Appendix A for the Objection filed by NEC, AWR, WD and Y2U against the ANF Aspen Restoration Project.

Appendix A contains relevant cited portions of the following reports and/or publications.

Bartos, D. and R. Campbell. 1998. Decline of quaking aspen in the Interior West-examples from Utah. Rangelands 20 (1), February 1998.

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Decline of Quaking Aspen in the Interior West—Examples from Utah

Dale L. Bartos and Robert B. Campbell, Jr.

Quaking aspen (*Populus tremuloides*) are unique because, in contrast to most western forest trees, they reproduce primarily by suckering from the parent root system. Generally disturbance or dieback is necessary to stimulate regeneration of aspen stands. These self-regenerating stands have existed for thousands of years. If they are lost from the landscape, they will not return through normal seeding processes as do other tree species.

Aspen landscapes in the West provide numerous benefits, including forage for livestock, habitat for wildlife, watershed protection, water yield for downstream users, esthetics, sites for recreational opportunities, wood fiber, and landscape diversity.

Loss, or potential loss, of aspen on these lands can be attributed primarily to a combination of successional processes, reduction (or elimination) of fire, and long-term overuse by ungulates. Existing conditions indicate that most aspen stands will eventually be replaced by conifers, sagebrush, or possibly other shrub communities. The decline of aspen results in loss of water, forage, and biodiversity. Numerous landscapes throughout the West that were once dominated by aspen are in late successional stages dominated by mixed-conifer. If restoration treatments are to be successful, action must be taken soon.

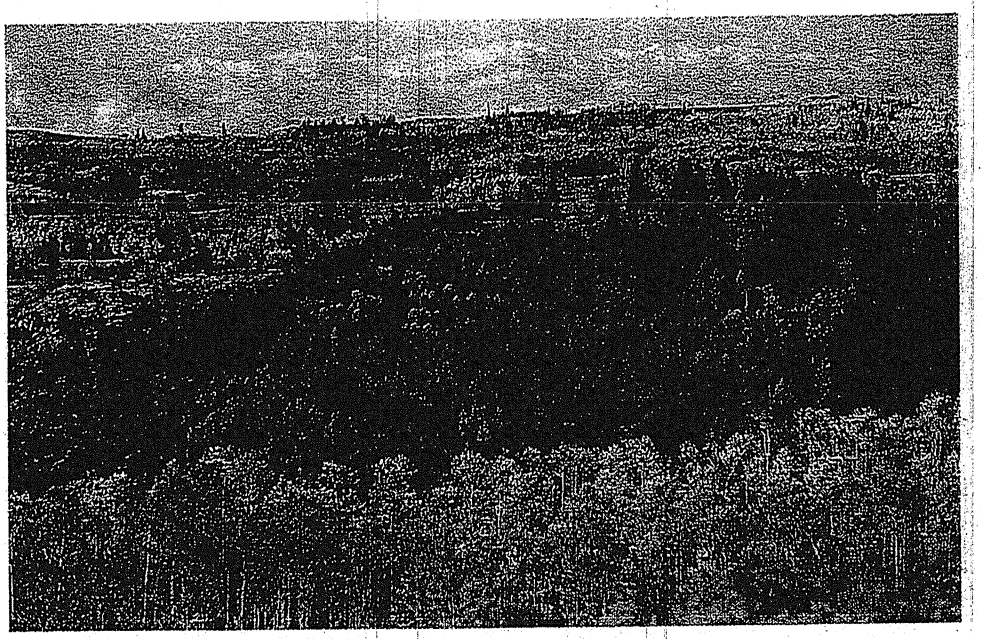


Fig. 1. *Aspen clones on the Fishlake National Forest have sparse regeneration (upper third) and show aspen being replaced by conifer (middle third). Photograph, Kreig Rasmussen.*

Aspen Decline

Figures 1–4 illustrate aspen decline as it currently exists in southern Utah.

Authors are Ecologist, U.S. Dept. of Agriculture, Forest Service. Bartos is with the Rocky Mountain Research Station, Logan, Ut. 84321 and Campbell is with the Fishlake National Forest, Richfield, Ut. 84701.

Changes in the abundance of aspen dominated landscapes have occurred over the past 125+ years partly as a result of livestock grazing, wildlife use, and a reduction in fires. The historical fire regime was altered in the mid-1800's after European settlement. Fire exclusion resulted from a combination of excessive grazing, timbering, and people extinguishing wildland fires. Grazing removed the fine fuels which generally carried the fires. Most of the historical fires were low-intensity ground fires and were not stand replacing.

Combined information related to these changes in Utah is shown in Figure 18a. There are about 2.1 million acres of National Forest Systems land in Utah (1.1 million acres

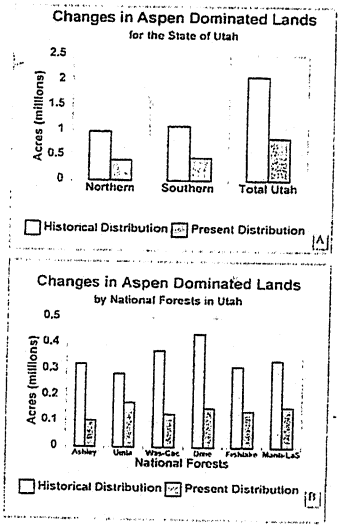


Fig. 18. *Loss of aspen in Utah from its historical distribution. Unpublished data supplied by the Survey Project (Renee O'Brien of the USDA Forest Service), Rocky Mountain Research Station, Ogden, Ut.*

in southern Utah) that contain aspen either living or dead. Most of this acreage was probably once dominated by aspen and has converted to conifer or sagebrush. Currently, of this total amount, only 800,000 acres (450,000 acres in southern Utah) is classified as aspen forest type. Figure 18b shows aspen decline for each of the six national forests in Utah. An approximately 60% decline in aspen dominated landscapes has occurred on National Forest System lands across Utah.

When aspen dominated landscapes convert to mixed-conifer losses occur. For every 1,000 acres that convert:

(1) Some 250 to 500 acre-feet of water is transpired into the atmosphere and not available for streamflow or undergrowth production. (Modified from Gifford,Humphries, and Jaynes 1984. A preliminary quantification of the impacts of aspen to conifer succession on water yield—II. Modeling results. Water Resources Bulletin 20(2):181–186).

(2) An estimated 500 to 1,000 tons of undergrowth biomass is not produced. Usable forage ranges between 40 and 70% of the undergrowth biomass.

(3) Numbers and kinds of plants and animals in the area decline appreciably.

Five risk factors for aspen dominated landscapes are:

(1) Conifer understory and overcover >25%

(2) Aspen canopy cover <40%

(3) Dominated aspen trees >100 years of age

(4) Aspen regeneration <500 stems/acre (5–15 feet tall)

(5) Sagebrush cover >10%

Any of these factors may indicate that the landscape is not in properly functioning condition.

Reversing the Trend

New research in aspen communities should have the following objectives:

(A) Evaluate the effects of various treatments to return late successional aspen stands to a composition that will perpetuate aspen's role in these ecosystems.

(B) Supplement our knowledge about what occurred on these various sites prior to European settlement.

Research should test the following hypotheses:

(1) Regeneration of aspen suckers differs with the management alternatives of burning, cutting, or a combination of the two.

(2) Soil characteristics change with treatments (burning and cutting) as well as with succession.

(3) Aspen root density has a direct relationship with regeneration success

(4) Pre-European settlement fire frequency within the mixed-conifer/aspen zone was generally uniform across various mountain ranges in southern Utah.

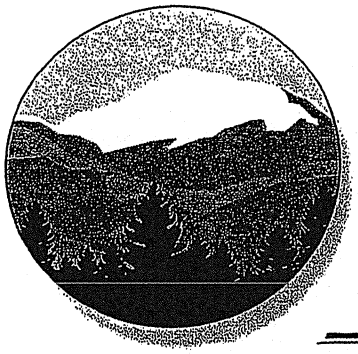
(5) Fire frequencies have significantly decreased in the mixed-conifer stands in southern Utah during the past 150 years.

(6) Landscape diversity has changed over the past 150 years.

(7) Use by livestock and wildlife limits the survival of aspen regeneration.

Successful aspen regeneration does not occur in some areas of the Interior West because of browsing by wildlife and/or livestock. Figure 19 is an example of the impact of cattle grazing on aspen regeneration. The entire area pictured was partially logged and broadcast burned several years earlier. Wildlife has equal access on both sides of this allotment pasture fence. Cattle also grazed these two pastures but with different intensities. Actions (treatments) to induce suckering must not be initiated until excessive browsing is controlled (Pamphlet from USDA-Forest Service, Southwest Region, 1994).

Fig. 19. *Fence-line contrast between allotment pastures showing effects of livestock usage. Photograph, Robert Campbell.*



Development of a Protocol for the Ecological Assessment of a Special Species

David Burton[1]

Abstract—*Developing consistent inventory and assessment protocols is important to people working on aspen issues in California and Nevada. Efforts have focused on identifying key indicators of ecological condition within aspen stands. The protocols have incorporated a range of factors that create or affect those indicators. Resulting ecological assessments conducted through the protocols describe stand structure, indicate unique stand management conditions, and record factors that might be putting stands at risk. Protocols for determining ecological condition were developed and field-tested by review groups in 2002. To date, units from seven state and federal agencies have collected data using the same protocols and field form.*

Overview

There has been heightened interest in aspen issues in the Far West as governmental mandates like the Sierra Nevada Forest Plan Amendment (U.S. Forest Service 2001) have increasingly directed land managers to protect the biodiversity of native flora and fauna. Many sub-units of management agencies have been inventorying, assessing, and treating aspen stands throughout their range, but prior to 2002 these efforts were not coordinated (U.S. Forest Service, Region 5, Aspen Survey Summary, May 2002).

The Aspen Delineation Project—a collaborative effort of the U.S. Forest Service's Pacific Southwest Region, California Department of Fish and Game, and the California Office of Bureau of Land Management—has began to develop a consistent approach for collecting data about ecological condition of aspen stands on agency lands. During workshops and field seminars co-sponsored by the Aspen Delineation Project, resource managers from multiple agencies and scientific disciplines identified the importance of developing and integrating consistent inventory, assessment, monitoring, and research protocols. The overriding objectives of this package of protocols were threefold: (1) to improve the scientific basis of management decisions, (2) to share data more easily within and between agencies, and (3) to produce more consistent evaluation of management practices, thus resulting in more effective adaptive management.

Because there was no consistency in how agencies were collecting data on location and condition of aspen stands, agency biologists and ecologists decided that the Aspen Delineation Project should initially develop a standardized protocol for (1) reconnaissance of the location of aspen stands and (2) assessment of a stand's ecological condition. Participants working on this project recognized that they would have to work fast due to time and budgeting constraints. It was agreed that the information gained using this protocol could lay the groundwork for establishing specific management

[1] Aspen Delineation Project, Penryn, CA.

objectives later on and be used as the foundation for decisions regarding more quantitative inventory or monitoring efforts.

A protocol and a data entry program were developed and field-tested by review groups during the summer of 2002. To date, sub-units from seven state and federal agencies—Forest Service, BLM, California F&G, California State Parks, California Tahoe Conservancy, Nevada State Parks, and Tahoe Regional Planning Agency—have collected data using the same protocol, field form, and data entry program.

Protocol for Establishing the Ecological Condition of Aspen Stands

Because the desired condition for aspen habitats across the Western landscape has been defined as fully functional, structurally diverse aspen stands (U.S. Forest Service 1994, U.S. Forest Service 2001), the principal objective of this protocol is to determine whether the age class, structural diversity, composition, and cover of individual aspen stands are within the desired range of natural variability for the vegetative community.

The range of naturally viable, fully functional aspen ecosystems does not express itself in any single simple form. Research has shown that aspen naturally expresses itself through the relationships between its reproduction processes; clones generating suckers off lateral roots; responses to natural environmental events; shade intolerance; growth hormonal balance mechanism; and disturbance dependency (Schier and others 1985).

However, there are a number of historic and current land management factors that have affected the natural viability of Western aspen habitats (Bartos 2001). The protocols focus on three thoroughly researched processes that have been shown to affect the natural viability of aspen: (1) effects of increased conifer encroachment caused by prolonged fire suppression, (2) failure of successful regeneration stemming from poor livestock management or unnatural wildlife stocking numbers, and (3) combinations of these factors. In order to see how these historic and current factors may have affected the current viability of individual stands, the protocol focuses on three key indicators of stand condition: overall stand structure, unique stand management issues, and a ranking of factors that might be putting stands at risk.

Stand Structure

The first key indicator in analyzing a stand's ecological condition is delineating its structure. The protocol's foundation is the use of a definition of "stand" (California Native Plant Society 2001) that calls for separating aspen units into units that are compositionally and structurally similar, i.e., if the mix of all species within the stand has structural and compositional integrity, it meets this definition (figure 1). This delineation of structure should establish not only the presence and condition of aspen age cohorts but also establish the relationship of those cohorts to the two most significant factors affecting stand viability: conifer encroachment and effects of browsing by wild and domestic ungulates.

The protocol divides the stand structure into three classes: overstory canopy cover (aspens and conifers >8 inches DBH), mid-level canopy cover (aspens and conifers 1 inch DBH to 8 inches DBH), and understory (aspens and conifers <1 inch DBH). The standard of 8 inches DBH was selected as

Figure 1—Structurally diverse stand with multiple age cohorts.

a candidate for the separation of overstory and mid-level classes after field-testing established that stems with 8 inches DBH were consistently shorter than the average height of the range of all larger DBH aged cohorts in the overstory. The first two classes—overstory and midlevel—focus on the presence or absence of aspen and conifer species and quantify their percentage presence and ratio to each other. The third class—understory—focuses on aspen regeneration and factors affecting its success. The protocol proceeds from class to class establishing a picture of the current structure of a stand and factors contributing to its functional condition.

The first step of the protocol is to identify what the actual percent of canopy cover is in the overstory and calls for five classes into which that percentage can be placed: 100-85 percent, 84-51 percent, 50-16 percent, 15-1 percent, and 0 percent. Then, the ratio of conifer:aspen is established within the canopy cover that does exist (figures 2, 3, 4). The classes for conifer: aspen ratio are 10:0, 8:2, 6:4, 5:5, 4:6, 2:8, and 0:10. The ratio establishes the dominance or co-dominance relationships between aspen and conifer and provides both an indication of current condition of the overstory canopy and hints of potential future conditions.

Next, after the actual overstory canopy cover is determined, the protocol calls for establishing the presence and condition of the actual mid-level canopy cover. As in the overstory, the protocol calls for establishing the presence or absence of a mid-level canopy cover (1 inch DBH to 8 inches DBH). If present, the actual percent of canopy cover in the mid-level canopy and the ratio of conifer:aspen in the mid-level canopy are established using the same size classes that were used in evaluating the overstory.

Finally, in the third class—the understory(<1 inch DBH)—the protocol calls for establishing the presence or absence of regeneration and identifying the presence or absence of browsing. Greater than 500 stems per acre is the class size used for signifying the presence of a distinctive regeneration stem aspen class. Five hundred stems per acre are used because it is believed that fewer stems per acre will not provide enough stems to establish a fully stocked stand (Mueggler 1989).

The protocol then calls for indicating the presence of significant browse on current year's terminal growth. Browse of 20 percent of current terminal

Figure 2—100-85 percent canopy cover and a ratio of conifer:aspen of 0:10. No other aspen or conifer age cohorts present.

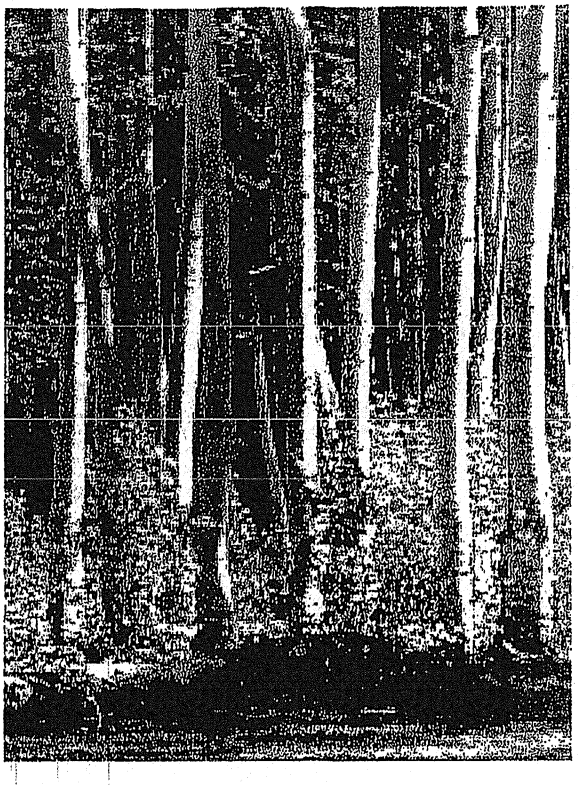


Figure 3—85-51 percent overstory canopy cover and a conifer:aspen ratio of 0:10 present. A midlevel canopy cover of 100-85 percent with a ratio of conifer:aspen of 10:0.

leader growth was selected because it is the current browse standard for woody stem vegetation found in the SNFPA (USDA Forest Service 2001) as well as in the standard used by the Bureau of Land Management in California (BLM—California State Office 2002).

A final note regarding stand structure: the protocol has an indicator for establishing the presence of a "decadent stand." Decadent stands are qualitatively described in this protocol as stands that have more dead or down aspen stems than living stems.

Management Issues

The second key indicator of a stand's ecological condition is delineation of any special stand management issues that provide clues to factors playing a role in the current status of the stand, a potential role in the future condition of the stand, or a role in potential stand management options. Included in the management issues are:

- location of stands within or adjacent to biologically sensitive habitats like meadows, riparian corridors, or springs;
- location of stands within or adjacent to geological refugia—environments conducive to stand

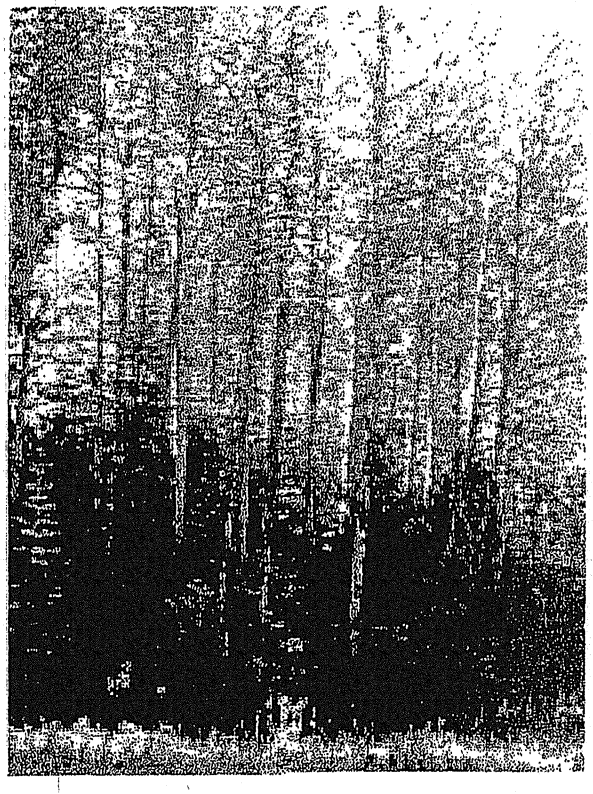


Figure 4—Two age cohorts with an overstory canopy cover of 85-51 percent and a ratio of conifer:aspen of 2:8 with an understory age cohort of >500 stems per acre.

continuation because of protection from browsing pressures or conifer encroachment or because of location on a site with particular geological characteristics such as lava flow, talus slope, rock outcrop, or moraine material;

- indication of significant (>20 percent) insect or disease damage;
- presence of significant human impacts such as regulated or non-regulated camping or structures;
- indication of signs of recent wildfires within or adjacent to the stand;
- indication of prior management treatment such as conifer removal within a stand or prescribed burns;
- indication of the presence or historic effect of beaver populations.

Risk Loss Analysis

The third key step in indicating a stand's ecological condition is evaluating combinations of factors observed in a stand and rating its potential for being lost. This procedure, which appears in the protocol as "Stand Loss Risk Factor," was initially set forth in a prioritized key for risk factors developed by Bob Campbell, Fish Lake NF, and Dale Bartos, RMRS (2001). The process, as it appears in the protocol, is the result of adaptation of the Campbell and Bartos work by the Eagle Lake Ranger District, Lassen National Forest and extensive field-testing of the end product.

Five classes of risk loss are found in the protocol:

Figure 5—Risk to stand is "Highest." This stand is being lost from above and not being replaced from below.

Highest: The clone is being lost from above and is not being replaced from below (figure 5).

- Conifer crowns have overtopped the aspen crowns, (primary risk factor), and conifer species comprise at least half the canopy (primary risk factor), ***and***
- Regeneration is absent or unsuccessful because of excessive browsing or shading from conifer encroachment (primary risk factor).

High: The clone is being lost from above or is not being replaced from below (figure 6).

Moderate: One or more risk factors are present, but the clone is not in immediate danger. This class may include one or more of the factors below:

- conifer closure >25 percent, but <50 percent (if >50 percent, rating is High or Highest)
- aspen cover <40 percent
- dominant aspen are decadent
- aspen regeneration 5-15 feet tall is <500 stems per acre
- regeneration is excessively shaded by conifers
- browsing is limiting extent and numbers of successful (>5 feet tall) regeneration

Figure 6—Risk to stand is "High." This stand is being lost from below through conifer encroachment.

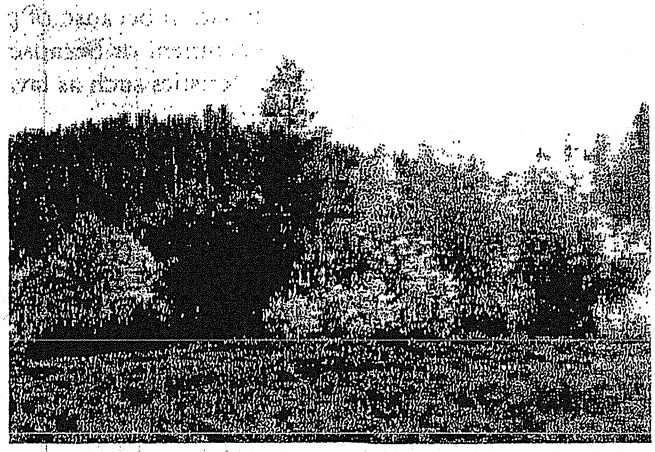


Figure 7—Risk to stand is "Low." Multiple age cohorts are present and few risk factors (<15 percent) exist.

Low: The clone is essentially healthy, containing mature trees and/or healthy and vigorous regeneration; there are no obvious signs that the clone has receded; and <15 percent of the clone is affected by risk factors (figure 7).

None: None of the above risk factors are present. Mature trees are vigorous, and regeneration 5-15 feet tall ≥500 stems per acre.

Support Material

To date, the Aspen Delineation Project has developed three support programs for facilitating effective and efficient use of the protocol:

- A Data Entry Program to simplify the process of moving hard copy (field form) data into GIS shape files as well as providing a vehicle for merging interagency aspen data into some centralized data source.
- A field crew training program to help crews understand and interpret goals and objectives of the assessment process and develop consistent interpretations of stand conditions.
- A CD containing PDF copies of the protocol and field form, a copy of the Data Entry Program, and an interactive guide to the protocol.

Copies of the CD or details about the protocol and the Aspen Delineation Project's field crew training program can be obtained by contacting peregrines@prodigy.net or the Aspen Delineation Project, P.O. Box 348, Penryn, CA 95663.

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1993

Aspen

HABITAT SUITABILITY FOR WILLIAMSON'S SAPSUCKERS IN MIXED-CONIFER FORESTS

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Abstract: Williamson's sapsuckers (*Sphyrapicus thyroideus*) have narrow habitat requirements and are sensitive indicators of change in intensively managed forest habitats of western North America. Thus, we studied habitat suitability for Williamson's sapsuckers at 99 4-ha sites (33 nest sites, 66 non-use sites) in mixed-conifer forests in Arizona during 1991. Nesting success of sapsuckers was high in this habitat (93.2% nest success, 0.0014 daily mortality, $n = 724$ nest days), and they preferred to nest in tall ($P < 0.05$) aspen snags ($P < 0.001$) near the bottom ($P = 0.012$) of snow-melt drainages with 0–20% of the canopies dominated by aspen. Sapsucker nest sites had particularly large ($P < 0.05$) live aspen and aspen snags in the surrounding area. Nest sites also had high ($P < 0.05$) snag densities ($\bar{x} = 7.65$ snags/ha) in the surrounding area, and these snag densities exceeded those commonly used in forest management plans. Effective snag management should concentrate snags in groups within low-lying areas and conserve large-sized snags. A Habitat Suitability Index (HSI) correctly predicted that Williamson's sapsuckers should generally prefer drainages over ridgetops, but the model could not distinguish between use and non-use sites within drainages. Future HSI models for Williamson's sapsucker should continue to stress snag density, but should consider aspen snag density separately from density of other snags, incorporate height and diameter of aspen snags, and use a more liberal definition of aspens contributing to overstory canopy cover.

snag density at nest = 3/ac

J. WILDL. MANAGE. 57(2):322–328

Forest managers are faced with trying to maintain viable populations of vertebrates while simultaneously managing for other resource needs such as timber production. This delicate balancing act depends upon knowing the effects of land management practices on habitat suitability for wildlife. The first step in attaining this knowledge is to identify habitat features that influence habitat suitability. Moreover, identification of habitat features that directly affect reproduction and survival is the most effective means of indexing habitat suitability because maintenance of viable populations depends on sustaining these fitness components (Martin 1992).

Cavity-nesting birds are a particularly appropriate group of species for examining effects of land management on habitat suitability because they are often affected by land management practices (Gysel 1961, Haapanen 1965, Hunter 1990). For instance, snags are less abundant in managed than unmanaged forests (Cline et al. 1980), and availability of snags commonly limits populations of cavity-nesting birds by limiting nesting opportunities (for review, see Li and Martin [1991], Martin and Li [1992]). In response, explicit policies for snag conservation have been implemented for managed stands (Hunter 1990). Yet, management for presence of snags is not sufficient; size, species, and age of snags, plus distribution of nest trees and vegetation in surrounding patches can affect presence and even reproductive success of birds (Flack 1976, Raphael and White 1984, Sedgwick and Knopf 1990, Li and Martin 1991). Key habitat features that influence habitat suitability, particularly with regard to reproduction or survival, need to be identified for individual species.

Williamson's sapsucker is the least numerous and most ecologically and genetically specialized species in its genus (Crockett and Hadow 1975, Johnson and Zink 1983). They are most common in middle-elevation forests of ponderosa pine (*Pinus ponderosa*) and Douglas-fir (*Pseudotsuga menziesii*), 2 forest types that are managed intensively for timber extraction in western North America (Thomas et al. 1979*b*). Within these intensively managed forests, Williamson's sapsuckers have narrow habitat requirements and limited versatility, making them sensitive indicators of environmental changes (Thomas et al. 1979*b*). Consequently, a Habitat

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overstory

Aspen / Birds

BIRD POPULATIONS OF ASPEN FORESTS IN WESTERN NORTH AMERICA

BY

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Department of Zoology
University of Wisconsin

ORNITHOLOGICAL MONOGRAPHS NO. 19

PUBLISHED BY

THE AMERICAN ORNITHOLOGISTS' UNION

1976

TABLE 1

BREEDING BIRDS OF ASPEN FORESTS
SUMMARY DATA FOR ALL STANDS SAMPLED, 1966–1969

	Relative abundance in each stand									
	Arizona					N. M.		Colorado		
Birds by nesting guild†	1	2	3	4	5	6	7	8	9	10
Canopy										
Western Wood Pewee	8	8	14	4			8	8	4	8
American Robin	8	8	4	6		2	6	10	10	8
Warbling Vireo	16	10	14	10	12	12	12	14	14	14
Audubon's Warbler	12	6	10	2		4	8	8	8	8
Western Tanager				2						
Evening Grosbeak	2		4	6						
Cassin's Finch										2
Shrubs										
Empidonax flycatcher		2	4	4			2	6	4	4
Black-headed Grosbeak			2	2			2	2		
Chipping Sparrow	4	2		2			2			
Song Sparrow										2
Holes										
Red-shafted Flicker			4	2				2		2
Yellow-bellied Sapsucker		2		2						
Hairy Woodpecker	2									
Downy Woodpecker	2		2	2			2			
Violet-green Swallow	6		12	4						6
Black-capped Chickadee										2
White-breasted Nuthatch	2									
Pigmy Nuthatch			2							
House Wren	8		14	6			6		4	10
Western Bluebird	4	2	4							
Mountain Bluebird	4	2	4	2			2	6	4	2
Ground										
Hermit Thrush	2	2	2	2	4	4	4	2	2	2
Townsend's Solitaire							2			
Gray-headed Junco	4	4	4	2	4	6	6	4	6	6
Summation Totals										
No. species	15	11	17	16	3	5	13	10	9	14
No. individuals	84	48	102	58	20	28	62	62	56	76
Others										
American Kestrel			P							
Great Horned Owl			2							
Long-eared Owl							2			
Broad-tailed Hummingbird	6	2	2	2	2	1	4	2	1	1
Brown-headed Cowbird	4	2	1	2						
Pine Siskin	2		2			3		2	2	4

Manti-La Sal National Forest Restoration and Fuels Reduction Prescribed Fire Project

DRAFT Terrestrial Biological Evaluation and Wildlife Specialist Report

NO 4046-20

Prepared by:
Angela Gatto, Wildlife Biologist
USDA Forest Service, Enterprise Program

for:
Manti-La Sal National Forest

September 28, 2021

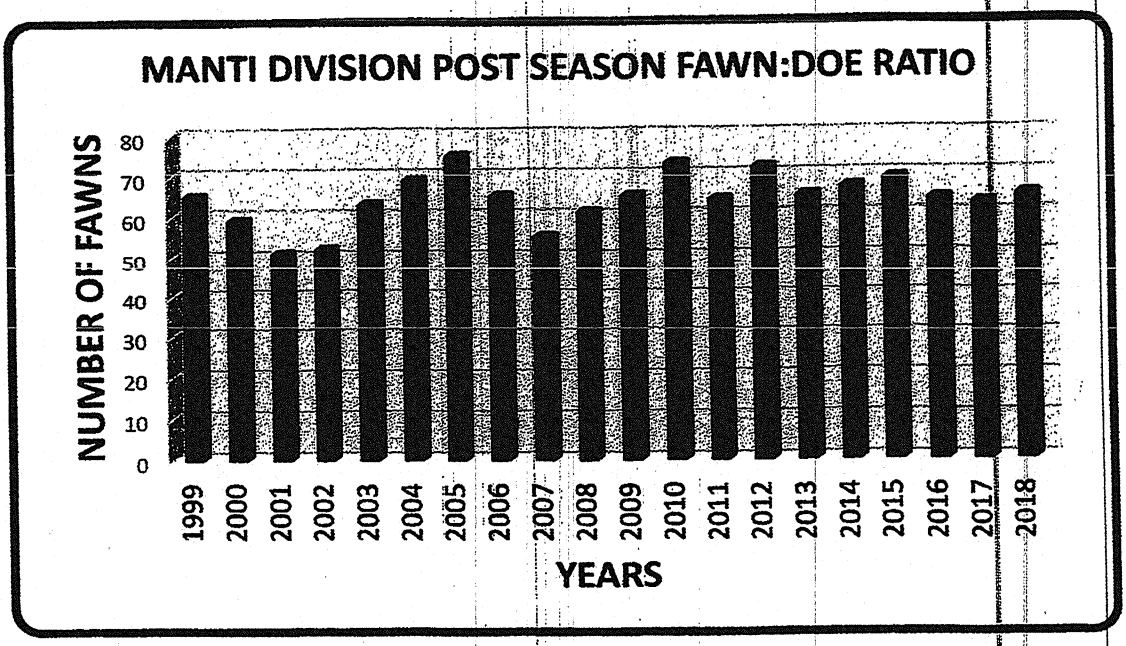

Figure 4. Post season fawns/100 does on the Manti Division from 1999 through 2018 (Bernales et al. 2018)

Direct and Indirect Effects: Adults and most fawns are expected to escape harm during project activities due to their high mobility. Pre-treatment and prescribed burning would create a short-term disturbance for individuals if present. However, habitat is not limited, and deer would likely move into undisturbed adjacent habitat during project implementation. Wood (2004) found that deer did not show avoidance of even heavily burned areas and heavily burned areas when adjacent to unburned areas could be beneficial to deer populations by increasing important forb and grass species cover while still providing adequate adjacent cover. Prescribed burning is expected to benefit deer in the long-term. Prescribed fire can restore and improve deer foraging habitat. This proposed project would not impact deer population trends.

Northern Goshawk

Goshawks have been found in a variety of forest ecosystems including lodgepole pine, aspen, ponderosa pine, Douglas-fir, and mixed forests throughout much of the northern hemisphere. Goshawk nest sites are usually located in dense mature forests with relatively large trees, near water, and on benches of relatively little slope (Graham et al. 1999). Closed canopies are important for protection and thermal cover, and relatively open under-stories are important to allow maneuverability during foraging. Data (district records) collected from the Wasatch Plateau between 1989 and 2013 show that over 80% of goshawk nests are in stands with a mixture of aspen and conifer species, with the remaining nest stands comprised of mixed conifer (primarily Engelmann spruce/sub-alpine fir) without aspen. Nearly 70 percent of all known nests have been in aspen trees, with proportionally fewer in Douglas-fir and spruce. Nests are often used year after year, but nest stands usually contain a number of alternate nests.

Goshawks are sensitive to human disturbance and have abandoned nests and young due to human activities that take place too close to their nest. The MLNF Land and Resources Management Plan directs that Forest Service management activities be restricted within a 30-acre area around

active goshawk nests. Restrictions within the 30-acre buffer around active nest sites would normally extend from 1 March through 30 September. In the 1980s an evaluation of 20-acre buffers around nest sites indicated that these small areas were not adequately protecting nest areas; in 1992 more comprehensive management recommendations suggested that managing for 6,000 acre territories to protect nest sites and provide adequate foraging habitat was more appropriate (Graham et al. 1999).

Suitable goshawk habitat is often heterogeneous, which supports a broad range of prey species; particularly those preferred by the goshawk: small mammals and birds including rabbits, squirrels, chipmunks, grouse, woodpeckers, jays and robins. Important forest components in Utah include snags, multiple canopies, and down woody debris (Graham et al. 1999).

Vegetation structural stages (VSS) are stages of forest growth and maturity and are used in defining the components of goshawk habitat. VSS 1 is classified as grass/forb with 0-1" dbh trees, VSS 2 is seedling/sapling with 1-5" dbh trees; VSS 3 is young forest with 5-12" dbh trees, VSS 4 is mid-age forest with 12-18" dbh trees, VSS 5 is mature forest with 18-24" dbh trees, and VSS 6 is old forest with 24" or greater trees. There are no VSS 6 within the treatment units, which could be a result of historic logging in this area.

Aspen regeneration is an important component in the life history of northern goshawks on the Manti-La Sal National Forest. The loss of aspen is concerning as well as the acreage of beetle killed spruce. Many nests have been lost due to wind or snow load causing dead spruce to fall over onto nest trees.

Population Trends: The goshawk population on the Manti-La Sal National Forest has remained relatively stable, although occupancy declined between 2008 - 2009. Looking at the number of active nests over the years would give the impression that the goshawk population on the Forest has steadily increased since the late 80s. A better indication of how the goshawk population is doing on the Forest would be the percent of monitored nests that were occupied each year, which is illustrated in Figure 5 below. The Manti-La Sal National Forest Plan Goshawk Amendment (USDA Forest Service 1986) states that a 20% decline in occupancy over a 3-year period is an acceptable range. We are currently within that range.

Aspen - Red Squirrels

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RESOURCE SELECTION AND PREDATION OF NORTH AMERICAN RED SQUIRRELS IN DECIDUOUS FOREST FRAGMENTS

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The North American red squirrel (*Tamiasciurus hudsonicus*) typically is regarded as having strong affinities for coniferous forests throughout its geographic range. In the state of Indiana, the red squirrel has expanded its geographic range concurrent with fragmentation of deciduous forests and widespread plantings of conifers. We undertook a radiotelemetry study to assess resource selection and survival of this species in 2 woodlots dominated by deciduous trees in west-central Indiana. Squirrels selected habitats with a high proportion of black walnut (*Juglans nigra*), avoided other hard mast-producing species, and avoided conifers. Squirrels exhibited overlapping home ranges, consistent with other studies on eastern populations. Individuals whose core areas contained conifers experienced higher survival rates than those individuals whose core areas did not contain conifers. We conclude that the increased safety conferred by conifers, as well as the presence of black walnut in these forests, are likely to contribute strongly to this species' persistence in Indiana.

Key words: deciduous forest, habitat, home range, red squirrel, resource selection, survival, *Tamiasciurus hudsonicus*

The North American red squirrel (*Tamiasciurus hudsonicus*; hereafter red squirrel) has expanded its geographic range into the state of Indiana concurrent with anthropogenic fragmentation of forests since the mid-1800s (Mumford and Whitaker 1982). Range expansion of the red squirrel is ongoing (R. K. Swihart, T. E. Nupp, J. R. Goheen, and S. Schnelker, in litt.) and probably has been facilitated by its superior dispersal ability relative to a fragmentation-sensitive competitor, the gray squirrel (*Sciurus carolinensis*—Goheen et al. 2003a; Nupp and Swihart 2001). In addition, range expansion of the red squirrel has coincided with the widespread propagation of conifer plantations in Indiana (Mumford and Whitaker 1982).

Red squirrels have strong coevolutionary ties with coniferous trees throughout the boreal forests of northern and western North America (Benkman 1995; Smith 1970), where they maintain and aggressively defend conifer cones from conspecifics in a central cache or larder (Steele 1998 and references therein). In eastern North America, dependence on conifer seeds is not as pronounced as in western populations, but eastern red squirrels display affinities for coniferous habitats nonetheless (Dempsey and Keppie 1993; Layne 1954; Riege 1991). Thus, range expansion of red squirrels also might be due in part to the availability of a resource that was absent historically.

Two aspects of coniferous habitats might facilitate persistence and further range expansion of red squirrel populations in Indiana. First, conifers might provide a resource preferred over other foods (e.g., hard mast) or habitats (e.g., deciduous trees), or a resource used as a "buffer" when preferred foods are scarce or unavailable (e.g., mast failure). In other regions of their geographic range, red squirrel populations are limited by food availability (Klenner and Krebs 1991; Sullivan 1990; but see Rusch and Reeder 1978), and survival of individuals has been correlated with food abundance (Halvorson and Engeman 1983). Second, coniferous trees might confer to squirrels some increased measure of safety from predators. Some authors have speculated that the permanent canopy cover and interlocking branches of coniferous trees afford more opportunities for squirrels to escape potential predators (Smith 1968; Vahle and Patton 1983). Rusch and Reeder (1978) suggested that predation drove differential survival rates between squirrels inhabiting coniferous and deciduous habitats in Alberta, with individuals in coniferous habitats experiencing lower mortality. Augmentation of food and decreased risk of predation are not mutually exclusive hypotheses, and they might operate synergistically with one another to promote range expansion of red squirrels into Indiana.

We conducted a radiotelemetry study of red squirrels in a highly fragmented portion of the central hardwoods region in west-central Indiana. Our primary objective was to quantify resource selection of, and determinants of home range size for,

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species might outweigh the benefits (Deutch 1978). In addition, the ability to visually detect and thus defend against intruders might be impaired in deciduous habitats (Deutch 1978).

Previous work in our system has demonstrated a positive relationship between presence of conifer and density of red squirrels (Nupp 1997) and, throughout Indiana, the occurrence of red squirrels in forest patches is strongly and positively related both to occurrence of conifer and of black walnut (J. R. Goheen and R. K. Swihart, in litt.). Because this pattern was well established prior to our study, we attempted to elucidate its causal mechanisms by conducting an intensive study within 2 forest patches differing in the occurrence and abundance of coniferous species. Although site differences partly explained variation in home range and core size, resource selection and factors influencing predation were similar between sites, lending support to our conclusions.

In nature, habitats containing high quality or quantities of food often are more risky with respect to predation (Houston et al. 1993). Although conifer was avoided within forest patches and was correlated with increased core sizes, squirrels whose core areas contained conifers had higher survival rates than those individuals having core areas in which conifers were unavailable. This was likely due to a reduction in the risk of raptor predation, either because of some increased measure of safety provided by conifers or because raptors preferred to hunt in deciduous habitats. Steury and Murray (2003) documented a decreased risk of predation with increasing territory size for red squirrels in coniferous forests in Idaho; however, this was likely due to a positive correlation between resource availability and territory size (Steury and Murray 2003).

Our study demonstrates that conifers were not a preferred resource for red squirrels, although it was not conducted long enough to assess the potential role of conifers as a buffer food source (see "Introduction"). Nonetheless, red squirrels did derive benefits from conifers. Throughout the state, the occurrence of red squirrels in forest patches is significantly related to the presence of conifer (J. R. Goheen and R. K. Swihart, in litt.). Thus, increased safety from predators derived from the presence of conifer may contribute to the range expansion and persistence of this species into Indiana.

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Aspen

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FEEDING ECOLOGY OF SHARP-SHINNED HAWKS NESTING IN DECIDUOUS AND CONIFEROUS FORESTS IN COLORADO[1]

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Abstract. Feeding ecology of 11 Sharp-skinned Hawk (*Accipiter striatus*) pairs nesting in aspen (*Populus tremuloides*), conifer (*Abies, Picea* spp.), and mixed aspen-conifer habitats in southwest Colorado was investigated during 1988–1989. Small birds ($\bar{x}$ = 20.9 g, SE = 0.8 g) and mammals ($\bar{x}$ = 41.1 g, SE = 3.3 g) comprised 91 and 9% of 513 prey identified at nests that fledged at least one young, respectively. Sixty percent of the birds eaten during the hawks' nestling and fledgling stages were nestlings or fledglings. Accordingly, median mass of birds eaten decreased from 17.4 g during incubation to 12.1 g during the nestling stage. Although more birds were consumed than mammals during all nesting stages (birds = 91.1%; mammals = 8.9%), the proportion of birds relative to mammals in diets progressively decreased from incubation through fledging. Taxa of birds in the diet were consumed in proportion to their occurrence in the most abundant of three different habitats surrounding nests, whereas some mammalian taxa were consumed in greater proportion than their relative "availability" in these habitats. This suggested that Sharp-shinned Hawks foraged opportunistically for birds, but may have selectively foraged for mammals. Differences in the habits of mammals (e.g., fossorial vs. terrestrial behavior), and hence their relative availability, may explain the apparent selection for certain species of mammals by Sharp-shinned Hawks.

Key Words: Accipiter striatus; *Sharp-shinned Hawk; diet; food habits; feeding ecology; aspen forest; conifer forest; Colorado.*

INTRODUCTION

Sharp-shinned Hawks (*Accipiter striatus*) are small, forest-dwelling raptors that feed primarily on small birds (10–30 g) associated with forest canopies (Craighead and Craighead 1956, Storer 1966, Reynolds and Meslow 1984). In Colorado, Sharp-shinned Hawks breed in quaking aspen (*Populus tremuloides*) and conifer (*Abies, Picea, Pseudotsuga*) forests. Forests in which hunting Sharp-shinned Hawks have been observed include mature aspen, conifer, and mixed aspen-conifer. Within mature aspen forests, however, these hawks nest only in patches of conifers (S. M. Joy and R. T. Reynolds, pers. observ.).

Smith and MacMahon (1981) demonstrated that avian species diversity and community composition differed among aspen, spruce, and fir forests in Utah and Idaho. If bird and mammal numbers differ among these forests in Colorado and if Sharp-shinned Hawks forage opportunistically, then diets of the hawks might reflect the relative abundance or availability of prey in forest types surrounding nests. Foraging opportunism would allow Sharp-shinned Hawks to use a variety of forest types in spite of different faunal communities, provided that sufficient food and nest sites were available.

We examined the diets of nesting Sharp-shinned Hawks in mature forests of aspen, conifer, and mixed aspen-conifer in Colorado during 1988–1989. We tested for differences in prey

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sizes and prey numbers among nesting pairs, among nesting stages, and between avian and mammalian prey. To evaluate the hypothesis that Sharp-shinned Hawks are opportunistic foragers, taxonomic composition and the size-frequency distribution of birds and mammals in Sharp-shinned Hawk diets were compared with estimates of relative abundances of birds and mammals in each of the three forest types.

STUDY AREA

The study was conducted on 28,750 ha of the Gunnison and Grand Mesa National Forests in west-central Colorado and included portions of Gunnison, Delta, and Mesa counties. Elevations ranged from 2,750–3,200 m. In western Colorado, aspen forests comprise as much as 25% of forested lands (Green and Van Hooser 1983). Within our study area, aspen existed as both seral and stable (contained no conifers) communities in small (<10 ha) patches and extensive (>500 ha) forests. In the study area, seral aspen patches and forests (often with emergent conifers in the aspen canopies) are successional to Englemann spruce (*P. engelmannii*) and subalpine fir (*A. lasiocarpa*), and at lower elevations blue spruce (*P. pungens*) (Langenheim 1962, Morgan 1969). Situated within these aspen forests were small to large areas of pure, or nearly pure, conifer forests. As a result, forests on the study area were composed of a mosaic of large (>500 ha), medium (200–500 ha), and small (<200 ha) patches of aspen, conifer, and mixed aspen-conifer forests. Throughout the study area, the majority (>70%) of aspen forests were in mature (70–120 year) and old-growth (120+ year) age classes (Shepperd 1990).

Vegetative composition of the understory in these forests varied with elevation, slope, aspect, and type of forest. Aspen and, to a lesser extent, mixed aspen-conifer had tall (>1 m), well-developed, herbaceous understories with a minor shrub component. Dominant forbs included butterweed groundsel (*Senecio serra*), white geranium (*Geranium richardsonii*), Barbey larkspur (*Delphinium barbeyi*), white-flowered peavine (*Lathyrus leucanthus*), and monkshood (*Aconitum columbianum*). Prominent low-growing plants included elk sedge (*Carex geyeri*), wild strawberry (*Fragaria ovalis*), yellow prairie violet (*Viola nuttallii*), and fringed brome (*Bromus ciliatus*). The shrub component consisted of snowberry (*Symphoricarpos* spp.), chokecherry (*Prunus virginiana*), and mountain-mahogany (*Cercocarpus montanus*).

Conifer understories were sparsely vegetated and dominated by low-growing (<1 m) herbs and shrubs including heart-leafed arnica (*Arnica cordifolia*), field horsetail (*Equisetum arvense*), elk sedge, myrtle blueberry (*Vaccinium myrtillus*) and kinnikinnik (*Arctostaphylos uva-ursi*). Plant names follow Weber (1976). Sheep and cattle foraged freely in the study area during July and August.

METHODS

DIET DETERMINATION

Areas intensively searched for nests within the study area were identified from aerial photographs (1:24,000) and topographic maps (7.5-min series) following Joy (1990) and included patches, as well as extensive areas, of aspen, conifer, and mixed aspen-conifer. Nest searches began in May in both 1988 and 1989 and were conducted throughout the nesting season. Prey remains (feathers, bills, feet, fur, skull fragments, and regurgitated pellets) were collected from nest sites as least once every seven days, and as often as every two days. Four days prior to changes in nesting stage (incubation to nestling, nestling to fledgling), prey remains were collected every two days. Activities of pairs (food exchanges between males and females, prey handling, feeding at nests, and roosting behavior) were recorded from blinds near all nests. Areas of approximately 5 ha, centered on nest trees, that contained prey-handling sites were thoroughly searched for prey remains during each visit to nests. Searches for prey remains often required 1–3 hr/visit. Prey remains were collected from nest sites until fledged young dispersed or nests failed.

Prey remains were sorted by nest and collection date. Avian remains (remiges, contour feathers, rectrices, feet, bills, and bones) were reconstructed and compared with National Museum of Natural History (Washington, D.C.) specimens for identification and a minimum count of individuals was determined as described by Reynolds and Meslow (1984). Single feathers of a species were excluded. Mammalian remains (fur, feet, tails, skull fragments, and teeth) were identified in a similar manner at the U.S. Fish and Wildlife Service museum in Fort Collins, Colorado.

Diet determinations based on the collection of prey remains may not include all prey consumed

viduals in 8-min circular variable plot censuses (Reynolds et al. 1980) at 15 stations (100-m spacing) in each plot. Stations in each of the 15 census plots were censused five times during June of each year (1988–1989). All censuses occurred between 05:30–12:30 MDT (D. M. Finch and R. T. Reynolds, pers. comm.).

Relative abundances of small mammals in each forest type were estimated by totaling the number of individuals per species captured once (excluding recaptures) during late-July to mid-September, 1988–1989, in the 15 plots (R. T. Reynolds and D. M. Finch, pers. comm.). Mammal trapping grids (10 × 10 at 15-m spacing) were nested within the larger bird census grids and had one Sherman trap (7.6 cm × 8.9 cm × 23.0 cm) at each of 100-grid points per plot (100 Sherman traps/plot) and one circular pit-fall trap (160 cm depth, 130 cm dia.) at every-other grid point (25 pitfall traps/plot). Small mammal plots were live-trapped for one session of six consecutive nights per year (750 trap nights/plot^{-1} year^{-1} × 2 year = 22,500 total trap nights) (R. T. Reynolds and D. M. Finch, unpubl. data). We assumed that these abundance estimates approximated the relative "availability" of birds and mammals to Sharp-shinned Hawks in the three forest types within our study area.

Forests within 2-km radii of each Sharp-shinned Hawk nest were classified as aspen (≥90% aspen trees in overstory), conifer (≥90% conifer trees in overstory), or mixed aspen-conifer (mixed stands with <90% aspen and <90% conifer trees in overstory) from aerial photographs (1:24,000) taken in October 1978 when the color contrast between aspen and conifers was at a maximum. The 2-km radius was selected because 2 km is about half the nearest-neighbor distance between nests of this hawk (Reynolds and Wight 1978, Clarke 1984) and circles with this radius encompass all estimates of home-range size in this species (see review in Reynolds 1983). Around each nest, the three forest types were assigned to one of three habitat-dominance categories (dominant, secondary, or limited) based on the proportion of the 2-km radius circle each forest type occupied. Forest types comprising ≥50% of the area around nests were considered "dominant". "Secondary" habitats were defined as forest types occupying >5% and <50% of the area. Forest types comprising ≤5% of the area were termed "limited".

Program PREFER (Johnson 1980) was used to examine whether birds or mammals in diets (n = 11 nests) reflected the relative abundance—by taxa (bird and mammal families) and size-class categories (after Storer 1966)—in the (1) dominant, (2) secondary, and (3) limited habitat-dominance categories surrounding nests. Null-hypotheses tested were that: 1) avian and 2) mammalian taxa were eaten in proportion to their abundance in each of the three habitat-dominance categories (dominant, secondary, and limited); and that 3) avian and 4) mammalian size-classes were eaten in proportion to their abundance in each of the three habitat-dominance categories. PREFER requires that the number of "preference components" (i.e., taxa and size-class categories) not exceed the number of "individuals" (i.e., nests) in tests (Johnson 1980). Thus, taxonomic categories of prey containing fewer than 2 individuals at all nests combined (families Scolopacidae, Trochilidae, Mimidae, and Mustelidae) were omitted from analyses. Mammalian prey in size-class categories nine (166–216 g), eight (125–166 g), six (64–91.1 g), and two (8–15.6 g) (Storer 1966) were also omitted because they were not tallied in prey remains or during census counts. Diet and relative-abundance data were paired by year to eliminate differences in prey abundance between years. Waller and Duncan's (1969) multiple comparison procedure was used in PREFER to assess relative "preferences".

Finally, we examined the overall pattern of food-resource use of nesting Sharp-shinned Hawks by comparing the hawks' diets to the entire range of bird and mammal sizes that occurred in the study area.

RESULTS

Diets of 11 Sharp-shinned Hawk nesting pairs were determined, four nests during 1988 and seven nests during 1989. Ten (91%) nests were in small (1–14 ha), insular conifer patches surrounded primarily by aspen (n = 8) or mixed aspen-conifer (n = 2) forests, and one nest was within a large, contiguous conifer forest. Prey were collected from all nest sites during all nesting stages in 1988. In 1989, five of seven nests failed prior to fledging. Prey remains were collected from all seven nest sites during incubation, from four nest sites during the nestling stage, and two nest sites during the fledgling stage. A total of 686 prey items was identified (Appendix 1), including 53 species (39 genera, 14 families) of

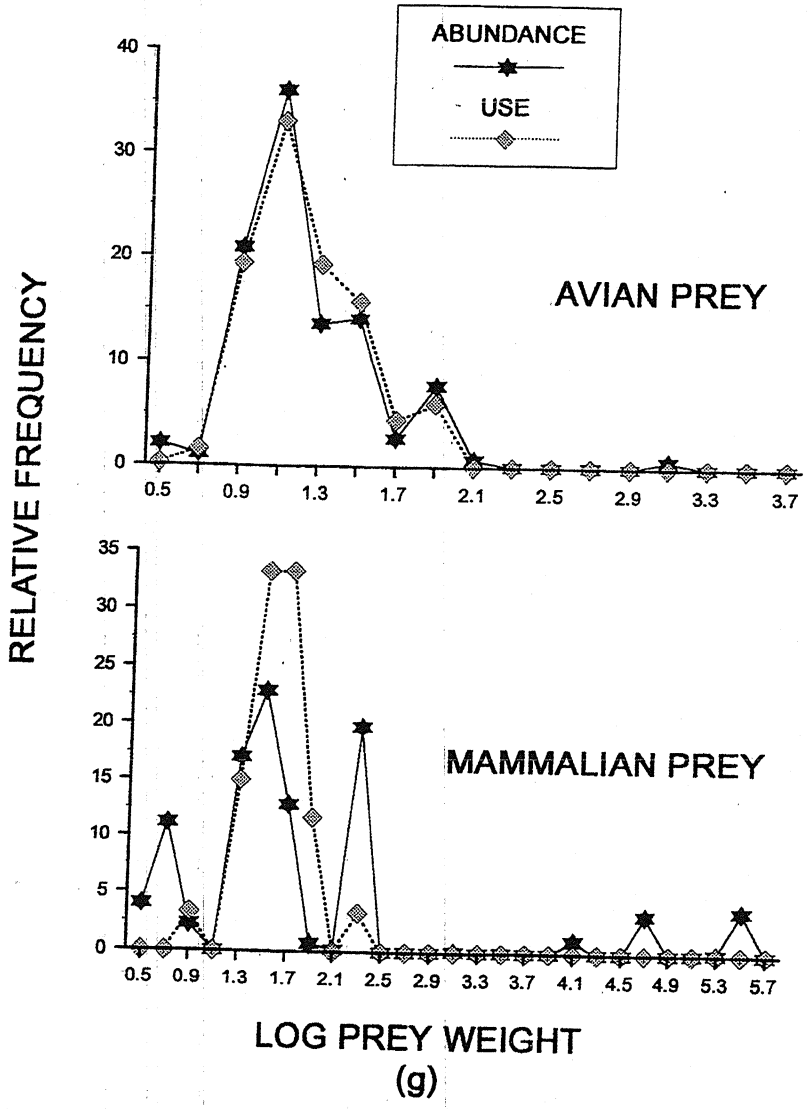

FIGURE 5. Relative abundance of birds and mammals by size class and their relative frequency in diets of Sharp-shinned Hawks in aspen and conifer forests in Colorado, 1988–1989.

ing fossorial vs. terrestrial behavior of Geomyidae and Muridae).

Although aspen and mixed aspen-conifer habitats comprised the majority of dominant and secondary habitats surrounding nests, all Sharp-shinned Hawks nested in conifer trees. The singular use of conifers, which have long and dense crowns, for nesting is probably related to a need for this small hawk to hide its nest from predators (Reynolds 1989).

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REPORTS

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Decline of the North American avifauna

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Species extinctions have defined the global biodiversity crisis, but extinction begins with loss in abundance of individuals that can result in compositional and functional changes of ecosystems. Using multiple and independent monitoring networks, we report population losses across much of the North American avifauna over 48 years, including once common species and from most biomes. Integration of range-wide population trajectories and size estimates indicates a net loss approaching 3 billion birds, or 29% of 1970 abundance. A continent-wide weather radar network also reveals a similarly steep decline in biomass passage of migrating birds over a recent 10-year period. This loss of bird abundance signals an urgent need to address threats to avert future avifaunal collapse and associated loss of ecosystem integrity, function and services.

Slowing the loss of biodiversity is one of the defining environmental challenges of the 21st century (*1–5*). Habitat loss, climate change, unregulated harvest, and other forms of human-caused mortality (*6, 7*) have contributed to a thousand-fold increase in global extinctions in the Anthropocene compared to the presumed prehuman background rate, with profound effects on ecosystem functioning and services (*8*). The overwhelming focus on species extinctions, however, has underestimated the extent and consequences of biotic change, by ignoring the loss of abundance within still-common species and in aggregate across large species assemblages (*2, 9*). Declines in abundance can degrade ecosystem integrity, reducing vital ecological, evolutionary, economic, and social services that organisms provide to their environment (*8, 10–15*). Given the current pace of global environmental change, quantifying change in species abundances is essential to assess ecosystem impacts. Evaluating the magnitude of declines requires effective long-term monitoring of population sizes and trends, data which are rarely available for most taxa.

Birds are excellent indicators of environmental health and ecosystem integrity (*16, 17*), and our ability to monitor many species over vast spatial scales far exceeds that of any other animal group. We evaluated population change for 529 species of birds in the continental United States and Canada (76% of breeding species), drawing from multiple standardized bird-monitoring datasets, some of which provide close to fifty years of population data. We integrated range-wide estimates of population size and 48-year population trajectories, along with their associated uncertainty, to quantify net change in numbers of birds across the avifauna over recent decades (*18*). We also used a network 143 weather radars (NEXRAD) across the contiguous U.S. to estimate long-term changes in nocturnal migratory passage of avian biomass through the airspace in spring from 2007 to 2017. The continuous operation and broad coverage of NEXRAD provide an automated and standardised monitoring tool with unrivaled temporal and spatial extent (*19*). Radar measures cumulative passage across all nocturnally migrating species, many of which breed in areas north of the contiguous U.S. that are poorly monitored by avian surveys. Radar thus expands the area and the proportion of the migratory avifauna that is sampled relative to ground surveys.

Results from long-term surveys, accounting for both increasing and declining species, reveal a net loss in total abundance of 2.9 billion (95% CI = 2.7-3.1 billion) birds across almost all biomes, a reduction of 29% (95% CI = 27-30%) since 1970 (Fig. 1 and Table 1). Analysis of NEXRAD data indicate a similarly steep decline in nocturnal passage of migratory biomass, a reduction of 13.6 ± 9.1% since 2007 (Fig. 2A). Reduction in biomass passage occurred across the eastern U.S. (Fig. 2, C and D), where migration is dominated by large numbers of temperate- and boreal-breeding songbirds; we observed no consistent trend in the Central or Pacific flyway regions (Fig. 2, B to D, and table S5). Two completely different and independent monitoring techniques thus signal major population loss across the continental avifauna.

Species exhibiting declines (57%, 303/529) based on long-

term survey data span diverse ecological and taxonomic groups. Across breeding biomes, grassland birds showed the largest magnitude of total population loss since 1970—more than 700 million breeding individuals across 31 species— and the largest proportional loss (53%); 74% of grassland species are declining. (Fig. 1 and Table 1). All forest biomes experienced large avian loss, with a cumulative reduction of more than 1 billion birds. Wetland birds represent the only biome to show an overall net gain in numbers (13%), led by a 56% increase in waterfowl populations (Fig. 3 and Table 1). Surprisingly, we also found a large net loss (63%) across 10 introduced species (Fig. 3, D and E, and Table 1).

A total of 419 native migratory species experienced a net loss of 2.5 billion individuals, whereas 100 native resident species showed a small net increase (26 million). Species overwintering in temperate regions experienced the largest net reduction in abundance (1.4 billion), but proportional loss was greatest among species overwintering in coastal regions (42%), southwestern aridlands (42%), and South America (40%) (Table 1 and fig. S1). Shorebirds, most of which migrate long distances to winter along coasts throughout the hemisphere, are experiencing consistent, steep population loss (37%).

More than 90% of the total cumulative loss can be attributed to 12 bird families (Fig. 3A), including sparrows, warblers, blackbirds, and finches. Of 67 bird families surveyed, 38 showed a net loss in total abundance, whereas 29 showed gains (Fig. 3B), indicating recent changes in avifaunal composition (table S2). While not optimized for species-level analysis, our model indicates 19 widespread and abundant landbirds (including 2 introduced species) each experienced population reductions of >50 million birds (data S1). Abundant species also contribute strongly to the migratory passage detected by radar (*19*), and radar-derived trends provide a fully independent estimate of widespread declines of migratory birds.

Our study documents a long-developing but overlooked biodiversity crisis in North America—the cumulative loss of nearly 3 billion birds across the avifauna. Population loss is not restricted to rare and threatened species, but includes many widespread and common species that may be disproportionately influential components of food webs and ecosystem function. Furthermore, losses among habitat generalists and even introduced species indicate that declining species are not replaced by species that fare well in human-altered landscapes. Increases among waterfowl and a few other groups (e.g., raptors recovering after the banning of DDT) are insufficient to offset large losses among abundant species (Fig. 3). Importantly, our population loss estimates are conservative since we estimated loss only in breeding populations. The total loss and impact on communities and ecosystems could be even higher outside the breeding season if we consider the amplifying effect of "missing" reproductive output from these lost breeders.

Extinction of the Passenger Pigeon (*Ectopistes migratorius*), once likely the most numerous bird on the planet, provides a poignant reminder that even abundant species can go extinct rapidly. Systematic monitoring and attention paid to population declines could have alerted society to its pending extinction (*20*). Today, monitoring data suggest that avian declines will likely continue without targeted conservation action, triggering additional endangered species listings at tremendous financial and social cost. Moreover, because birds provide numerous benefits to ecosystems (e.g., seed dispersal, pollination, pest control) and economies (47 million people spend 9.3 billion U.S. dollars per year through bird-related activities in the U.S. (*21*)), their population reductions and possible extinctions will have severe direct and indirect consequences (*10, 22*). Population declines can be reversed, as evidenced by the remarkable recovery of waterfowl populations under adaptive harvest management (*23*) and the associated allocation of billions of dollars devoted to wetland protection and restoration, providing a model for proactive conservation in other widespread native habitats such as grasslands.

Steep declines in North American birds parallel patterns of avian declines emerging globally (*14, 15, 22, 24*). In particular, depletion of native grassland bird populations in North America, driven by habitat loss and more toxic pesticide use in both breeding and wintering areas (*25*), mirrors loss of farmland birds throughout Europe and elsewhere (*15*). Even declines among introduced species match similar declines within these same species' native ranges (*26*). Agricultural intensification and urbanization have been similarly linked to declines in insect diversity and biomass (*27*), with cascading impacts on birds and other consumers (*24, 28, 29*). Given that birds are one of the best monitored animal groups, birds may also represent the tip of the iceberg, indicating similar or greater losses in other taxonomic groups (*28, 30*).

Pervasiveness of avian loss across biomes and bird families suggests multiple and interacting threats. Isolating spatio-temporal limiting factors for individual species and populations will require additional study, however, since migratory species with complex life histories are in contact with many threats throughout their annual cycles. A focus on breeding season biology hampers our ability to understand how seasonal interactions drive population change (*31*), although recent continent-wide analyses affirm the importance of events during the non-breeding season (*19, 32*). Targeted research to identify limiting factors must be coupled with effective policies and societal change that emphasize reducing threats to breeding and non-breeding habitats and minimizing avoidable anthropogenic mortality year-round. Endangered species legislation and international treaties, such as

the 1916 Migratory Bird Treaty between Canada and the United States, have prevented extinctions and promoted recovery of once-depleted bird species. History shows that conservation action and legislation works. Our results signal an urgent need to address the ongoing threats of habitat loss, agricultural intensification, coastal disturbance, and direct anthropogenic mortality, all exacerbated by climate change, to avert continued biodiversity loss and potential collapse of the continental avifauna.

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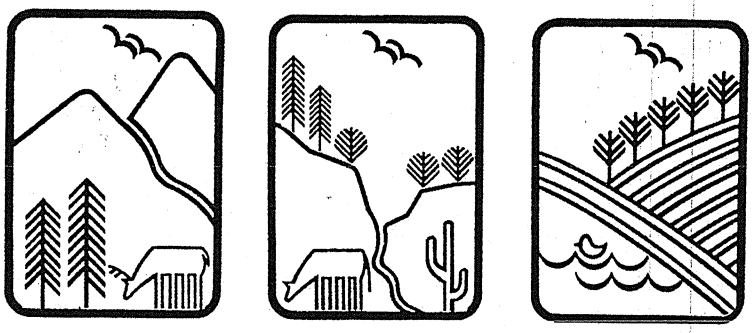
Table 1. Net change in abundance across the North American avifauna, 1970-2017. Species are grouped into native and introduced species, management groups (landbirds, shorebirds, waterbirds, waterfowl), major breeding biomes, and nonbreeding biomes (see data S1 in (*18*) for assignments and definitions of groups and biomes). Net change in abundance is expressed in millions of breeding individuals, with upper and lower 95% credible intervals (CI) shown. Percentage of species in each group with negative trend trajectories are also noted. Values in bold indicate declines and loss; those in italics indicate gains.

Species Group	Number of Species	Net Abundance Change (Millions) & 95% CI			Percent Change & 95% CIs			Proportion Species in Decline
		Change	LC95	UC95	Change	LC95	UC95	
Species Summary								
All N. Am. Species	529	**-2,911.9**	-3,097.5	-2,732.9	**-28.8%**	-30.2%	-27.3%	57.3%
All Native Species	519	**-2,521.0**	-2,698.5	-2,347.6	**-26.5%**	-28.0%	-24.9%	57.4%
Introduced Species	10	**-391.6**	-442.3	-336.6	**-62.9%**	-66.5%	-56.4%	50.0%
Native Migratory Species	419	**-2,547.7**	-2,723.7	-2,374.5	**-28.3%**	-29.8%	-26.7%	58.2%
Native Resident Species	100	*26.3*	7.3	46.9	*5.3%*	1.4%	9.6%	54.0%
Landbirds	357	**-2,516.5**	-2,692.2	-2,346.0	**-27.1%**	-28.6%	-25.5%	58.8%
Shorebirds	44	**-17.1**	-21.8	-12.6	**-37.4%**	-45.0%	-28.8%	68.2%
Waterbirds	77	**-22.5**	-37.8	-6.3	**-21.5%**	-33.1%	-6.2%	51.9%
Waterfowl	41	*34.8*	24.5	48.3	*56.0%*	37.9%	79.4%	43.9%
Aerial Insectivores	26	**-156.8**	-183.8	-127.0	**-31.8%**	-36.4%	-26.1%	73.1%
Breeding Biome								
Grassland	31	**-717.5**	-763.9	-673.3	**-53.3%**	-55.1%	-51.5%	74.2%
Boreal forest	34	**-500.7**	-627.1	-381.0	**-33.1%**	-38.9%	-26.9%	50.0%
Forest Generalist	40	**-482.2**	-552.5	-413.4	**-18.1%**	-20.4%	-15.8%	40.0%
Habitat Generalist	38	**-417.3**	-462.1	-371.3	**-23.1%**	-25.4%	-20.7%	60.5%
Eastern Forest	63	**-166.7**	-185.8	-147.7	**-17.4%**	-19.2%	-15.6%	63.5%
Western forest	67	**-139.7**	-163.8	-116.1	**-29.5%**	-32.8%	-26.0%	64.2%
Arctic Tundra	51	**-79.9**	-131.2	-0.7	**-23.4%**	-37.5%	-0.2%	56.5%
Aridlands	62	**-35.6**	-49.7	-17.0	**-17.0%**	-23.0%	-8.1%	56.5%
Coasts	38	-6.1	-18.9	8.5	-15.0%	-39.4%	21.9%	50.0%
Wetlands	95	*20.6*	8.3	35.3	*13.0%*	5.1%	23.0%	47.4%
Nonbreeding Biome								
Temperate North America	192	**-1,413.0**	-1,521.5	-1,292.3	**-27.4%**	-29.3%	-25.3%	55.2%
South America	41	**-537.4**	-651.1	-432.6	**-40.1%**	-45.2%	-34.6%	75.6%
Southwestern Aridlands	50	**-238.1**	-261.2	-215.6	**-41.9%**	-44.5%	-39.2%	74.0%
Mexico-Central America	76	**-155.3**	-187.8	-122.0	**-15.5%**	-18.3%	-12.6%	52.6%
Widespread Neotropical	22	**-126.0**	-171.2	-86.1	**-26.8%**	-33.4%	-19.3%	45.5%
Widespread	60	-31.6	-63.1	1.6	-3.7%	-7.4%	0.2%	43.3%
Marine	26	**-16.3**	-29.7	-1.2	**-30.8%**	-49.1%	-2.5%	61.5%
Coastal	44	**-11.0**	-14.9	-6.7	**-42.0%**	-51.8%	-26.7%	68.2%
Caribbean	8	-6.0	1.4	-15.7	12.1%	-2.8%	31.7%	25.0%

Rocky Mountains | Southwest | Great Plains



Aspen

Research Note RM-482

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Breeding Birds and Small Mammals in Pole-Sized Lodgepole Pine and Small Inclusions of Aspen in Central Colorado

Virgil E. Scott[1] and Glenn L. Crouch[2]

Numbers of birds and mammals in small aspen stands within a pole-sized lodgepole pine forest were compared with those found in the surrounding conifer forest. Some birds and one mammal that are usually associated with conifer forests were more abundant in the aspen than in the lodgepole pine. However, the aspen stands appear too small to provide habitat for aspen obligates.

Keywords: wildlife habitat, lodgepole pine, aspen

65 45 LPP

Introduction

Aspen (*Populus tremuloides*) is the major deciduous tree species in the central Rocky Mountains. In the western United States, aspen is a component of many forest types, but rarely covers extensive areas as the dominant overstory species except in Colorado and Utah (Daniel 1979). In the past, aspen in the West has had little market potential but demand has increased in recent years. The importance of the aspen community to forest birds has been documented by several authors (Flack 1976, Salt 1957, Winternitz 1976). Information about small aspen stands intermixed with coniferous forests and how they influence diversity and density of birds and small mammals is not available. This note compares species composition and densities of birds and small mammals in small, isolated aspen stands (≅0.1 acre) within a lodgepole pine (*Pinus contorta*) forest with those found in the surrounding lodgepole pine.

Study Area

The study site is near the main entrance to the Fraser Experimental Forest in Colorado, at about 9,000 feet elevation (Alexander et al. 1985). The terrain is relatively gentle, and the overstory consists mostly of 65-year-old lodgepole pine, with scattered inclusions of small, isolated patches of aspen.

Understory vegetation in the lodgepole stands resembles that described by Hess and Alexander (1986) and others as occurring in *Pinus contorta/Vaccinium scoparium* plant associations. On the Fraser Experimental Forest, these lodgepole pine dominated communities are seral to *Abies lasiocarpa/Vaccinium scoparium* plant associations. Understories are dominated primarily by *Vaccinium* spp. but also include a few other low growing shrubs, a sparse cover of graminoids (mostly sedges), and a few forbs (Crouch 1986).

The small patches of aspen are also seral to the conifer associations. They contain a few conifers and somewhat more herbaceous cover than the lodgepole stands.

Methods

Inventory Plots

Fourteen near-circular stands of aspen, averaging about 0.1 acre, were selected for study. Fourteen similar-sized sites were selected in lodgepole pine, at least 400 feet from the aspen stands. Within each study site, canopy cover was determined with a spherical densitometer, tree diameters at breast height were measured with a diameter tape, and heights were determined with a clinometer.

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Birds

Birds were counted on a 0.1–acre (37–foot radius) circular plot centered in each study site between sunrise and 3 hours after sunrise, in June and early July in 1982 and 1983. Five inventories were conducted each year to assure that peak breeding activity was encompassed. Means of the counts of each species were used to estimate population densities.

Mammals

Small mammals were trapped at five stations on each bird survey plot. One station was placed at the plot center and one 25 feet from the center in each cardinal direction. One rat trap and two museum specials were placed at each station and were baited with a peanut butter-oatmeal mix. Traps were checked twice daily for 3 consecutive days. Data from each stand were considered one observation for analyses.

Data Analyses

Differences ($P = 0.10$) in densities of birds and mammals between aspen and lodgepole sites were tested using two-way analyses of variance.

Results and Discussion

Overstory Characteristics

Tree diameters and heights tended to be greatest in the lodgepole and smallest in the aspen plots (table 1). The largest number of trees per acre (2,442) occurred in the aspen and the fewest (1,384) in the lodgepole plots. Aspen accounted for 82% and 1% of the trees in the aspen and lodgepole plots, respectively. Lodgepole pine accounted for virtually all of the trees in the conifer and 12% of those in the small aspen stands. A few subalpine fir (*Abies lasiocarpa*) and Engelmann spruce (*Picea engelmannii*) trees were present in both overstory categories.

Table 1.—Overstory characteristics of study plots in pole-sized lodgepole pine and small aspen stands on the Fraser Experimental Forest, Colorado.

	Lodgepole pine plots	Small aspen stands
Canopy cover (%)	85	92
Basal area (ft.^{2}/acre)	188	189
Total trees (no./acre)	1,384	2,442
Composition (%)		
Aspen	1	82
Lodgepole pine	98	12
Engelmann spruce	0	1
Subalpine fir	1	5
Tree heights (ft.)	46	34
Diameters (in.)	4.5	3.3
Snags > 6 in. d.b.h. (no./acre)	13	6

Six and 13 snags per acre, ranging from 6 to 9 inches d.b.h., respectively, were found in aspen and lodgepole stands. No snags over 10 inches d.b.h. were found.

Bird Densities

Total numbers of birds were higher in aspen stands than in lodgepole pine plots (table 2). Species diversity was not different between overstory categories (fig. 1).

Three of the four species of birds that were more abundant in aspen are generally considered birds of conifer forests. Of these, yellow-rumped warblers,[3] usually found in conifers (Johnsgard 1979, Terres 1980), also prefer open woodlands or edges and inhabit aspen woodlands in the Rocky Mountains (Flack 1976). Most of the yellow-rumped warblers observed in aspen stands were using aspen trees. Red-breasted nuthatches and ruby-crowned kinglets also were more numerous in aspen; but more than one-half of the observed kinglets were in conifers in the aspen plots.

Dark-eyed juncos are ground feeding and ground nesting birds and generally prefer forests with grass-forb understories (Diem 1980). More juncos were counted in aspen than lodgepole plots.

Warbling vireos, birds of deciduous forests, were not observed in lodgepole plots and were counted in only three (11%) of the aspen plots.

In the aspen plots, 81% of the mountain chickadees observed were in aspen trees, and the remainder were in the few conifers growing there.

No differences were detected among other species.

Small Mammals

The total number of small mammals caught was greater in aspen than in lodgepole pine plots (table 3).

Two species of shrews (*Sorex cinereus, S. monticolus*) were caught in aspen, but not in lodgepole plots (table 3). Few least chipmunks were caught, and there was no difference in numbers between overstory categories. Few deer mice were caught, and only on lodgepole plots. Southern red-backed voles are usually associated with conifer forests (Armstrong 1972) and prefer areas with slash or woody material on the ground (Gashwiler 1970). Regardless, red-backed voles were more abundant in aspen than lodgepole plots in this study. The number of montane voles caught on aspen and lodgepole plots was not different.

Conclusions

Results indicate that the presence of small aspen stands within a nearly pure pole-sized lodgepole pine forest may result in an increase in the overall density of birds and mammals. However, data[4] from a 20–acre aspen stand

[3] *Scientific names of birds and mammals are listed in tables 2 and 3.*

[4] *Data in the files of the senior author.*

Table 2.—Estimated number of birds per 100 acres (standard error) in pole-sized lodgepole pine and small aspen stands on the Fraser Experimental Forest, Colorado.[1]

	Means of 1982 and 1983	
	Lodgepole pine	Aspen
Gray jay *(Perisoreus canadensis)*	19 (10.6)	12 (8.3)
Mountain chickadee *(Parus gambeli)*	50 (17.1)	82 (13.8)
Red-breasted nuthatch *(Sitta canadensis)*	2b (2.4)	14a (6.2)
Ruby-crowned kinglet *(Regulus calendula)*	24b (4.5)	62a (19.1)
Townsend's solitaire *(Myadestes townsendi)*	0 (0.0)	4 (2.4)
Hermit thrush *(Catharus guttatus)*	12 (5.9)	12 (4.9)
Swainson's thrush *(Catharus ustulatus)*	0 (0.0)	5 (4.7)
American robin *(Turdus migratorius)*	7 (5.2)	10 (5.6)
Warbling vireo *(Vireo gilvus)*	0 (0.0)	14 (9.8)
Yellow-rumped warbler *(Dendroica coronata)*	12b (5.9)	59a (14.5)
Dark-eyed junco *(Junco hyemalis)*	26b (8.5)	71a (20.0)
Pine grosbeak *(Pinicola enucleator)*	4 (2.4)	0 (0.0)
Total birds	156b (23.5)	345a (49.2)

[1]*Within species, means followed by no letter or the same letter are not different (P = 0.10).*

Table 3.—Numbers (standard error) per plot of small mammals caught in pole-sized lodgepole pine and small aspen stands on the Fraser Experimental Forest, Colorado.[1]

	Means of 1982 and 1983	
	Lodgepole pine	Aspen
Shrews *(Sorex sp.)*	0b (0.00)	0.1a (0.04)
Least chipmunk *(Tamias minimus)*	0.3 (0.07)	0.3 (0.08)
Deer mouse *(Peromyscus maniculatus)*	0.4 (0.08)	0 (0.00)
Southern red-back vole *(Clethrionomys gapperi)*	1.6b (0.26)	2.9a (0.29)
Montane vole *(Microtus montanus)*	0.1 (0.03)	0.4 (0.23)
Total mammals	2.4b (0.27)	3.7a (0.41)

[1]*Within species, means followed by no letter or the same letter are not significantly different (P = 0.10).*

near the study area, and from two other aspen study sites in western Colorado, suggest that the small aspen patches described here were too small to attract aspen obligate birds and mammals. These stands, although aspen dominated, do not have the area or the understories that are typical of well-developed, larger-sized stands.

However, based on numbers of species and numbers of birds and mammals, the small aspen stands studied here are much more productive than the lodgepole pine forest that surrounds them.

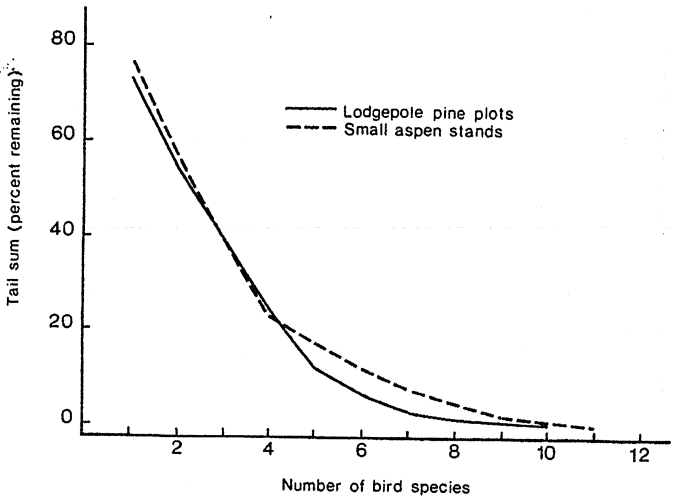

Figure 1.—Species diversity profile of bird communities in pole-sized lodgepole pine and small aspen stands on the Fraser Experimental Forest, Colorado. Lines in the profile that do not cross indicate differences in species diversity (Patil and Taille 1979).

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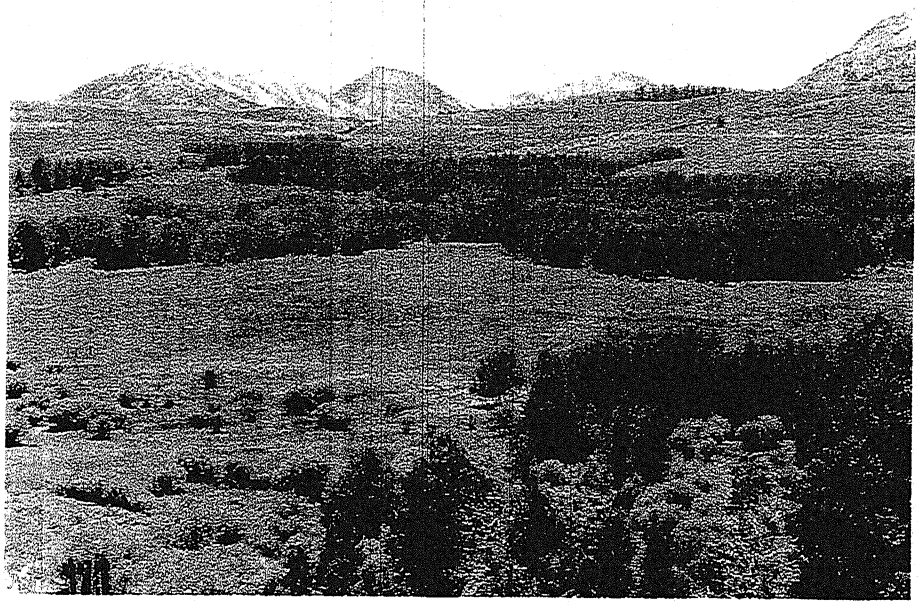
Ecology, Management, and Restoration of Aspen in the Sierra Nevada



Final Report
Collaborative Work Agreement
between
USDA, Forest Service
Lake Tahoe Basin Management Unit
and
USDA, Forest Service
Rocky Mountain Research Station

January 20, 2006

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Aspen regeneration—It appears that light grazing while the herbaceous understory is lush and succulent is less likely to damage aspen regeneration than grazing late in the season after the herbaceous plants cure and become less palatable (DeByle 1985a). We acknowledge, however, that early season grazing may cause more intense soil compaction, thereby limiting growth in many species. It has been shown that domestic livestock consume aspen with increasing pressure through summer and early fall as preferred forage decrease in volume and nutritional quality (DeByle 1985a; Fitzgerald and others 1986). This browsing can be very severe, especially on young succulent sprouts and can be site specific by all ungulates. Repeated heavy browsing will lead to bushy, multi-stemmed aspen shrubs, leaving them susceptible to browsing year after year until use ceases, or the aspen eventually disappear.

Conifer encroachment on rangelands—Much of the aspen rangeland in our area of interest is found in the fire-adapted ecosystems of the Sierra Nevada, Southern Cascades, and Modoc Plateau (Taylor 1998; Taylor 2000). Fire regimes began to change radically with the advent of intense grazing during the late 1800s and early 1900s. Grazing reduces "fine fuels" that would carry frequent low intensity fires through the understory of forests and shrubland ecosystems. The advent of fire suppression resulted in an initial increase in shrub components followed by conifer invasion that eventually led to the development of dense conifer canopy covers, which burn as high-intensity crown fires. Bartos (2001) documented loss of vegetative diversity in aspen forests because of fire suppression and subsequent conversion to conifer dominated systems. Conifer encroachment shades out understory plants, lowering plant cover, forage productivity, and species diversity. Mueggler (1988), in his study of aspen communities of the Intermountain West, found that production of understory herbaceous and shrubby cover decreased as conifer overstory increased. This effect becomes apparent when as little as 15 percent of the basal area is made up of conifers (Mueggler 1988). Furthermore, DeByle (1985b) reported that an Arizona study found aspen habitats contained 15 times more forage than adjacent ponderosa pine forests. Succession of aspen to conifers therefore results in a considerable lowering of the grazing capacity and a preference by both wild and domestic ungulates for aspen communities over conifer communities. This issue is significant because with no changes in stocking intensity, livestock and wildlife are forced to compete for diminishing resources.

Livestock/Wildlife Interactions—DeByle (1985a) summarized studies that showed removal of forage as well as disruption of nesting site cover had negative effects on both large and small mammals. Kie and others (1991) reported that deer increased their home range in California as cattle grazing increased. Loft and others (1987) found that deer and fawn hiding cover decreased with increases in browsing intensity by domestic livestock.

Livestock grazing is limited to stringer meadow, riparian areas, brush fields, and transitional areas in mixed-conifer forest at mid-elevations on the

western slope of the Sierra Nevada (for example, areas created after wildfires or plantations after clear cutting or small group selections). Many of the aspen on the Western slope of the Sierra Nevada are found in these stringer meadows and riparian areas. Loft and others (1993) found that as the grazing season progressed, cattle were attracted to the patchily distributed meadow-riparian and aspen habitats where herbaceous forage was most available. Therefore, we feel confident in stating that grazing management is a critical issue in aspen type in the Sierra Nevada and that timing and intensity of livestock use is critical as it relates to aspen regeneration and wildlife values.

A final consideration is that elk herds have recently become established in the Southern Cascades and Warner Mountains (Personal Comm., Mary Flores and Tom Rickman, Wildlife Biologists, USDA Forest Service, Modoc NF and Lassen NF). Elk can put added pressure on aspen habitats, especially when aspen are found on winter ranges (DeByle 1985c; Baker and others 1997; Suzuki and others 1999; Barnett and Stohlgren 2001). We conclude it is essential that management of both domestic and wild animals be closely linked to forage carrying capacity to avoid adverse effects on the aspen resource.

Forest Health Protection

Numbered Report 10 - 03 April 2010

Damage Agents and Condition of Mature Aspen Stands in Montana and Northern Idaho

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Introduction

Forest, range, and wildlife managers in the western United States have documented a 50–96% decline in total aspen (*Populus tremuloides* Michx.) forest acreage since European settlement (Bartos 2001). Data from the U. S. Department of Agriculture, Forest Service (USFS) Forest Inventory and Analysis (FIA) unit in Ogden, Utah suggest aspen acreages within Montana and Idaho are down 64% and 61% since settlement, respectively (Bartos 2001).

Aspen stand health has also shown declines since the 1970's. Two primary forces are most commonly cited as contributing to this decline; changes in fire regimes since European settlement and heavy ungulate browsing leading to inadequate regeneration (for example see Romme et al. 1995, Kay 1997, Bartos and Campbell 1998). More recently, severe and rapid dieback and mortality of aspen in Colorado, as well as Alberta, Saskatchewan, and Manitoba, Canada have been tied to drought (Hogg et al. 2008, Worrall et al. 2008). Forest diseases and insects are often notable as potential contributing or inciting factors (Frey et al. 2004) but play a largely undefined role in the decline of aspen.

Published data from long-term permanent monitoring plots established by the USFS FIA unit in Ogden confirmed the severity and extent of suspected decline symptoms and deterioration of aspen forests throughout its range in the Rocky Mountains from Canada to Mexico (Shaw 2004). The publication also recommended establishment of additional off-plot sites to further define extent and severity of decline in aspen clone health and examine the role of various damage agents.

Funding provided by USFS Evaluation Monitoring (project INT-F-06-01) allowed establishment of permanent monitoring plots in aspen stands in Nevada, Utah, southern Idaho and western Wyoming (USFS Region 4) in 2006 and 2007, and west and central Montana, and northern Idaho (USFS Region1) in 2008. Surveys were to supplement established FIA Forest Health Monitoring plot system efforts by providing additional data on forest damage/decline agents in aspen forests. Only results from Region 1 are reported here.

significantly between plots east and west of the CD (11.1 and 9.6%, respectively) (P>0.05) (Table 4), nor was mortality level related to elevation, slope position, clone stability, successional status, or degree of conifer competition. The highest percent of recent mortality (bark still tight on tree) was 29% found on one plot on the Lolo National Forest.

Shepperd (2008) describes sudden aspen decline (SAD) as rapid (1-2 years) mortality of mature trees with a lack of new sprouting after overstory mortality. SAD is not normal stand succession, but may be related to climate/drought. In some areas with SAD symptoms, a third or more of mature stems died within a couple of years. With 'recent', on-plot mortality at a 29% maximum (7% average), our plots do not fit the definition of SAD. In studies of SAD in Colorado and Arizona, higher mortalities were associated with lower elevations. Evaluation of the limited mortality observed in this survey failed to detect any association between recent mortality and elevation.

Aspen density averaged 626 trees per acre (tpa) across all plots and ranged from 140 to 2140 tpa (Table 4). For aspen ≥5 in. dbh, density was slightly higher in plots east of the CD (342 tpa) than west (285 tpa), though that difference was not statistically significant. Densities of aspen saplings (2-4.9 in. dbh), averaged 314 tpa, with averages of 281 tpa east of the CD and 345 tpa west of the CD. Plot elevation was positively correlated with aspen density (r = 0.24, P=0.037). Aspen density did not vary by slope position, clone stability, successional status, or degree of conifer competition.

DeByle (1985) notes that dense, even-aged stands of aspen that have at least 400 stems per acre when they reach 13 ft tall are usually healthy enough to withstand considerable tree losses. Bartos and Cambell (1998) describe stands having fewer than 500 regeneration stems per acre (5-15 feet tall) as being at risk. Stands under complete regeneration (e.g. clear-cut or stand replacing fire) are likely to have more stems than stands regenerating under a mature overstory. Thus, it is unclear if the 400-500-stem threshold is also appropriate for uneven-aged stands. Comparison is also difficult as our size classes were based on diameter rather than height. However, if we consider the number of saplings (2-4.9 in. dbh) a reasonable surrogate, 55 (72%) plots would not meet the 400 stem threshold: (11 of these registered no regeneration). If the regeneration <2 in. dbh but greater than 5 ft tall were included, the stems-per-acre count would be great enough to meet the threshold on many plots.

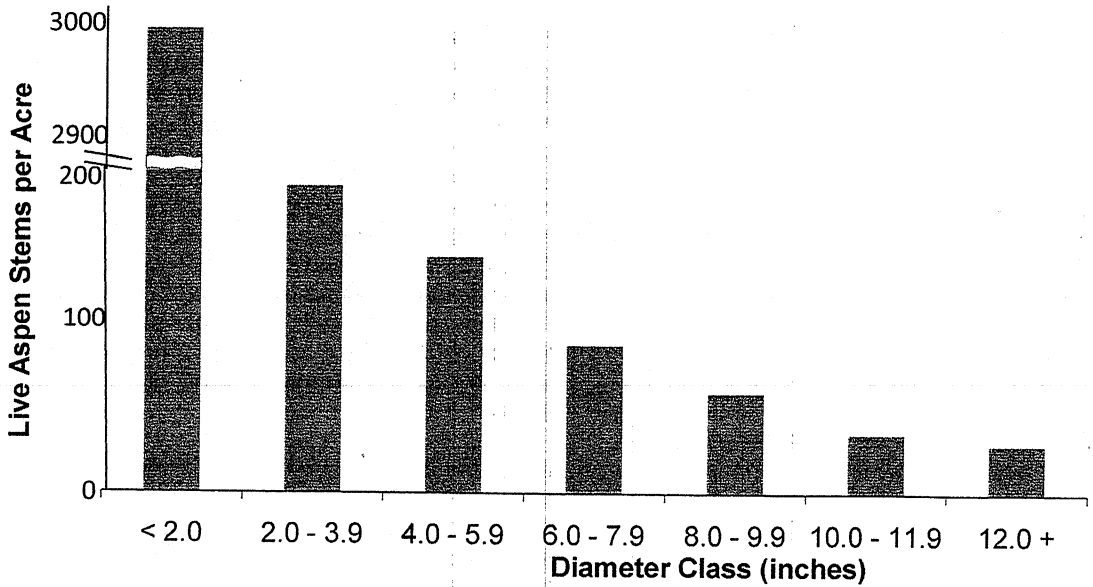

Figure. 3 . Mean stem density per acre of live aspen by 2-in. diameter classes in 76 permanent plots.

Regeneration

Aspen regeneration was present on 86% of plots. Aspen sprout densities ranged from 0 to 31,600 sprouts per acre (spa); 46% of plots had 1,500 or more spa. Plots east of the CD had an average of 3,511 spa while plots west of the CD had an average of 2,510 spa, but those densities were not significantly different. Although dead sprouts cannot be separated from live in our data, the percentage of dead was low (personal observation). Seedlings of tree species other than aspen were also present on 53% of plots with densities ranging from 0 to 5300 seedlings per acre. What constitutes a stand of healthy regeneration in terms of spa is unclear. However, with several plots having no measured regeneration and others with numbers well above 2,500, it is clear we have a range of conditions. Although the count of live and dead sprouts cannot be separated in this study, agent severity often reflects the level of mortality attributed to that agent; agents attributed with causing sprout mortality were given the highest severity ranking.

Principal Insects

Defoliating insects were found damaging aspen sprouts on 32% of plots, and 10% of all sprouts had damage from these leaf eaters (Fig. 10). Most defoliators on regeneration were the same as those seen in the canopy of saplings and trees (Fig. 6 A-C). As with the larger stems, we did not see outbreak activity in the regeneration by any of the defoliating insects during our survey.

Minor foliar insects were recorded on 9% of sprouts (Fig. 6 D-K). As noted with the trees and saplings, impact of these insects is relatively low. Their unusual activity can sometimes draw attention, and if present in large numbers on a small sprout they can have an impact on sprout health (Jones et al. 1985).

Principal Diseases

Aspen shoot blight (*Venturia macularis* (Fr:Fr) E. Muller & Arx) was recorded on 33% of plots and 16% of aspen sprouts (Fig. 10 and 11 A-C). Shoot blight differs from other foliar diseases in that it is not restricted to the leaf. It also causes dieback of new terminal shoots and can kill aspen suckers (Hinds 1985). Significantly greater proportions of aspen sprouts were damaged by shoot blight in plots east of the CD (19%) compared to plots west of the CD (3.5%) ($P<0.05$). Of the 25 plots where shoot blight was infecting aspen sprouts, 3 had associated damage rated at the highest severity indicating that shoot blight was causing sprout mortality.

Foliar diseases, including ink spot and Marssonina leaf blight (Fig. 8 A-C), were present on 51% of plots and affected 28% of aspen sprouts (Fig. 10). The severity of damage to regeneration by foliar diseases was rated as low or moderate on all plots.

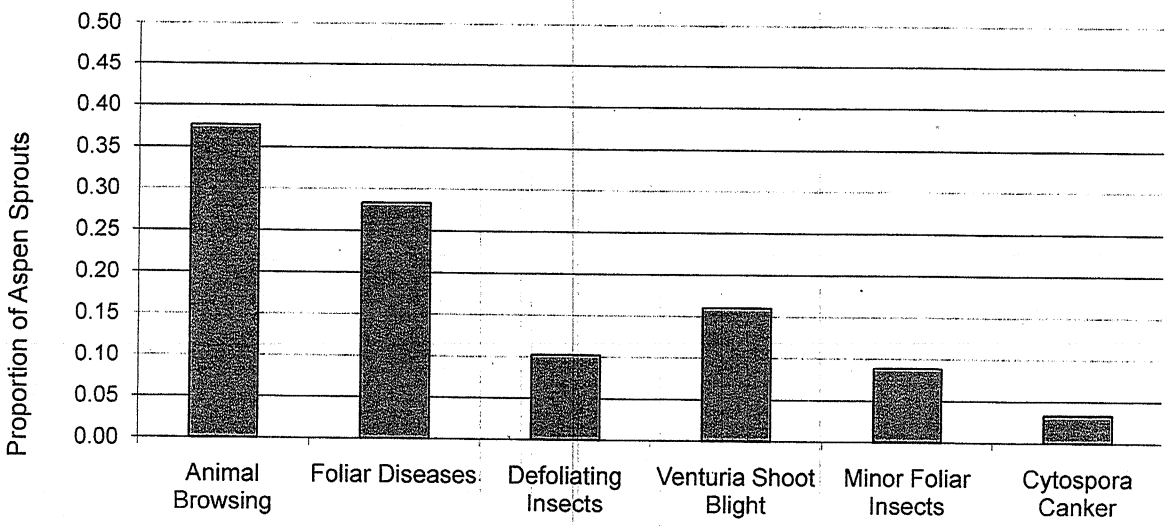

Figure 10. Most commonly recorded damaging agents on aspen sprouts in Montana and northern Idaho.

Conclusion

Our results suggest that rapid stand decline noted in other regions (SAD) is not prevalent in Montana and northern Idaho. We observed only low levels of mortality within plots, and it appeared to have occurred over many years rather than suddenly. Plot selection may have biased our surveys against dying stands, which likely would not have met the minimum threshold of 7 large stems within plot. However, in our travels we saw few stands with heavy, recent die-off. A couple stands south of Ennis, MT on the eastern edge of the Gravelly Range appeared to have most overstory stems with graying crowns, but no surveys were conducted due to lack of access. In the same general vicinity, NE of Earthquake Lake, other highly degraded stands were found to have significant infection by Ganoderma root disease. To better determine the health of our study stands, root condition surveys would probably need to be conducted (see Shepperd et al. 2001).

Although not sudden, Montana and northern Idaho appear to be experiencing aspen decline. In the absence of fire, advancing succession with increased conifer encroachment and aspen's increased susceptibility to diseases and insects has resulted in declining aspen stands. In addition, heavy ungulate grazing detected where regeneration was present, has resulted in stands that appear unable to regenerate themselves. If climate change results in less precipitation and higher temperatures, we could find recurring drought to be an important factor in future mortality (Rehfeldt et al. 2009; Hogg et al. 2008). Some authors suggest that extreme drought events provide a stimulus for mortality, adding the 'sudden' factor into previously slow declines (Rehfeldt et al. 2009).

Without fire, aspen stands in the West are often invaded by conifer species, which will predominate after 80-120 years (Hinds 1985; Mueggler 1994). Mixed aspen stands may have up to 50% of the stocking in coniferous species, without apparent detrimental effect to the below-ground root system. Above this level, impacts to aspen stem growth can be significant (Shepperd et al. 2001). In fact, conifer competition has been noted as a principle cause of decline in western aspen stands (Bartos 2001). West of CD, our aspen stands tended to be small, isolated patches surrounded by heavily encroaching conifers. Stands to the east had less conifer competition and tended to be larger in area. Nevertheless, Wirth and others (1996), evaluating aspen east of CD in southwest MT (Gravelly Range), found an approximately 47% decline in aspen area from 1947-1992, largely attributable to conifer invasion.

Sprouting can occur in stands with significant conifer competition. However, many of these areas also have high ungulate pressure and slower sprout growth rates, resulting in significant sprout loss and energy cost to the already stressed clone (DeByle 1985, Shepperd et al. 2001). These same ungulates can also cause bark damage from antler rubbing and bark chewing leading to damage and death of trees otherwise too large to be browsed. These stressors help lead to the ultimate demise of aspen within conifer stands.

Future work related to this study may include remeasurement of these permanent plots to determine change in aspen condition over time. A field guide to identify the many biological agents that affect aspen is also under construction and should be available in the near future. In addition, we are also collaborating with other studies to look at climate impacts on observed aspen mortality and decline throughout the western United States.

CHARACTERISTICS OF OLD-GROWTH FORESTS

IN THE INTERMOUNTAIN REGION

COMPILED BY

RONALD C. HAMILTON
REGIONAL GENETICIST

APRIL 1993

U.S.D.A. - FOREST SERVICE

SAF COVER TYPE
ASPEN
CODE 217

Species: *Populus tremuloides* - Quaking aspen

Description

Aspen is the most widely distributed forest type in North America. Over its extensive range it associates with a number of tree species. Generally, aspen forests are considered seral to conifer forests over much of its range. However, aspen forests are also noted for their stability. Aspen in the Intermountain Region constitutes a typical topo-edaphic climax at the lower elevations of the upper montane and subalpine forests. Extensive stands are usually attributed to repeated wildfires and in some instances to excessive grazing. Aspen is an aggressive pioneer usually dominating sites until replaced. That replacement process may take up to 1,000 years of fire exclusion.

In the Intermountain Region aspen is considered a permanent (stable) type on some sites. Conifers would invade the type if seed trees were available. In some instances the un-evenaged character in some aspen stands indicates a *de facto* climax.

The principal separation between seral and stable (*de facto* climax) aspen stands seems to be tied to the presence of specific indicator species, Muegeller (1988) in "Aspen Community Types of the Intermountain Region" identified types as: stable, seral to conifers, and grazing disclimax. The stable and grazing disclimax situations represent long-term persistent aspen stands where the conditions tied to the Forest Service generic definition of old-growth can be met.

Seral stands have vigorous and abundant conifer reproduction in the tree canopy layer that constitutes more than 10 percent of the layer. The stable stands have little or no conifer reproduction, thus less than 10 percent of the tree canopy is conifer. A grazing disclimax is much more difficult to determine, but is of little concern because of the lack of conifers.

Old-age is a relative term with aspen. Stands of aspen seldom persist for more than 200 years. However, root systems, where most regeneration occurs, persist up to 8,000 years, research suggests.

Aspen aerial shoots are moderately short lived. Life span is controlled primarily by site capability, elevation, and biologically damaging agents. Aspen stands older than 100 years are usually affected by heart and butt rotting fungi, plus species of several canker. These stable aspens stands are typically characterized as late mature or seral stage five (see appendix A). The time in this stage is usually short because of the number and virulence of pathogens.

Area of Application

The old-growth aspen definition applies only to the stable (aspen is both seral dominant and potentially climax-dominant) and grazing disclimax (severe alteration from past intense grazing) stands. The seral stands may contain what may qualify as old-growth, but its maintenance would require a treatment to remove the competing conifers. Stable and disclimax stands are located primarily in northern Wyoming, southeastern Idaho, Utah, and Nevada.

Standard Summary of Old-growth Characteristics

Vegetative Series: Quaking Aspen
SAF Cover Type: Aspen
Applicable Area: Southern Idaho, N. Wyoming, Utah, and Nevada
Site Productivity: N/A

Live Trees

Main canopy			Variation in tree diameter		Tree decadence	Tree canopy layers
DBH*	TPA*	Age*	6-in Classes	TPA-DBH	Number	
≥12	10 dry 20 mesic	100	≥2	N/A	≥2	

Dead Trees

Standing		Down	
DBH/ht ft	TPA	Small diameter (in)	Pieces/acre length in ft
≥10-15	2	8	≥10-10

Description of Attributes

Live trees

A 100-plus year old stand that contains trees greater than 12 inches in diameter; and ranges from 10 trees per acre on dry sites to 20 trees per acre on the more mesic environments. These large diameter trees dominate a canopy of smaller diameter trees that may overtop clumped or uniformly distributed seedling and sapling sizes. Two or more 6-inch diameter classes will be common.

Dead trees

On the average four snags at least 10 inches in diameter and 15 feet or greater in height per acre will occur in the stand. The snags would be created primarily by various fungi killing trees. Clumps of dead trees are common.

Dead material that is on or near the ground is composed primarily of 10 large trees broken or blown down that are larger than 8 inches at the small end and 10 feet in length.

Stand Size and Shape

Ten acres is the minimum area identifiable with normal inventory procedures. Viable old-growth stand sizes have not been determined at this point. Surrounding stand situations plus area management objectives should be considered in determining old-growth stand size.