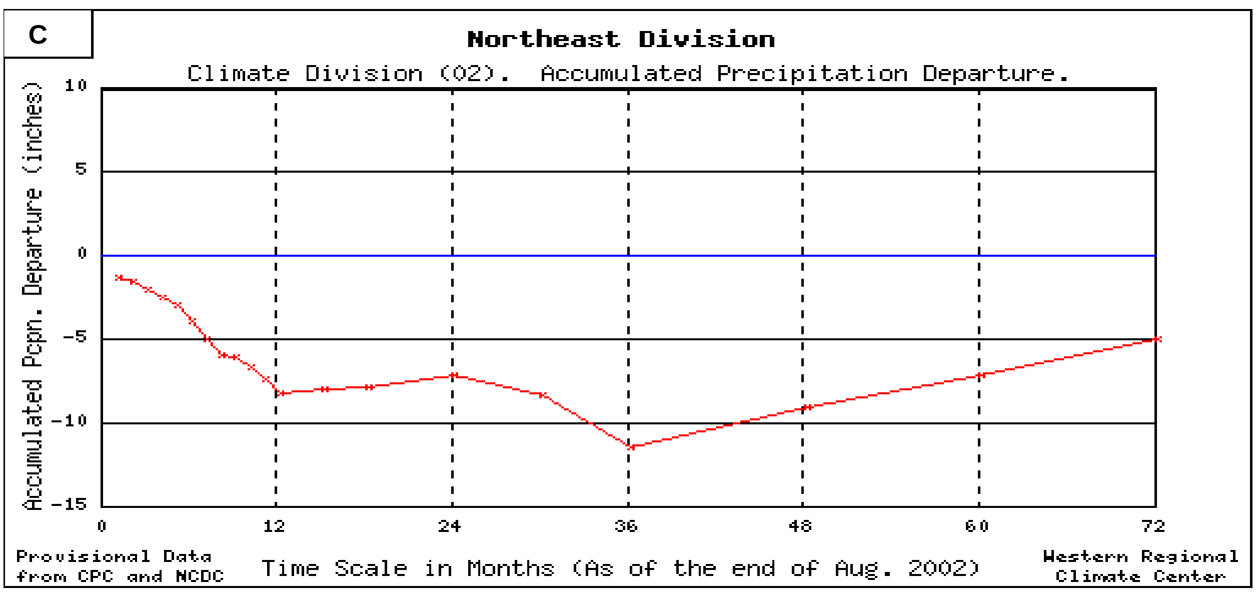


Figure 31C. Accumulated Precipitation Departure from Normal: Each point on the graph represents the difference of 'total accumulated precipitation' minus 'normal accumulated precipitation' for the climate division for the number of months shown on the X-axis, and ending at the end of the indicated month. For example, the value at '12' shows the total departure over the past 12 months from the average sum over all similar 12-month periods since 1895.

Figure 31. Historical precipitation summaries from the Western Regional Climate Center, administered by the National Oceanic and Atmospheric Administration Data oversight is provided by the National Climatic Data Center, a part of the National Environmental Satellite, Data, and Information Service. Graphs were obtained from www.wrcc.dri.edu.

Management Actions

There are 41 grazing allotments on the KNF. Summary information was available for allowable stocking levels from 1971, 1983, and 2002 and is presented below (Table 27). The maximum allowable stocking level has declined throughout this period. Today's stocking levels represent a 34 percent decrease in obligated animal unit months over the last 31 years. During that time, production-utilization surveys led to a decrease in allowable use on 28 allotments; increases in stocking rates on eight allotments; and five allotments have remained unchanged.

Table 27. Changes in the total number of obligated Animal Unit Months by Ranger District, Kaibab National Forest.

Ranger District	Animal Unit Months		
	1971	1983	2002
Williams	61,295	41,255	38,895
North Kaibab	16,720	14,590	12,265
Tusayan	30,530	30,530	20,245
Forest Total	108,545	86,375	71,405

There was a concurrent increase in range restoration as stocking numbers were being lowered in the 1970s and 1980s. Between 1978 and 1983, about $1.5 million were invested in range management improvements on the Williams RD. Range improvements included: over 21,000 acres of pinyon-juniper control on grassland (mollisol) soils; over 10,000 acres seeded to grass; about 120 miles of fencing installed; the construction of cattleguards and water tanks; and about 1,000 acres of fuelwood sales. Although this work was accomplished before the Forest Plan, the ramifications of some of this work affects the data examined below both positively (e.g., an increase in grass cover and grass species) and negatively (e.g., re-growth of pinyon and juniper on grassland soils that did not receive subsequent maintenance).

Parker surveys are laid out in a variety of habitat types in order to monitor forage conditions across the allotments. Some transect clusters were placed in forest openings where forage conditions indicated heavier grazing pressure could potentially occur. Only Parker surveys that were conducted in areas thought to represent historic grasslands were selected while evaluating grassland conditions on the KNF. This selection was based on soil types and was done to try to limit noise in the data that could be introduced from comparing vegetation recorded from surveys done in grassland communities and mixing in vegetation surveys from pine forest associations (Table 28).

Table 28. Soil types from the Terrestrial Ecosystem Survey Map Units used to classify grasslands on the Kaibab National Forest.

Terrestrial Ecosystem Number	Soil Classification	Terrestrial Ecosystem	Acres
3	Fluventic Ustochrepts, fine-loamy, mixed, mesic	Fourwing saltbush, Western wheatgrass, Pinyon pine	4,265
5	Pachic Udic Argiborolls, fine-loamy, mixed	Kentucky bluegrass, Sheep Fescue, Nodding brome	2,710
6	Pachic Argiborolls, fine, montmorillonitic	Kentucky bluegrass, Arizona fescue	9,190

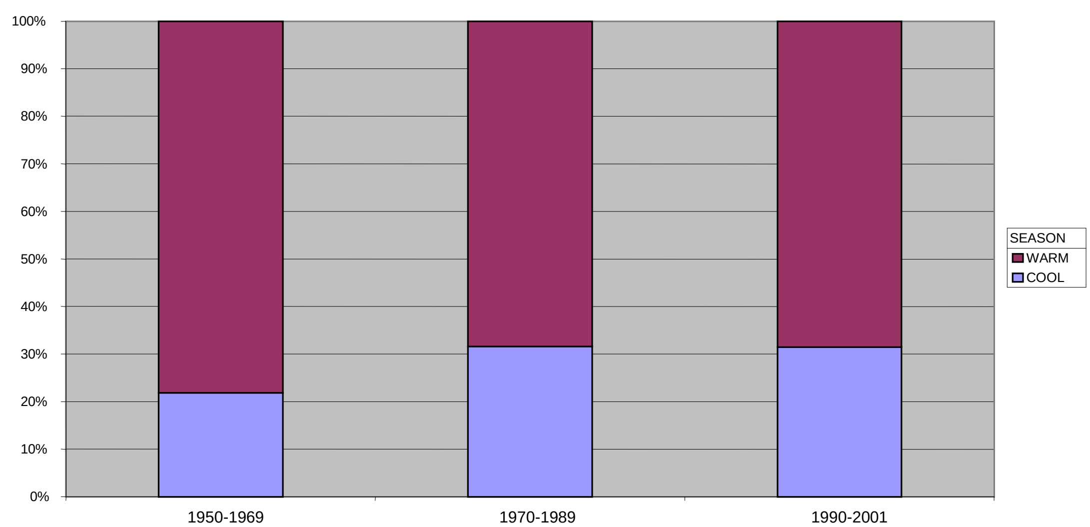
7	Cumulic Haplustolls, fine-loamy, mixed	Western wheatgrass, Pinyon pine	3,635
9	Cumulic Haploborolls, fine-loamy, mixed Cumulic Haploborolls, loamy-skeletal, mixed	Kentucky bluegrass, Western wheatgrass, Ponderosa pine	3,620
11	Cumulic Haploborolls, fine- loamy-mixed	Kentucky bluegrass, Mountain muhly	1,530
20	Vertic Haplaquolls, very fine, montmorillonitic, frigid	Carex, Spike rush, Solomonseal	2,230
23	Fluventic Ustochrepts, fine-loamy, mixed, mesic Fluventic Ustochrepts, loamy-skeletal, mixed, mesic	Big sagebrush, Blue grama, Pinyon pine	10,190
32	Fluventic Ustochrepts, fine-loamy, mixed, mesic Fluventic Ustochrepts, loamy-skeletal, mixed, mesic	Pinyon pine Utah juniper, Big sagebrush	4,055
35	Argic Cryaquolls, fine-loamy, mixed Argiaquic Cryoborolls, fine, mixed	Carex, Spike rush	610
36	Pachic Argiustolls, fine, mixed, mesic	Rubber rabbittbrush, Western wheatgrass, Pinyon pine	4,395
37	Aquic Haploborolls, loamy-skeletal, mixed	Kentucky bluegrass, Carex Arizona fescue	3,130
41	Typic Argiustolls, clayey-skeletal, montmorillonitic, mesic	Western wheatgrass Pinyon pine	1.660
251	Lithic Ustochrepts, loamy-skeletal, mixed, mesic	Pinyon pine, Utah juniper, Big Sagebrush, Neddle-and-Thread	27,225
255	Lithic Ustochrepts, loamy-skeletal, mixed, mesic	Fourwing saltbush, Needle-and-Thread Blue grama, Pinyon pine	8,890
507	Vertic Argiborolls, fine, montmorillonitic Vertic Argiborolls, clayey-skeletal, montmorillonitic	Rubber rabbittbrush, Arizona fescue Blue grama	31,170
513	Typic Argiborolls, clayey-skeletal, montmorillonitic Pachic Argiborolls, fine, montmorillonitic	Arizona fescue, Mountain muhly	27,675
514	Vertic Argiborolls, fine, montmorillonitic Vertic Argiborolls, clayey-skeletal, montmorillonitic	Rubber Rabbittbrush, Galleta Pinyon pine	31,175
518	Lithic Argiborolls, clayey-skeletal, montmorillonitic Pachic Argiborolls, fine, montmorillonitic	Arizona fescue, Mountain muhly	5,740
542	Vertic Argiborolls, fine, montmorillonitic Udic Chromusterts, fine, montmorillonitic	Rubber Rabbittbrush, Blue grama Pinyon pine	3,770
591	Petrocalcic Calciustolls, loamy, carbonatic, mesic Typic Calciustolls, fine-loamy, carbonatic, mesic	Fourwing saltbush, Needle-and-Thread Blue grama, Pinyon pine	7,920

592	Typic Calciustolls, fine loamy, carbonatic, mesic Petrocalcic Calciustolls, loamy, carbonatic, mesic	Pinyon pine, One-seed juniper Needle-and-Thread	14,260
599	Typic Argiustolls, fine, montmorillonitic, mesic Typic Argiustolls, fine-loamy, mixed, mesic	Blue grama, Pinyon pine	14,760
630	Lithic Eutroboralfs, clayey-skeletal, mixed Mollic Eutroboralfs, clayey-skeletal, montmorillonitic	Arizona fescue, Mountain muhly	1,115
632	Lithic Ustochrepts, loamy-skeletal, carbonatic, mesic Aridic Ustochrepts, loamy-skeletal, carbonatic, mesic	Fourwing saltbush, Blue grama Needle-and-Thread	14,095
634	Typic Ustochrepts, loamy-skeletal, carbonatic, mesic Lithic Ustochrepts, loamy-skeletal, carbonatic, mesic	Big sagebrush, Creasted wheatgrass Needle-and-Thread, Pinyon pine	14,775
636	Aridic Ustochrepts, loamy-skeletal, carbonatic, mesic Aridic Ustochrepts, fine-loamy, carbonatic, mesic	Fourwing saltbush, Blue grama Winterfat	6,290
637	Lithic Ustochrepts, loamy-skeletal, carbonatic, mesic	Fourwing saltbush, Blue grama Winterfat	6,330
642	Typic Eutrochrepts, loamy-skeletal, mixed, frigid Lithic Eutrochrepts, loamy-skeletal, mixed, frigid	Sheep Fescue, Nodding brome Mountain muhly	550
655	Argic Cryoborolls, fine-loamy, mixed	Sheep Fescue, OatgrassMountain muhly	890
672	Typic Haplustalfs, clayey-skeletal, montmorillonitic, mesic Typic Haplustalfs, fine, montmorillonitic, mesic	Big sagebrush, Crested wheatgrass Gambel oak	4,990
677	Lithic Ustochrepts, loamy-skeletal, carbonatic, mesic Typic Ustochrepts, loamy-skeletal, carbonatic, mesic	Fourwing saltbush, Needle-and-Thread Crested wheatgrass, Pinyon pine	13,065
682	Typic Haplustalfs, fine-loamy, mixed, mesic	Big sagebrush, Crested wheatgrass Gambel oak	3,570
683	Lithic Ustochrepts, loamy-skeletal, carbonatic, mesic Typic Ustochrepts, loamy-skeletal, carbonatic, mesic	Big sagebrush, Crested wheatgrass Needle-and-Thread, Gambel oak	10,420

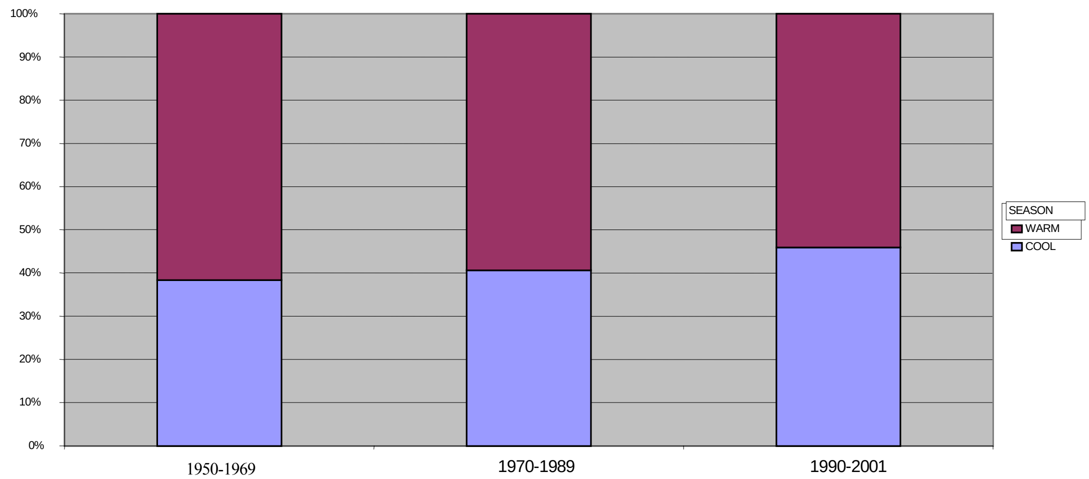
In evaluating changes in grassland conditions based on Parker survey results, transect clusters were selected that had repeated surveys since the 1950s. The KNF Parker three-step survey database is in the process of being converted into an electronic database. The data has not been fully entered nor has it been completely proofed for errors. However, simple queries can be made and the database is expected to be fully operational in fiscal year 2003. Parker surveys include permanent photographic points which will also be part of the electronic database. Parker survey data were grouped into time periods to try to identify potential trends and only the transect clusters that were read in each time period were included. Sample size was too small to group by decade, so 20-year periods were used instead. Unfortunately, the last "20-year period," i.e., the period from 1990 to 2001, constitutes only 12 years time. Further complicating this is the variable precipitation patterns that have been ongoing since the late1980s. Dry seasons, combined with a shorter time period, bias these data, especially when comparisons are made to the years of relatively abundant precipitation in the mid-1970s through the 1980s. Biases not withstanding, the data has yielded some interesting results.

The ratio of cool to warm season grasses increased by about 10 percent between the 1950s and the 1980s (Figure 32). There appears to be little change in the ratio of cool to warm season grasses since the 1980s. When examined by individual Ranger District, the increasing trend continues through 2001 on the NKRD, but appears unchanged on the Williams District. The Tusayan District shows an apparent decrease in cool season grasses during the last 12 years. The pattern in cool season grasses for individual RDs matches the general precipitation pattern across the Forest. Tusayan is the driest and the NKRD the wettest District on the KNF. The drought conditions occurring across north-central Arizona since the early 1990s may have amplified the differences in precipitation between Districts, resulting in the apparent patterns displayed.

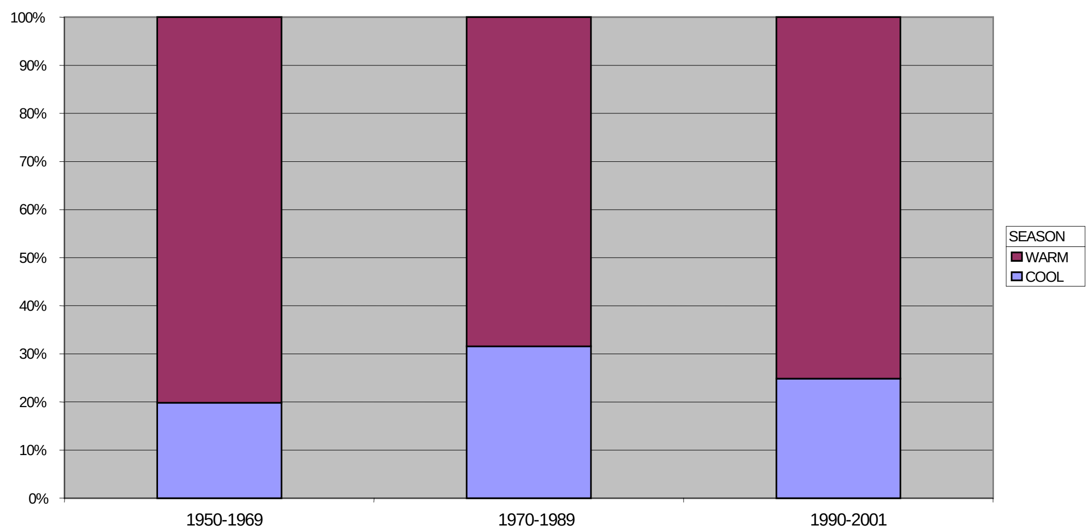
Forest Wide



North Kaibab



Tusayan



Williams

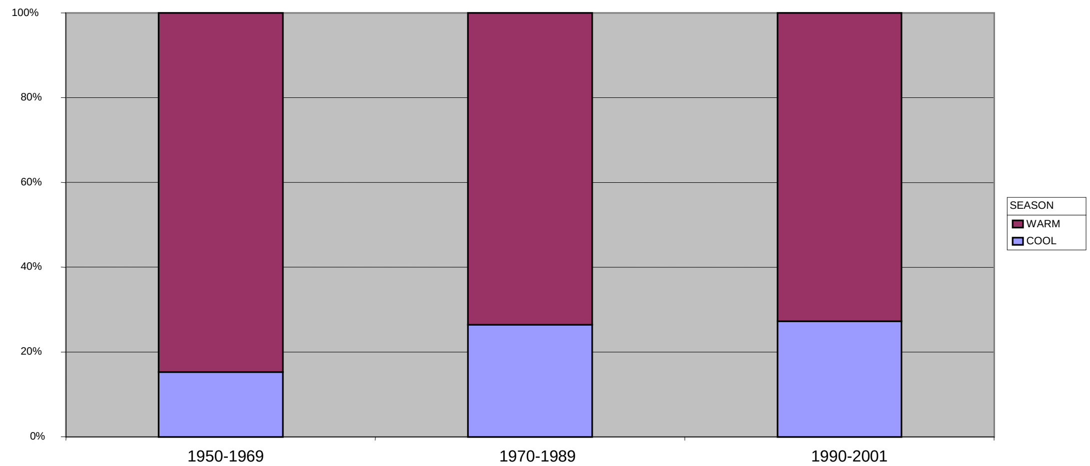
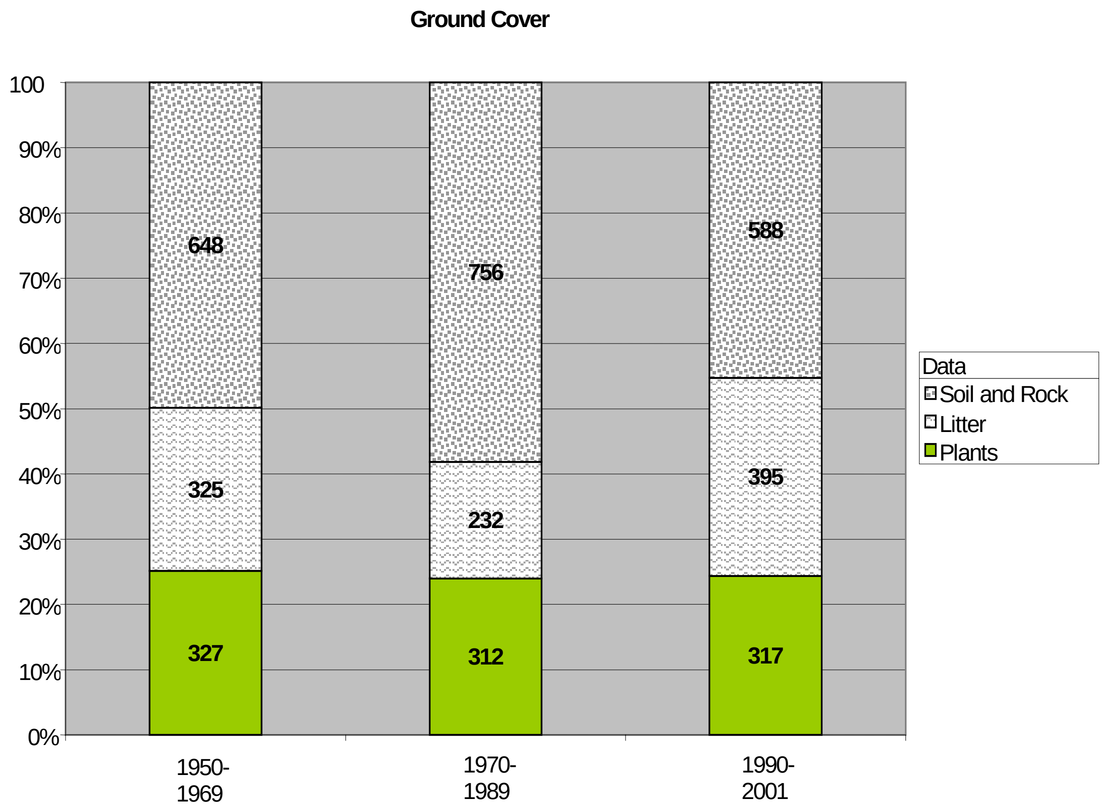


Figure 32. Change in ratio of cool season-warm season grasses by Forest and Ranger District, Kaibab National Forest. Samples represent transects surveyed at least once each time period using the Parker three-step survey method. Decades were combined to increase sample size (n = 93).

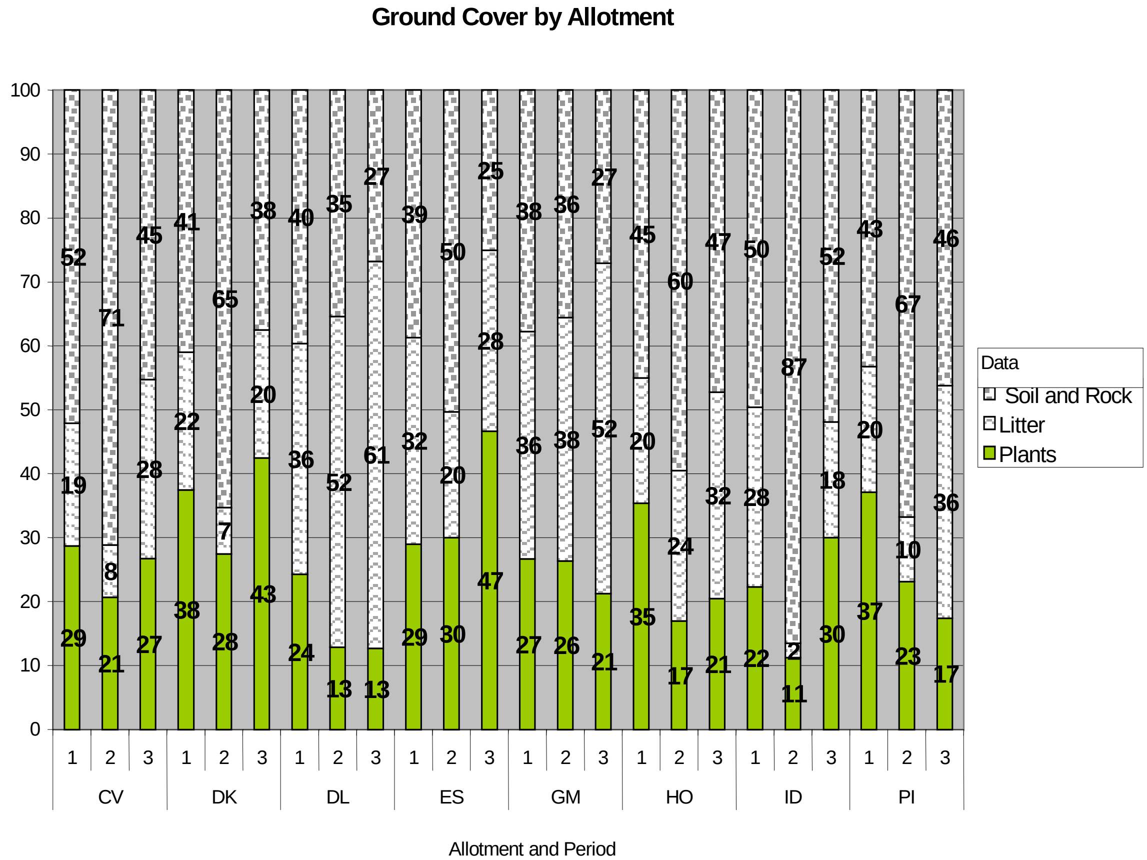
As the vegetation changes, the amount of bare ground and litter also changes. However, when these variables are tracked across the Forest, general patterns become difficult to discern (Figure 33). Forest wide, plant cover essentially remained unchanged from the 1950s/60s through 1990-2001. Despite the relatively constant level of plant cover, litter cover has increased in recent years. The increased litter levels may be the result of the decrease in livestock grazing pressure. Although the decrease in grazing pressure has not released plant development at the Forest scale, this can also be viewed as plant cover not decreasing despite the variable precipitation patterns that have persisted for much of the last decade.

Patterns become more difficult to explain when viewed at finer scales. Only eight allotments on the Williams RD had records from Parker surveys during each time period. Four of the allotments have a “U-shaped” pattern over the three time periods. However, not all “U’s” are created equal: two allotments have about equal levels of plant cover in the 1950s/60s (period 1) and in the 1990s through 2001 (period 3); one allotment has less plant cover in period 3 than in period 1; and one allotment has more plant cover in period 3 than in period 1. Three of the four remaining allotments show a decrease in plant cover since period one and the remaining allotment has an increase in plant cover. Increases in organic cover show no consistent patterns across these allotments. Cover patterns are different for each of the cover types on the two allotments with adequate records on the Tusayan RD. Conversely, each of the three allotments on the NKRD share a similar pattern across cover types. In short, identifying patterns in the development of organic ground cover cannot be accomplished at the RD level due to the wide range of variability between allotments. Assessing ground cover at the allotment level needs to also account for changes in grazing pressure from both domestic and wild ungulates, vegetative management within the allotments, and precipitation patterns that may vary within and between Districts. This level of analysis is beyond the current scope of this document. This report is intended to establish a baseline of Forest conditions for future trend analyses and, once the electronic database for the Parker surveys is completed and more allotments are available for comparison, results should become more meaningful.

KAIBAB NATIONAL FOREST

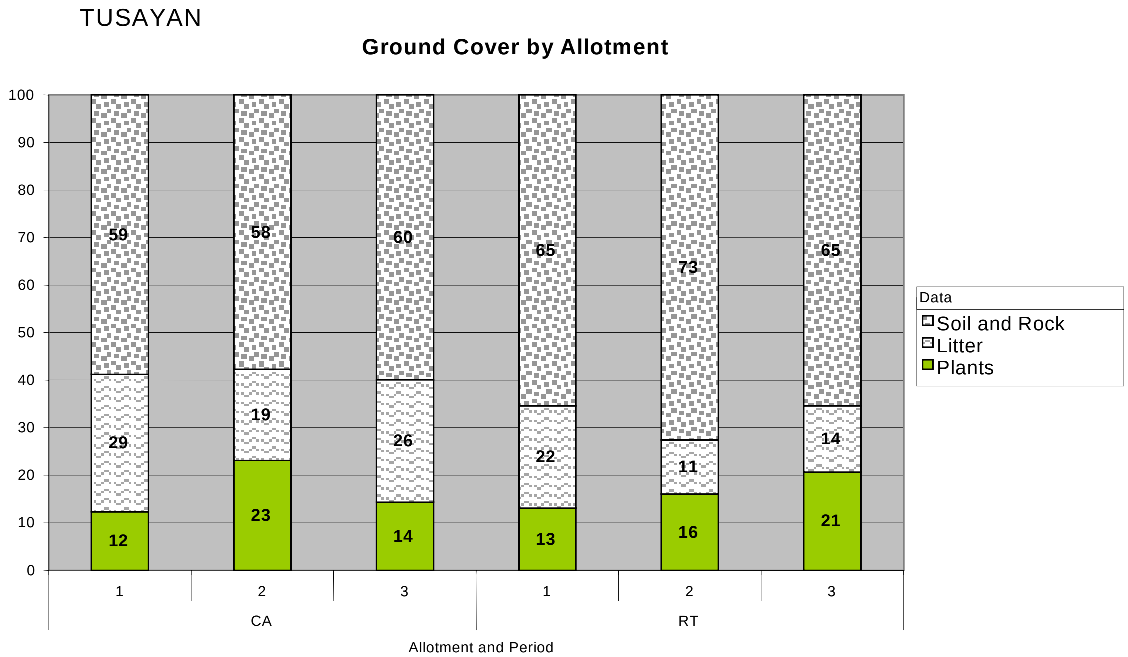


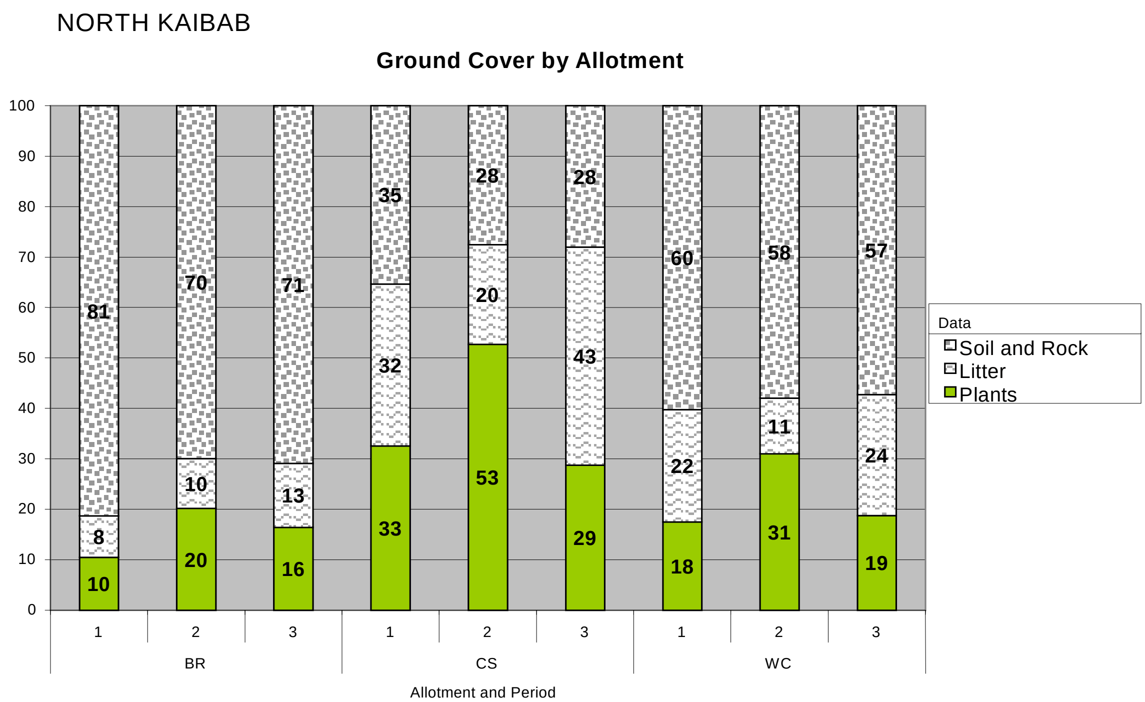
WILLIAMS



A

ALLOTMENTS: CV = CORVA; DK = DOG KNOBS; DL = DAVENPORT LAKE; ES = ELK SPRINGS; GM = GOVERNMENT MOUNTAIN; HO = HOMESTEAD; ID = IRISHMAN DAM; PI = PINE CREEK.
PERIODS: "1" = 1950-1969; "2" = 1970-1989; "3" = 1990-2001.
ALLOTMENTS : CA = CAMERON; RT = RAIN TANK.
PERIODS: "1" = 1950-1969; "2" = 1970-1989; "3" = 1990-2001.





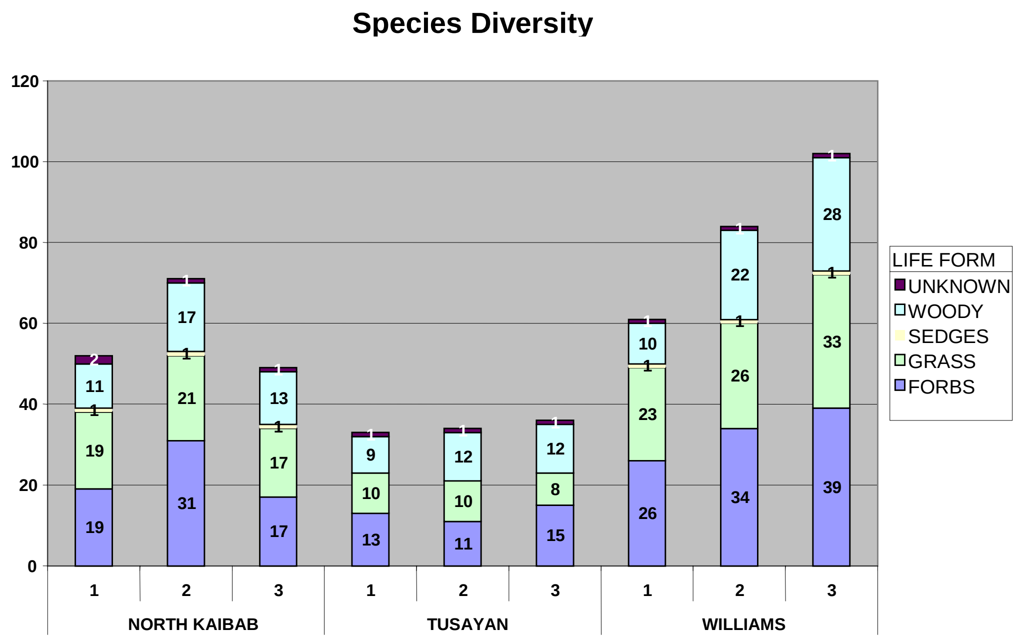
ALLOTMENTS: BR = BUFFALO RANCH; CS = CENTRAL SUMMER; WC = WILLIS CANYON.
PERIODS: "1" = 1950-1969; "2" = 1970-1989; "3" = 1990-2001.

Figure 33. Change in plant, litter, soil, and rock cover by Forest and Ranger District. Numerals in graph represent the number of times each cover type was encountered using the Parker three-step survey method. Samples were scaled to relative percentages. Only allotments that were surveyed each time period are included. Decades were combined to increase sample size (n = 93).

Another aspect of ground cover is the numbers and kinds of plants present. Trends in grass and forb cover are different for each District (Figure 34). We are making the assumption that an increase in grass and forb diversity indicates improved grassland conditions. However, at this point in time we are not identifying and tracking individual species of plants, but instead are merely trying to ascertain whether changes in the overall assemblage of plants have been occurring. Two factors are again suspect when viewing Figure 34: variation between RDs and in precipitation patterns that have likely affected period 3. Nevertheless, it is interesting to note that the numbers of species have increased in each of the vegetative classes.

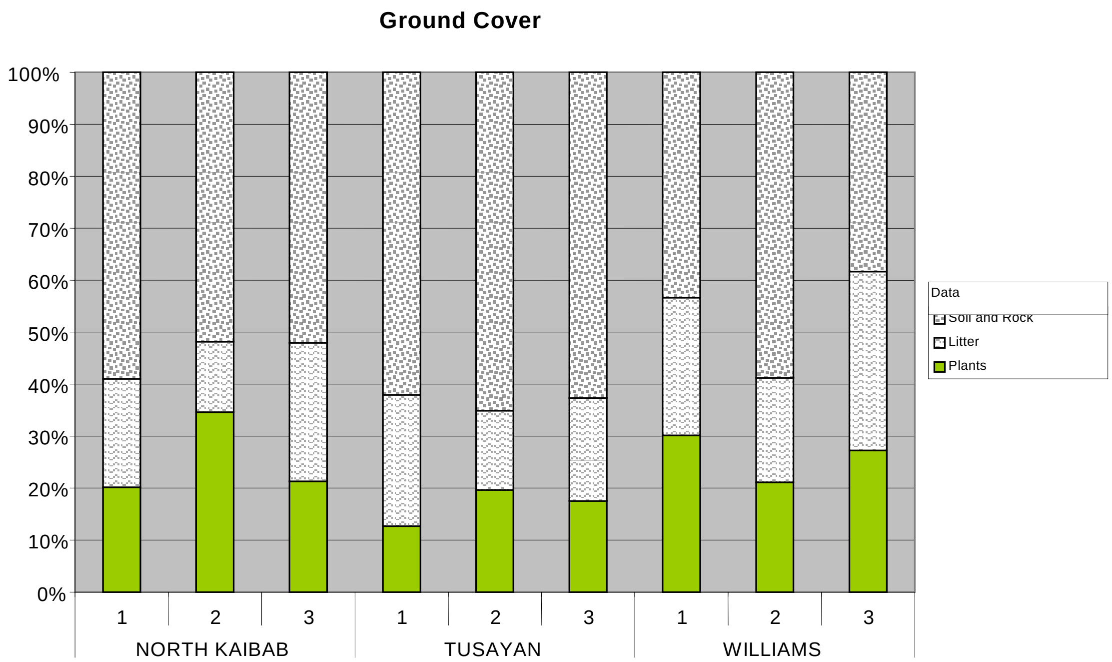
There is an ongoing assumption that woody species have been increasing with time. Figures 34 and 35 suggest that the changes in vegetative cover over time do not support the idea of woody species encroaching into grasslands as a Forestwide phenomenon. We can track the changes in plant types (Figure 34) as compared to changes in total plant cover (Figure 35). Parker survey data for grasslands indicate today's levels of woody vegetation represent a six percent increase on both the Tusayan and NKRDs over the 1950s/60s levels. The increase in total plant cover on the NKRD during period two was relatively evenly spread across each plant form. Although there appears to be an increase in relative and absolute percentages of woody species on the Tusayan RD, the increases in both cases are relatively minor, i.e. the increase in woody vegetation during period 2 is only eight percent of a 14 percent increase in total plant cover. However, it is important to note that averaging across RDs may not present an accurate picture of the situation. If these relatively limited increases in woody species are primarily occurring on certain allotments, than the impact to those grasslands or meadows may be greater than these percentages indicate.

Total plant cover on the Williams RD decreased in period 2, then increased but still remained at a level less than period 1 (Figure 35). The decline in period 2 is likely a result of historic grazing patterns. The increase in plant and litter cover in period 3, although dampened by the decreased precipitation, may result from range improvements accomplished during the late 1970s and early 1980s. Another influence on the species assemblage in period 3 was the decrease in allowable Animal Unit Months that occurred in the 1970s/80s. Although overall plant cover was lower in period 3 than in the 1950s/60s, more species of grasses, forbs, and trees and shrubs were encountered in period 3 than in any other time period. This is interpreted as an overall improvement in range conditions occurring during a period marked by drought. The relative percent of trees and shrubs increased during period 3 even while total plant cover was decreasing, indicating encroachment of grasslands by woody species on the Williams District (Figure 36). Interestingly, the bulk of this increase (10 percent) occurred during period 2, a time when more than 21,000 acres of pinyon-juniper was being pushed by bulldozers. It is important to note that, when evaluating increases in woody vegetation based on Parker transects, the location of the transects were originally laid out to monitor changes in forage conditions. By definition, transects that occurred in forest openings were likely removed from the dataset (i.e., based on soil types, they did not represent grassland or meadow vegetation associations) and it is also unlikely that many, if any, of the Parker transect clusters were located near the forest edge. Therefore, we expect this data represents a negative bias when used for assessing the status of woody vegetation in grasslands. While it does provide a view of changing conditions within the grasslands, it does not evaluate changes at the grassland-forest interface where we would expect the most significant impacts from tree invasion, i.e., those areas closest to a seed source.



PERIODS: "1" = 1950-1969; "2" = 1970-1989; "3" = 1990-2001.

Figure 34. Change in vegetative classes by Ranger District. Numerals in graph represent the number of species of each vegetative class encountered using the Parker three-step survey method. Only allotments that were surveyed during each time period are included. Decades were combined to increase sample size (n = 93).

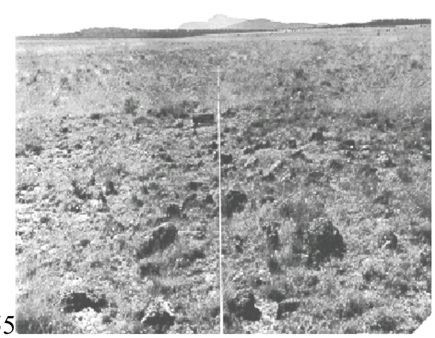


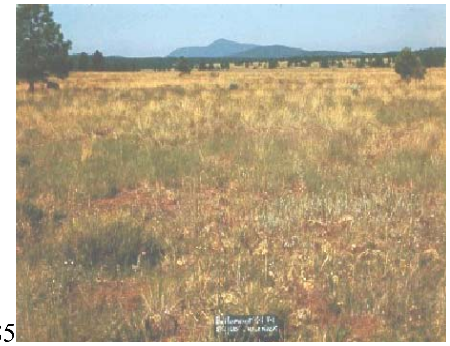
PERIODS: "1" = 1950-1969; "2" = 1970-1989; "3" = 1990-2001.

Figure 35. Change in plant, litter, soil, and rock cover by Ranger District. Graphs represent the number of times each cover type was encountered using the Parker three-step survey method and scaled to relative percentages. Only allotments that were surveyed each time period are included. Decades were combined to increase sample size (n = 93).

A. 9/17/53

B. 9/30/55



A. 2/24/65  B.8/21/85 

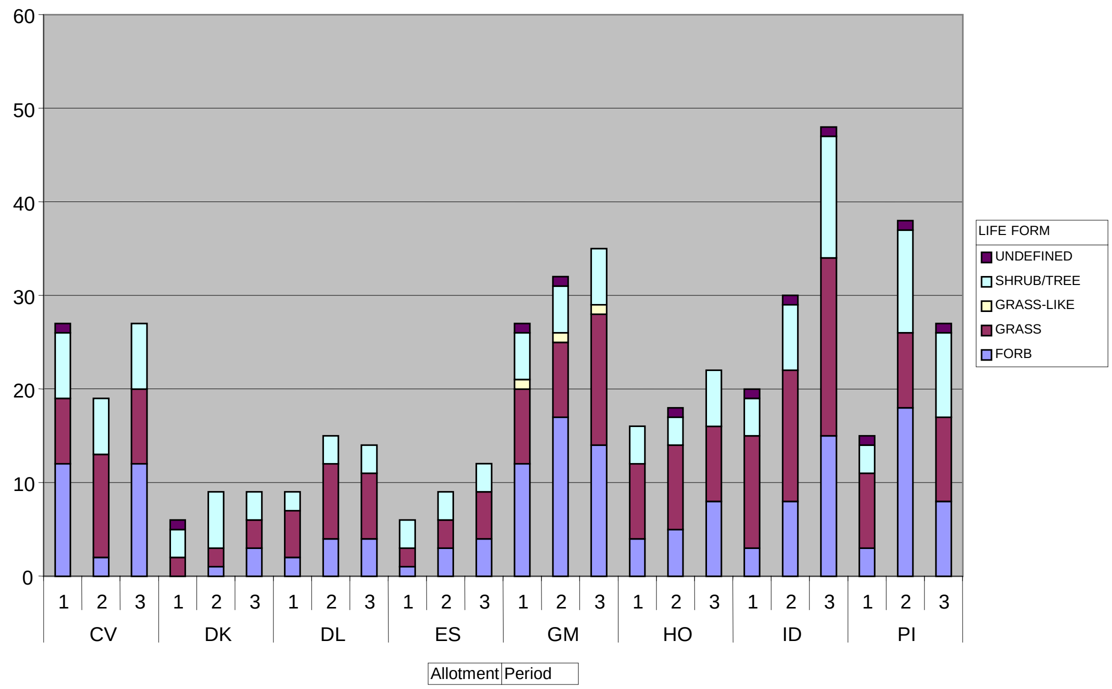
A. 1/9/78  C.10/4/62 

A. 9/11/93  C. 8/17/83 

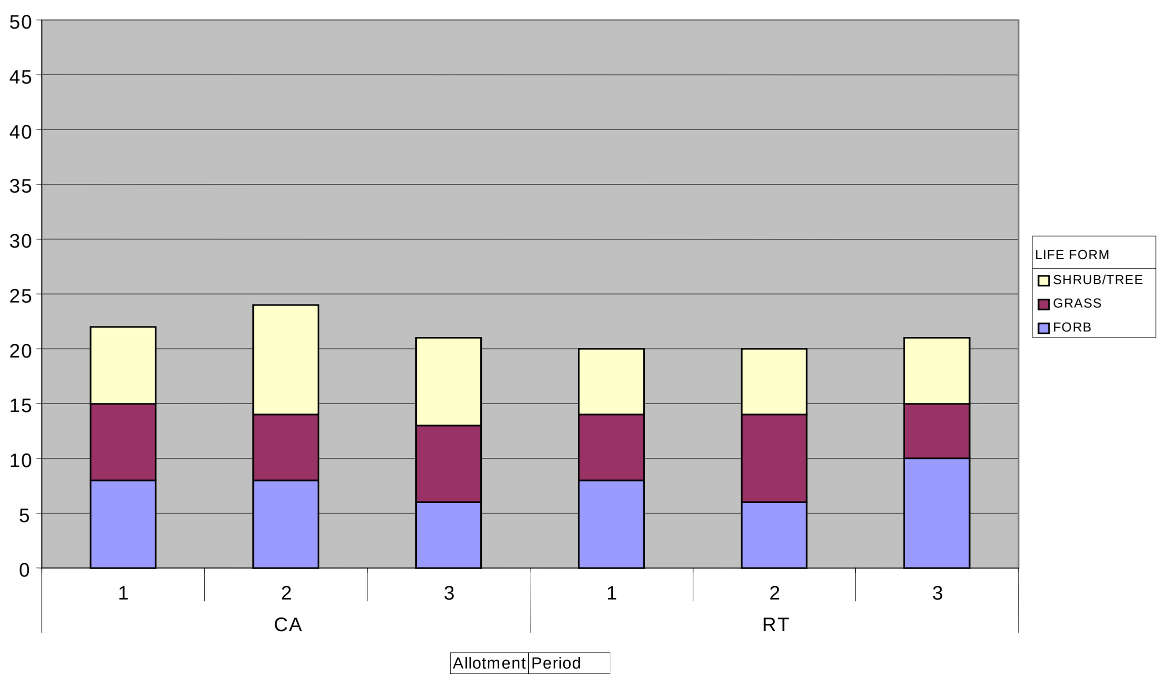
Figure 36. Views illustrating tree encroachment from fixed photo points on Parker three-step surveys, Kaibab National Forest. Pictures with the same letter denote the same photo point.

Similar data are presented below but organized by allotment (Figure 37). The variability noted within Ranger Districts is well illustrated at the allotment scale, i.e., even where the pattern is similar in overall plant cover, differences can be seen in the ratio of plant forms between allotments. An important factor influencing species diversity is the seasonality of when the Parker transects were read. Because of the dichotomy of cool season/warm season plants and a peak in species present during August and October, the time of year when transects are surveyed has a large influence on total species tallied (Figure 38). This bias has not been filtered from the results presented above.

WILLIAMS



TUSAYAN



NORTH KAIBAB

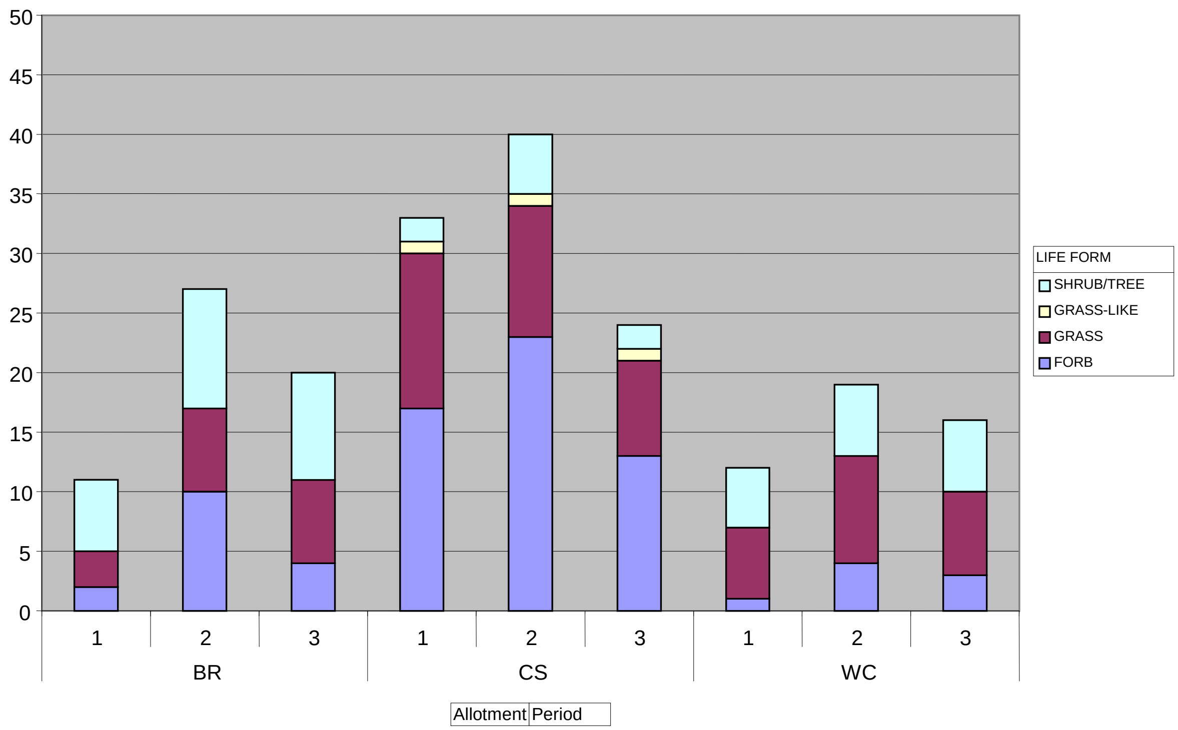


Figure 37. Change in vegetative classes by allotment. Bars represent the number of encounters by plant type. Only allotments that were surveyed during each time period are included. Allotments and time periods are defined in Figure 53. Decades were combined to increase sample size (n = 93).

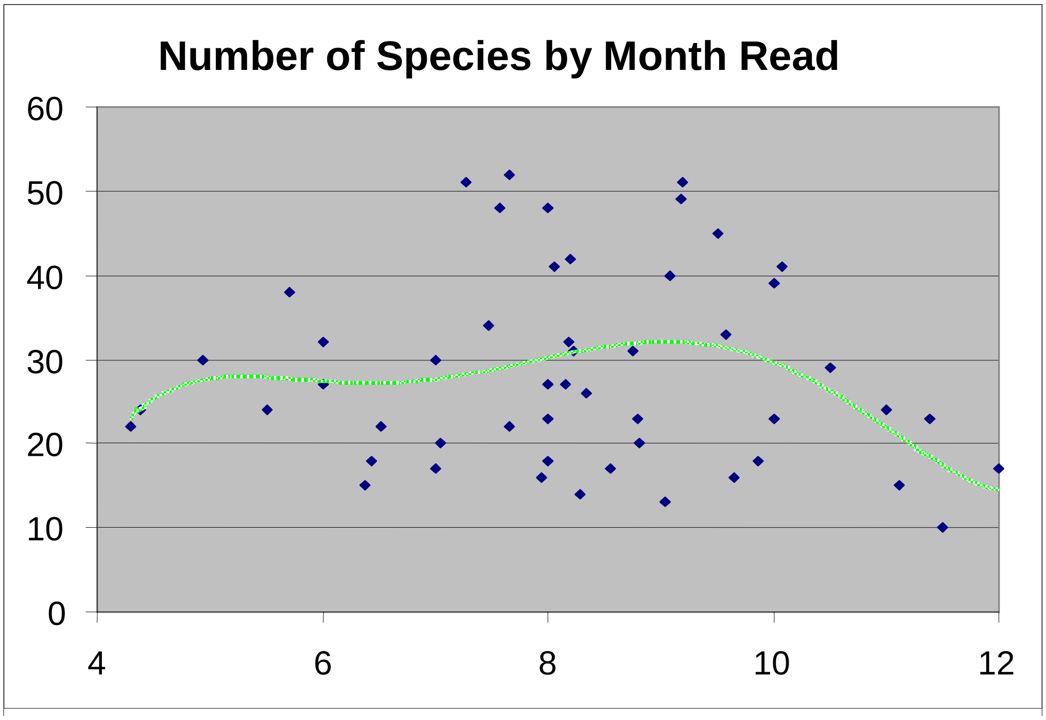


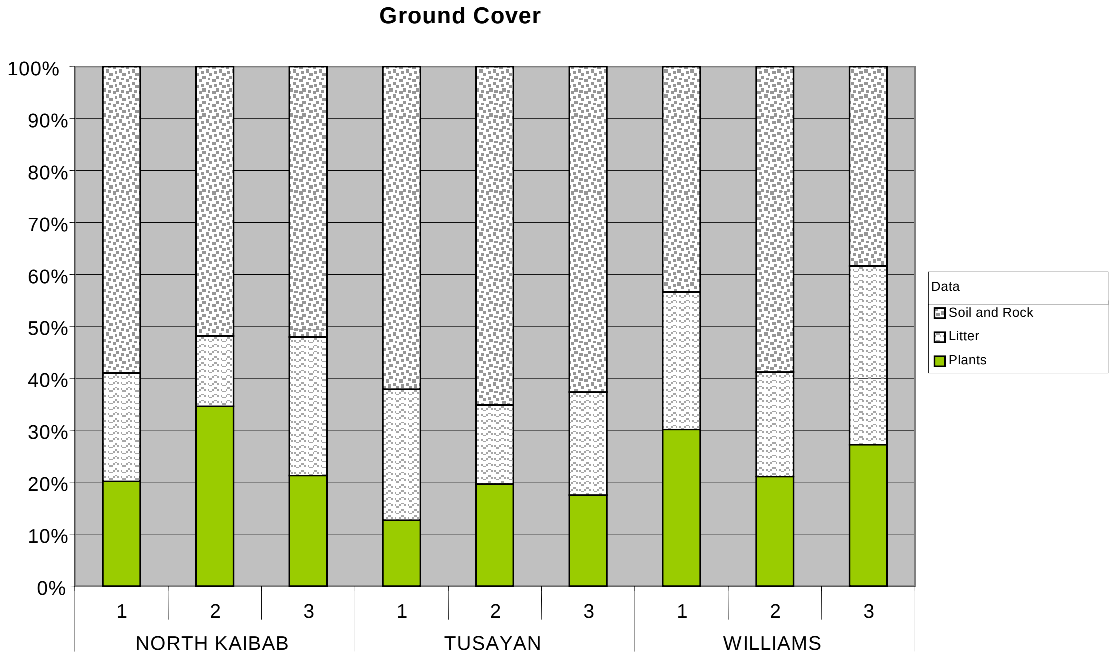
Figure 38. The number of species (Y-axis) identified on Parker surveys by month (X-axis), April through December. Average of mean values was fitted using a fifth-order polynomial. Each dot represents a separate Parker Cluster.

Grassland restoration has been a consistent goal of the KNF, particularly on the South Zone. Since 1987, the Williams and Tusayan RDs have applied prescribed fire to about 10,000 acres of grassland. About 500 acres dominated by ponderosa pine and 19,000 acres of pinyon-juniper woodland have been restored to grassland. About 11,800 acres of grassland have burned on the NKRD since 1987 due to wildfire. Combined, over 40,000 acres of grassland (16 percent of the total grasslands) have been restored or rejuvenated across the Forest.

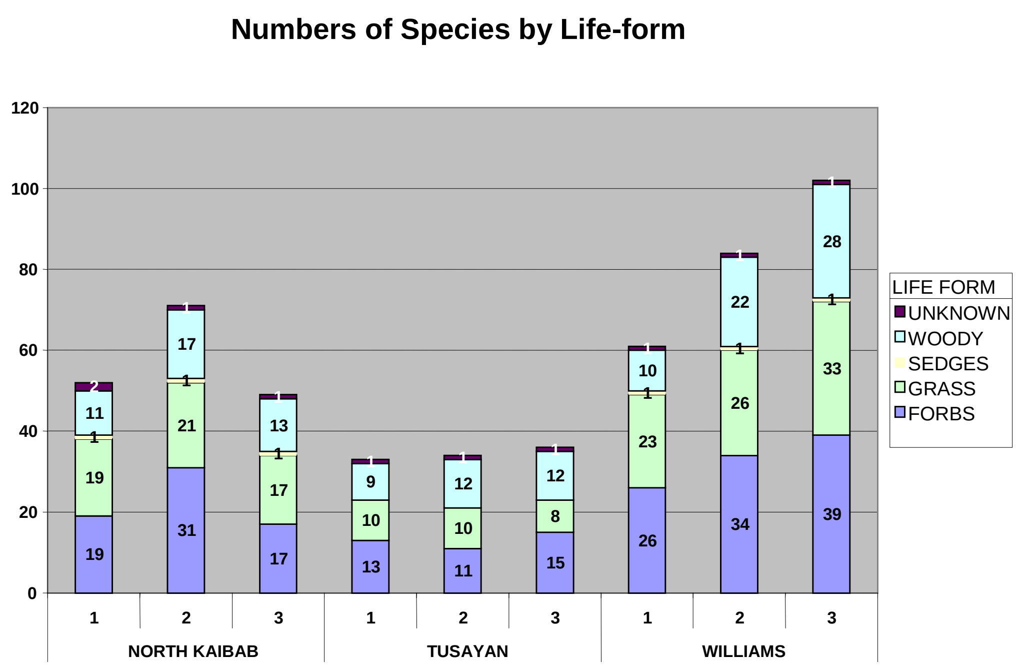
The only MIS for grasslands on the KNF is pronghorn antelope. Autenrieth edited the 1978 Pronghorn Antelope Workshop where specific recommendations and management guidelines were developed addressing key habitat components where pronghorn are selected as a featured species for management.

KAIBAB NATIONAL FOREST

A



B



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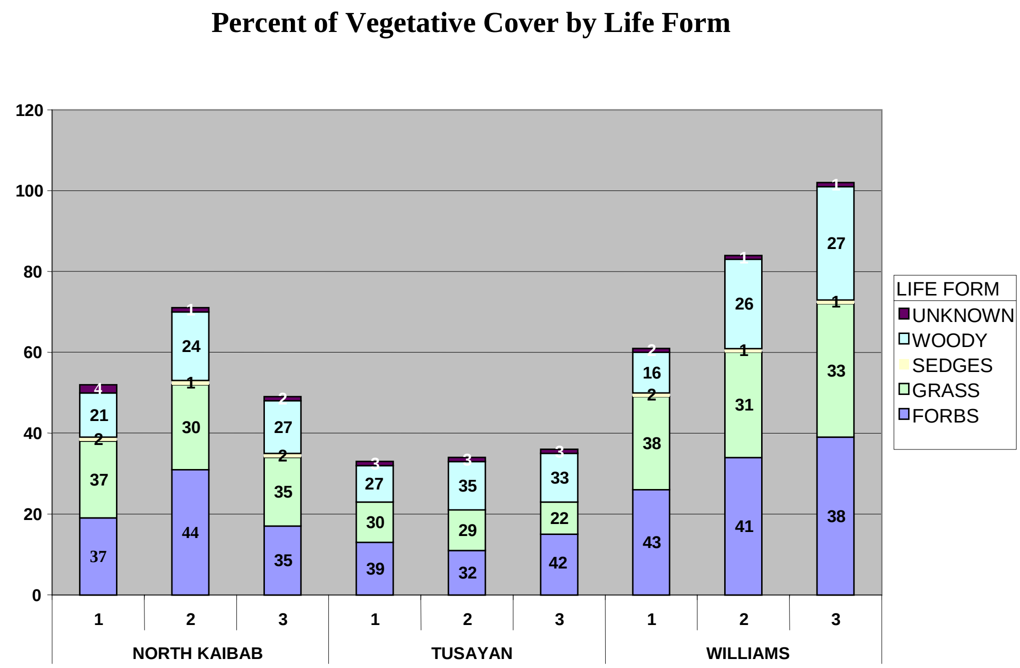


Figure 39. Habitat components identified as important for the management of pronghorn antelope (Autenrieth 1978): A is percent cover by cover type; B is numbers of species by vegetative class (values within bars are the number of species encountered); C is the relative ratio of plant forms (values within bars are the relative percent of each vegetation type); Time periods: 1 = 1950-69; 2 = 1970-89; 3 = 1990-2001.

According to Autenrieth (1978), "the best" pronghorn habitat should have:

- "at least 50 percent ground cover…"

On the KNF, only the Williams RD meet this recommendation (Figure 39A). On a District-wide average, Williams currently exceeds 60 percent cover. The NKRD almost meets this threshold but the Tusayan RD falls below 40 percent ground cover. Although Tusayan largely consists of relatively dry country, a survey of 10 relic (i.e., not grazed since at least the 1930s) sites in and around the Tusayan RD indicate maximum ground cover can range from 24 to 90 percent (U.S.D.A. 1986).

- "High diversity in vegetation is preferable: 5-10 grass species, 20-40 forb species, and 5-10 shrubs."

In general, the KNF is within the recommendations for plant diversity in pronghorn habitat, although the Tusayan District again appears marginal (Figure 39B).

- "…vegetative composition that averages 40-60 percent grasses, 10-30 percent forbs, and 5-20 percent shrubs"

None of the Districts achieve the 40 percent threshold for grasses, although when sedges are added, the Williams and NKRDs achieved 40 and 39 percent, respectively, in period 1 (Figure 39C). Currently the grass component ranges from 22 percent on the Tusayan District, 34 percent on Williams, and 37 percent on the NKRD. The forb component exceeds the recommended levels. Although trees and shrubs are combined, the shrub component is likely close to or within the guideline threshold.

- Fences can restrict the movement of pronghorn, cause changes in herd distribution, deaths, and the isolation of ranges. “Barbed wire fences with smooth bottom wires … allow pronghorn to pass.”

In recent years the Williams RD has targeted areas important to pronghorn for fence changes. At least nine miles of fencing has been removed from the RD and a minimum of seven miles of fencing were modified so that the bottom wire was pronghorn-friendly. Grazing practices are currently being changed in a key forage area (the ephemeral Davenport Lake) in part to provide direct benefits to pronghorn antelope, including the installation of 82 pronghorn crossings along the fence lines.

Impacts from grazing, logging, and fire suppression practices that started in the late1800s are still discernible on the landscape today (Covington and Moore 1992, Eddleman 1987). Impacts from grazing, evident in the1890s, continued into the 1970s. 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 that further dissected the landscape and contributed to a change in the size of the average fire and an interruption in the fire frequency (Covington and Moore 1994). Fire suppression, combined with grazing pressure, has resulted in a decrease in the overall diversity of grasses and forbs and allowed increases in the introduction and establishment of woody and exotic species. The history of grazing in Arizona is such that improvements in range conditions should, in general, benefit many wildlife species even where range improvements are not the same as ecological improvements. Range conditions in the Southwest have been monitored since the 1950s using the Parker three-step survey method. This same methodology has been used in the same locations on the KNF for nearly 50 years. Today’s obligated animal unit months are about 34 percent less than the targeted levels from 30 years ago.

In 2001, the Arizona Game and Fish elk population model estimated that over 7,000 elk occurred in the GMUs that overlap Williams and Tusayan RDs (these GMUs, and the elk that use them, also cover portions of the Coconino NF). Elk grazing in the four GMUs that encompass the Williams and Tusayan RDs (Figure 15) have gone from zero early in the century to an estimate of greater than 9,500 animals in the mid-1990s (Table 9).

Currently, cool season grass conditions have improved on the KNF since the 1950s/60s. Although the ratio of cool to warm season grasses appears steady since the 1970s/80s, the reoccurring drought conditions may be masking or inhibiting improvements in range conditions. This hypothesis is based on the 1980s being a decade of higher precipitation but, even with the drought of the 1990s continuing through today, the cool season grasses appear stable and there has been a congruent increase in species diversity for grasses and forbs. Tree encroachment has occurred on the Forest, but the level at which it occurs varies at the allotment scale. Tree encroachment appears to be more of an issue on the Williams RD, despite the history of pinyon-juniper pushes, recent and ongoing follow-up maintenance on the pushes, and prescribed fire conducted by the District. Exceptions to forest trends probably occur on each District and the results of changes in management may be masked by recent precipitation patterns, but Forestwide, there appears to be a stable to upward trend in grasslands on the KNF.
The Arizona Game and Fish Department identified one of the grasslands on the Williams Ranger District as representative of good quality pronghorn antelope habitat for north-central Arizona.

In general, plant diversity on the KNF is within the recommendations for pronghorn habitat (Autenrieth 1978), although the Tusayan District again appears marginal (Figure 39B). None of the Districts achieved the vegetative composition recommendation of 40 percent for grasses (Figure 39C). Currently the grass component ranges from 22 percent (Tusayan Ranger District) to 37 percent (NKRD). The forb component exceeds recommended levels. Although trees and shrubs were combined in the analysis, the shrub component is likely close to or greater than the guideline threshold. Fences can restrict the movement of pronghorn, cause changes in herd distribution, deaths, and the isolation of ranges. Pasture fencing is evaluated annually on the KNF and changes are made specifically to benefit pronghorn antelope. However, conditions on individual allotments vary widely and tree encroachment, especially on the Williams District, could be negatively affecting pronghorn habitat. The Williams District continues to work with Arizona Game and Fish during project planning to identify modifications that benefit pronghorn. Cattle do not directly compete with pronghorn for forage (Kitchen and O'Gara 1982), but may affect fawn mortality through reductions in hiding cover. Changes in range management implemented by the KNF, and the subsequent changes in grassland conditions, should improve pronghorn habitat. Nevertheless, as Kindschy et. al. (1982) noted, pronghorn benefit from range management when their needs are addressed in advance, not necessarily when left to chance or based on the assumption that "good range management is good for wildlife." Pronghorn habitat on the Kaibab appears to be stable, although the effects of reoccurring drought could easily lead to a declining trend.

Pinyon-Juniper Woodlands Cover Type

Pinyon-juniper (*Pinus edulis/Juniperus monosperma)* is the most abundant forest type in Arizona, covering about 7.7 million acres (56 percent) of the State's forested lands (O'Brien 2002). About 682,540 acres of pinyon-juniper habitat occurs on the KNF (Table 15). The NKRD has over 247,800 acres of pinyon-juniper woodland and the balance, 434,718 acres, occurs on the South Zone. Historically, pinyon-juniper has primarily been 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.

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 percent (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).

A proportional decrease in understory production occurs as the overstory develops. In addition to direct competition for soil moisture and nutrients, moisture availability for grasses and forbs is decreased by tree crown interception of precipitation. Allelopathic leachates from juniper roots further impact understory development. An average grassland in northern Arizona produces about 600 pounds of forage per acre, but forage production decreases to about 300 pounds per acre with 20 percent canopy cover and drops to less than 100 pounds of forage per acre under 60 percent canopy cover (Arnold et. al. 1964 *in* Doughty 1987). A prolonged decrease in understory biomass leads to decreases in the soil seed bank, limiting or delaying understory response after tree removal (Bedell 1987, Doughty 1987, Tiedemann 1987, Vaitkus and Eddleman 1987).

Pinyon-juniper forests are believed to have increased in both total landcover 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). Juniper counts of 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 (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.

Historically, the pinyon-juniper woodlands have been used extensively and intensively for cattle grazing. Grazing has contributed to the interruption in the fire sequence by reducing grass cover to the point where understory vegetation could no longer sustain fires. Herbivory has resulted in decreased biotic and genetic diversity of native species, contributing to the introduction and establishment of exotic species (Dahms and Geils 1997). 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.

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.

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 overwintering in pinyon-juniper habitat; radio-marked juvenile spotted owls used pinyon-juniper during both winter and dispersal movements (Joseph Ganey, personal communication, 2003). On the NKRD, pinyon-juniper habitat provide primary and critical winter ranges and transitional habitat during migration for the Kaibab deer herd (McCulloch and Smith 1991). Similarly, pinyon-juniper habitat on the South Zone 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.

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.

In many regards, changes in pinyon-juniper habitat are the mirror image of grassland conditions. West and Van Pelt's (1987) intercommunity cycle and management actions that have affected the dynamics of the ebb and flow between grassland and woodland habitats are discussed above in the grasslands evaluation. Parker Three-Step survey data were again used to evaluate changes in vegetative conditions in the woodland community. The analyses below focus on pinyon-juniper habitat on the South Zone of the KNF. Only allotments with Parker surveys completed in each of the three time periods were included in the analysis of trends in pinyon-juniper habitat. The NKRD was excluded because of the lack of consistent data. Decades in which Parker surveys were completed were combined to increase sample (i.e., period 1 includes the 1950s/60s, period 2 is the 1970s/80s, and period 3 encompasses the 1990s through 2001), but there were no allotments with Parker survey readings from each time period for the NKRD. Only about eight percent of the pinyon-juniper habitat on the KNF occurs on the NKRD and the remaining 92 percent is found on the South Zone.

Parker survey data have inherent biases when used for ecological assessment (see the grasslands discussion above), but they also represent the best long-term data set for understory conditions on the KNF. Queries of the Parker survey data were based on soil type (Table 29). Each soil type used in the analyses was unique to either grasslands or woodlands. However, four soil types that developed under grassland or savannah conditions now support pinyon-juniper habitat (Table 29). These soil types total about 80,110 acres, i.e., about 15 percent of the current woodland habitat on the KNF developed from grassland or savannah vegetation.

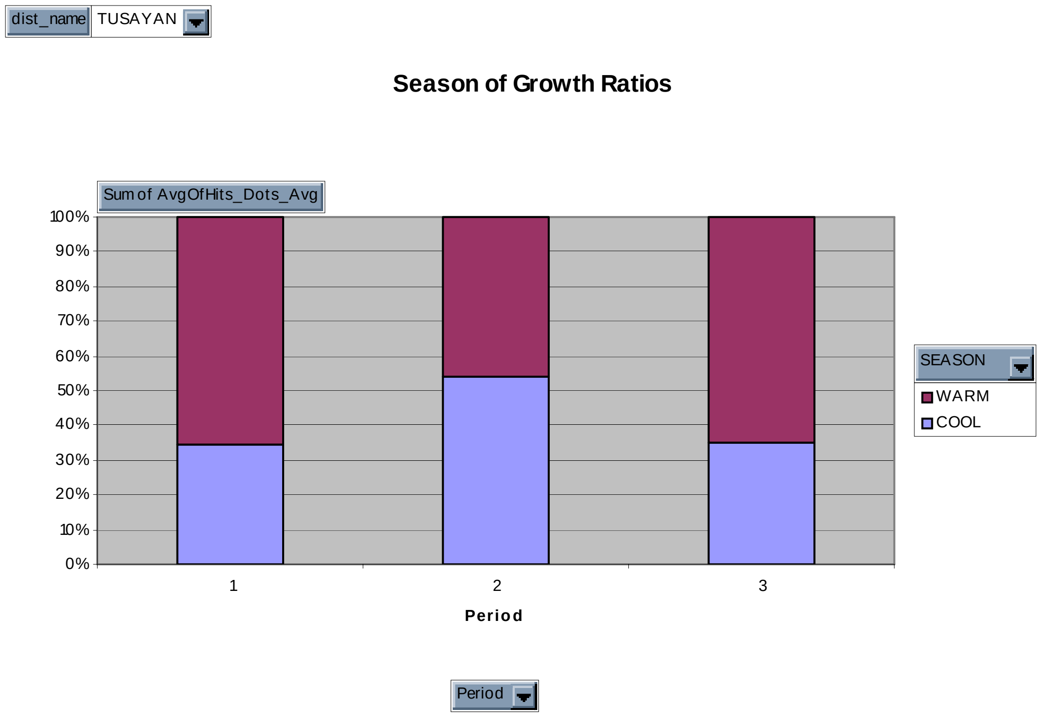
Table 29. Soil types from the Terrestrial Ecosystem Survey (TES) Map Units used to identify pinyon-juniper woodlands on the Kaibab National Forest.

TES NO.	SOIL CLASSIFICATION	TERRESTRIAL ECOSYSTEM	ACRES
162	Typic Haplustalfs, fine, montmorillonitic, mesic	Pinyon pine/one-seed juniper	3,319
165	Typic Haplustalfs, clayey-skeletal, montmorillonitic, mesic Lithic Haplustalfs, clayey-skeletal, montmorillonitic, mesic	Pinyon pine/one-seed juniper	4,202
166	Typic Haplustalfs, clayey-skeletal, montmorillonitic, mesic Lithic Haplustalfs, clayey-skeletal, montmorillonitic, mesic	Pinyon pine/one-seed juniper	4,302
167	Typic Haplustalfs, mesic Lithic Haplustalfs, mesic	Pinyon pine/one-seed juniper	2,590
172	Lithic Ustochrepts, calcareous, loamy-skeletal, mixed, mesic	Pinyon pine/one-seed juniper/needle and thread	3,796
251	Lithic Ustochrepts, calcareous, loamy-skeletal, mixed, mesic	Pinyon pine/Utah Juniper/big sagebrush/needle and thread	34,764
252	Lithic Ustochrepts, calcareous, mesic Typic Ustochrepts, calcareous, mesic Rock Outcrop	Pinyon pine/Utah Juniper/big sagebrush/needle and thread	75,581
257	Typic Haplustalfs, fine-loamy, mixed, mesic	Pinyon pine/Utah juniper/Gambel oak/big sagebrush	2,114
260	Lithic Ustochrepts, calcareous, loamy-skeletal, mixed, mesic Typic Ustochrepts, loamy-skeletal, carbonatic, mesic	Pinyon pine/Gambel oak/big sagebrush/needle and thread	53,978
261	Lithic Ustochrepts, calcareous, loamy-skeletal, mixed, mesic Rock Outcrop	Pinyon pine/Gambel oak/big sagebrush/needle and thread	4,970
263	Lithic Ustochrepts, calcareous, loamy-skeletal, mixed, mesic Typic Ustochrepts, loamy-skeletal, carbonatic, mesic	Pinyon pine/Utah juniper/big sagebrush/needle and thread	62,112
264	Lithic Ustochrepts, calcareous, loamy-skeletal, mixed, mesic Typic Ustochrepts, loamy-skeletal, carbonatic, mesic Rock Outcrop	Pinyon pine/Utah juniper/big sagebrush/needle and thread	31,860
272	Typic Haplustalfs, clayey-skeletal, montmorillonitic, mesic	Pinyon pine/Utah juniper/Gambel oak/big sagebrush	31,029
273	Typic Haplustalfs, clayey-skeletal, montmorillonitic, mesic	Pinyon pine/Utah juniper/Gambel oak/big sagebrush	27,816
274	Typic Ustochrepts, calcareous, mesic Lithic Ustochrepts, calcareous, mesic Typic Haplustalfs, mesic Rock Outcrop	Pinyon pine/Utah juniper/big sagebrush	7,688
277	Lithic Ustochrepts, calcareous, loamy-skeletal, mixed, mesic Typic Ustochrepts, loamy-skeletal, carbonatic, mesic	Pinyon pine/one-seed juniper/needle and thread	26,259
281	Typic Ustochrepts, loamy-skeletal, mixed, mesic Typic Haplustalfs, fine,	Pinyon pine/Utah juniper/big sagebrush	5,864

	montmorillonitic, mesic		
287	Lithic Ustochrepts, calcareous, loamy-skeletal, mixed, mesic Typic Ustochrepts, loamy-skeletal, carbonatic, mesic	Pinyon pine/one-seed juniper/Gambel oak/needle and thread	33,811
288	Typic Haplustalfs, fine, montmorillonitic, mesic Typic Haplustalfs, clayey-skeletal, montmorillonitic, mesic	Pinyon pine/one-seed juniper	7,206
295	Lithic Ustochrepts, calcareous, loamy-skeletal, mixed, mesic Typic Ustochrepts, loamy-skeletal, carbonatic, mesic Rock Outcrop	Pinyon pine/one-seed juniper/needle and thread	8,496
296	Lithic Ustochrepts, calcareous, loamy-skeletal, mixed, mesic Typic Ustochrepts, loamy-skeletal, carbonatic, mesic Rock Outcrop	Pinyon pine/one-seed juniper/needle and thread	1,775
476	Typic Haplustalfs, mesic Lithic Haplustalfs, mesic	Pinyon pine/one-seed juniper	1,271
495	Typic Haplustalfs, fine, montmorillonitic, mesic	Pinyon pine/one-seed juniper	19,546
496	Typic Haplustalfs, fine, montmorillonitic, mesic Lithic Haplustalfs, clayey-skeletal, montmorillonitic, mesic	Pinyon pine/one-seed juniper	15,484
543[a]	Vertic Haplustalfs, fine, montmorillonitic, mesic	Pinyon pine/one-seed juniper	29,418
586[a]	Typic Argiustolls, fine, montmorillonitic, mesic Typic Argiustolls, clayey-skeletal, montmorillonitic, mesic	Pinyon pine/one-seed juniper	16,141
587[a]	Lithic Argiustolls, clayey-skeletal, montmorillonitic, mesic Vertic Argiustolls, clayey-skeletal, montmorillonitic, mesic	Pinyon pine/one-seed juniper	19,098
589[a]	Typic Argiustolls, clayey-skeletal, montmorillonitic, mesic Typic Argiustolls, fine, montmorillonitic, mesic	Pinyon pine/one-seed juniper	15,453

[a]These soil types developed under grassland or savannah conditions but currently support pinyon-juniper habitat.

Range conditions can be evaluated in terms of the ratio of cool- to warm-season plants. Historic overgrazing has reduced the frequency and occurrence of cool-season grasses (typically C_3 plants), allowing warm-season grasses (C_4 plants) to expand. Although pre-settlement ratios are not known, the preference for C_3 plants that herbivores exhibit have led to a general rule of thumb that increases in cool-season plants will improve range conditions. Substantial increases in cool-season grasses occurred on the KNF during the 1970s/80s but were followed by declines in the 1990s through 2001 (Figure 40). Little improvement is evident between the 1950s/60s and the 1990s/2001, despite significant changes in the number of livestock permitted on the South Zone allotments (Table 27) and changes in range management. The pattern of change is very similar for both South Zone Districts, although the percentage of cool-season grasses is much lower on the Williams RD. The increases and subsequent decreases in cool-season grasses may largely be responses to concurrent changes in precipitation. Nevertheless, changes in range management, combined with the many range improvements instilled during the 1970s/80s, are expected to have ameliorated recent decreases in precipitation (Figure 40).



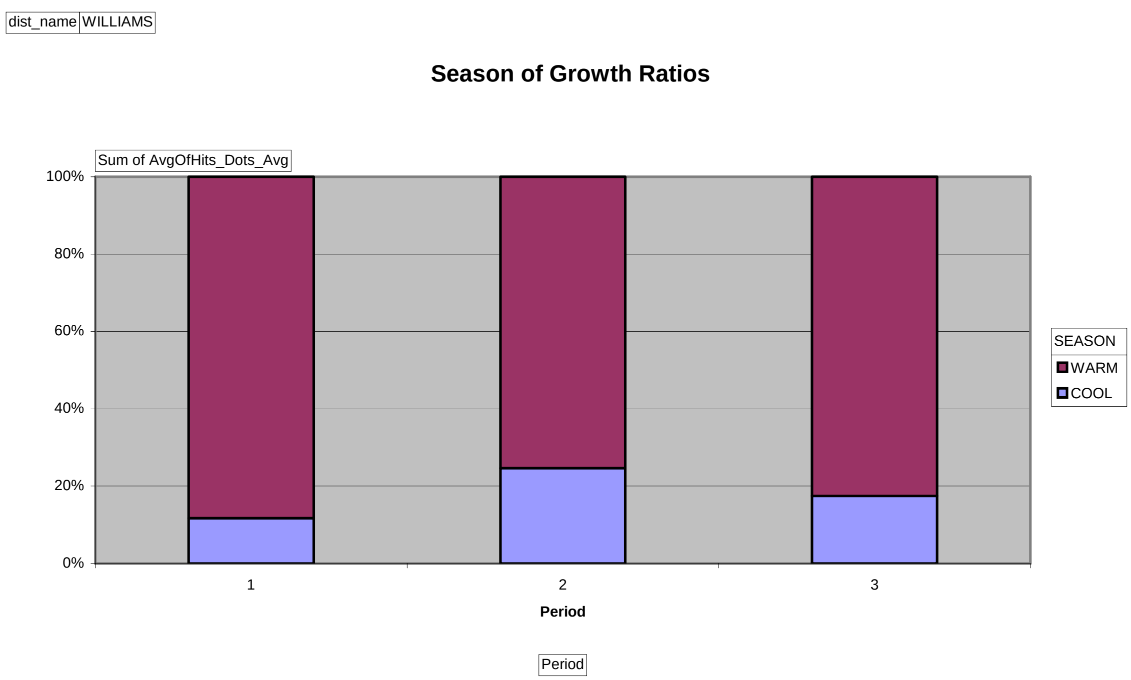


Figure 40. Changes in the ratio of cool- to warm-season grasses on the South Zone of the Kaibab National Forest. Only Parker surveys sampled during each time period are included (n = 7 allotments). Decades were combined to increase sample size (period 1 = 1950s/60s; period 2 = 1970s/80s, and period 3 = 1990s through 2001).

The pattern for total organic cover, i.e., plant and litter cover, over time is similar to that for the warm season grasses. Whereas organic cover decreased in the 1970s/80s, cover increased in the 1990s/2001 (Figure 41). Both South Zone Ranger Districts were combined due to their nearly identical time period values. Although increases in litter are due in part to overstory contributions, plant cover also increased in the 1990s/2001. Figure 31 indicates less winter and more monsoon precipitation fell during the 1990s/2001. Given the ratios of warm- to cool-season grasses and the winter/monsoon precipitation patterns, increases in the plant cover are probably due to increases in warm season grasses. It appears that the increases in plant and litter cover are a result of increases in pinyon, juniper, and warm season grasses. Nevertheless, increases in plant and litter cover should better protect woodland soils from erosion and aid in soil moisture retention.

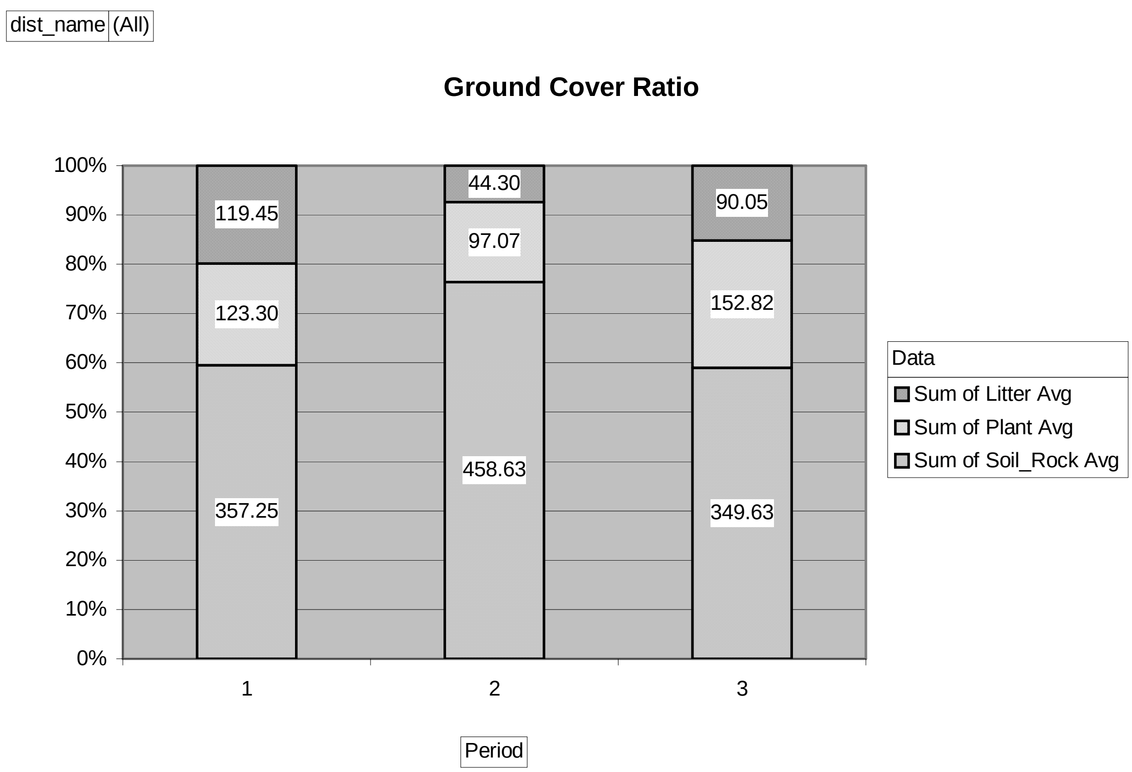
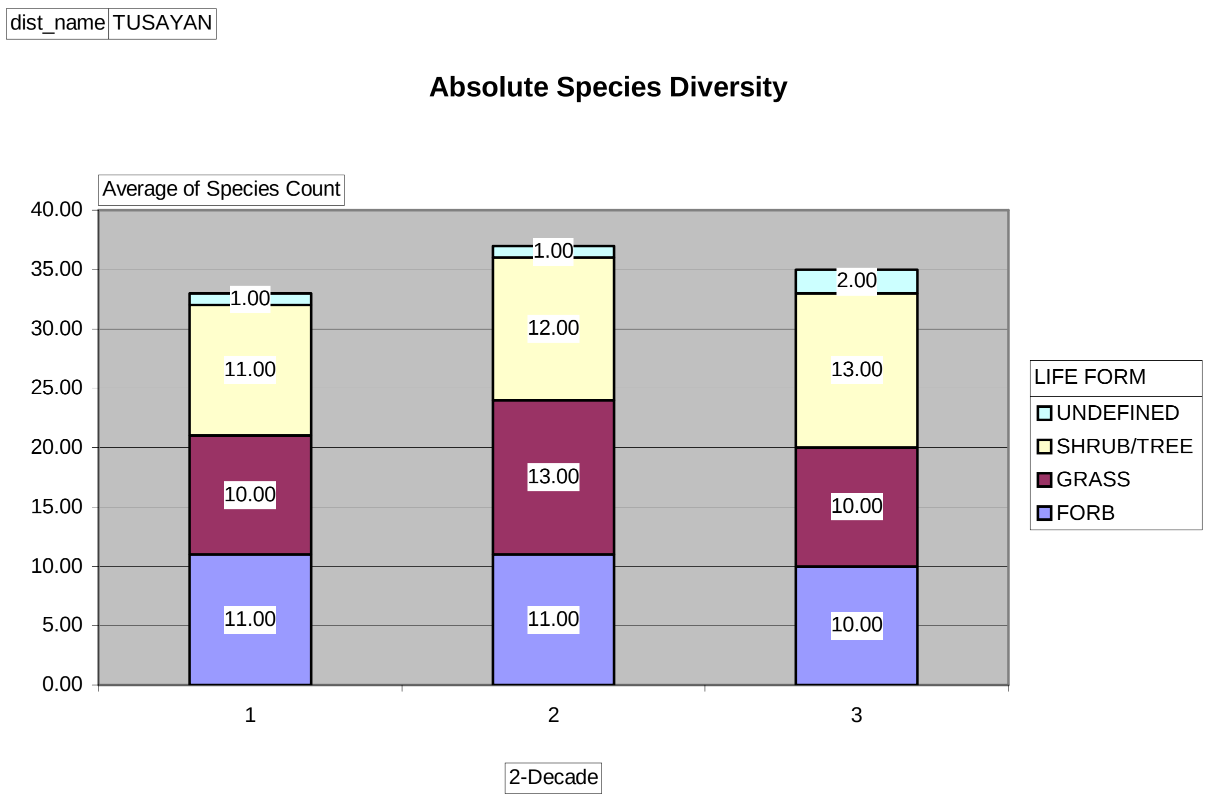


Figure 41. Changes in litter, plant, and bare ground for the South Zone of the Kaibab National Forest. Numbers inside graph bars represent the number of times each cover type was encountered during Parker transect surveys. Values were then scaled to relative percentages. Only allotments surveyed during every time period were included (n= 7); decades were combined to increase sample size (period 1 = 1950s/60s; period 2 = 1970s/80s, and period 3 = 1990s through 2001).

The increase in plant cover was accompanied by changes in plant species. Unlike the changes in ground cover, there are apparent differences between the Williams and Tusayan Districts (Figure 42). The total count of woody species increased in occurrence while grass species decreased on the Tusayan District. The increase in woody species occurred within stands of pinyon-juniper, which is different from and in addition to the increase of woody species in grasslands reported in the grasslands write-up. However, the scale of change is small in all categories. Increases in the numbers of species in all vegetative strata occurred on the Williams District between the

1950s/60s and the 1970s/80s. There was little change in total species between the 1970s/80s through the 1990s/2001, but there was an increase in the number of forb species.



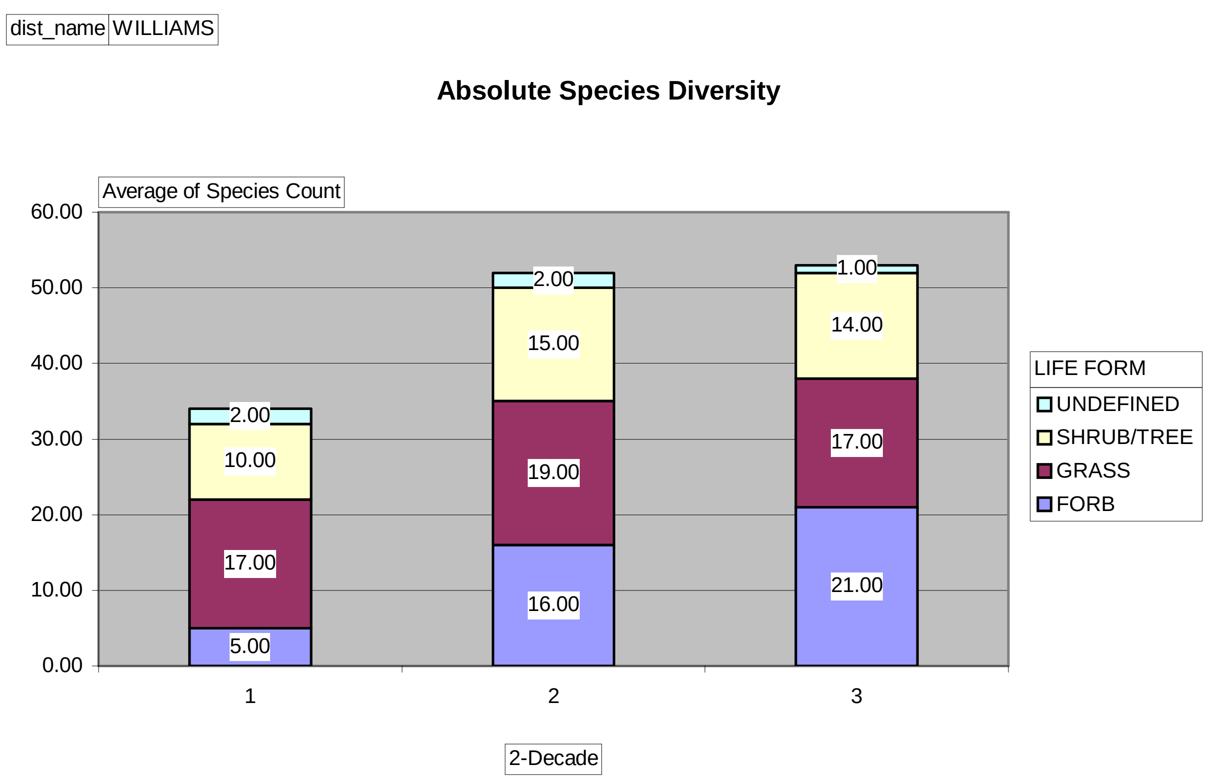


Figure 42. Changes in species count by vegetative class. Numbers inside graph bars represent the number of species encountered by vegetative class during Parker transect surveys. Only allotments surveyed during each time period were included (n= 7); decades were combined to increase sample size (period 1 = 1950s/60s; period 2 = 1970s/80s, and period 3 = 1990s through 2001).

The number of species encountered when sampling is related to the date that transects are read. The species total begins increasing in conjunction with the summer monsoon rains, typically in July, and peaks in the autumnal months (Figure 43). Therefore, it is important to sample throughout the growing season to capture trends in all vegetation types. Reading more Parker transects in spring would help to better track cool-season plant conditions.

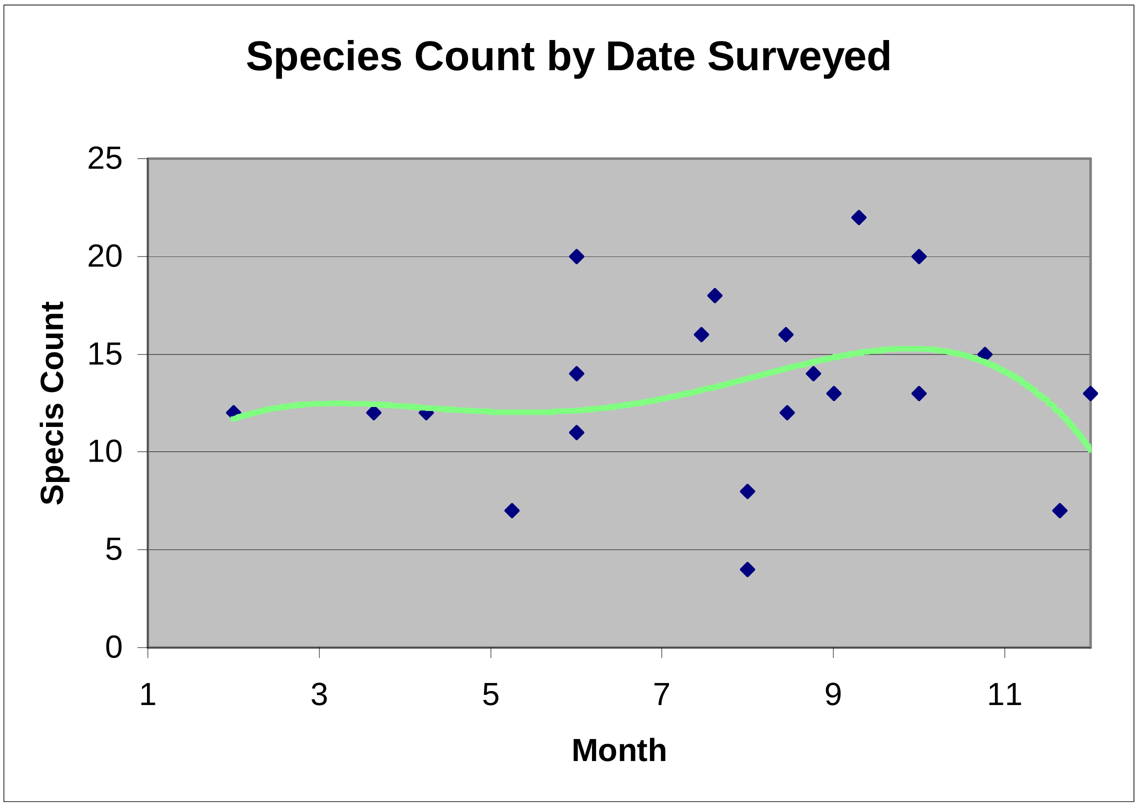


Figure 43. Number of species encountered by month sampled for the South Zone of the Kaibab National Forest. Each point represents an individual Parker transect cluster.

Management activities on the NKRD have been limited in the pinyon-juniper cover type. There have been about 320 acres treated to reduce tree densities. Understory removal of woody vegetation occurred on about 220 acres. Another 100 acres were treated through fuelwood sales. Fuelwood removal occurred in spot treatments of a ¼ acre or less. Wild fires on the NKRD have burned over 97,000 acres of pinyon-juniper woodland since 1987. Invasion of exotic and noxious plant species is a concern on these vast tracts of disturbed ground.

Conversely, management of pinyon-juniper on the South Zone has treated about 10,000 acres and no wildfires over an acre or so have occurred since 1987. Thinning in pinyon-juniper, totaling about 5,200 acres, has been done to release understory vegetation and enhance cover-forage ratios. Prescribed burns have been conducted across 1,500 acres of pinyon-juniper and patch cuts have been accomplished through fuelwood sales across 3.300 acres of woodland habitat.

The ongoing drought, combined with increased and inter-related bark beetle activity, is currently greatly reducing the pinyon component within pinyon-juniper habitat on the KNF. The U.S.D.A. Forest Service Forest Health Protection Arizona Zone Office coordinates annual forest insect and disease aerial detection surveys. Surveys in Arizona estimated that bark beetle activity increased by 3.6 percent in 2000 and by 185 percent in 2001 and 2002. Over 800,000 dead trees were detected on 60,000 acres since the 2001 flight survey was completed. Most mortality occurs on dryer sites with thinner soils in vegetation transition zones. Flight surveys focused only on areas immediately adjacent to ponderosa pine, leaving the majority of the pinyon-juniper habitat unsurveyed. Areas where follow-up ground surveys were conducted indicate pinyon pine mortality levels as high as 90 percent. The *Ips* bark beetles that attack pinyon trees have multiple generations per year and populations that can grow exponentially in a single season. This means that areas surveyed earlier in the season could potentially experience up to a 300 percent increase before the end of the season (Anhold 2002). Although it is not accurate to extrapolate these results across the woodland cover type, we do know that pinyon mortalities are increasing at an increasing rate.

This loss of the pinyon component represents a loss of food and structure for a variety of wildlife species, including Merriam's turkeys and migratory birds, and is expected to have long-term effects on species that nest, roost, and forage on pinyon trees. Although this is creating an abundance of snags in the short-term, the long-term expectation is for a future bottleneck of pine snags in the pinyon-juniper woodlands.

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. Improving existing range conditions has been complicated by ongoing drought conditions. Over the last 30 years the KNF has reduced the obligated animal unit months by about 34 percent and has developed new water areas to better distribute livestock impacts. Other modifications, e.g., refiguring pastures and adjusting timing of use, have been implemented to improve range conditions. Changes detected in understory conditions through long-term Parker Three-Step surveys include an increase in the ratio of warm-season grasses relative to cool-season grasses and an increase in overall litter and plant cover. The latter may be due to increases in both warm season grasses and woody vegetation. Plant species diversity has shown little change on the Tusayan RD, but forb diversity has increased on the Williams District. In general, pinyon-juniper habitat and stem density/basal area within woodland habitat are believed to have increased across the Forest.

Increases in pinyon-juniper habitat are generally at the expense of grasslands, shrublands, savannahs, and openings within woodland habitat. Although these shifts in cover types likely benefit true woodland species, it is at a cost to open habitat species. Grassland avifauna, pronghorn antelope, and Gunnison's prairie dogs are all likely negatively impacted. The increase in forb diversity on the Williams RD suggests a potential improvement in pronghorn forage. Mule deer and juniper titmice are the pinyon-juniper Management Indicator Species.

The broad level at which this analysis was conducted does not address the detail necessary for truly evaluating mule deer habitat. This is particularly true for the Parker survey results where multiple allotments are combined to generate data for analysis. However, variability between allotments may be significant and may be masked by grouping survey results. Anecdotal observations indicate that cliffrose is commonly in a decadent state and is not successfully reproducing. Conversely, sagebrush appears to be increasing in total cover. Both shrub species are key forage items, although cliffrose is a higher quality food. 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 mule deer habitat on the Kaibab since 1987 appear to be relatively stable.

Juniper titmice are management indicators for late-seral pinyon-juniper habitat and snags. Except for actions aimed at restoring grasslands and savannah, the low levels of active management of pinyon-juniper woodland have allowed stands to increase in area, density, and has allowed seral succession to continue. The results have presumably increased the amount of habitat, created more snags through competition, and to date, management has only converted a small percentage of the woodland cover type to early successional stages. 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 and pinyon snag availability. Overall, there has probably been an increasing trend in juniper titmouse habitat quantity and quality since the signing of the KNF Forest Plan.

Riparian Associations

About 270,200 acres of riparian habitat exists within Arizona's 73 million acres (Latta et. al. 1999). Riparian areas act as water, cover and food sources and as migration corridors for a wide range of wildlife species (Latta et. al. 1999). Riparian areas are perhaps the most heavily impacted habitat type in Arizona. Millions of people depend on and recreate in riparian habitats and about 65 percent 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).

Using the general descriptions provided by the Arizona Partners in Flight (AZPIF) (Latta et al. 1999), riparian associations occur in or adjacent to drainageways and/or floodplains, and are characterized by species and/or life forms that are distinctly different from the immediately surrounding non-riparian habitat. 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.

Low elevation riparian 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. Elevations range from 3,500 to 6,000 feet and about 80 percent of the area has slopes greater than 40 percent. Unlike the Kaibab Plateau, the climate is semi-arid with the mean annual precipitation ranging from eight to 12 inches. The entire Forest Service portion of the drainage is a designated Wilderness Area. 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. Water flow has been restricted by diversions upstream of the Forest boundary. Tamarisk is established throughout the watershed, with abundant seed sources upstream of the Forest Service boundary. Tamarisk is out competing and displacing native woody vegetation in Kanab Creek. Tamarisk control treatments are currently being evaluated. Tamarisk control could extend to about 565 acres and would include mechanical, cultural, herbicidal, and biological treatments.

Lucy's warblers, yellow-breasted chats, and aquatic macroinvertebrates are the MIS for riparian habitat in this Ecosystem Management Area. Aquatic macroinvertebrates are identified as MIS for 11 of 14 Ecosystem Management Areas across the KNF. However, they are only effective in North Canyon Creek where the ratio of aquatic insects can be related to changes in water quality. Unlike North Canyon Creek (see below), where water quality is an essential element in the health of that particular ecosystem, the mere presence of water and any associated riparian vegetation are the keys to evaluating habitat conditions for most riparian associations on the KNF. Aquatic macroinvertebrates are not effective management indicators when stream courses have cycles of spring runoff that subside into slow or stagnant reaches of warm, isolated, receding waters, as in Kanab Creek. Similarly, tracking water quality in stock tanks through the use of aquatic insects does not address any substantive management issues.

Lucy's warblers and yellow-breasted chats prefer to nest in willow and mesquite thickets (Johnson et. al. 1997, Cornell Laboratory of Ornithology 2000). Lucy's warblers also nest in cottonwood gallery forests (bosques), cottonwood-willow forests, densely vegetated dry washes, and ash-walnut-sycamore-live oak forests (Johnson et. al. 1997, Latta et al. 1999). Neither species is an appropriate management indicator for low elevation riparian areas on the KNF because of the absence of nesting habitat. Woody vegetation in the riparian zones is dominated and compromised by tamarisk within the Kanab Creek watershed. The sometimes extensive tamarisk stands are homogeneous, do not provide good foraging habitat, and are increasing in distribution. Isolated pockets of willow and cottonwood do exist within the drainage, but are limited in extent (Figure 44). No cottonwood galleries occur in Kanab Creek (or anywhere else on the KNF). Bird surveys conducted in Kanab Creek in 2001 failed to detect any of the riparian MIS. However, a single red-naped sapsucker, a MIS for aspen, was identified.

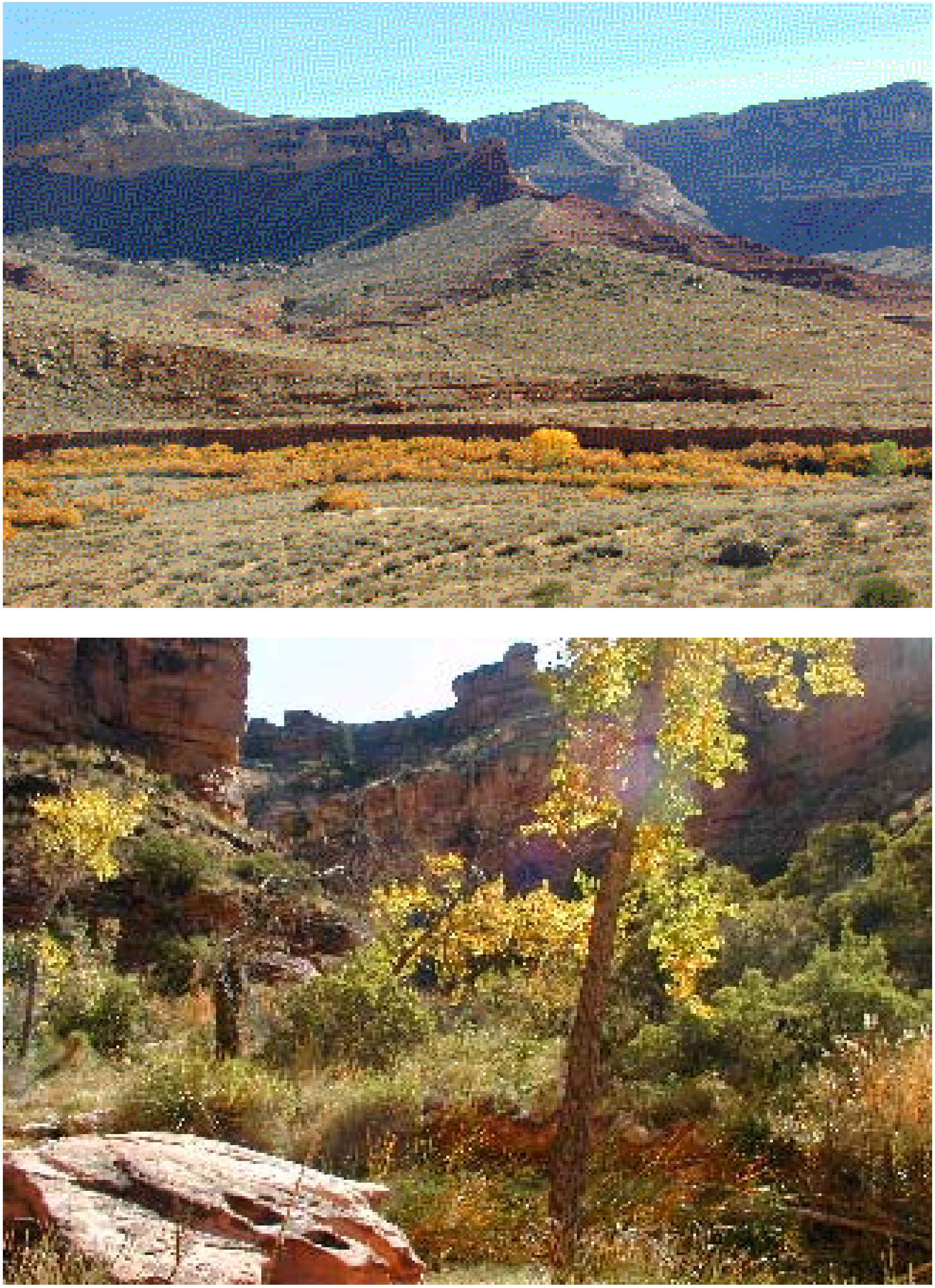


Figure 44. Woody riparian vegetation along Kanab Creek. Cottonwood trees are scarce in the drainage and lack the gallery structure required by many riparian-dependent species. Photographs courtesy of Thomas E. Hooker.

Although the current MIS are not applicable to the Kanab Creek drainage, habitat changes can still be evaluated. 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 45). One way to evaluate range conditions is to categorize plants by their response to grazing (Table 28). Native perennial vegetation normally present in climax communities is classified as increasers (plants that increase under heavy grazing pressure), decreasers (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 decreasers.

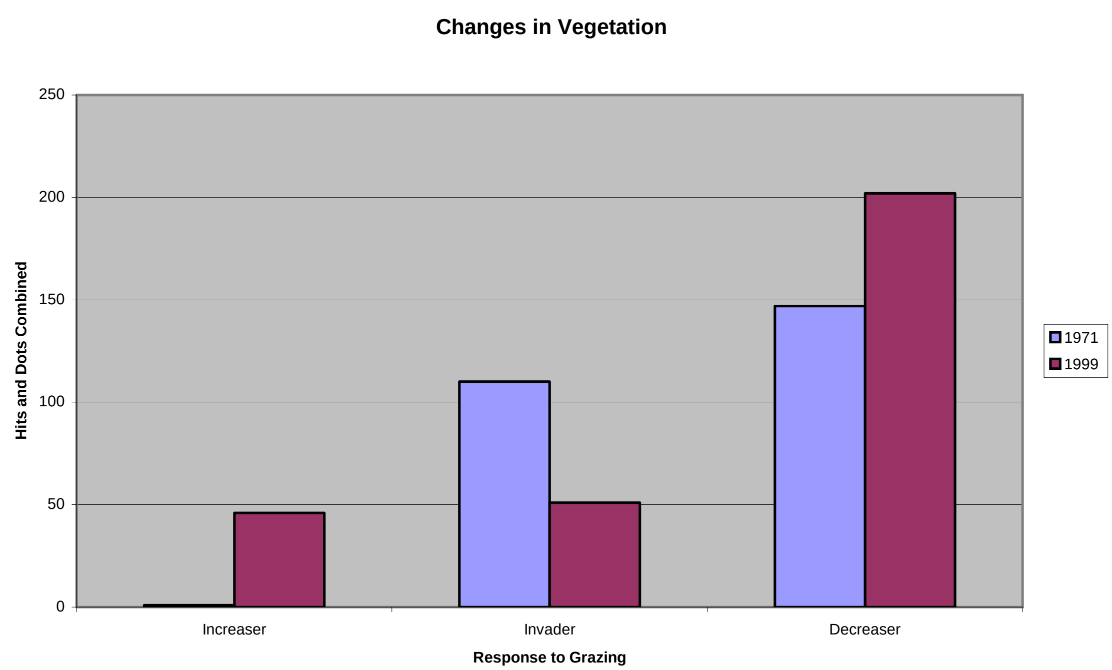


Figure 45. Changes detected in perennial species in Kanab Creek using both hits and dots along a permanent Parker Transect Cluster. Increasers, decreasers, and invaders refer to the plants response to grazing.

Table 30. Perennial plants encountered along the Parker Transect Cluster in Kanab Creek, Kaibab National Forest. Dots and Hits represent techniques used to record plants along established transects.

Kanab Parker Transect Cluster

SPECIES	COMMON NAME	GRAZING RATING	Dots and Hits Combined Survey Year 1971	Survey Year 1999
Atriplex canescens	4-wing saltbrush	Decreaser	0	1
Bouteloua eriopoda	Black grama	Decreaser	85	104
Ephedra spp	Mormon tea	Invader	1	32
Gutierrezia spp	Snakeweed	Invader	3	19
Hilaria jamesii	Galleta	Increaser	1	39
Kochia americana	Greenmolly	Increaser	0	7
Poa fendleriana	Muttongrass	Decreaser	1	0
Spartina spp	Cordgrass	Invader	59	0
Sporobolus cryptandrus	Sand dropseed	Decreaser	61	97
Tridens pulchella	Fluffgrass	Invader	24	0
Zygadenus spp	Deathcamas	Invader	23	0

Changes in management since the signing of the Kaibab National Forest Land Management Plan (Forest Plan) indicate an improving trend in low elevation riparian habitat.

Riparian Associations: High Elevation Riparian

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. These waters flow for a combined maximum distance of about 8 to 9 miles, but average distances range between 2 to 5 miles. Flows and stream lengths 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 North Canyon Springs and the stream course continues, on average, about 2 to 4 miles. The length of the creek varies, depending on winter snowpack and seasonal temperature and precipitation patterns. A survey in June 1990 found 4 to 4.5 miles of perennial water, but flow was measured at 1 to 4 cubic feet per second. Slopes vary tremendously, from about 60 percent near the springs, to 4 to 5 percent 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. 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 are open, with exposed rock and soils and scattered pine and juniper trees. 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 the 1960s. 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 may also allow added options in future management decisions.

Active monitoring and management occurs with 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 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.

Trail work has occurred in 2003 to help stabilize the soils and limit sedimentation into the creek. A fuels reduction project is being planned above North Canyon Creek, outside of, but along the edge of the Saddle Mountain Wilderness Area. The goal of this project is to reduce the threat of stand replacing fires from occurring within the North Canyon drainage. This threat was identified when the Outlet fire escaped from the Grand Canyon National Park in 2000. 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. Data collection and analysis for the fuels reduction project are expected to occur in 2004 and is identified in the 3-year program-of-work for the NKRD fuels program.

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 to 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 RD occur several miles down the drainage where Forest Service road 109 crosses the Big Spring draw. 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.

MIS for riparian habitat on the KNF include aquatic macroinvertebrates, Lucy's warbler, yellow-breasted chat, and cinnamon teal. Aquatic macroinvertebrates are an effective management tool for monitoring water quality (see the Aquatic Macroinvertebrates section of the Species Description above) in North Canyon Creek. However, no habitat occurs within the high elevation riparian associations for Lucy's warbler, yellow-breasted chat, or cinnamon teal.

Nevertheless, management activities in the high elevation riparian areas are minor and have been further restricted since the signing of the Forest Plan. Habitat trends are considered to be stable to improving for high elevation riparian areas on the KNF.

Riparian Association: Freshwater Marshes

Similar to the rest of Arizona, freshwater marshes are an anomaly on the KNF. 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 marshlands 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. Marshland habitat on the KNF tends to be a result from dams, diversions, and/or consists of ephemeral waters dependent on annual weather cycles. Ephemeral marshlands might only produce the defining vegetation once every several years. Marsh habitat on the KNF primarily occurs on the Williams RD (Table 31). The Tusayan RD does not have any marshland.

Table 31. Primary marsh habitat on the Kaibab National Forest. Note that acres of water refers to total area covered by water and is included for scale; actual marsh habitat will be a subset of this value in most instances.

Ranger District	Location	Acres of Water
North Kaibab	Franks Lake Botanical Area	2+
North Kaibab	Three Lakes	10
Williams	Coleman Lake	80
Williams	Davenport Lake	285
Williams	Duck Lake	50
Williams	J.D. Dam	30
Williams	Scholz Lake	65
Williams	Sunflower Flats	30

Franks Lake Geologic-Botanical Area and Three Lakes are the major marshy areas on the NKRD. Franks Lake is perennial whereas Three Lakes is an ephemeral feature. Franks Lake is a series of three limestone sinks with a mosaic of wet and dry meadows, marsh, 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 Arizona 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 to 90 percent in the meadow sample plots and the species composition consisted nearly entirely of native plants with only dandelions occasionally recorded. The shoreline had filled in with native vegetation, the bog had expanded in size, and the vegetation had recovered in areas previously trampled by cattle.

Three Lakes has an excavated ring around the exterior of the pond to provide open water. Emergent vegetation occupies the center portion of the lake. Although other examples of this habitat exist on the NKRD, they tend to be less than 2 acres 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 marshy areas including small patches or limited linear strips of marsh vegetation, e.g., Big Spring and Keyhole Sink, to broad marsh and wet meadow complexes such as Sunflower Flat. Many of these areas are microsites that are only fractions of an acre in size and support small patches of marshy vegetation, such as Dow Springs, Bear Springs, and Pomeroy Tanks. These perennial springs are more prevalent near the Mogollon Rim where melted snowpack collects and comes to the surface. Coleman Lake, JD Dam, Scholz Lake and Sunflower Flat are the most extensive perennial marshes on the Forest (30 to 80 acres). The amount of marsh vegetation and water are highly variable, depending on the seasonal weather patterns, including winter snows and monsoon rains. As a group, they provide habitat for resident, wintering, and stopover waterfowl, osprey, bald eagles, peregrine falcons, goshawks, a range of migratory passerines, and the only known great blue heron rookery on the Forest. Sunflower Flat is currently owned by the Arizona Department of Game and Fish and is included in a proposed land exchange between the State of Arizona and the U.S.D.A. Forest Service. The entire parcel is 160 acres. If ownership transfers to the Forest Service, the KNF would continue to manage the area for wildlife values. Specifically, livestock would continue to be excluded from the entire parcel and wildlife would continue as the beneficial use for any water rights transferred into Federal ownership (U.S.D.A. Forest Service 2001).

Ephemeral wetlands such as Davenport, Duck, and Dry Lakes occur when adequate amounts of winter snows accumulate. These precipitation patterns are unpredictable and may occur once every five 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 primarily stopover waterfowl enroute to annual nesting areas. Use by resident waterfowl is limited even in wet years (Table 32). Many of the major marshlands have been fenced to exclude livestock and most exclude off-road vehicles. Off-road vehicles are prohibited at most of the microsites too.

Table 32. Average number of waterfowl spotted at Davenport Lake during wetter than normal years (1987, 1988, and 1995) from records at the Williams Ranger District Office.

February – May	June	July	August-October	November	December-January
≈ 3400	10	1	0	8	0

Riparian habitat protection is largely dependent on fencing. Although problematic, the overwhelming majority of the fencelines 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 in time.

The variety of riparian features across the KNF creates a range of marsh conditions. However, most of the marshlands are limited in size, occur as isolated features across a xeric landscape, and many are ephemeral. None of the marshlands provide habitat for Lucy's warbler or yellow-breasted chat and none of the marshlands provide habitat to support populations of cinnamon teal. Although this habitat is key in supporting individual pairs or family groups of resident, riparian-dependent avifauna, there is not sufficient habitat to support viable *populations*. Using populations of riparian-dependent avifauna to monitor riparian health is ineffective under these conditions. Cinnamon teal habitat is present but limited both temporally and in area.

Nevertheless, marsh habitats are expected to be stable to improving as a result of ongoing fencing projects since the signing of the Forest Plan.

Riparian Associations: Open Water Habitat

The most common example of open water habitat on the KNF is the ubiquitous stock tanks created for livestock and wildlife. There are also a handful of lakes, including Cataract, Dogtown, Jacob, JD, Kaibab, Scholz, and Whitehorse. Most of the lakes identified above include significant developed recreation sites (Scholz Lake, a designated wildlife area, is an exception). The impacts of human disturbance and the associated changes in vegetation lower the wildlife values of these lakes, although the lakes still provide key habitat for waterfowl, wading birds, and foraging raptors. Lakes such Duck, Dry, and Davenport are ephemeral and may not hold standing water for years at a time. When flooded, these areas can provide excellent habitat.

Stock tanks tend to be created by building berms around water collection points. 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. When vegetation does grow on the berms, it typically consists of terrestrial species. Most stock tanks are devoid of riparian vegetation. 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 cattle and, to a lesser degree, deer and elk. The drier the habitat, the less ground cover there tends to be in the vicinity of the tanks.

Habitat immediately surrounding the tanks is generally poor for ground nesting, but the availability of the water itself is important to a wide range of avian and mammalian species, including a wide range of bat species (Mollhagen and Bogan 1997). A bat project on the NKRD netted isolated waters in August of two successive years and recorded over 20 species of bats, capturing a total of 2,282 individual bats (Painter 2003). Benefits to resident *populations* of birds due to the presence of stock tanks are unknown. The many tanks created in areas that were historically devoid of open water are assumed to enhance populations of upland bird species. Big Springs is a relatively unique perennial water on the NKRD that flows into two created ponds before 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 support submergent vegetation. An introduced population of rainbow trout has been established in these ponds for about 30 years.

The variety of pools, tanks, ponds, and lakes across the KNF create a wide range of habitat conditions. However, the consistent attribute associated with these wide ranging conditions is the scale of the habitat: riparian habitat occurs as a spatially and temporally limited resource on the KNF. There may be habitat to support individual pairs or family groups of riparian dependent species, but the habitat is too limited and too scattered to support viable populations. Monitoring riparian health through the presence of populations of avian species is not effective in areas that cannot consistently support populations of those species. In terms of MIS for riparian habitats, none of the open water features support the habitat required by Lucy's warbler or yellow-breasted chat, nor do they maintain enough habitat to support populations of cinnamon teal. The Forest Plan addresses stock tanks with stable water levels and the capacity to grow emergent vegetation. In ponderosa pine, mixed conifer, and spruce-fir cover types, 30 percent of the tank shoreline is to be fenced to exclude livestock. In pinyon-juniper habitat, 70 percent of the tank shoreline is to be fenced to exclude livestock and up to 5 acres of adjacent habitat is to be seeded to low height ground cover species. These goals have not yet been achieved, but fencing issues are addressed every year, both in terms of maintenance of past fencing projects as well as fencing new areas. The total number of stock tanks that meet these fencing goals for emergent vegetation is not available at the time of this writing, but the KNF database on structures for the South Zone identifies 160 fenced water sources of which 132 have been visually checked and had fencing repairs performed since the year 2000. The database does not indicate how many unfenced tanks may exist that should be fenced. Water availability may be more restricted than during "average" conditions due to the extended drought across much of north-central Arizona, but based on the fencing repair schedule, it is estimated that, in terms of management activities, open water habitat is at least stable across the KNF.

Stopover Ecology of Migratory Landbirds and the Oasis Theory

Given the mostly dry habitat cover types common to the KNF, areas with water and particularly with riparian vegetation are relatively unique. An assessment of their ecologic value should include any potential impacts to migratory as well resident birds. Conservation of long-distance migratory birds is difficult because of their complicated life-history characteristics and the spatial scales over which these species are associated (Yong et al. 1998). Typically, populations of birds are more vulnerable to negative changes in their rate of growth during specific seasons, such as nesting or post-fledging, but any time of year can act to limit populations of long-distance migratory birds (Hutto 1998). Populations can be affected during migration when inhospitable habitats must be crossed by species adapted to entirely different environments, especially when changes have occurred food, cover, or water availability (Hutto 1998, Yong and Finch 2002). In terms of this habitat assessment, the question becomes what, if any, impacts does management have towards the survival of landbirds resting en route on the KNF during seasonal migrations?

Relatively recent 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 that exceed a certain minimum size; the value of the habitat used by migratory birds appears closely related to food production at those sites.

Isolated oases (100 to 250 ac) can host more species of migrating birds than intermediate-sized oases (300 to 450 ac) or large (over 5,000 ac) corridors of riparian forest. However, larger size-classes can support more total birds and numbers of birds are more consistent between years than in isolated oases (Skagen et al.1998). There is no corridor of riparian habitat on the KNF and Davenport Lake (about 285 ac) is the only riparian association that is large enough to qualify as an isolated oasis. Complicating the situation is the fact that Davenport Lake is an ephemeral lake and the habitat available one year may bear little resemblance to the habitat found there the next year.

Insufficient energy stores may be the most limiting factor for individual birds during migration. Food availability, which correlates to the rate at which birds deposit fat, appears to be a key factor in stopover site selection (Yong et al. 1998, Yong and Finch 2002). Most marshy habitat on the KNF is less than 50 acres in size and much of it is ephemeral in nature. Stock tanks typically provide under a half-acre of surface water and are devoid of riparian vegetation. These are not the conditions that are likely to consistently support an abundant food base (e.g., arthropods) needed to attract migrant birds and allow them to rebuild fat and protein reserves. Also, the vast expanse of xeric habitat between the isolated patches of riparian habitat also decreases the likely value of these locations as stopover sites (Skagen et al. 1998, Yong et al. 1998, Hutto 2000, Kelly et al. 2000, Yong and Finch 2002). Similar to the discussion above relating to supporting individuals versus populations, the inherent value of the habitat must be viewed in terms of scale. Migrating birds have been documented using riparian habitats on the KNF (e.g., a flock of brown pelicans was reported resting at Davenport Lake on the Williams RD in spring 1993), but the riparian associations are not likely to be key in supporting populations of any given species.

Although the riparian associations across the KNF may not provide an adequate food base to consistently attract resting flocks of migratory birds, the stock tanks, marshes, ponds and lake shores that define riparian associations on the KNF are probably prime mosquito-breeding habitats compared to the surrounding xeric ponderosa pine and pinyon-juniper habitats. West Nile virus was identified for the first time in Arizona during 2003. If the virus does become established in the mosquito populations associated with waters on the KNF, it may have a significant affect on resident birds. However, due to the limited role these waters are thought to serve in relation to migrating birds, the potential establishment of the West Nile virus is not expected to affect populations of migratory birds.

Open water in the form of stock tanks is the most prevalent riparian association across the KNF. The wildlife values and uniqueness of the riparian features on the KNF remain high and are undoubtedly important at the local scale. However, these habitats are too limited in size, distribution, and occurrence to conclude that they support populations of migrating avifauna.

Management Indicator Species for Riparian Associations

Ephemeral wetlands support a number of migrant and resident species that could not otherwise exist across most of the Forest. However, the presence of individual pairs or family groups of birds does not mean that the scattered wetland habitat on the KNF supports viable populations of those species. In fact, after reviewing the current and potential wetland/marshland habitat available on the Forest, it is my conclusion that cinnamon teal, yellow-breasted chat, and Lucy's warblers are all inappropriate MIS for the KNF. Lucy's warblers and yellow-breasted chats prefer to nest in habitat that either does not exist on the KNF or exists only as limited, isolated patches.

Marsh habitat exists on both the NKRD and the SZ, although mostly as isolated areas (typically 2 to 100 acres each), fragments of habitat (i.e., measured in feet rather than acres), or ephemeral habitat dependent upon annual and/or seasonal weather patterns. Grazing can directly affect nesting cover, but most of the large, suitable areas are fenced to exclude livestock. The main areas not protected by fencing and that do offer potential teal habitat are ephemeral wetlands. These areas, e.g., Davenport and Dry Lakes, cannot support viable populations if they cannot provide consistent habitat. The temporary presence of an otherwise absent habitat provides clear benefits to the individual birds that may nest there during a wet year, but this is very different from habitat that can consistently support viable populations of a given species. Where nesting habitat does occur near open water, as required by cinnamon teal, it 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 well serve as sinks due to the temporal limitations and vulnerability to nest depredations rather than providing quality habitat.

It is important to note that the species portion of this document concludes that the above MIS are generally stable on the KNF. This portion of the document was written before I started on the Forest. Although the work was done in good faith, upon review of the documents upon which the conclusions were based and field visits to most of the key riparian areas and many of the stock tanks on the KNF, I do not agree with the current conclusions and will be addressing the inaccuracies in the individual species assessments as time and workload allow. The unsuitability of the riparian-dependent MIS on the KNF does not detract from or negate the inherent value of the riparian habitats. However, when assessing the values of these habitats, it is important to maintain a perspective on scale. The current MIS do not lend themselves to this kind of assessment.

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. The development of stock tanks across much of the Forest may have greatly expanded suitable habitat for resident birds and mammals, particularly for the bat species. Even ephemeral stock tanks can provide critical water needs for bats occupying arid environments (Mollhagen and Bogan 1997).

One low elevation riparian area exists on the KNF. Two high elevation riparian waters occur on the Forest: on the SZ. The remaining riparian classification is open water and consists primarily of stock tanks and several lakes that are recreation focal points, thus limiting habitat effectiveness.

Low Elevation Riparian (Kanab Creek): Management changes since the signing of the Forest Plan have resulted in an improving trend.

High Elevation Riparian (North Canyon Creek and Big Spring): Habitat trends have been stable to improving for both high elevations and riparian areas due to changes in the grazing programs.

Marshlands: 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.

<u>Overall:</u> Riparian habitat conditions across the KNF are stable to improving.

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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	*Turdus migratorius*	Stellar's jay	*Cyanocitta stelleri*
Blue grouse	*Dendragapus obscurus*	Williamson's sapsucker	*Sphyrapicus thyroideus*
Band-tailed pigeon	*Columba fasciata*	Chipmunks	*Tamias spp*
Hairy woodpecker	*Picoides villosus*	Cottontail rabbits	*Sylvilagus spp*
Mourning dove	*Zenaida macroura*	Mantled ground squirrel	*Citellus lateralis*
Northern flicker	*Colaptes auratus*	Red squirrel	*Tamiasciurus hudsonicus*
Red-naped sapsucker	*Sphyrapicus nuchalis*	Tassel-eared squirrel	*Sciurus aberti*

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: 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: 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	1996 to 2005 Trend	1996 to 2005 p	1996 to 2005 N	1996 to 2005 - C. I.	1996 to 2005 + C. I.	1996 to 2005 R.A.	1966 to 1978 Trend	1966 to 1978 p	1966 to 1978 n	1980 to 2005 Trend	1980 to 2005 p	1980 to 2005 n
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		
					Trend	p	N	- C. I.	+ C. I.	R.A.	Trend	p	n	Trend	p	n
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		
					Trend	p	N	- C. I.	+ C. I.	R.A.	Trend	p	n	Trend	p	n
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		
					Trend	p	N	- C. I.	+ C. I.	R.A.	Trend	p	n	Trend	p	n
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.

Year	Species	Habitat: Aspen	Mixed Conifer	Montane Grassland	Ponderosa Pine	Woodland / Grassland	Forest Wide
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

	Western wood-peewee		4	4	70	1	79
2006	Abert’s towhee				1		1
	American cow	1	2	1	1	8	13
	American robin		28	4	34	2	68
	Ash-throated flycatcher				17	59	76
	Audubon’s warbler	1	257	20	115	2	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	68	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	34	73
	Dark-eyed junco	5	96	25	157	7	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.